COMMUNITY ECOLOGY 9(2): 217-227, 2008
1585-8553/$20.00 © Akadémiai Kiadó, Budapest
DOI: 10.1556/ComEc.9.2008.2.11

Impact of the Austrian Agri-Environmental Scheme on diversity of landscapes, plants and birds

T. Wrbka[1], S. Schindler[1,4], M. Pollheimer[1,2], I. Schmitzberger[1,2] and J. Peterseil[1,3]

[1] *Department of Conservation Biology, Vegetation & Landscape Ecology, University of Vienna, Rennweg 14, A-1030 Vienna, Austria*
[2] *coopNATURA, Consulting Engineers for Ecology and Nature Conservation, Kremstalstrasse 77, A-3500 Krems an der Donau, Austria*
[3] *Austrian Environmental Protection Agency, Spittelauer Lände 5, 1090 Vienna, Austria*
[4] *Corresponding author. E-mail: stefan.schindler@univie.ac.at, Tel.: +43 6504605771*

Keywords: Agriculture, Agri-environmental measures, Austria, Biodiversity, Birds, Landscape structure, Landscape change, monitoring, ÖPUL, Vascular plants.

Abstract: Agricultural management is a major driver of changes in floral and faunal species richness of anthropogenic landscapes. Counteracting the negative impact of industrialized agriculture by providing subsidies to farmers for environmentally friendly agricultural practices, agri-environmental schemes (AES) are the most important policy instruments to protect European biodiversity in agricultural landscapes. However, as they are rarely cost-effective, there is an urgent need for evaluation and improvement. To assess the environmental effects of the Austrian AES, we mapped landscapes and vascular plants in 1998 and 2003 and birds in 2003. The sampling areas were located in the three most important types of Austrian agricultural landscape, i.e., grassland in alpine valleys and basins, mixed agriculture in mountain areas, and eastern arable land. We investigated the agri-environmental measures (AEMs) in a parcel-wise manner and analyzed their effects on landscape values and biodiversity. Reduction of agrochemicals showed positive effects on biodiversity of vascular plants in grassland and birds in arable land. Targeted measures that directly address threatened species were most effective, but had much less coverage. Contradicting developments became apparent for landscape structure and ecological infrastructures, but effects of the AES were generally larger in simple than in complex landscapes. We conclude that AEMs are currently not targeted enough to effectively halt biodiversity losses, and recommend better regionalization by offering landscape-context specific measures, stronger focus on maintenance and improvement of landscape diversity, avoidance of counterproductive development, and improvement of the coverage of specific conservation measures.

Abbreviations: AEM–Agri-environmental measure(s), AES–Agri-environmental scheme(s), CAP–Common Agricultural Policy, ÖPUL–Österreichisches Programm für umweltgerechte Landwirtschaft (i.e., Austrian AES), SHDI–Shannon Diversity Index.

Nomenclature: Bauer et al. (2005ab).

Introduction

Mosaics of seminatural habitats characterize many European landscapes (Blondel and Aronson 1999, Ernoult et al. 2003, Billeter et al. 2008). Over time, many species of wildlife have adapted to these extensively managed and highly variable landscapes resulting in the development of many anthropogenetic species-rich ecosystems (Baldi et al. 2005, Carrete and Donázar 2005, Billeter et al. 2008). The primary habitat of some species has been lost entirely, with those species becoming dependent on secondary habitats for survival (Kleijn et al. 2006). During the last few decades, agricultural changes aimed at increasing the cost-efficiency of farming have had adverse effects on wildlife (Donald et al. 2001, Benton et al. 2003). These changes in farming practices and associated habitat loss, fragmentation and landscape homogenization have led to biodiversity loss (Wilson et al. 1999, Jongman 2002, Stoate et al. 2001). The growing awareness of the negative effects of intensive agricultural practices on the environment, coupled with the costs of regulating of agricultural markets have led to the introduction of agri-environmental schemes (AES) since the mid 1980s (Primdahl et al. 2003, Herzog et al. 2005).

By countering the negative impact of industrialized agriculture on the environment through the provision of financial compensation to farmers for environmentally friendly agricultural practices, AES are considered the most important instruments to protect biodiversity in agricultural landscapes (EEA 2004). Following the EU Common Agricultural Policy (CAP), all member states are obliged to implement agri-environmental programs (Kleijn et al. 2006). In the period 2007-2013, the indicated budget for the implementation of CAP is € 375 billion, with € 88 billion for the second pillar which is, in part, dedicated to the AES. Based on the Austrian share of these funds and additional national subsidies, € 0.6 billion are currently available for the Austrian AES ("ÖPUL") per year. The coverage is extensive with 75% of the farms and 88% of the agriculturally used area participating (BMLFUW 2007).

Numerous studies recommend that AES should be assessed from a landscape perspective (e.g., Abensperg-Traun et al. 2004, Tscharntke et al. 2005) and can be considered as valuable real world landscape experiments (Herzog 2005). However, impact and effectiveness of AES have rarely been thoroughly analyzed (see Kleijn and Sutherland 2003 for a review), and recent studies reported limited success and a need for improving the effectiveness of the AES (Kleijn et al. 2001, 2006, Herzog et al. 2005, Sepp et al. 2005, Tscharntke et al. 2005, Knop et al. 2006, Aviron et al. 2007). In line with this debate, and based on our findings (see Wrbka et al. 2004, Haberl et al. 2005), there was a need to evaluate the effectiveness of the current Austrian agri-environmental subsidies system in order to examine financial sinks and to propose improvements for the halt of biodiversity loss. We included an evaluation of effects on landscapes, including linear biotopes, nature value and land use diversity. As further indicators, we chose vascular plants and birds, considering them the most useful surrogate taxa for general biodiversity in Austrian agricultural landscapes (Sauberer et al. 2004, but see also Kati et al. 2004 for a Mediterranean region), and given their increasing comparability with other European evaluations (Kleijn and Sutherland 2003, Herzog et al. 2005, Kleijn et al. 2006). As bioindication at the level of communities has several advantages (King et al. 1998, Batáry et al. 2007a), we also included functional groups and character guilds (Wilson 1999) such as nitrophilous plants and ground breeding birds in our assessments. The aims of the current study are 1) to evaluate and compare the effects of the various agri-environmental measures on landscape, plant and bird diversity, and 2) to analyze the effectiveness of the AES in landscapes of different complexity (see also Tscharntke et al. 2005).

Methods

Study area

The study area was the entire Austrian agricultural landscape. The base data set consisted of 39 landscape plots of 1 km^2 which were selected according to a stratified sampling procedure (Peterseil et al. 2004) and initially investigated in 1998. Ten of these plots were selected (Fig. 1) to optimally represent the three main agricultural landscape types of Austria (i.e., grassland in alpine valleys and basins, mixed agriculture in mountain areas, and eastern arable land) and reinvestigated in 2003 (Table 1).

The sampling areas were further classified according to the complexity of their landscape (i.e., heterogeneity, texture, patch shape irregularity) and to their type and subtype of land use (Table 1, Fig. 2).

Sampling strategy and data collection

We followed two complementary sampling approaches. In the paired design, we compared spatial units with and without AEM, in the repeated measurement design we demonstrated changes under the influence of certain measures between the years 1998 and 2003.

