Estimating the biodiversity of hay meadows in north-eastern Switzerland on the basis of vegetation structure

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Abstract

Biological diversity is a major criterion in evaluating the effectiveness of measures to enhance the ecological quality of rural areas. Assessments of biodiversity based on selected groups of indicator organisms, including both animal and plant groups, are time-consuming and require a high level of expert knowledge: simpler methods are therefore needed. The biodiversity of 18 hay meadows in north-eastern Switzerland (Schaffhauser Randen, Canton Schaffhausen) was investigated using three indicator groups (angiosperms, spiders and true bugs). Simple structural parameters describing the vegetation canopy were investigated in terms of their usefulness as surrogates for biodiversity measures.

Eighteen sites varied widely in terms of species richness, diversity and abundance for all indicator groups. Numbers of angiosperm species varied from 21 to 57, spiders 25 to 45, and true bugs 12 to 37. Species numbers of the three indicator groups were significantly correlated with each other (angiosperms versus spiders: $r = 0.53$, $P = 0.02$; angiosperms versus true bugs: $r = 0.59$, $P = 0.01$; true bugs versus spiders: $r = 0.61$, $P = 0.01$). Management had a strong influence on species richness, the sites under extensive management showing the highest species number and diversity. The strongest correlations were those between true bugs and the two other indicator groups. However, the number of angiosperm species per $120\,m^2$, which was easy to assess, was also an acceptable predictor of the other two indicator groups.

Several parameters of the vegetation canopy were significantly related to species numbers and multiple regression models based on these parameters explained 25–60% of the variance in species richness and composition. Canopy density, plant biomass distribution and plant height were the most important parameters for all indicator groups.

The results indicate that parameters describing the structure of the vegetation provide useful information about the relative species richness of sites. In combination with parameters describing the surrounding land-use and an inventory of the angiosperms present, they offer a promising method for assessing biodiversity on a large scale.
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1. Introduction

Biological diversity has diminished rapidly in rural areas of Switzerland over the last 70 years as a result of the intensification of agriculture and associated changes in farming methods (Isler-Hübscher, 1980; Mühleberg and Slowik, 1997). Since 1992, there have been major changes in Swiss agricultural policy aimed at liberalising the market and promoting sustainable production. An important objective has been to break the traditional link between prices of agricultural products and farmers incomes by

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making direct payments to farmers related to environmental criteria. Farms which receive financial support are obliged to allocate at least 7% of their land for so-called ecological compensation areas. Financial incentives are also available for other management practices which promote conservation. Although, these schemes have been widely adopted, their effectiveness in promoting biological diversity is difficult to assess due to the lack of suitable evaluation methods. A straightforward, time-saving methodology is needed to measure the efficiency of such management practices.

Usual biodiversity assessment procedures, which involve making inventories of selected indicator groups, are much too expensive for large-scale surveys, especially when animal taxa are involved. Hänggi (1989) calculated the labour costs for an evaluation based on the spider fauna of 10 sites (three traps per site with seven capture periods from May to August) as follows: 2.5 days field work, 7 days sorting traps, 10 days identification, 1.5 days analysis of the results. Thus, the total labour required for a modest survey of 10 sites was 21 days. Previous attempts to develop simpler methods of evaluating biodiversity on farmland have been based only on the assessment of angiosperm species, and have proved inadequate in predicting invertebrate diversity (Zwölfer et al., 1984; Gloor, 1996).

In this study, the species richness of 18 grassland sites was investigated using three indicator groups. The taxa chosen were the angiosperms—being the group most commonly recorded in biodiversity studies—and the true bugs (Heteroptera) and spiders (Araneae), both of which have proved to be useful indicators of biodiversity in agricultural areas (Wachmann, 1989; Kiechle and Trautner, 1992; Duelli and Obrist, 1998).

The study also aimed at testing whether the structure of the vegetation can be used as an alternative to making inventories of indicator species. Several components of the vegetation structure influence invertebrate abundance (Lawton, 1983; Brown, 1991). Variation in canopy components and interactions between them may produce different habitat possibilities for invertebrates (Denno and Roderick, 1991). Various structural parameters of the vegetation were measured with a view to detect those which most closely reflected differences in grassland biodiversity.

