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Vascular plants as surrogates of butterfly and grasshopper diversity on two Swiss subalpine summer pastures

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Abstract Summer pastures in the Swiss Alps are currently affected by land-use changes that cause a decrease in biodiversity. Although these habitats make up one-third of the whole Swiss agricultural area, direct payments dedicated to support their management are very low. Current political instruments do not support efforts to conserve the biodiversity in these areas, but a vegetation-based approach such as the one implemented in the permanently utilized agricultural areas is under discussion. However, available studies evaluating the surrogate value of vascular plants for other (particularly animal) taxa have yielded inconsistent results, and very few have been conducted in habitats at high elevations. We investigated the extent to which vascular plants are adequate surrogates for butterfly and grasshopper diversity, examining the congruence of species richness and community similarity in two heterogeneous subalpine pastures in the Swiss Alps. Results at the species richness level (Spearman's rank correlation) varied widely according to the study site and taxa assessed. In contrast, at the community similarity level (Procrustean randomization tests with Bray–Curtis similarity), congruencies between vascular plant and invertebrate taxa were generally highly significant. We therefore recommend the use of community similarity as a basis for estimating biodiversity patterns. Our results suggest that conservation measures aimed primarily at enhancing the floristic diversity of subalpine grasslands are also likely to benefit butterfly and grasshopper diversity, at least at the local scale.

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Introduction

Agricultural activities have shaped Swiss alpine landscapes over centuries, and contributed
greatly to their biological diversity (Bätzing 2003). Despite their anthropogenic origin, the
subalpine grasslands used as summer pastures have a very high conservation value, har-
bouring up to three times as many species as the forest they replaced (Zoller and Bischof
1980). However, changing social and economic conditions have affected land use in alpine
regions, with subalpine pastures developing in diverging ways; whereas productive and
accessible areas are now managed more intensively, remote, steep and less productive
areas are underused or even abandoned. Both trends tend to cause a decrease in biodi-
versity (MacDonald et al. 2000; Tasser and Tappeiner 2002), and without measures to
counteract them, the summer pastures in the forest belt will soon lose much of their
conservation value (Mack et al. 2008).

To halt the decline of biodiversity in agricultural landscapes, many European countries
including Switzerland have introduced agri-environmental schemes (Kleijn and Sutherland
2003). Since 1999, Swiss farmers have been eligible for direct payments provided they
manage their permanently utilized agricultural area according to specified criteria (BLW
1998). One of these is that at least 7 % of the agricultural land should be managed as
ecological compensation areas (ECAs), which may include extensively managed meadows
and pastures, and traditional orchards. Furthermore, an ordinance introduced in 2001
entitles farmers to receive result-oriented financial support for ECAs that meet certain
standards of ecological quality (BLW 2001). In this context, ecological quality is defined
by the presence in sufficient quantity of certain plant taxa considered to be of conservation
value. However, alpine summer pastures are not supported by these instruments because
they are not regarded as permanently used agricultural land (BLW 2007). Thus, although
the pastures in the Swiss Alps and Jura Mountains make up one-third of the total Swiss
agricultural area (Baur et al. 2007), they accounted for <4 % of direct agricultural pay-
ments in 2010 (BLW 2011). It is becoming increasingly clear that more financial support
will be needed to ensure that these pastures continue to be used for grazing during the
summer period (Mack and Flury 2008), and an extension of the national agri-environ-
mental schemes to include the ecological quality of summer pasture areas is currently
under discussion. Such a development would help protect the biodiversity of subalpine
grasslands while supporting the livelihoods of farmers who use these areas (Klimek et al.
2008; Wittig et al. 2006).

The administration of such a scheme, however, would present practical difficulties
because monitoring biodiversity in subalpine grasslands is no simple matter. As the whole
of biodiversity cannot be recorded (McGeoch 1998; Noss 1990), surrogate indicators are
widely used in conservation biology and planning (Caro and O'Doherty 1999; Favreau
et al. 2006). Although this approach saves time and money, it depends upon a crucial
assumption—that the indicator taxa can serve as a surrogate for a broader range of taxo-
nomic diversity (Howard et al. 1998; Prendergast et al. 1993). Vascular plants are com-
monly used because they are relatively easy to sample and identify, react sensitively to
environmental conditions, and support a large number of animal species (e.g. Marini et al.
2007; Niemelä and Baur 1998). The use of selected vascular plants for assessing ecological

quality is the basis of the result-based approach in both Switzerland and Germany (Oppermann and Gujer 2003; Wittig et al. 2006). This approach has been shown to be both efficient and well accepted by farmers (Klimek et al. 2008), though studies investigating the surrogate value of plants in general, and subsets of plant species in particular, for other (primarily animal) taxa have produced results in support (Duelli and Obrist 1998; Nipperess et al. 2012; Panzer and Schwartz 1998; Pharo et al. 1999; Sauberer et al. 2004) and in contradiction (Billeter et al. 2008; Chiarucci et al. 2007, 2005; Niemelä and Baur 1998; Santi et al. 2010; Vessby et al. 2002). Furthermore, very few of these studies have considered the relationships between plant and invertebrate diversity in subalpine areas (Favreau et al. 2006).