Landscape and plant diversity. Landscape elements and vascular plants were mapped within each of the ten 1×1 km^2 sampling areas. In 1998 and 2003, field surveys were carried out using the same methods and during the same time of the year (April through to September), to ensure compatibility across the data. To characterize landscape diversity, we applied the SINUS approach (Peterseil et al. 2004), mapping each area completely and recording for the landscape elements habitat type (after Essl et al. 2004), type and intensity of land use, type of crop and nature value. (Nature value is a measure of naturalness, using the concept of hemeroby to assign the deviation of a given vegetation type from the potential natural vegetation based on standardized criteria and expert knowledge, Zechmeister et al. 2003.) Data on plant species richness were recorded by relevés using the Braun-Blanquet (1964) method. At least 20 vegetation relevés were completed in each sampling area in 1998 following the guidelines of the British Countryside Survey that recommends surveying at least one relevé per habitat type. Using field maps, plot descriptions, aerial photographs and a Garmin GPS with a positional accuracy of 5 m these plots were revisited in 2003 and the survey methods repeated. Minor spatial shifts in relocating the plots did not cause any bias, due to the known location of relevés in homogeneous habitats like cereal fields. These relevés were supplemented in

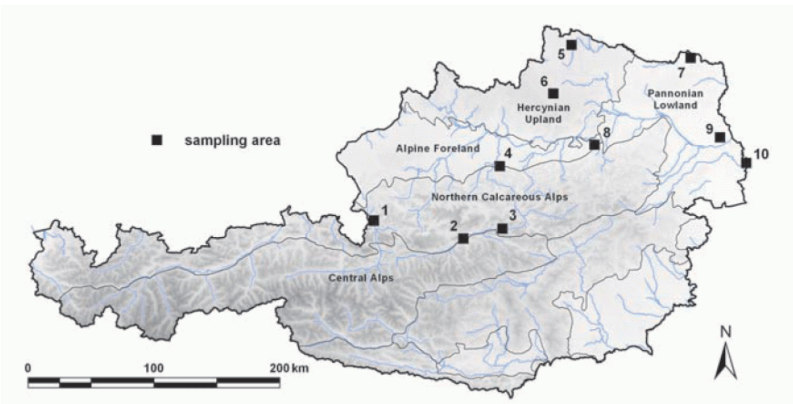

Figure 1. Locations of the ten sampling areas in the Austrian agricultural landscape.

Table 1. Characterization and classification of the sampling areas. Nr = Number in Figure 1.

Nr	Name of sampling area	Ecoregion	Type of agri-cultural landscape	Type of land use	Subtype of land use	Landscape complexity
1	Unterlangenberg	Alpine valleys and basins	Grasslands in valleys	Grasslands	live-stock farming	Intermediate
2	Irdning	Alpine valleys and basins	Grasslands in valleys	Grasslands	live-stock farming	Intermediate
3	Niederhofer	Alpine valleys and basins	Grasslands in valleys	Grasslands	live-stock farming	Intermediate
4	Post	Alpine valleys and basins	Mixed agriculture in mountainous areas	Grasslands	mixed agriculture	Intermediate
5	Edlitz / Thaya	Hercynian uplands	Mixed agriculture in mountainous areas	Arable Land - fine grain	mixed agriculture	Complex
6	Annatsberg	Hercynian uplands	Mixed agriculture in mountainous areas	Arable Land - fine grain	mixed agriculture	Complex
7	Zeiserlberg	Pannonian Lowlands	Eastern arable land	Arable Land - fine grain	cash cropping	Complex
8	Saudorf	Alpine foreland	Eastern arable land	Arable Land - coarse grain	fodder cropping	Simple
9	Teichhof	Pannonian Lowlands	Eastern arable land	Arable Land - coarse grain	cash cropping	Simple
10	Karlhof	Pannonian Lowlands	Eastern arable land	Arable Land - coarse grain	cash cropping	Simple

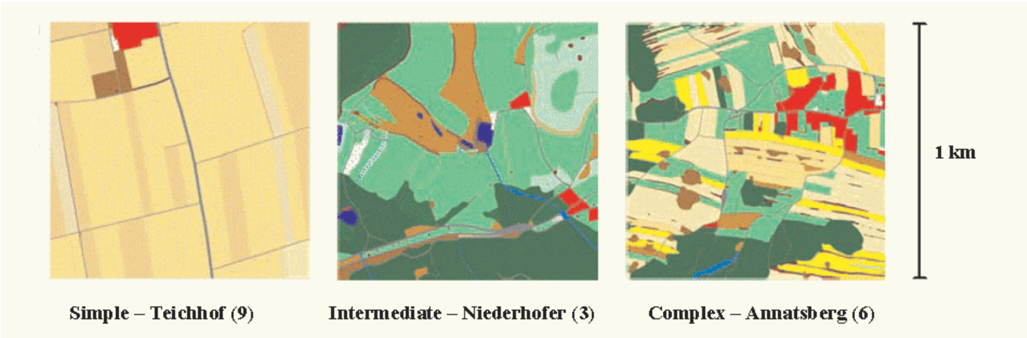

Figure 2. 1 km^2 sample areas of simple, intermediate and complex landscape structure.

2003 by 20 more relevés for simultaneous analyses. Following Braun Blanquet (1964), the relevé area was 25 m^2 in grasslands, 4 m^2 in arable land and elongated plots of 10×1 m^2 in linear features.

Bird census. To account for the high mobility of birds and to enlarge sample size, the sampling areas were extended to 3 km^2, encompassing the respective 1 km^2 area for mapping of landscape and plant diversity. Breeding birds were mapped during 3 visits between April and June 2003. Applying the territory mapping technique (c.f. Bibby et al. 1998), the route within each sampling area approached to within approximately 50 m of every point on the plot. Following this rule, three trained observers needed 5 hours each to cover the 3 km^2 of one sampling area. All bird observations were recorded on maps of 1:5000 scale with a high degree of precision and related to the investigated field parcels. For the Eurasian Skylarks (*Alauda arvensis*), start and landing points of singing males were used instead of song flights to assign more accurately to field parcels.

Data on agri-environmental measurements. Information on the status and type of AEM for each individual parcel of land was retrieved from the official database INVEKOS (Integrated Management and Control System). In total, there were 32 AEMs with different eligibility depending on the ecological and administrative context. To assess their effectiveness, we grouped them into bundles (and sub-bundles) depending on their expected effects on biodiversity. The sub-bundle "Specific conservation measures for species" (Table 2), for instance, includes set aside cropland and remnants of extensively managed grassland. Finally, we classified parcels of land according to the AEM bundles they participated in.

Data analysis and statistical treatment

Landscape diversity. To account for small scale differences in landscape characteristics and participation of farmers in the AES, sampling areas were divided into smaller raster grids of 0.25 km^2. This grid cell size reflected the average area of an Austrian farm (BMLFUW 2007). Depending on the share of area covered by the AEM "basic payment", each grid cell was assigned to one of three categories: <50% area covered, 50-75% and >75 %. Unevenness across these three categories was necessary because 88% of Austrian agricultural land is covered by AEM. The 40 raster grids across the 10 sampling areas were assigned to the categories grassland ($n = 16$), fine-grained arable land ($n = 12$) and coarse grained arable land ($n = 12$) according to the type of land use (Table 1). We used as indicators of landscape diversity: amount of several types of special linear biotopes, mean nature value of linear biotopes, amount of fallow land and mean nature value of the agriculturally utilized area, and Shannon Diversity Index (SHDI) of the land use types of the agriculturally utilized area and of the crop types of the arable land (see Table 3).

Table 2. Bundles of AEM evaluated in this study.

Expected Effect	Main bundles and sub-bundles of AEM
Conservation of small biotopes in agricultural landscape	Basic payment
Enhancement of diversity due to agricultural extensification	Reduction measures (including ecopoints and total renouncement)
	Reductions of pesticides & fertilizers in grassland
	Reduction of fertilizers in arable land
	Reduction of pesticides in arable land
	Renouncement of agrochemicals during critical periods
	Alternative classification:
	Total renouncement
	Ecopoints (reduction assumed)
	Reduction only, but no total renouncement
Conservation of endangered species and habitats	Specific conservation measures
	Specific conservation measures for species (e.g. set aside cropland)
	Specific habitat conservation measures - for bird habitat

These indicators were calculated for each raster grid and results were compared statistically between 1998 and 2003 using paired *t*-tests and Wilcoxon rank tests for paired samples.