2. Material and methods

2.1. Study area and study sites

The investigation was carried out in 1998 in the Merishauser Randen (Swiss coordinates: 683,000/ 291,500–687,500/293,000, world-wide coordinates 8°36′E/46°44′N), which lies at the north-eastern end of the Swiss Jura. Average yearly rainfall in the nearby town of Schaffhausen (437 m a.s.l.) is 866 mm, with the highest values in summer and the lowest in late winter. The mean annual temperature is 7.8 °C, with a maximum mean monthly temperature in July (23.2 °C) and a minimum in January (−3.9 °C) (SMA, 1999). The region is mainly forested but with some agricultural land including a large number of traditional hay meadows between 780 and 830 m a.s.l. (total area of agricultural land: 243 ha). The soils overlie a nutrient-poor limestone and are shallow and freely draining (skeleton-free top layer of 10–15 cm; orthic rendzina according to the FAO classification).

Three types of hay meadows were distinguished.

- Extensive (E) sites ($n = 6$), not fertilised, cut once or twice per year.
- Low (L) intensive sites ($n = 6$), lightly fertilised with manure, cut twice per year.
- Moderate (M) intensive sites ($n = 6$), regularly fertilised with slurry, cut two to three times per year.

Eighteen meadows, six per management intensity, were selected within a 1.5 km-radius of each other. In each of the 18 meadows, a 120 m^2 plot was used to sample the species in the three indicator groups and to measure the vegetation parameters.

2.2. Assessment of indicator groups and vegetation parameters

The choice of the true bugs (Heteroptera) as a group for study was based on Duelli and Obrist, 1998, conclusion that they are the best indicators of invertebrate biodiversity. This is because they include phytophagous, saprophagous and predatory species (Dolling, 1991), larvae and adults live in the same habitat and respond strongly to environmental changes (Morris and Lakhani, 1979; Otto, 1996, true bug species nomenclature follows Günther and Schuster, 1990).

Spiders are also regarded as a useful indicator group for biodiversity (Marc et al., 1999). Many authors (Duffey, 1966; Conrady, 1987; Scheidler, 1990; Uetz, 1991; Barthel, 1997; Dennis et al., 1998) have documented associations between spider and plant communities, and reported how the architecture of vegetation influences the species richness of the spider community. Pozzi et al. (1998) developed an evaluation method using spiders to quantify the conservation value of particular sites in Switzerland (spider species nomenclature follows Maurer and Hänggi, 1990).

Spiders and true bugs were sampled using a standardised sweep-net method (Remane, 1958; Otto, 1996). The sweep-net had a diameter of 40 cm and was fitted with a heavy cloth. Samples were collected every 4 weeks from May to September (5 samplings per plot) by making 50 sweeps over 50 m. Sampling was performed between 11 a.m. and 4 p.m., on days when the sun was shining and the temperature was at least 17 °C. Barber pitfall traps (Adis, 1979; Curtis, 1980) were used to sample surface-active arthropods. This method is believed to provide a good estimate of the number of spider species in a community (Luff, 1975; Thomas and Marshall, 1999). The pitfall traps consisted of 200 ml plastic cups inserted into the ground and fitted a funnel set flush with the substrate. Each trap contained a 100 ml solution of 75% alcohol. Traps were protected from rain by transparent covers (17 cm × 17 cm) supported by a metal arch 10 cm above the ground. At each site, three pitfall traps were placed 2.5 m apart in a row. Sampling was carried out from 8 May to 12 June and from 3 July to 7 August (i.e. 10 weeks, cf. Duelli et al., 1999). Traps were emptied once a week, the spiders and true bugs being preserved in 70% alcohol in polyethylene tubes. All collections were combined to one sample per plot ($n = 18$) for each indicator group.

Shortly before the first cut, a species list of angiosperms was made for an area of $120 \, m^2$ at each site; the percentage cover of species was estimated for an area of $16 \, m^2$ (nomenclature following Lauber and Wagner, 1996). In July, various parameters describing the structure of the vegetation were recorded (Table 1).

2.3. Data analysis

Statistical analyses were performed using the Statistica statistical package (Version 6.0, 98th Edition) and Canoco (Version 4, 98th Edition). Data were evaluated for auto-correlation, colinearity, normality and homogeneity of variances. When necessary, the transformation $\ln(x + 1)$ was used to normalise the data (normality of the distribution was tested with Shapiro Wilks W-Test). Unless otherwise stated, the level accepted for statistical significance was 0.05.

Species number and Simpson's diversity index were used as parameters describing the diversity of a site. Mean values were used for statistical analysis throughout the study.

Rankings were calculated to reflect three different types of criteria for evaluating the sites. The first criterion was species number, sites being ranked separately for each indicator group (three rankings). The second criterion was Simpson's index, also for each indicator group separately. The third criterion expressed conservation value. The number of angiosperm species characteristic of nutrient-poor grassland was used as an index of the conservation value of the plant community (referred to as val. plant index). A second index of conservation value, referred to as val. spider index, was based on an evaluation method developed by Hänggi (1987) and Pozzi et al. (1998). These authors assigned all spider species in Switzerland a value from 1 to 36 to reflect their conservation importance in terms of rarity and specificity (habitat fidelity). The mean value of the scores for the spiders present at any site provided an index of the conservation status of the site.