The aim of this study was to investigate the adequacy of various measures of plant species richness and composition as surrogates for the diversity of diurnal butterflies and grasshoppers in the summer pasture habitat. We examined the small-scale congruencies of species richness and community similarity patterns in two highly heterogeneous subalpine pastures in the Swiss Alps.

Methods

Study sites and sampling design

The investigation was carried out on two subalpine pastures around the villages of Mesocco (46°23′31″N/9°13′58″E) and Guarda (46°46′33″N/10°08′59″E) in the Canton of Grisons in southeastern Switzerland (Fig. 1). The mean annual precipitation measured at weather stations close to the survey sites between 1961 and 1990 was 1,864 and 693 mm year^{-1} for Mesocco and Guarda, respectively (Schweizerische Meteorologische Anstalt 2008). Crystalline underlying rocks dominate at both sites (Reinhard et al. 1962). Both pastures have a south-west aspect, are ∼1.5 km^2 in area, grazed by cattle, and situated in marginal locations just above the present tree line (1,762–2,064 m elevation in Mesocco and 2,097–2,410 m in Guarda). The vegetation is composed of a mosaic of grasses and herbs, mainly *Nardus stricta*, and dwarf ericaceous shrubs and junipers (*Juniperus* spp.). Fifty-four plots of 30 × 30 m (900 m^2) were located with a systematic sampling (regular grid with a 155 m resolution; see Fig. 1) on each pasture. Hazardous regions of the pasture had to be excluded from the survey resulting in a final sample of 49 plots at Mesocco and 50 plots at Guarda.

Plant, butterfly and grasshopper sampling

Data collection was carried out between June and September 2010. To account for the heterogeneity in the plots, plants were surveyed in nine quadrats of 1 m^2 placed systematically within the 900 m^2 plot (Fig. 1). The plant cover was estimated using the Braun-Blanquet scale (Braun-Blanquet 1964) and the taxonomy follows Lauber and Wagner (2001). We also recorded the presence of any additional species absent from the nine quadrats by searching the whole plot for 30 min. Two plant data sets were derived in this way: a "cover data set" from the nine 1 m^2 quadrats consisting of abundances acquired from the Braun-Blanquet scale, and a "presence data set" of all species recorded in the 900 m^2 plot. We recorded all adult butterflies (Rhopalocera, Hesperiidae and Zygaenidae) and grasshoppers (Ensifera and Caelifera) by walking over the 900 m^2 plot in a serpentine-like transect with a net (see Balmer and Erhardt 2000; Fig. 1). Both taxa were identified
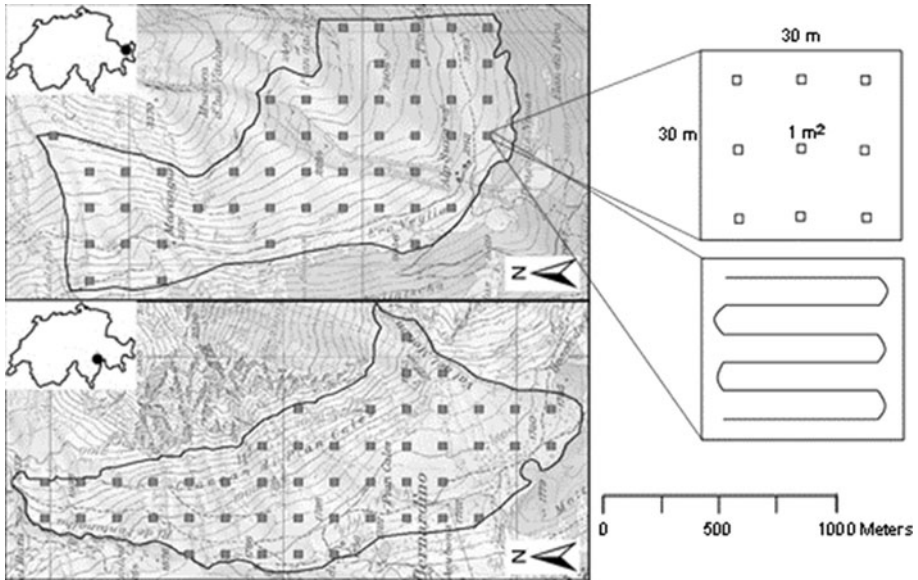
Fig. 1 On the left, the pasture areas in Guarda (*top*) and Mesocco (*bottom*) showing the boundary and the systematically located plots. The Topographic Map 1:25,000 (2012 ©Swisstopo) is shown in the *background*. On the *right*, the sampling design used for plants (*top*) and insects (*bottom*)