Plant diversity. Each plant relevé was assigned to a land parcel (and therefore linked to the respective AEM bundles applied to each land parcel). Relevés were pooled for each subtype of land use (Table 1), i.e., live-stock farming, mixed agriculture – grassland, mixed agriculture – arable land, cash cropping and fodder cropping. The total number of species, the number of species listed in the Austrian Red List (Niklfeld and Schratt-Ehrendorfer 1999), the mean weighted N-value after Ellenberg et al. (1992) and Shannon's Diversity Index were calculated. Differences in trends (1998 vs. 2003) between plots in parcels with and without certain AEM bundles were analyzed using Mann-Whitney U-tests and simultaneous analyses (i.e., paired design, 2003 only) among several measures were done using ANOVA combined with Games Howell post hoc tests.

Bird diversity. Bird species were classified into character guilds based on their breeding strategy and according to their vulnerability status, i.e., Red List Austria (Frühauf 2005) and Species of European Conservation Concern (BirdLife International 2004). Special focus was given to the character guilds (following Bauer et al. 2005ab) of ground breeders, e.g., Grey Partridge (*Perdix perdix*), Eurasian Skylark, Whinchat (*Saxicola rubetra*), breeders within the herb layer, e.g., Marsh Warbler (*Acrocephalus palustris*), Reed Bunting (*Emberiza schoeniclus*), and breeders within small remnants of reed, e.g., Common Grasshopper-Warbler (*Locustella naevia*), and Sedge Warbler (*Acrocephalus schoenobaenus*). These communities serve as good indicators as they are heavily affected by agricultural treatment (e.g., mowing, harvesting). Densities of bird individuals on parcels with and without AEM in 2003 were compared using Mann-Whitney U-tests and significance values were corrected applying sequential Bonferroni correction.

ArcGIS (ESRI Inc.) and Microsoft ACCESS were used for management and manipulation of data and SPSS was used for the statistical analyses.

Results

Within the ten sampling areas mapped and classified, there were a total of 3023 (1998) and 3343 (2003) landscape elements, consisting of 1073 (1998) and 1323 (2003) agricultural fields and 1177 (1998) and 1286 (2003) small biotopes, with the remaining landscape elements predominantly field roads, forests, and urban land. We sampled 268 (1998) and 468 (2003) permanent plots for vascular plants and detected 827 plant species and 5478 territories (548 ± 194 per sampling area) of 100 breeding bird species.

Landscapes

From the indicators of landscape diversity used in this study, only few indicated significant effects of AEMs (Table 3). Comparing trends from 1998 to 2003 in raster grids with low (<50%) and high (>75%) shares of AEM, we detected for grasslands with low shares of AEM, a significant negative trend for small linear biotopes (paired *t*-test: $n = 6$ raster grids, $T = -3.28$, $p = 0.022$), grass dominated linear biotopes ($n = 6$, $T = -3.12$, $p = 0.026$) and mean nature value of the whole agriculturally utilized area ($n = 8$, $T = -2.24$, $p < 0.001$). In grassland areas with high shares of AEM, these significant negative trends could be alleviated for the mean nature value of the whole agriculturally utilized area ($n = 3$, $T = -1.32$, $p = 0.317$), eliminated for small linear biotopes ($n = 4$, $T = 2.06$, $p = 0.132$) and converted to a positive trend for grass dominated linear biotopes ($n = 4$, $T = 3.57$, $p = 0.035$). In the coarse grained arable land, we detected that the negative changes in linear small biotopes with few AEM area ($n = 2$, $T = -0.18$, $p = 0.889$) could be converted to a nearly significant positive trend in the grids with much AEM area ($n = 8$, $T = 2.30$, $p = 0.055$). Similar effects were detected for the mean nature value (areas <50% AEM: $n = 2$, $T = 2.00$, $p = 0.295$; areas >75% AEM: n = 8, F, p = 0.062) and the percentage of fallow land across the agriculturally utilized area (areas <50% AEM: $n = 2$, $T = 0.45$, $p = 0.655$; areas >75% AEM: $n = 8$, $T = 2.03$, $p = 0.043$).

On the other hand, several indicators of landscape diversity showed a loss of ecologically important habitats, regard-

Table 3. Trends in landscape values of Austrian agricultural landscapes between 1998 and 2003, comparing across 0.25 km^2 raster grids with different share of AES (basic payment). Low = <50% of area, med = 50-75 % and high >75%. "+" and "-" = direction of change between 1998 and 2003, "~" = no change at all. *n* is given for each table cell as superscript; Clusters: Grass = grasslands of intermediate complexity, Arable-complex = complex arable land – fine grained, Arable-simple = simple arable land – coarse grained. Significance levels of paired *t*-tests are indicated by * for $p < 0.05$, ** for $p < 0.01$ and *** for $p < 0.001$.

reference area	Indicator	Grass			Arable-complex			Arable-simple		
		low	med	high	low	med	high	low	med	high
linear biotopes	Total small biotopes	^{6}– *	^{6}+	^{4}+	0	^{9}~	^{3}–	^{2}–	^{2}+	^{8}+
	Wood dominated	^{6}+	^{4}+	^{4}~	0	^{9}–	^{3}–	^{2}~	^{2}+	^{3}+
	Grass dominated	^{6}– *	^{5}+ *	^{4}+ *	0	^{9}+	^{3}–	^{2}–	^{2}+	^{8}+
	Tree rows	^{3}–	^{4}–	^{3}–	0	^{4}+	^{3}–	^{2}~	^{2}–	^{4}–
	Water body	^{6}+	^{3}+	^{3}+	0	^{6}–	^{1}~	^{2}–	^{1}~	^{3}–
	Mean nature value	^{6}+	^{6}–	^{4}~	0	^{9}+ *	^{3}+	^{2}–	^{2}+	^{8}–
agriculturally utilized area	% of fallow land	^{6}–	^{5}+	^{4}+	0	^{9}+	^{3}+	^{2}~	^{2}+	^{7}+ *
	Mean nature value	^{8}– ***	^{5}~	^{3}–	^{5}–	^{4}–	^{3}–	^{2}~	^{2}+	^{8}+
	SHDI of land use types	^{8}+	^{5}+	^{3}+	^{5}~	^{4}~	^{3}+	^{2}~	^{2}+	^{8}–
arable land	SHDI of crop types	^{6}~	^{6}+	^{4}+	0	^{9}~	^{3}+	^{2}–	^{2}+	^{8}–