Correlations between species richness of the three indicator groups were calculated to determine whether a single group could be used as a surrogate for the total biodiversity. Further analysis of the data was carried out by stepwise multiple linear regression. In interpreting the results, attention was paid to the limitation of such models when many explanatory variables are correlated (multi-colinearity).

To investigate which canopy parameters had a significant influence on species composition, a partitioning of the variance using canonical correspondence analysis was carried out. This technique identifies the linear combination of explanatory variables which maximises the dispersion of the species scores (for further explanation to CCA see Ter Braak, 1986). The statistical significance of the variables was determined by a Monte Carlo permutation test.

Table 1
Parameters used for describing the 18 hay meadows (plot size: $16\,m^2$) in Merishauser Randen, Switzerland

Categories	Criteria[a]	e or m[b]
Site location	Exposition and inclination	m
	Height of site above sea level	m
	Adjacent surrounding (six categories)	m
Qualitative description of the canopy	Height of canopy (five categories)	e
	Density of canopy (five categories)	e
	Homogeneity of canopy (five categories)	e
Quantitative description of the canopy	Maximum height of canopy (cm)	m
	Average height of canopy (cm)	m
	Maximum height of flowering herbs (cm)	m
	Average height of flowering herbs (cm)	m
	Number of angiosperm species on $16\,m^2$ and $120\,m^2$	m
	Maximum height of flowering grass species (cm)	e
Distribution of plant cover and biomass	Plant cover above 60 cm (%)	e
	Plant cover between 21–60 cm (%)	e
	Plant cover between 0–20 cm (%)	e
	Plant biomass above 60 cm (%)	e
	Plant biomass between 21–60 cm (%)	e
	Plant biomass between 0–20 cm (%)	e
	Total plant cover (%)	e
Cover and biomass of functional groups	Herb cover (%)	e
	Grass cover (%)	e
	Legume cover (%)	e
	Herb biomass (%)	e
	Grass biomass (%)	e
	Legume biomass (%)	e
Cover of further soil covering parts	Gap cover (%)	e
	Moss cover (%)	e
	Litter cover (%)	e
	Other covers (%), e.g. stones, woody species	e
Classification based on plant height	Height of canopy component $1-n$ (cm)	m
	Biomass of canopy component $1-n$ (%)	e
	Plant cover of canopy component $1-n$ (%)	e
Light absorption by the canopy	PAR 0 cm	m
	PAR 20 cm	m
	PAR 30 cm	m
	PAR 60 cm	m
	PAR 80 cm	m
Heterogeneity of the canopy	S.D. PAR 0 cm (intra-plot standard deviation)	m
	S.D. PAR 20 cm	m
	S.D. PAR 30 cm	m
	S.D. PAR 60 cm	m
	S.D. PAR 80 cm	m

[a] PAR = photosynthetic active radiation; S.D. = standard deviation.
[b] m = measured; e = estimated.

3. Results

Eighteen meadows represented a range of grassland types from species-rich plant communities dominated by *Bromus erectus* to species-poor communities with *Arrhenatherum elatius*. A total of 100 angiosperm species, 112 spider species (from 21,658 adult specimens) and 94 true bug species (from 5,666 adult specimens) were recorded.

Angiosperm species richness and composition differed considerably according to management. The communities of the extensive sites were dominated

by *B. erectus*, which had a mean cover of 25%, followed by *Salvia pratensis* and *Trifolium campestre*, both with about 5% cover. The mean number of angiosperm species per $120 \, m^2$ was 58. The most abundant species in the low intensive sites were *Trifolium pratense*s.l. (15% cover), *A. elatius* (8%) and *S. pratensis* (7%), and there was a mean of 40 species per $120 \, m^2$. Meadows under moderately intensive management had approximately equal amounts (ca. 10%) of *Dactylis glomerata*, *Trisetum flavescens* and