visually, and grasshoppers also acoustically, during sunny days with little or no wind between 10:00 and 17:00. The butterfly survey was performed on five occasions, and the grasshopper survey on two occasions. The higher survey effort for butterflies reflects the fact that some butterflies became active earlier than most grasshoppers, and the observation period was therefore longer (May–September for butterflies and July–September for grasshopers). Some butterflies (mostly individuals belonging to the Genera *Erebia*, *Pyrgus*, *Adscita* and *Zygaena*) were retained for accurate genital identification, but all other insects were released where they had been caught. Butterfly nomenclature followed Bühler-Cortesi (2009) and grasshopper nomenclature Baur et al. (2006). To consider the overall species diversity throughout the whole summer period, data of different sampling times were pooled at the plot level prior to data analysis for both insect taxa.

Measures of species richness and quality

Four plant measures were used to test congruence of the butterfly and grasshopper data at the species richness level: (1) total number of plant species, hereafter species richness, derived from the "presence data set"; (2) the number of plant target and characteristic species of nature conservation value considered to indicate ecological quality of Swiss agricultural surfaces (BAFU and BLW 2008), hereafter plant 'quality species'; (3) the average number of species within the nine 1 m² quadrats (α diversity), and (4) the number of species among quadrats (β diversity), derived by subtracting α diversity from the total number of species (γ diversity) in the nine quadrats (Formula 7 in Lande 1996). For the butterfly and grasshopper taxa two measures were used for comparison: (1) species richness (i.e. total number of species), and (2) the number of 'quality species' (i.e. number of target and characteristic species).

Comparisons between the four plant measures and the two measures for butterflies and grasshoppers were made by means of pairwise Spearman's rank correlation tests as the data were not normally distributed and this measure only assumes monotonic instead of linear relationships. Because only one grasshopper quality species, *Gomphocerus sibiricus*, was recorded at Mesocco, comparisons using the number of grasshopper quality species were not performed for this site. Spatial autocorrelation in the data was examined using a Mantel test with 10,000 permutations. Only species data in Guarda were found to be spatially autocorrelated, so for statistical inference partial Spearman's rank correlations between the variables controlling for the effect of geographical variables were determined in all cases (Dale and Fortin 2009). Analyses were performed using the "ppcor" (Kim 2011) and "vegan" (Oksanen et al. 2012) packages of the statistical program R (R Core Team 2012). We did not apply sequential Bonferroni corrections for correlations in agreement with the objections of Moran (2003) and similar studies dealing with surrogate indicators (Fattorini 2010; Mandelik et al. 2012).

Community similarity

We used procrustean randomization tests (PROTEST; Jackson 1995) for pairwise community similarity comparisons between the plant and insect taxa at each site. This approach was preferred to the commonly used Mantel test because of its greater power in detecting congruence between two community matrices (Gioria et al. 2011; Peres-Neto and Jackson 2001). PROTEST attempts to minimize the sum-of-squared deviations between two data matrices by stretching and rotating the points in one matrix, while still maintaining the relative distance between points within each matrix (Jackson 1995). Both the cover and presence plant data sets were used for comparisons with the insect abundance data. Cover data of plants and abundance data for insects were square root-transformed prior to data analysis to improve normality. Bray–Curtis similarity matrices for the species data were then calculated using the transformed cover/abundance data as well as presence data. Because a Mantel test (10,000 permutations) between species assemblages (Bray-Curtis similarity) and the geographic distance matrix revealed spatial autocorrelation in the community data of both the Mesocco and Guarda sites, a partial PROTEST following the approach of Peres-Neto and Jackson (2001) was performed. This method permits the degree of association of two matrices to be calculated using PROTEST while fixing the potential effect of geographical variables. The matrices were standardized using non-metric multidimensional scaling (NMDS), and information on the stress value was used to define the number of axes to use. Four NMDS axes were retained for both the species and geographical matrices. Two steps were used to perform partial PROTEST (Peres-Neto and Jackson 2001): First, for each axis retained from species NMDS analysis, multiple regressions with all geographical axes were performed. Then the residuals obtained from these multiple regressions were used to carry out pairwise partial PROTEST analysis. The Procrustes correlations $r = \sqrt{1 - m}$ were derived from the symmetric Procrustes residuals (m), which are the sum-of-square residuals between matrices in their optimal superimposition. Because of too many zero values in the matrix of butterfly and grasshopper quality species, it was not possible to carry out a reasonable NMDS and thus PROTEST analysis. Analyses were performed using the "vegan" package (Oksanen et al. 2012) in R (R Core Team 2012).