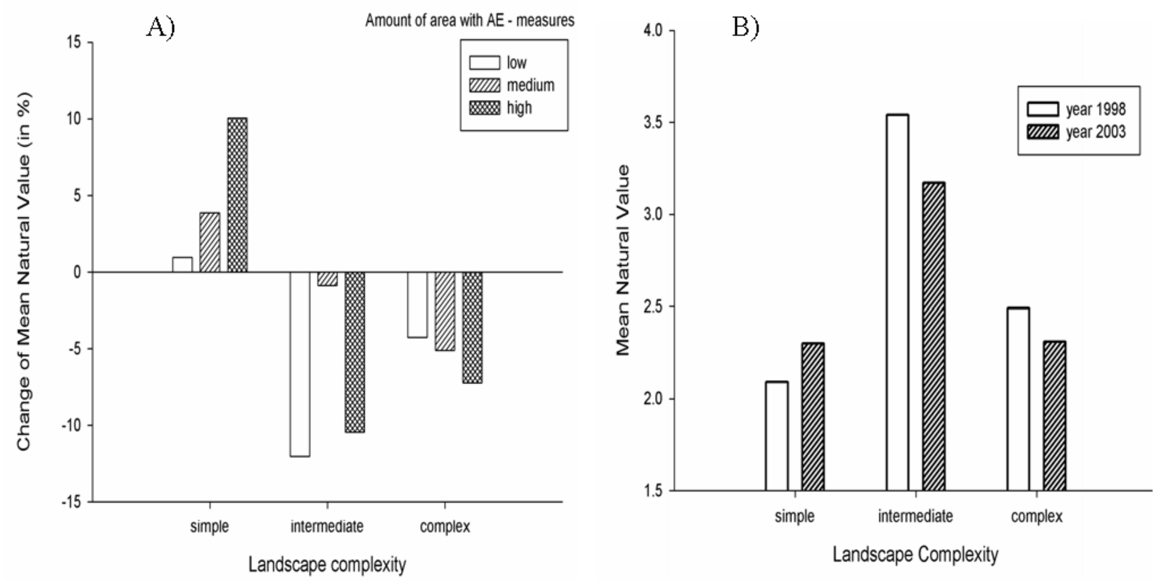

Figure 3. Effectiveness of the Austrian AES (basic payment) in landscapes of different complexity. Landscape units: 0.25 km^2 raster grids. **A**) Comparison of the effect of the amount of area with measures (low = < 50%; medium = 50-75%; high > 75%) on the change of the mean nature value between 1998 and 2003. *n* is ranging from 2 to 8 raster grids per vertical bar. **B**) Comparison of the absolute mean nature value for areas with > 75% AEM. *n* ranges from 3 to 8 per vertical bar.

less of the fact that >75% of the area was covered by the basic payment. In fine grained arable land, in particular 3 out of 4 types of linear biotopes decreased, although not significantly (Table 3). Other indicators that showed a probable, but not significant negative change from 1998 to 2003 in raster grids with a high share of AES included tree rows (in grasslands and arable land), linear water bodies (in coarse grained arable land), the mean nature value of linear biotopes (in coarse grained arable land) and of the entire agricultural utilized area (grassland and fine grained arable land) as well as the Shannon Diversity Index of land use and crop types in coarse grained arable land (Table 3).

Comparing the effects of subsidies in landscapes of different complexity, we detected that in very simple landscapes (coarse grained arable land) the highest effects were obtained, and in the most complex landscapes (fine grained arable land) the least effects. In the simple arable land the mean nature value of the raster grids with a high share covered by the AES (>75%) increased from 1998 to 2003, while it decreased in the complex arable landscapes (Fig. 3A). While in grasslands (intermediate complexity) the decline was less in grids with >75% AEM than in those with <50% AEM, in complex arable land the decline of the mean nature value was stronger in grids with a high share of measures than in those with a low share (Figure 3A). The absolute mean nature value was consistently higher in grasslands (intermediate complexity), because ploughing in arable fields lowers the value by one unit (Fig. 3B).

Table 4. Trends in plant communities in Austrian agricultural landscapes between 1998 and 2003 under the effect of different bundles of AEMs. "+" and "-" = direction of change between 1998 and 2003, "~" = no change at all. *n* = number of relevés, RL species = Number of Species included in the Austrian Red List of Plants, N-value = N-value after Ellenberg. Significance levels of Mann-Whitney U-tests are indicated by * for $p < 0.05$, ** for $p < 0.01$ and *** for $p < 0.001$.

AEM	Land use	n	Species Number	RL species	N-value	SHDI
Reduction measures (including renouncement, ecopoints)						
Reduction of pesticides & fertilizers in grassland						
	grassland - livestock farming – intensive	67	+ *	~	–	+ *
	grassland - mixed agriculture - extensive	54	+ *	~	– *	+ *
Reduction of fertilizers in arable land						
	arable land - mixed agriculture - extensive	38	+	~	~	~
	arable land - fodder cropping – intensive	21	+	~	~	~
	arable land - cash cropping – intensive	87	~	~	~	~
Reduction of pesticides in arable land						
	arable land - mixed agriculture - extensive	38	–	~		–
	arable land - fodder cropping – intensive	21	~	~		~
	arable land - cash cropping – intensive	87	~	~		~
Renouncement of agrochemical during critical periods						
	arable land - mixed agriculture - extensive	38	+ **	~	–	+ **

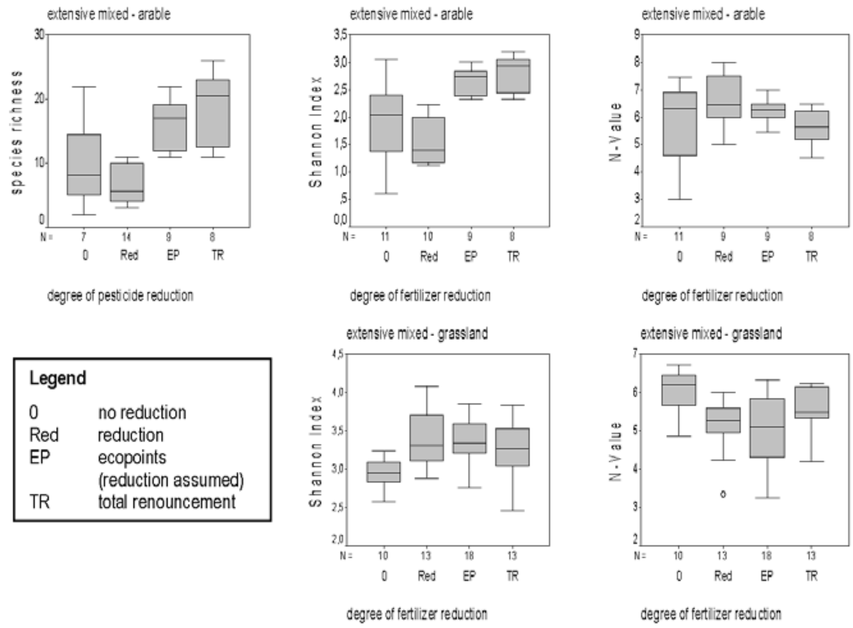

Figure 4. Simultaneous comparison of the effects of the sub-bundles "reduction", "ecopoints" and "total renouncement" (all included in the bundle "reduction") on species richness, diversity and N-value of vascular plants in arable and grassland areas of the cluster extensive mixed agriculture in mountainous areas (year 2003). The boxplots represent median, quartiles and the range of the distribution.

Vascular plants

A reduction of agrochemicals in grasslands led to an increase in the number of vascular plant species (Table 4), in both the intensive livestock farming cluster (*n* = 67 vegetation sampling plots, T = 2.35, *p* = 0.022) and in the extensive mixed agriculture cluster (*n* = 54, T = 2.20, *p* = 0.032). This increase is also reflected in the Shannon diversity index (intensive livestock farming: T = 2.21, *p* = 0.042, extensive mixed agriculture: T = 2.50, *p* = 0.002). In extensive mixed agriculture an effect on the nitrogen level could also be detected, the Ellenberg N-value being lower in parcels with reduction measures than the those without (T = –2.58, *p* = 0.013). In arable land, a slight positive effect from reductions of fertilizers on species richness could be shown in one cluster (extensive mixed agriculture), whereas in the more intensively used arable areas no effect of these measures could be detected. Nor did we detect any positive effect of the reduction of pesticides in arable land. However, most efficiently was "total renouncement in critical periods". This measure led to a highly significant increase in plant species richness (*n* = 38, T = 3.52, *p* = 0.004) accompanied by lover N-values and higher SHDI of plants (T = –3.05, *p* = 0.004) in the only cluster were it was applicable, i.e., extensive mixed agriculture (Table 4).