A. elatius, and a mean of 32 angiosperm species per sample site.

For each group, there were significant differences in species number (ANOVA; Fig. 1) between the extensive and moderately intensive sites, but no significant differences in species diversity as measured by Simpson's index. Angiosperm species number also differed significantly between extensive and low intensive managed meadows. Neither Simpson's diversity index nor species richness differed between low and
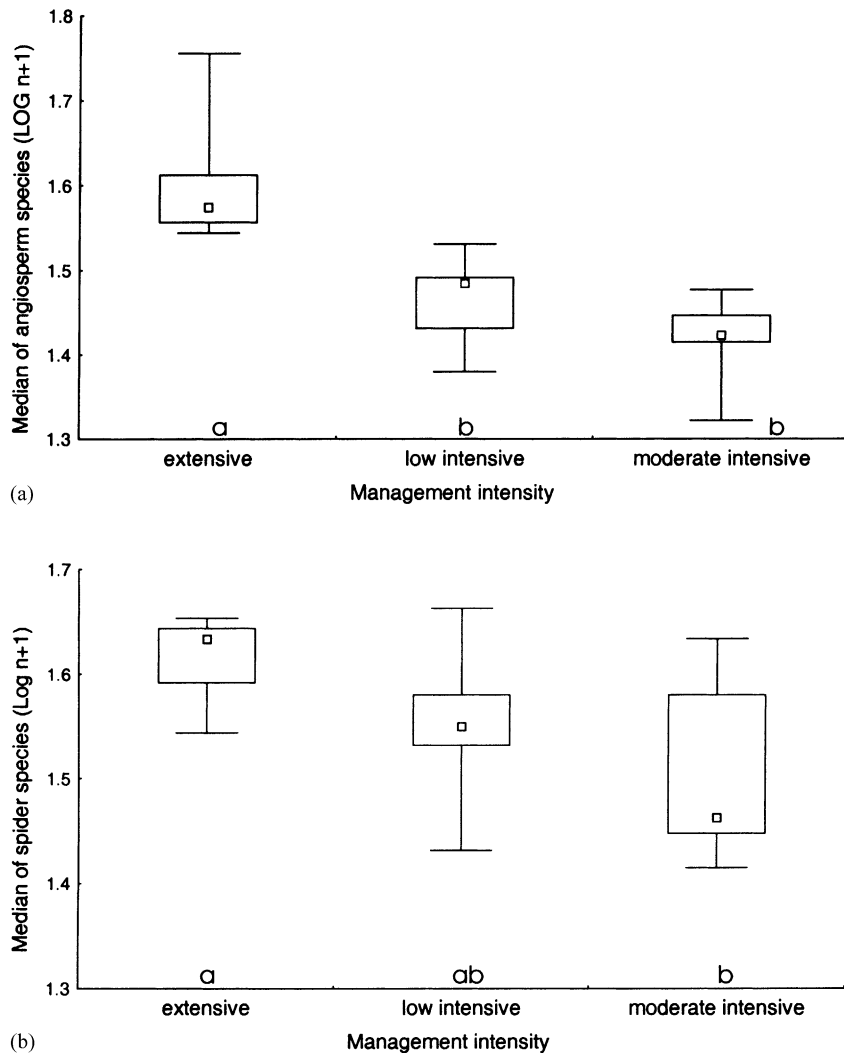

Fig. 1. Responses of indicator species to management intensity (median $\pm$ min/max). Bars with the same letter were not significantly different. More angiosperm (a) (ANOVA: $P < 0.001$), spider (b) (ANOVA: $P < 0.06$), and true bug species (c) (ANOVA: $P < 0.05$) found in extensive than in moderate intensive meadows.
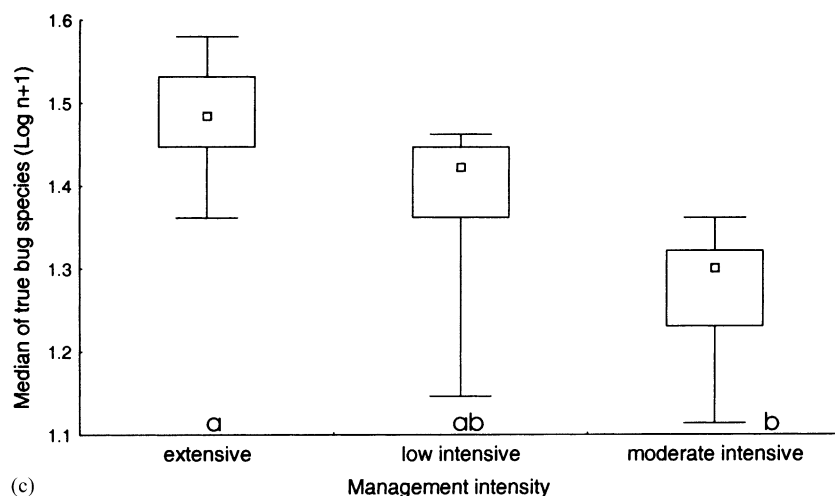
Fig. 1. (*Continued*).

moderately intensive management. There were significant correlations between the total species numbers of every pair of taxonomic groups, the strongest correlations involving the number of true bug species (true bugs versus spiders: $r = 0.61$, $P = 0.001$; angiosperms versus true bugs: $r = 0.59$, $P = 0.01$; angiosperms versus spiders: $r = 0.53$, $P = 0.02$).