Results

In Mesocco 318 plant, 44 butterfly and 8 grasshopper species were recorded and identified. In Guarda 332 plant, 52 butterfly and 14 grasshopper species were recorded. More exhaustive species and abundance data for the three taxa are summarized in Table 1. In general, plant, butterfly and grasshopper species richness and abundance at the two sites were similar.

Measures of species richness and quality

The various measures of plant diversity—species richness, number of 'quality' species, α and β diversity—generally showed moderate to strong positive correlations with both the butterfly total species and quality species richness at both study sites (Table 2; Fig. 2). At Mesocco, however, the correlations between the α diversity of plants and both total and quality species richness of butterflies did not quite reach the level for significance ($p = 0.064$ and $p = 0.113$). At Guarda, the correlations between the four plant measures and species richness of grasshoppers were not significant (Table 2; Fig. 3).

Community similarity

Cover as well as presence community data of vascular plants coincided well with insect abundance at both sites. Apart from the insignificant relationships between plant species cover and grasshopper abundance as well as plant quality species cover and butterfly and grasshopper abundance at Mesocco, all relationships between plant and insect community data showed highly significant positive correlations at both sites (Table 3; Figs. 4, 5).

Discussion

In this study of two subalpine pastures, we found generally strong and positive congruence between various measures of plant diversity (species richness patterns, α diversity and β diversity) and the species richness of butterflies and grasshoppers, though with some differences between the insect taxa and the sites assessed. However, congruencies between plant and invertebrate communities were consistently strong for both taxa and sites regarded.

One explanation for the congruence between plants and butterflies is their producer–consumer relationship. Most butterflies are dependent upon one or a few plant species during their larval stage, while adult butterflies also show more or less strong associations with certain flowering plants for nectar (e.g. Tudor et al. 2004). These relationships are well reflected in our results obtained, which agree with previous studies and confirm that vascular plants can be taken as a good surrogate for butterflies both at the levels of species richness (Panzer and Schwartz 1998; Pearman and Weber 2007) and community similarity (Santi et al. 2010; Su et al. 2004).

The results for grasshoppers are less clear. A significant positive correlation between vascular plants and grasshoppers was found in a study of low altitude agricultural landscapes in Austria (Sauberer et al. 2004). However, our data only partially support their findings, with differences between the two study areas, particularly at the level of species richness. Although grasshoppers also depend upon vascular plants, most species are

Table 1 Summary statistics for the plant, butterfly and grasshopper taxa collected in 900 m² plots at the study sites Mesocco ($n = 49$) and Guarda ($n = 50$). Insect data were pooled over the summer season (five surveys for butterflies and two for grasshoppers)

	Measure	Sampling area[b]	Mesocco			Guarda		
			Min–max	Mean ± SE	Total	Min–max	Mean ± SE	Total
Vascular plants	Species richness (total number of species)	9 × 1 m² and 30 min walk	36–110	67.1 ± 2.7	318	41–116	76.1 ± 2.3	332
	Number of quality species[a]	9 × 1 m² and 30 min walk	5–36	19.6 ± 1.2	94	5–44	24.6 ± 1.1	100
	Alpha diversity (mean number of species)	9 × 1 m²	6–23	13.3 ± 0.7	–	12–30	21.7 ± 0.6	–
	Beta diversity	9 × 1 m²	12–56	30.9 ± 1.5	–	18–62	37.7 ± 1.5	–
Butterflies	Species richness	900 m² plot	3–14	7.8 ± 0.5	44	4–22	10.1 ± 0.5	52
	Number of individuals	900 m² plot	8–76	31.0 ± 2.1	1,517	7–93	48.6 ± 3.2	2,429
	Number of quality species[a]	900 m² plot	0–4	0.9 ± 0.2	19	0–10	4.0 ± 0.3	25
Grasshoppers	Species richness	900 m² plot	2–6	3.5 ± 0.1	8	2–10	5.5 ± 0.2	14
	Number of individuals	900 m² plot	1–316	65.4 ± 8.7	3,207	22–329	127.4 ± 9.6	6,369
	Number of quality species[a]	900 m² plot	0–1	–	1	0–6	2.7 ± 0.2	7

SE Standard Error

[a] Number of species indicating ecological quality in Swiss agricultural landscapes (BAFU and BLW 2008)