Table 5. Effects of Austrian Agro-environmental measures on breeding birds. Pairwise comparison of parcels of land with (+M) and without (-M) a certain AEM. n = number of parcels of land. Values represent the mean number of bird individuals per 10 ha ± SE. RL = Red List, SPEC = Species of European Conservation Concern. Significance levels of Mann-Whitney U-tests are indicated after sequential Bonferroni correction by * for $p < 0.05$, ** for $p < 0.01$ and *** for $p < 0.001$.

Bundles of AEMs	cluster	n		RL Austria			SPEC			Ground breeder			Breeder of the herb layer			Reed breeder		
		-M	+M	-M	+M	p	-M	+M	p	-M	+M	p	-M	+M	p	-M	+M	p
reduction of production means	arable	409	428	1.8 ±0,3	2,5 ±0,3	*	10,3 ±0,9	13,9 ±0,9	***	13,2 ±1,1	16,1 ±1.0	***	2,3 ±0.5	1.6 ±0.4	ns			
	grass	71	316	1,3 ±0,4	2,6 ±0,3	ns	2,9 ±0,7	2,2 ±0,3	ns	4,9 ±1,0	4,4 ±0,5	ns	1.8 ±0.8	2.6 ±0.5	ns	0.6 ±0.4	0.2 ±0.01	ns
Specific conservation measures for species	arable	781	56	2,1 ±0,2	3,2 ±0,9	ns	11,1 ±0,6	27,6 ±4,0	***	13,7 ±0,7	28,6 ±4,0	**						
	grass	229	24	2,7 ±0,4	8,9 ±4,2	ns	1,8 ±0,3	3,6 ±2,1	ns	4,3 ±0,6	7,1 ±3,3	ns	3,1 ±0,8	11,1 ±6,2	ns	2,8 ±0,8	11,3 ±4,9	***
Specific habitat conservation measures for bird habitats	arable	314	13	0,9 ±0,7	2,5 ±0,3	ns	7,8 ±4,0	13,9 ±1,0	ns	8,0 ±4,4	16,6 ±1,0	*						
	grass	41	45	2,6 ±0,4	10,2 ±4,6	ns	1,8 ±0,3	4,2 ±2,3	ns	4,3 ±0,6	8,1 ±3,6	ns	3,3 ±0,7	7,8 ±5,3	ns	0,3 ±0,1	3.0 ±3.0	ns

Comparison reduction vs. renouncement. For the cluster mixed agriculture in mountainous areas, we analyzed the effects of the degree of agrochemical reduction on vascular plants, applying the simultaneous approach and data from 2003 (Fig. 4). The bundle included "reduction only", "total renouncement" and "ecopoints" (degree of reduction not clearly defined) of fertilizer and pesticide application. When comparing their effects on bioindicators in grasslands, the variance of the control group was small and all types of reduction led to an improvement. In arable land, the variance of the control group was very large, with "ecopoints" and "total renouncement" performing slightly better than the control group and "reduction only" performing slightly worse (Fig. 4). There "total renouncement" was statistically significantly outperforming "reduction only" with regard to species richness (Games-Howell post-hoc: p = 0.001) and SHDI (p = 0.001).

Birds

The results for birds suggested that effects of the AEMs were generally stronger in arable fields than in grasslands. In arable fields, even the less specific "reduction measures" led to significantly higher densities of ground breeders and endangered species. The effects of the specific conservation measures were clear, although sometimes not statistically significant due to the very low number of parcels that participated in these measures (Table 5). In contrast, in grasslands only the bundle "specific conservation measures for species" led to significantly higher densities of reed breeders. The bundle "specific habitat conservation measures for bird habitats" led to higher densities of species included in the Red List Austria, but the p-value of 0.039 was not significant after sequential Bonferroni correction.

Discussion

Some measures widely used in Austria showed clearly positive effects such as the reduction of agrochemicals on plant diversity in grasslands and on bird density in arable land. The general pattern that we detected suggests that the more specific a measure, the more positive the effect, but the lesser the coverage. The most important among such specific measures were total renouncement of agrochemicals during critical periods and set-aside of cropland (in the bundle specific conservation measures). Set-aside of land is also proved to be especially effective by various other studies (Marggraf 2003, Van Buskirk and Willi 2004, 2005, Askew et al. 2007, Billeter et al. 2008).

Landscapes

Landscape heterogeneity is an important driver of biodiversity in agroecosystems (Wrbka et al. 1999, Benton et al. 2003, Billeter et al. 2008). This study has uncovered contradictory trends across different landscape indicators. The increase of setting aside of agricultural land contributed to positive effects and in arable landscapes in particular, habitat diversity was greatly enhanced with this measure. However, the loss of linear infrastructures in some areas despite AEMs proved to be the main shortcoming as evidenced by this research. Fine grained arable land, characterized by very complex landscape structure, did not benefit from the AES due to ongoing land consolidation processes in richly structured cultural landscapes of peripheral regions. A further reason for the assessed inefficiency could be that the control group without AEM has a high variance for many indicators (see Fig. 4), which may be caused by traditionalist and part-time farmers who do not participate in the subsidy system (Schmitzberger et al. 2005). This limitation could be addressed through a tailored focus of the scheme, particularly given that such land is often rich in biodiversity but prone to intensification, land abandonment and uncertain succession (Schmitzberger et al. 2005, Knop et al. 2006, Billeter et al. 2008).

Another obvious reason for the low efficiency in complex landscapes was the high starting value of landscape indicators (Wrbka et al. 2005). In complex landscapes, a net gain in naturalness was difficult to obtain, while in simple landscapes even soft measures like a reduction of fertilizers can lead to net gains and significant improvements. By comparing the conceptual model suggested by Tscharntke et al. (2005) who describe the response of the effectiveness of AES to landscape complexity as a hump-backed curve with our findings a different relationship is revealed. In our sample, which includes the most simple as well as the most complex agricultural landscapes of Austria, the effectiveness of the measures is declining continuously with increasing landscape complexity (Fig. 3). Generally, due to historical, topographical and biogeographical reasons some northwest European concepts are not compatible with other parts of Europe (Baldi et al. 2005, Kleijn and Báldi 2005, Batáry et al. 2007b, Billeter et al. 2008). The difference may be caused by the specific situation in northwest European countries, where large farm size (Eurostat in BMLFUW 2007) combined with a high level of agricultural industrialization, a dramatic increase in disturbance and a dramatic decrease of biodiversity over the last century (e.g., Maes and Van Dyck 2001).

Plants

As plant species richness and plant diversity indices showed positive effects of reduction measures in grasslands, we interpret these findings as a community rather than a single species response. Red List plant species richness appeared to be a suboptimal indicator for uncovering the effects of AEMs, because the agricultural flora consists mainly of ruderals and nitrophilous plants, while specialist Red List species are rarely present (Herzog et al. 2005, Kleijn et al. 2006). Generally, an increase in biodiversity of rather common species is reached relatively easily, while this is not generally the case for endangered species (Kleijn et al. 2006). We conclude that the community ecological approach and the development of a specific "leitbild" for the management of landscapes and biotic communities should benefit the efficiency of subsidy systems and their evaluations (Potschin and Haines-Young 2003, Bastian 2004). Comparing the impact of the different measures, the most targeted bundle "renouncement of agrochemicals during critical periods" appeared to perform the best. These results are in line with findings of other research groups across Europe (e.g., Herzog et al. 2005). Especially in the more extensively managed mountainous areas a simple reduction of fertilizers applied to arable land had no positive effects on plant species diversity, whereas total renouncement was reflected by higher plant diversity. In such marginal areas, the commitment to "reduction of fertilizers" did not result in a real change in management intensity, because the poorer soils of these areas do not reward higher fertilizer application.