The sites were ranked using three different criteria evaluating their biological diversity (Table 2). The sum of the eight rankings provided an overall index of biodiversity. Extensively used meadows had scores of 33–60, while low and moderately intensive sites achieved scores of 69–83 and 85–117, respectively. Table 3 shows the Spearman rank correlation coefficients between all pairs of rankings. The parameters involved in the largest number of correlations (four out of a possible seven) were number of true bug species, val. plant and val. spider index.

Correspondence analyses were calculated for each indicator group separately (Table 4). Sites clustered according to management type into three groups, site E7 being an outlier in all analyses. The first axis scores of all three correspondence analyses were significantly correlated (angiosperms versus spiders: $r = 0.95$, angiosperms versus true bugs: $r = 0.92$, and true bugs versus spiders: $r = 0.89$; all $P < 0.00$). The second axes were also correlated significantly for angiosperms versus true bugs and true bugs versus spiders ($r = 0.47$, $P = 0.05$; $r = 0.77$, $P < 0.00$, respectively).

To identify which measure of vegetation structure best reflected species diversity, the canopy parameters listed in Table 1 were tested for their correlation with species numbers in the three indicator groups. Canopy parameters with significant correlations (Table 5) were used in a multiple linear regression model. The parameters which significantly influenced the species richness of the three groups are listed in Table 6. Half of the variability in angiosperm species richness was explained by the structural parameters, (photosynthetic active radiation)'PAR 20 cm' being the most important variable. Close to 50% of the variation in the total number of spiders was explained by the amount of 'angiosperm biomass 21–60 cm' in combination with the 'height of canopy component 3'. Nearly 70% of the variation in the true bug data was explained by the 'maximum height of flowering herbs' and the 'standard deviation of the PAR 0 cm'. Parameters not used for the regression either did not correlate with the number of species or were correlated with parameters already included in the regression equation. The relationships between observed and expected values showed that the combination of variables used in the model achieved a good fit to the data, with no obvious signs of non-linearities (Fig. 2).

Management intensity influenced all three indicator groups and explained 74% of the variance for angiosperm species richness, 31% for spiders, and 51% for true bugs.

Table 2

Number of indicator species for 18 hay meadows in Merishauser Randen (Switzerland) based on rankings of species number, diversity and conservation value, inclusive sum of ranks and overall rank[a]

	E1	E3	E5	E7	E9	E11	L13	L15	L17	L19	L21	L23	M25	M27	M29	M31	M33	M35
Species number																		
Angiosperms	35	41	37	57	36	38	30	24	31	34	27	31	27	28	21	26	26	30
Rank	6	2	4	1	5	3	10	17	8	7	13	9	14	12	18	15	16	11
Spiders	38	42	42	44	34	43	36	37	26	33	45	33	28	28	25	42	37	27
Rank	7	4	5	2	11	3	10	8	17	12	1	13	14	15	18	5	9	16
True bugs	33	37	27	29	22	30	27	27	13	28	24	22	20	19	22	19	16	12
Rank	2	1	6	4	10	3	6	6	17	5	9	10	13	14	10	14	16	18
Simpson diversity index																		
Angiosperms	0.82	0.94	0.93	0.88	0.91	0.91	0.94	0.91	0.94	0.85	0.92	0.94	0.93	0.94	0.93	0.88	0.91	0.90
Rank	18	1	6	15	10	10	1	10	1	17	9	1	6	1	6	15	10	14
Spiders	0.63	0.69	0.55	0.87	0.83	0.59	0.41	0.50	0.42	0.49	0.59	0.68	0.42	0.63	0.61	0.74	0.67	0.47
Rank	7	4	12	1	2	10	18	13	16	14	10	5	16	7	9	3	6	15
True bugs	0.90	0.69	0.75	0.88	0.64	0.93	0.86	0.83	0.93	0.92	0.91	0.80	0.93	0.88	0.76	0.90	0.72	0.85
Rank	6	17	14	8	18	1	10	12	1	4	5	13	1	8	13	6	15	11
Conservation index (value)																		
Angiosperms	14	12	11	16	15	14	7	11	8	11	7	8	7	6	5	5	5	5
Rank	3	5	6	1	2	3	11	6	9	6	11	9	11	14	15	15	15	15
Spiders	5.03	4.88	4.93	6.55	5.24	5.07	4.11	4.43	4	4.79	4.38	4.67	4.54	3.86	3.12	4.02	4.68	3.22
Rank	4	5	6	1	2	3	13	11	14	7	11	8	10	16	18	14	8	17
Sum of ranks	53	39	59	33	60	36	79	83	83	72	69	68	85	87	107	88	95	117
Overall rank	4	3	5	1	6	2	10	11	11	9	7	8	13	14	17	15	16	18

[a] E = extensive; L = low intensive; M = moderate intensive management.