[b] Vascular plant species were recorded in 9 × 1 m² quadrats within the 900 m² plots as well as during targeted search for further 30 min walk through the plot. To calculate alpha and beta diversity only the 9 × 1 m² quadrats were considered. Insects were recorded on a serpentine-like transect covering the whole plot

Table 2 Pairwise comparisons between four plant measures and two measures for butterflies and grasshoppers in the study sites Mesocco ($n = 49$) and Guarda ($n = 50$). Partial Spearman's rank correlation and significance values (p, in brackets; significant values in bold) are reported

Study areas	Comparisons	Plant measures			
	Sampling area	Species richness 900 m²	Quality species 900 m²	α-Diversity 9 × 1 m²	β-Diversity 9 × 1 m²
Mesocco	Butterfly species richness	**0.362 (0.009)**	**0.337 (0.016)**	0.266 (0.064)	**0.396 (0.004)**
	Butterfly quality species	**0.302 (0.034)**	0.273 (0.057)	0.230 (0.113)	**0.353 (0.011)**
	Grasshopper species richness	**0.387 (0.005)**	**0.408 (0.003)**	**0.492 (<0.001)**	**0.375 (0.007)**
	Grasshopper quality species[a]	–	–	–	–
Guarda	Butterfly species richness	**0.579 (<0.001)**	**0.668 (<0.001)**	**0.551 (<0.001)**	**0.550 (<0.001)**
	Butterfly quality species	**0.426 (0.001)**	**0.472 (<0.001)**	**0.460 (<0.001)**	**0.404 (0.003)**
	Grasshopper species richness	0.245 (0.087)	0.164 (0.259)	0.139 (0.343)	0.270 (0.057)
	Grasshopper quality species	0.227 (0.114)	0.148 (0.309)	0.052 (0.722)	0.224 (0.120)

[a] This comparison was not made because of the presence of only one quality grasshopper species at the Mesocco site
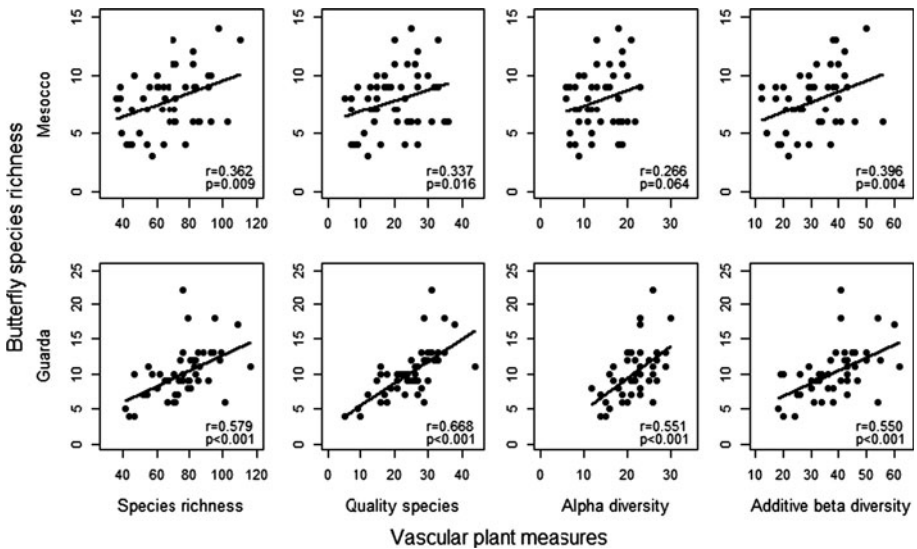

Fig. 2 Four plant measures against butterfly species richness in Mesocco ($n = 49$) and Guarda ($n = 50$). Spearman's rank correlations (r) and significances (p) are reported in the single graphs. Regression lines are shown to simplify interpretation