The slightly lesser effects of the AES on plants than on birds could partly have been caused by the weather conditions. Whereas the early summer was hot and wet in 1998, it was hot and extremely dry in 2003 (ZAMG 2003). This resulted in a low rate of establishment of arable weeds, and even in conventional plots, little application of herbicides was necessary in 2003. Due to their mobility, birds should react faster to habitat changes than plants and can potentially obtain positive effects from a broader share of landscape, but comparable studies revealed much more pronounced effects on plants than on birds (Kleijn et al. 2006). Repeated monitoring schemes would minimize the effects from variations of weather on the results.

Birds

In Austrian landscapes, species richness of endangered birds correlates positively with the human appropriation of net productivity of ecosystems (Haberl et al. 2005). The commonly applied reduction measures have significant positive effects on the tested bird communities in arable fields, but not in grassland. One explanation is that reduction of pesticides in arable land may lead to a sparser and less homogenous crop sward structure and higher densities of prey animals and thus allow for higher densities of breeding and foraging birds (Vickery et al. 2001, Donald et al. 2002, Kleijn and Sutherland 2003, Wilson et al. 2005). Postponed harvesting due to reduced fertilizers is also of significant advantage for several character species (e.g., Grey Partridge, Eurasian Skylark) that mainly start breeding in March and April. Further explanations can be found in an overview in Benton et al. (2003, Fig. 1). Alternatively, characteristic grassland species such as Whinchat, Common Grasshopper-Warbler and Marsh Warbler normally breed in May and June. Thus, first mowing of grasslands delayed by the reduction measure to May, is not postponed enough, and such measures could also lead to ecological traps, providing grasslands which appear attractive to birds initially, but which are ultimately unsuitable for successful breeding (Müller et al. 2005, Tanneberger et al. 2008). Depending on geographical position and altitude, postponing first mowing to no earlier than July would enable a successful breeding for most Austrian lowland populations of the Whinchat (cf. Müller et al. 2005). The only AEMs that clearly promoted the investigated grassland bird communities are bundled to "Specific habitat conservation measures for bird habitat". But these very specific measures have such limited coverage that they cannot guarantee the continuous survival of populations. However, the few parcels with such measures indicate strong positive effects towards sustainable agriculture, and we highly recommend that increased participation of farmers in these measures should be encouraged.

Implications for biodiversity management, AES design and future perspectives

Increased specificity of the AEMs is a key recommendation from this study for improving the effectiveness of the AES (see also Báldi et al. 2005, Herzon and O'Hara 2007). Also Gottschalk et al. (2007) detected contrasting effects and concluded that the heterogeneous responses of birds and carabids at different localities suggest the need for spatially targeted subsidy schemes. Clearly though there is a trade-off between ecological advantage of highly specific measures and the effort required for their evaluation. The Austrian AES is very complex with several detailed measures that were pooled to bundles in this study for comprehensiveness and comparability with other studies. In some cases, the bundles included measures with contradictory effects on certain indicators or in certain areas, and this was largely unavoidable. For instance, the bundle reduction combines “reduction” and “total renouncement”, two measures that had different effects of plant diversity for extensive mixed arable land (see Fig. 4). However, our results suggest that complexity of a specified and targeted scheme in a landscape perspective and regional context leads to higher effectiveness (Abensperg-Traun et al. 2004, Kleijn and Báldi 2005, Aviron et al. 2007, Knop et al. 2006, Batáry et al. 2007abc). There should be the sufficient political interest for targeted schemes and for sound evaluations. In our study design we tried to account for the sampling effect (i.e., measures being preferentially located in parcels with high biodiversity, see Kleijn and Sutherland 2003) by including temporal evaluations. However, robustness and precision of assessments and cost-efficiency of schemes could be raised by evaluations at broader spatial scales and in shorter temporal intervals (Carey 2001, Carey et al. 2003, Kleijn and Sutherland 2003, Kleijn et al. 2006, Carey and Pywell 2007).

Currently, cost efficiency of the Austrian AES is low and it is not targeted adequately to effectively halt biodiversity losses and landscape simplification. We recommend stronger focus on landscape-context specific measures, maintenance and improvement of landscape diversity, avoidance of counterproductive development and improvement of the participation of farmers in specific conservation measures. Higher effectiveness can also be obtained by incentives for the achievement of standardized goals (Herzog and Walter 2005) and not solely for paying farmers for the application of lesser targeted measures. However, a very efficient measure seems to be the set aside of proportions of used land, especially in simple structured and intensive use agricultural areas. Recent changes in EU policy to diminish set aside areas for promoting biofuel production, could contribute considerably to the miss of the 2010 target to halt biodiversity loss (Billeter et al. 2008). This policy serves as good example of the lack of awareness regarding the consequences of precipitous global change mitigation measures for biodiversity, ecosystem functions and services (EPBRS 2008).

Acknowledgements: We thank G. Zethner for project coordination, A. Stocker-Kiss, C. Ott, B. Thurner, J. Pollheimer, J. Oberwalder and A. Danzl for field work and A. Bartel, E. Schwaiger and G. Banko for data acquiring and management especially regarding the AEM. We are also grateful to M. Prinz for support in preparing the figures and F. M. Grünweis, M. Pavlicev, C. Renetzeder, O. G. Schindler, and two anonymous reviewers for their helpful comments on former manuscript versions. In particular, we thank N. Wales for a sorrow review of the English and many linguistic improvements. This project was funded by the Austrian Federal Ministry for Agriculture and Forestry, Environment and Water (BMLFUW).

References

Abensperg-Traun, M., T. Wrbka, G. Bieringer, R. Hobbs, F. Deininger, B. York Main, N. Milasowszky, N. Sauberer and K.P. Zulka. 2004. Ecological restoration in the slipstream of agricultural policy in the old and new world. *Agric. Ecosyst. Environ.* 103:601-611.

Askew, N.P., J.B. Searle and N.P. Moore. 2007. Agri-environmental schemes and foraging of barn owls (*Tyto alba*). *Agric. Ecosyst. Environ.* 118:109-114.

Aviron, S., Ph. Jeanneret, B. Schüpbach and F. Herzog. 2007. Effects of agri-environmental measures, site and landscape conditions on butterfly diversity of Swiss grassland. *Agric. Ecosyst. Environ.* 122:295-304.

Báldi, A., P. Batáry and S. Erdős. 2005. Effects of grazing intensity on bird assemblages and populations of Hungarian grasslands. *Agric. Ecosyst. Environ.* 108:251-263.

Bastian, O. 2004. Functions, leitbilder and red lists – expression of an integrative landscape concept. In: J. Brandt and H. Vejre (eds), *Multifunctional Landscapes Vol. I. Theory, Values and History.* WITPress Southampton, Boston. pp. 75-94.

Batáry, P., A. Báldi and S. Erdős. 2007a. Grassland versus non-grassland bird abundance and diversity in managed grasslands: local, landscape and regional scale effects. *Biodivers. Conserv.* 16:871-881.

Batáry, P., A. Báldi and S. Erdős. 2007b. The effects of using different species conservation priority lists on the evaluation of habitat importance within the Hungarian grasslands. *Bird Conserv. Internat.* 17:35-43.

Batáry, P., K.M. Orci, A. Báldi, D. Kleijn, T. Kisbenedek and S. Erdős. 2007c. Effects of local and landscape scale and cattle grazing intensity on Orthoptera assemblages of the Hungarian Great Plain. *Basic Appl. Ecol.* 8:280-290.

Bauer, H.-G., E. Bezzel and W. Fiedler. 2005a. *Das Kompedium der Vögel Mitteleuropas. Alles über Biologie, Gefährdung und Schutz. Band 1: Nonpasseriformes – Nichtsperlingsvögel.* Aula-Verlag, Wiesbaden.

Bauer, H.-G., E. Bezzel and W. Fiedler. 2005b. *Das Kompedium der Vögel Mitteleuropas. Alles über Biologie, Gefährdung und Schutz. Band 2: Passeriformes – Sperlingsvögel.* Aula-Verlag, Wiesbaden.