Table 3
Spearman Rank order correlation coefficients and significance between 8 rankings of 18 hay meadows in Merishauser Randen, Switzerland[a]

	Angiosperm species number	Angiosperm species diversity	Val. plant index	Spider species number	Spider diversity	Val. spider index	True bug species number	True bug diversity
Angiosperm species number	–							
Angiosperm diversity	−0.07 n.s.	–						
Val. plant index	0.81***	−0.21 n.s.	–					
Spider species number	0.43 n.s.	−0.30 n.s.	0.49*	–				
Spider diversity	0.25 n.s.	−0.23 n.s.	0.27 n.s.	0.38 n.s.	–			
Val. spider index	0.76***	−0.33 n.s.	0.87***	0.62**	0.44 n.s.	–		
True bug species number	0.60**	−0.19 n.s.	0.75***	0.64**	0.18 n.s.	0.67**	–	
True bug diversity	−0.01 n.s.	−0.19 n.s.	−0.01 n.s.	0.00 n.s.	−0.45 n.s.	−0.121 n.s.	−0.01 n.s.	–

[a] n.s. = not significant.
* $0.01 < P < 0.05$.
** $0.001 < P < 0.01$.
*** $P < 0.001$.

Table 4
Correspondence analyses of angiosperm, spider and true bug species compositions of 18 hay meadows in Merishauser Randen, Switzerland[a]

		Axes				Total inertia
		1	2	3	4	
Angiosperms	Eigenvalue	0.4	0.257	0.21	0.14	1.738
	Cumulative % variance of species data	23	37.8	50.2	58.2	
Spiders	Eigenvalue	0.3	0.166	0.12	0.12	1.346
	Cumulative % variance of species data	20	32.1	41.1	49.8	
True bugs	Eigenvalue	0.3	0.225	0.21	0.19	2.124
	Cumulative % variance of species data	15	25.9	35.6	44.7	

[a] Eigenvalues measure the importance of the axes (values between 0 and 1) and the total inertia is the total variance in the species data as measured by the chi-square of the sample-by-species table divided by the table's total.

Table 5
Pearson-Product–Moment correlations coefficients and significance between the number of indicator species (angiosperms, spiders and true bugs) and vegetation parameters describing 18 hay meadows in Merishauser Randen, Switzerland[a]

	Angiosperms	Spiders	True bugs
Management intensity	−0.86***	−0.56*	−0.73***
Total plant cover	−0.61**	−0.25 n.s.	−0.60**
Plant biomass 0–20 cm	0.64**	0.56*	0.58*
Plant biomass 21–60 cm	−0.65**	−0.57*	−0.59*
PAR 20 cm	0.70***	0.46*	0.70***
PAR 30 cm	0.63***	0.42 n.s.	0.62***
Angiosperm species number (120 m^2)	–	0.53*	0.60
Plant cover of canopy component 2	0.46 n.s.	−0.59*	−0.45 n.s.
Plant biomass of canopy component 2	−0.57*	−0.42 n.s.	−0.59*
Maximum height of flowering herbs	−0.58*	−0.39 n.s.	−0.80***

[a] n.s. = not significant;
* $0.01 < P < 0.05$;
** $0.001 < P < 0.01$;
*** $P < 0.001$.

Table 6
Multiple regression models predicting species richness of angiosperms, spiders and true bugs in 18 hay meadows in Merishauser Randen, Switzerland[a]

	Constant	Variable 1	Variable 2
Angiosperms			
		PAR 20 cm	
Regression coefficient	19.5	38.9	
Cumulative r^2		0.5	
F-value at end		$F(1, 16) = 15.9$, $P < 0.001$, S.E. $= 8.4$	
Spiders			
		Plant biomass 21–60 cm	Height of canopy component 3
Regression coefficient	22.0	−0.3	0.4
Cumulative r^2		0.32	0.5
F-value at end		$F(1, 16) = 7.7$, $P < 0.01$, S.E. $= 5.6$	$F(2, 15) = 6.9$, $P < 0.01$, S.E. $= 5.1$
True bugs			
		Maximum height of flowering herbs	S.D. PAR 0 cm
Regression coefficient	39.2	−0.3	79.6
Cumulative r^2		0.6	0.7
F-value at end		$F(1,16) = 23.4$, $P < 0.000$, S.E. $= 4.4$	$F(2, 15) = 17.1$, $P < 0.000$, S.E. $= 4.0$