generalists and their diets may include a relatively wide range of plant species. Thus, aspects such as biomass production, vegetation structure or microclimate are more important in shaping grasshopper distribution than plant species richness per se (Detzel
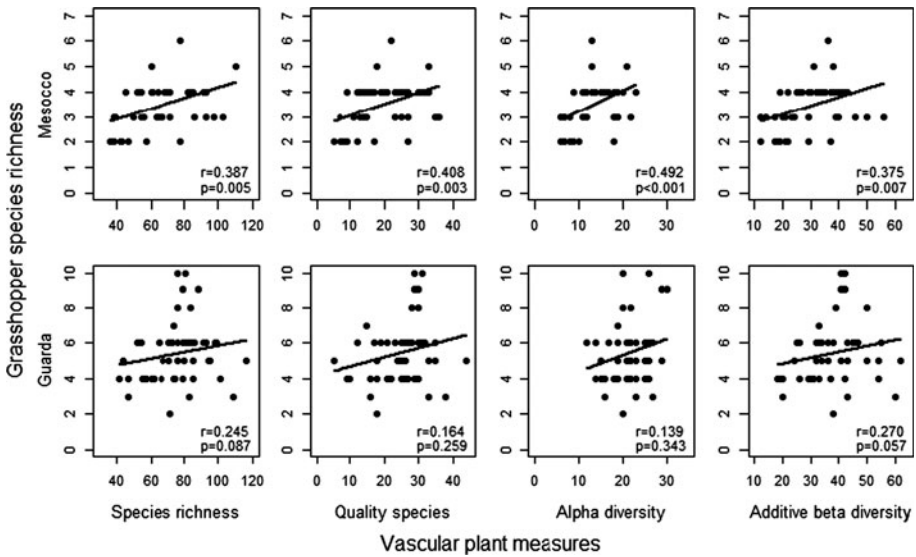
Fig. 3 Four plant measures against grasshopper species richness in Mesocco ($n = 49$) and Guarda ($n = 50$). Spearman's rank correlations (r) and significances (p) are reported in the single graphs. Regression lines are shown to simplify interpretation

Table 3 Correlation and significance values (p, in brackets; significant values in bold) of partial PROTEST with 1,000 permutations between (Bray–Curtis) community similarity matrices among vascular plants, butterflies and grasshoppers in Mesocco ($n = 49$) and Guarda ($n = 50$). Cover data for plants and abundance data for insects were square-root transformed prior to analysis

Study areas	Comparisons	Plant measures			
	Sampling area	All species cover 9×1 m^2	All species presence 900 m^2	Quality species cover 9×1 m^2	Quality species presence 900 m^2
Mesocco	Abundance of butterfly species	**0.516 (0.001)**	**0.480 (0.001)**	0.164 (0.290)	**0.447 (0.001)**
	Abundance of grasshopper species	0.285 (0.078)	**0.329 (0.007)**	0.092 (0.559)	**0.355 (0.004)**
Guarda	Abundance of butterfly species	**0.593 (0.001)**	**0.577 (0.001)**	**0.576 (0.001)**	**0.537 (0.001)**
	Abundance of grasshopper species	**0.712 (0.001)**	**0.649 (0.001)**	**0.607 (0.001)**	**0.584 (0.001)**

1998; Szövényi 2002). Furthermore, the number of grasshopper species surveyed at these higher altitudes was relatively low. Results for the species richness and their interpretation should therefore be treated with caution as the small species pool might have affected the statistical outcomes, at least in terms of reduced power to obtain significant results. Despite the few species, the total numbers of grasshoppers at both sites were high, making community similarity a more appropriate measure for comparison. Oertli et al. (2005) also found stronger correlations between grasshoppers and bees using community similarity data instead of species numbers.
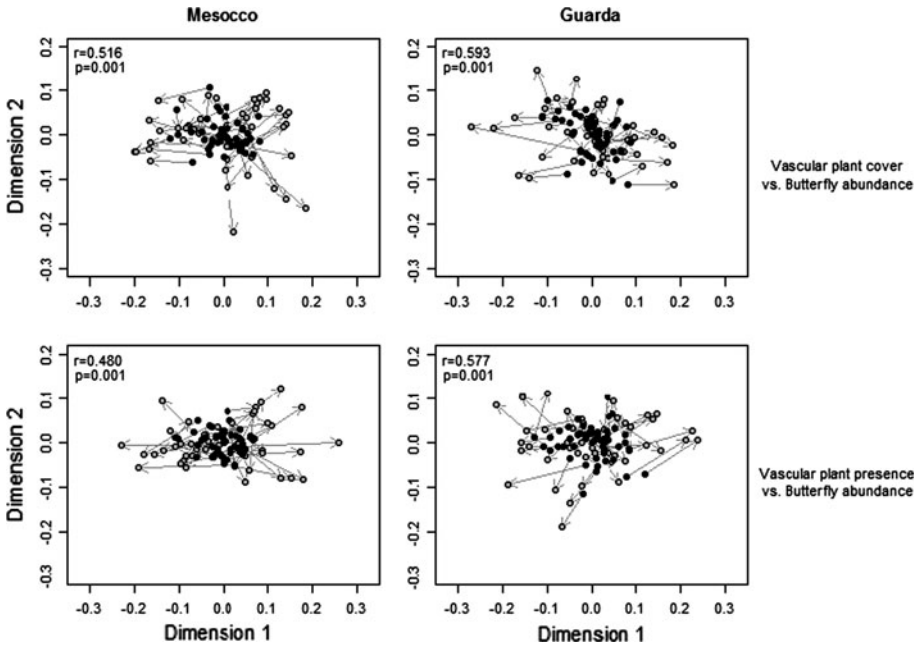
Fig. 4 Procrustean superimposition plots between vascular plant cover or presence and butterfly abundance in Mesocco ($n = 49$) and Guarda ($n = 50$). Procrustes correlations (r) and significances (p) are reported in the graphs. The procrustean errors are shown in the graphs as *arrows*, the length of which illustrates the amount of difference between the community similarities (Bray–Curtis similarity) of the vascular plant and butterfly taxa. The *arrows* start from the original rotated data (butterflies, *black circles*) and points to the target data (vascular plants, *greycircles*)