Benton, T.G., J.A. Vickery, J.A. and J.D. Wilson. 2003. Farmland biodiversity: is habitat heterogeneity the key? *Trends Ecol. Evol.* 18:182-188.

Bibby, C., M. Jones and S. Marsden. 1998. *Expedition Field Techniques. Bird Surveys.* Expedition Advisory Centre, Royal Geographical Society, London.

Billeter, R., J. Liira, D. Bailey, R. Bugter, P. Arens, I. Augenstein, S. Aviron, J. Baudry, R. Bukacek, F. Burel, M. Cerny, G. De Blust, R. De Cock, T. Diekötter, H. Dietz, J. Dirksen, C. Dormann, W. Durka, M. Frenzel, R. Hamersky, F. Hendrickx, F. Herzog, S. Klotz, B. Koolstra, A. Lausch, D. Le Coeur, J.P. Maelfait, P. Opdam, M. Roubalova, A. Schermann, N. Schermann, T. Schmidt, O. Schweiger, M.J.M. Smulders, M. Speelmans, P. Simova, J. Verboom, W.K.R.E. van Wingerden, and M. Zobel. 2008. Indicators for biodiversity in agricultural landscapes: a pan-European study. *J. Appl. Ecol.* 45:141-150.

BirdLife International. 2004. *Birds in Europe: Population Estimates, Trends and Conservation Status. BirdLife Conservation Series No. 12.* Cambridge, UK.

Blondel, J. and J. Aronson. 1999. *Biology and Wildlife of the Mediterranean Region.* Oxford University Press, Oxford.

BMLFUW (Federal Ministry for Agriculture, Forestry, Environment and Water). 2007. Grüner Bericht 2007. Vienna, The Republic of Austria. www.gruenerbericht.at

Braun-Blanquet, J. 1964. *Pflanzensoziologie. Grundzüge der Vegetationskunde*. 3. Auflage, Springer-Verlag, Wien.

Carey, P.D. 2001. Schemes are monitored and effective in the UK. *Nature* 414:687.

Carey, P.D. and R.F. Pywell. 2007. An up-to-date benefit analysis of English agri-environmental schemes: their impact at the landscape scale and the cost of adequate monitoring. In: R.G.H Bunce, R.H.G Jongman, L. Hojas and S. Weel (eds), *25 Years of Landscape Ecology: Scientific Principles in Practice. Proceedings of the 7th IALE World Congress, Part 1, IALE Publication Series 4.* Wageningen, The Netherlands. pp. 70-71.

Carey, P.D., C. Short, C. Morris, J. Hunt and A. Priscott. 2003. The multi-disciplinary evaluation of a national agri-environment scheme. *J. Environ. Manage.* 69:71-91.

Carrete, M. and J.A. Donázar. 2005. Application of central-place foraging theory shows the importance of Mediterranean dehesas for the conservation of the cinereous vulture, *Aegypius monachus*. *Biol. Conserv.* 126:582–590.

Donald, P.F., R.E. Green and M.F. Heath. 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. *Proc. Roy. Soc. Lond.* B. 268:25-30.

Donald, P.F., G. Pisano, M.D. Rayment and D.J. Pain. 2002. The Common Agricultural Policy, EU enlargement and the conservation of Europe's farmland birds. *Agric. Ecosyst. Environ.* 89:167-182.

EEA. 2004. *High Nature Value Farmland – Characteristics, Trends and Policy Challenges.* European Environmental Agency, Copenhagen.

Ellenberg, H., H.E. Weber, R. Düll, V. Wirth, W. Werner and D. Paulissen. 1992. Zeigerwerte von Pflanzen in Mitteleuropa. *Scripta Geobotanica Vol. 18*, Göttingen.

Ernoult, A., F. Bureau and I. Poudevigne. 2003. Patterns of organization in changing landscapes: implications for the management of biodiversity. *Landsc. Ecol.* 18:239–251.

Essl, F., G. Egger, G. Karrer, M. Theiss and S. Aigner. 2004. *Rote Liste der gefährdeten Biotoptypen Österreichs. Grünland, Grünlandbrachen und Trockenrasen, Hochstauden- und Hochgrasfluren, Schlagfluren und Waldsäume, Gehölze des Offenlandes und Gebüsche. Monographien Umweltbundesamt, vol. 167.* Wien.

EPBRS (European Platform for Biodiversity Research Strategies). 2008. Recommendations of the meeting of the European Platform for Biodiversity Research Strategy held under Slovenian Presidency of the EU. 15-18 January 2008, Brdo, Slovenia. http://www.epbrs.org.

Frühauf, J. 2005. Rote Liste der Brutvögel (*Aves*) Österreichs. In: K. P. Zulka (ed.), *Rote Listen gefährdeter Tiere Österreichs. Checklisten, Gefährdungsanalysen, Handlungsbedarf.* Böhlau Verlag, Wien. pp. 63-165.

Gottschalk, T.K., T. Diekötter, K. Ekschmitt, B. Weinmann, F. Kuhlmann, T. Purtauf, J. Dauber and V. Wolters. 2007. Impact on agricultural subsidies on biodiversity at the landscape level. *Landsc. Ecol.* 22:643-656.

Haberl, H., C. Plutzar, K.-H. Erb, V. Gaube, M. Pollheimer and N.B. Schulz. 2005. Human appropriation of net primary production as determinant of diversity in Austria. *Agric. Ecosyst. Environ.* 110:119-131.

Herzog, F. 2005. Agri-environmental schemes as landscape experiments. *Agric. Ecosyst. Environ.* 108:175-177.

Herzog, F., S. Dreier, G. Hofer, C. Marfurt, B. Schlüpbach, M. Spiess and T. Walter. 2005. Effects of ecological compensation areas on floristic and breeding bird diversity in Swiss agricultural landscapes. *Agric. Ecosyst. Environ.* 108:189-204.

Herzog, F. and T. Walter. 2005. *Evaluation der Ökomassnahmen, Bereich Biodiversität. Schriftenreihe der FAL, 56.* Agroscope FAL Reckenholz, Zürich.

Herzon, I. and R.B. O'Hara. 2007. Effects of landscape complexity on farmland birds in Baltic States. *Agric. Ecosyst. Environ.* 108:297-306.

Jongman, R.G.H. 2002. Homogenisation and fragmentation of the European landscape: ecological consequences and solutions. *Landscape Urban Plann.* 58:211-221.

Kati, V., P. Devillers, M. Dufrene, A. Legakis, D. Vokou and Ph. Lebrun. 2004. Testing the value of six taxonomic groups as Biodiversity Indicators at a local scale. *Conserv. Biol.* 18:667-675.

King, J.R., Andersen, A.N. and A.D. Cutter. 1998. Ants as bioindicators of habitat disturbance: validation of the functional group model for Australia's humid tropics. *Biodivers. Conserv.* 7:627-638.

Kleijn, D. and A. Báldi. 2005. Effects of set-aside land on farmland biodiversity: comments on Van Buskirk and Willi. *Conserv. Biol.* 19:963-966.

Kleijn, D., R.A. Baquero, Y. Clough, M. Díaz, J. De Esteban, F. Fernández, D. Gabriel, F. Herzog, A. Holzschuh, R. Jöhl, E. Knop, A. Kruess, E.J.P. Marshall, I. Steffan-Dewenter, T. Tscharntke, J. Verhulst, T.M. West and J.L. Yela. 2006. Mixed biodiversity benefits of agri-environmental schemes in five European countries. *Ecol. Lett.* 9:243-254.

Kleijn, D., F. Berendse, R. Smit and N. Gilissen. 2001. Agri-environmental schemes do not effectively protect biodiversity in Dutch agricultural landscapes. *Nature* 413:723-725.