[a] Independent variables (parameters of the vegetation structure).
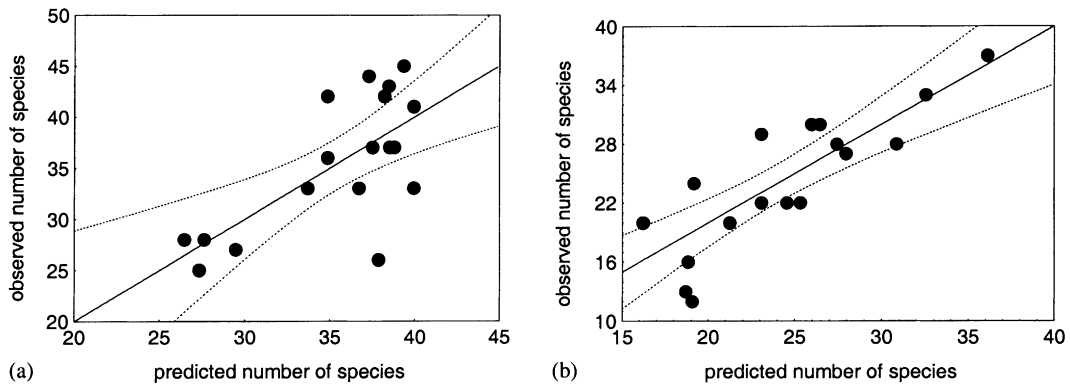
Fig. 2. Multiple regression models fitting actual data (observed richness), regression line and 95% confidence interval, (a) number of spider species; (b) number of true bug species.

4. Discussion

The results presented here show a high level of concordance between the indicator groups in their ability to discriminate between the 18 sites (e.g. correspondence analyses, Table 4). The trends in species composition in relation to land use history and present-day management of the sites are similar for the three groups. For each group, increased intensity is associated with reduced species richness. Management intensity has the strongest effect on the number of angiosperm species, explaining 74% of the variability of species richness. The mainly foliage-dwelling true bugs (51% of variation explained by management) appear to be more affected by regular cutting than the spiders (31%), reflecting the fact that nearly 99% of all spider individuals captured were soil-dwellers. This conclusion is consistent with the results of Otto and Dorn (1995) and Di Giulio et al. (2001). Management intensity had rather little influence on the actual species composition of the community, explaining 18% of the variability for angiosperms, 29% for

spiders, and 17.5% for bugs. However, some spider species are promoted by intensive cutting and a high number of species were exclusively found in sites of the same management type.

Conclusions about the value of grassland sites depend on the criteria used for evaluation (Table 2). Bàldi and Kisbenedek, 1997, argue that an evaluation should take account of the conservation importance of the species present. Significant correlations were found between the ranking of the number of angiosperm species and that of the two conservation indices (i.e. val. plant index and val. spider index, Table 3). The number of angiosperm species may, therefore, be considered a reasonable predictor of the conservation value of hay meadows.

As a first step towards a robust and rapid method for evaluating grasslands based on the vegetation structure, parameters with a significant influence on species richness were identified. Table 6 shows that angiosperm species richness was clearly related to the PAR. However, from a practical point of view it does not make sense to replace the traditional plant species list with an indirect measure such as PAR if it provides no significant time saving. For invertebrates, however, indirect methods are potentially very useful, as the faunistic assessments are time-consuming, weather dependent and demand expert knowledge. The species richness of spiders was influenced most strongly by the density of the canopy, while true bug richness was also affected by plant species composition. The results suggest that the species richness of spiders tends to be low if the lower part of the plant canopy is dense. This finding is consistent with many other studies showing that the diversity of the spider community is strongly influenced by habitat structure (Lowrie, 1948; Hatley and MacMahon, 1980; Robinson, 1981; Downie et al., 1995; Anderlik-Wesinger et al., 1996). Canopy parameters can, hence, be used as predictors of species richness of spiders, especially at the family level (Duffey, 1966). The large group of Linyphiidae was influenced by the combination of the 'average canopy height' and 'angiosperm cover of canopy component 3' (72% of the variation in Linyphiidae species richness explained), emphasizing the importance of height and density of the vegetation for this family. The Gnaphosidae in contrast, were mainly influenced by the 'PAR 0 cm' (most

probably as a measure of the closeness of the vegetation), the 'legume biomass' and the 'maximum height of the flowering herbs' (84% variation explained).