In general, insect communities showed higher similarity among plots than plant communities. Particularly the eight rather generalist grasshopper species at the Mesocco site caused the species assemblages to be very homogeneous among plots with rather different vegetation communities (Fig. 5). These findings are comparable to observations of Murdoch et al. (1972), who found the Homoptera assemblages in three old fields to be more similar than those of plants. This suggests less site specificity in the insect communities, which seem to be relatively unaffected by differences in the plant communities. One explanation might be the greater mobility of butterflies and grasshoppers, which leads to less distinct boundaries between communities (Samways et al. 2012), reinforcing the argument for a vegetation-based approach.

Besides the direct associations of vascular plants with butterflies and grasshoppers, the taxa also share similar responses to environmental factors and management (e.g. Marini et al. 2009). Oertli et al. (2005) suggested that congruence of diversity patterns is more likely at local scales when there is a strong environmental gradient to which the taxa respond similarly. For example, Su et al. (2004) concluded that the congruence of plant, bird and butterfly communities was due to a hydrological gradient. The sites in our study were topographically very heterogeneous with very variable micro-climatic conditions, which are likely to have influenced the distribution of species and thus potentially underlie congruencies.

The spatial scale at which species diversity is studied may also affect the congruence among taxa (Favreau et al. 2006; Wolters et al. 2006), whilst local environmental
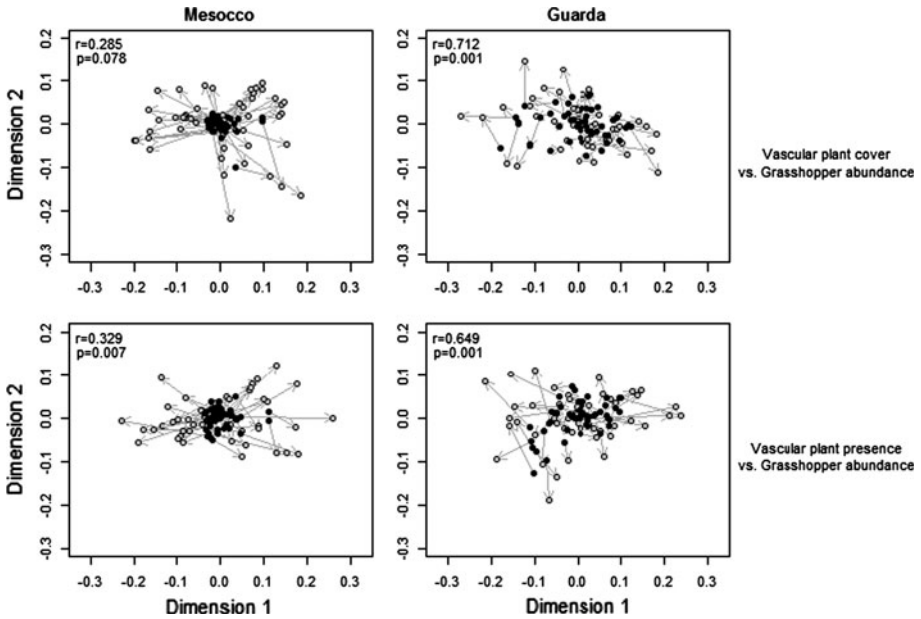
Fig. 5 Procrustean superimposition plots between vascular plant cover or presence and grasshopper abundance in Mesocco (*n* = 49) and Guarda (*n* = 50). Procrustes correlations (r) and significances (p) are reported in the graphs. The procrustean errors are shown in the graphs as *arrows*, the length of which illustrates the amount of difference between the community similarities (Bray–Curtis similarity) of the vascular plant and grasshopper taxa. The *arrows* start from the original rotated data (grasshoppers, *black circles*) and points to the target data (vascular plants, *grey circles*). It is worth to note the homogeneous grasshopper communities in Mesocco compared to Guarda

conditions are likely to be more important at the small scale, evolutionary, historical or climatic factors may be more influential at larger scales (Heino et al. 2009). Although it is reported that taxa show better correspondence in species richness at larger scales (Wolters et al. 2006), the importance of studying congruence among taxa at a small scale has been repeatedly emphasized (Vera et al. 2011; Weaver 1995). It can be argued that this is the scale at which conservation actions and management planning such as agri-environmental schemes are mostly carried out, and therefore the scale at which congruence analyses would be of most practical relevance. Even though ours was a small-scale study, vascular plants performed well as surrogates of butterflies and grasshoppers at the community similarity level. Despite divergent results of surrogate studies, vascular plants have been repeatedly recommended as one of the best indicators for overall biodiversity (Duelli and Obrist 1998; Kati et al. 2004; Sauberer et al. 2004).