Kleijn, D. and W.J. Sutherland. 2003. How effective are European agri-environmental schemes in conserving and promoting biodiversity? *J. Appl. Ecol.* 40:947-969.

Knop, E., D. Kleijn, F. Herzog and B. Schmid. 2006. Effectiveness of the Swiss agri-environmental scheme in promoting biodiversity. *J. Appl. Ecol.* 43:120-127.

Maes, D. and H. Van Dyck. 2001. Butterfly diversity loss in Flanders (north Belgium): Europe's worst case scenario? *Biol. Conserv.* 99:263-276.

Marggraf, R. 2003. Comparative assessment of agri-environmental programmes in federal states of Germany. *Agric. Ecosyst. Environ.* 98:507-516.

Müller, M., R. Spaar, L. Schifferli and L. Jenni. 2005. Effects of changes in farming of subalpine meadows on a grassland bird, the whinchat (*Saxicola rubetra*). *J. Ornithol.* 146:14-23.

Niklfeld, H. and L Schratt-Ehrendorfer. 1999. Rote Liste gefährdeter Farn- und Blütenpflanzen (*Pteridophyta* und *Spermatophyta*) Österreichs. 2. Fassung. In: H. Niklfeld (ed.), Rote *Listen gefährdeter Pflanzen Österreichs. 2. Aufl., Grüne Reihe des Bundesministeriums für Umwelt, Jugend und Familie, Band 10.* Austria Medien Service, Graz. pp. 33-152.

Peterseil, J., T. Wrbka, C. Plutzar, I. Schmitzberger, A. Kiss, E. Szerencsits, K. Reiter, W. Schneider, F. Suppan and H. Beissmann. 2004. Evaluating the ecological sustainability of Austrian agricultural landscapes – The SINUS approach. *Land Use Policy* 21:307-320.

Potschin, M.B. and R.H. Haines-Young. 2003. Improving the quality of environmental assessments using the concept of natural capital: a case study form southern Germany. *Landsc. Urban Plann.* 63:93-108.

Primdahl, J., B. Peco, J. Schramek, E. Andersen and J.J. Oñate. 2003. Environmental effects of agri-environmental schemes in Western Europe. *J. Environ. Manage.* 67:129-138

Sauberer, N., K.P. Zulka, M. Abensperg-Traun, H.-M. Berg, G. Bieringer, N. Milasowszky, D. Moser, C. Plutzar, M. Pollheimer, C. Storch, R. Tröstl, H. Zechmeister and G. Grabherr. 2004. Surrogate taxa for biodiversity in agricultural landscapes of eastern Austria. *Biol. Conserv.* 117:181-190.

Schmitzberger, I., T. Wrbka, B., Steurer, G. Aschenbrenner, J. Peterseil and H.G. Zechmeister. 2005. How farming styles influence biodiversity maintenance in Austrian agricultural landscapes. *Agric. Ecosyst. Environ.* 108:274-290.

Sepp, K., M. Ivask, A. Kaasik, M. Mikk and A. Peepson. 2005. Soil biota indicators for monitoring the Estonian agri-environmental programme. *Agric. Ecosyst. Environ.* 108:264-273.

Stoate, C., N.D. Boatman, R.J. Borralho, C. Rio Carvalho, G.R. de Snoo and P. Eden. 2001. Ecological impacts of arable intensification in Europe. *J. Environ. Manage.* 63:337-365.

Tanneberger, F., J. Bellebaum, T. Fartmann, H.-J. Haferland, A. Helmecke, P. Jehle, P. Just and J. Sadlik. 2008. Rapid deterioration of Aquatic Warbler (*Acrocephalus paludicola)* habitats at the western margin of the breeding range. *J. Ornithol.* 149:105-115.

Tscharntke, T., A.M. Klein, A. Kruess, I. Steffan-Dewenter and C. Thies. 2005. Landscape perspectives on agricultural intensification and biodiversity-ecosystem service management. *Ecol. Lett.* 8:857-874.

Van Buskirk, J. and Y. Willi. 2004. Enhancement of farmland biodiversity within set-aside land. *Conserv. Biol.* 18:987-994.

Van Buskirk, J. and Y. Willi. 2005. Meta-analysis of farmland biodiversity within set-aside land: reply to Kleijn and Báldi. *Conserv. Biol.* 19:967-968.

Vickery, J.A., J.R. Tallowin, R.E. Feber, E.J. Asteraki, P.W. Atkinson, R.J. Fuller and V.K. Brown. 2001. The management of lowland neutral grassland in Britain: effects of agricultural practices on birds and their food resources. *J. Appl. Ecol.* 38:647-664.

Wilson, J.B. 1999. Guilds, functional types and ecological groups. *Oikos* 86:507-522.

Wilson, J.D., A.J. Morris, B.E. Arroyo, S.C. Clark and R.B. Bradbury. 1999. A review of the abundance and diversity of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change. *Agric. Ecosyst. Environ.* 75:13-30.

Wilson, J.D., M.J. Whittingham and R.B. Bradbury. 2005. The management of crop structure: a general approach to reversing the impacts of agricultural intensification on birds? *Ibis* 147:453-463.

Wrbka, T., K.-H. Erb, N.B. Schulz, J. Peterseil, C. Hahn and H. Haberl. 2004. Linking pattern and process in cultural landscapes. An empirical study based on spatially explicit indicators. *Land Use Policy* 21:289-306.

Wrbka T., K. Reiter, M. Paar, E. Szerencsits, A. Stocker-Kiss and K. Fussenegger. 2005. *Die Landschaften Oesterreichs und ihre Bedeutung fuer die biologische Vielfalt. Monographien Umweltbundesamt, vol. 173*. Vienna.

Wrbka, T., E. Szerencsits, D. Moser and K. Reiter. 1999. Biodiversity patterns in cultivated landscapes: experiences and first results form a nationwide Austrian survey. In: M. Maudsley and J. Marshall (eds), *Heterogeneity in Landscape Ecology: Pattern and Scale.* IALE (UK), Bristol. pp. 3-21.

ZAMG (Zentralanstalt für Meteologie und Geologie). 2003. Witterungsübersicht Juni 2003. http://www.zamg.ac.at/

Zechmeister, H.G., I. Schmitzberger, B. Steurer, J. Peterseil and T. Wrbka. 2003. The influence of land-use practices and economics on plant species richness in meadows. *Biol. Conserv.* 114:165-177.

Received February 1, 2008
Revised June 21, 2008
Accepted October 18, 2008

The 1st European Congress of Conservation Biology (ECCB2006) was organized by the Society for Conservation Biology - European Section (SCB-ES) in Eger (Hungary) in 2006. It brought together scientists, practitioners and policy formers from all parts of Europe and beyond. It was a unanimous decision of this first Congress that the event should not be a one-off and a second Congress should be held in 2009 to maintain a regular forum for exchange on conservation science and nature conservation practice.

The 2nd European Congress of Conservation Biology (ECCB2009) will be held in the heart of Europe, in Prague on September 1 - 5, 2009. By studying the needs of biodiversity, deepening our dialogue with stakeholders and all citizens, and looking beyond the countdown 2010, the topic "Conservation biology and beyonce - from science to practice" reflects the fact that delivering effective conservation requires a range of actors. Conservation still suffers from these different actors being poorly coordinated and there is work to do to ensure a concerted effort. Conservation science needs to cover a broader range of disciplines than just biology to be relevant to practice and needs feedback from application on successes, problems faced and research needs. In addition, conservation biologists often remain poor at communicating the importance of their science to policy and practice; mechanisms for better communication exist but need to be agreed upon and invested in.

These are both exciting and challenging times for conservationists and we expect Prague 2009 to be an event that will be remembered as a milestone for nature conservation!