For the true bugs, the 'height of the canopy' and a measure of the canopy density appeared to be important. The negative correlation of 'maximum height of flowering herbs' can be explained by the fact that the presence of taller angiosperm species is generally associated with increased canopy density, which restricts the movement of flying insects such as true bugs. The standard deviation of the PAR on the ground is a parameter expressing the horizontal variability of the vegetation canopy of the habitat. An heterogeneous canopy offers more gaps than a dense, homogeneous one and may be the reason for the positive correlation between 'S.D. PAR 0 cm' and the number of true bug species.

The statistical partitioning of the data (Borcard, 1992) into structural (vegetation canopy) and management components showed that about 60% of variation (Fig. 3) in species richness can be explained by management (18% of the variance of the angiosperm data, 29% of the spiders and 17.5% of the true bugs), and by vegetation structure (33, 11 and 40% respectively). The interaction term, management × vegetation structure, explained a further 8–14% of the variance. Canopy parameters with significant impact on the abundance of the indicator groups were the number of 'angiosperm species in 120 m^2', 'height of flowering herbs', 'grass and herb biomass', 'PAR', 'height of the canopy' and 'surroundings'.

The results presented here show that canopy parameters explain at least 50% of the variation in species richness, a result comparable with that obtained in other studies (Hatley and MacMahon, 1980; Barthel, 1997; Dennis et al., 1998). Parameters related to canopy density are the most important for explaining the species richness of the indicator groups. An evaluation methodology based on both the numbers of angiosperm species and selected structural parameters would provide acceptable estimate of above-ground biodiversity. Such a methodology would be much faster than conventional biodiversity assessments. In our study, it took an average of 30 min for one person to make a list of the angiosperm species present at one site, the recording of plant height took only 1 min, while
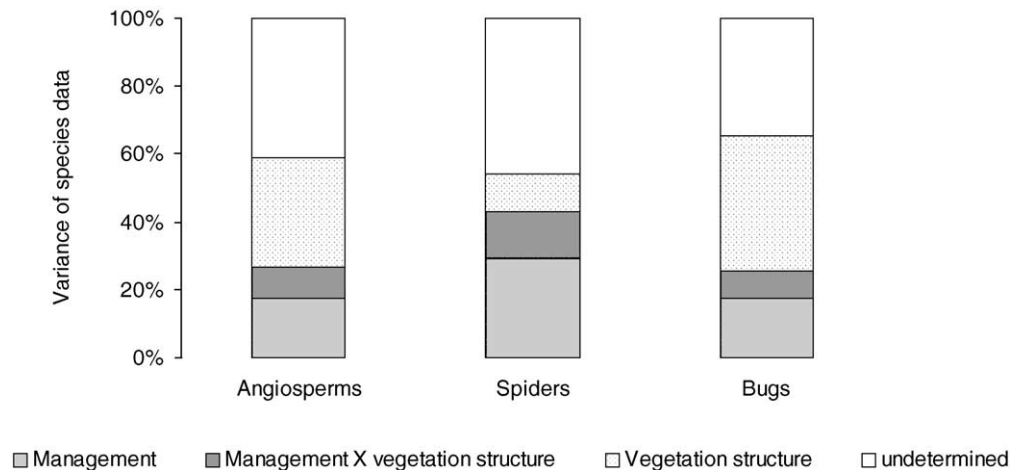
Fig. 3. Statistical partitioning of the parameters explaining most of the ecological variation.

the PAR profiles took 30 min. Thus, just over one hour was required to record an average site, and the survey of 18 sites took 2.5 days. This contrasts very favourably to the 21 days that Hänggi (1989) needed for the assessment of the spider fauna in 10 sites.

5. Conclusions

The use of bioindicators is a useful approach to evaluating biodiversity (Paoletti, 1999) and has the potential to make a major contribution to optimizing different farming systems. However, simpler and more time-efficient methods are needed for evaluating the success of initiatives to enhance biological diversity in the agricultural landscape. Based on the findings of this study, a rapid evaluation methodology may be recommended, composed of three main types of information. The first is a list of the angiosperm species present, since, it is the best single indicator of the biodiversity at a site. The second element is information about the structure of the canopy, likely to considerably improve predictions about the diversity of invertebrates. Finally, it is important to include information about the surrounding land use, since, this can strongly influence local species richness, perhaps through aerial photographs.

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