In order to capture different aspects of species diversity patterns, we also used measures of plant diversity other than species richness. Whilst Spearman rank correlations between the average plant diversity and invertebrate taxa within samples (α diversity) were lower, correlation values for the diversity among samples (β diversity) were as high as those for the species richness, and in some cases even higher. Not only the number of plant species present but also the spatial turnover of plant species among neighbouring areas is thus an important aspect in shaping the diversity of butterflies and grasshoppers in this subalpine habitat, possibly reflecting the considerable topographical heterogeneity in the areas studied. This outcome is in contrast to the results of Kessler et al. (2009), who compared

species richness and β diversity of plants and animals in Indonesia and found that β diversity of one taxon had very limited indicator potential for species richness of other taxa. The differing results could be explained by the fact that Kessler et al. (2009) included different habitat types, and that some taxa showed divergent preference for specific habitats, while our study only included alpine grasslands. Nevertheless, the community similarity approach, which not only incorporates the abundances of single species but also provides information on the species identity, yielded strong congruencies between plants and invertebrates. This agrees with previous studies on surrogate indicators that found measures based upon community similarity to be more effective than those based on species richness (Bilton et al. 2006; Lovell et al. 2007; Su et al. 2004).

As the whole of biodiversity cannot be assessed, specific taxa are often used as surrogates for other taxa to reduce the time and cost of monitoring in conservation planning. The effort can be reduced even further if a subset of species from one taxon, rather than recording all species, is used. This approach is already utilized in agri-environment schemes in Switzerland and Germany, where result-oriented payments to the farmers and thus biodiversity conservation of agricultural areas are based on a list of plant indicator species (Oppermann and Gujer 2003). Our results indicate that the plant 'quality species' represent well species richness and community similarity of butterflies and grasshoppers. These findings partly contradict results from Vellend et al. (2008) who found that the strength of correlations for species richness and composition are reduced if only a subset of species is used; on the other hand, they argue that the use of a subset can even maximize statistical power, and that little information is lost by eliminating a small proportion of rare species (e.g. below 10 %). In any case, the success of a subset relative to all species will highly depend on how the species are selected; whilst the subsets used by Vellend et al. (2008) were obtained by removing species randomly, the so-called quality species used here were selected specifically for their conservation value.

In summary, our investigation of subalpine Swiss summer pastures has shown strong overall correspondence between vascular plants and two insect taxa, butterflies and grasshoppers, especially when community diversity measures are used rather than simple species richness. We therefore recommend the use of community similarity data in studies evaluating surrogacy, as this measure takes species identity as well as species abundance into account, just as do standard diversity indices. Although our correlations do not reach the value of 0.75 recommended by Lovell et al. (2007) as a criterion for accepting vascular plants as a suitable surrogate for other taxa, given the consistently strong correlations at the community level, we are confident that vascular plants represent an acceptable surrogate for butterflies and grasshoppers (though caution is required when the species pool is small). This in turn implies that conservation measures aimed primarily at protecting the plant diversity of subalpine grasslands will also benefit butterflies and grasshoppers, at least at the local scale at which management decisions are usually made. We expect a similar profit for other invertebrate taxa such as bumblebees or syrphid flies that show strong interactions with plants. Nevertheless, animal groups not directly related to plants are likely to show poor congruence. Therefore, Finch and Löffler (2010) recommended the use of one group of invertebrates in addition to plants as indicator for conservation planning in alpine areas. Given the lack of studies in these areas, more effort should be dedicated to investigate surrogacy, especially for invertebrate groups not directly related to specific plant taxa. As the results of this study are based on 1 year survey data, further research including multi-year analyses would also be needed to confirm the pattern detected. Furthermore, we conclude that recording a subset of vascular plants (the 'quality' species) can substantially increase the efficiency of assessments without reducing their predictive value as surrogates

for other taxa. Our results suggest that the Swiss 'quality species' are well selected to represent insect species richness and community similarity. They, therefore, are well suited for assessing result-oriented policy.

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