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ORIGINAL PAPER

Assessing the effects of vegetation type on hoverfly (Diptera: Syrphidae) diversity in a Mediterranean landscape: implications for conservation

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Abstract Understanding the responses of insects to ecological variables provides information that is fundamental for their conservation. The present study took place in three different landscapes (three plots of 10 × 10 km each) in a typical Mediterranean ecosystem of a Spanish national park. Each landscape included three vegetation types, grasslands, scrublands, and woodlands, and was characterised by a dominant vegetation type. Our objectives were: (1) to assess how important the influence of the dominant vegetation type and the vegetation type of sites are on hoverfly (Diptera: Syrphidae) diversity at landscape scale; (2) to assess whether scrublands are contributing to the loss of hoverfly diversity in an ecosystem with a long history of human use. In order to achieve these goals, we compared hoverfly diversity among sampling sites by prospecting all three diversity levels, alpha, beta and gamma, at each landscape. We sampled adult hoverflies at 18 sites located in different vegetation types within the three landscapes. No evidence was found that demonstrated an effect of the dominant vegetation type on alpha or beta diversity of sites, but the vegetation type of sites did have an affect. The highest species richness was found in woodland sites, particularly in the grassland-dominated landscape. At each landscape, beta diversity among sampling sites contributed more to gamma diversity than alpha diversity did. Our results highlight the need to focus on the conservation of woodland remnants of grassland-dominated landscape and scrubland-dominated landscape in order to preserve a large proportion of the biodiversity of Cabañeros hoverflies, as well as on the maintenance of the mosaic landscape, which is linked to high beta diversity, typical in many Mediterranean ecosystems. We emphasise the importance of open clearings in the vast mass of scrubland in the scrubland-dominated landscape in order to provide extra resources for the hoverflies.

Keywords Alpha, beta and gamma diversity · Grassland · Scrubland · Woodland · National park · Spain

Introduction

Since the early Neolithic, the Mediterranean region has experimented critical changes in its landscapes as a result of agriculture and stockbreeding (Vaquero 1997). These human activities have continuously influenced the dynamics of the mosaic landscape that is typical of the Mediterranean region, where grasslands and scrublands merge with residual woodlands (Bignal and McCracken 2000). Recent abandonment of agriculture and stockbreeding activities (Hernández 1997; Verdú et al. 2000; Allen 2003) has had an impact on the flora and fauna of the region, which had adapted to human-regulated conditions over millennia (Zamora et al. 2006).

The Cabañeros National Park is in the "Montes de Toledo" mountain range (Central Spain). From 1246,

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"Montes de Toledo" was governed under feudal law, which restricted population development and allowed only low-intensity traditional activities. In the 15th century, oak forests began to be exploited but without any significant alteration of the wooded character of the area (Muñoz 1976). From the 16th to the 18th century, tall, dense scrublands occurred in the low and flat areas of Cabañeros, while forests occurred in valleys, bordering streams or on riverbanks. Until the 19th century the intensity of activities such as clearing, stockbreeding and charcoal production (charcoal was also extracted from scrub vegetation) moderately increased and scrublands began to open in some areas (Vaquero 1997). In the 19th century, the parcels with more productive potential, which were mainly colonized by Mediterranean scrub, went to private ownership and their vegetation changed, with the less productive areas (high and hilly) remained free of alteration. This was due to the direct influence of the freeing of encumbrances. Around the middle of the 20th century, extensive areas of native scrubland and woodland were cleared for agriculture and stockbreeding (Vaquero 1997). Moreover, during the 20th century the populations of predatory mammals such as the wolf, *Canis lupus* (L., 1758) and the lynx, *Lynx pardinus* (Temminck, 1827), became extinct or their numbers were critically reduced (Guzmán 1997). In the last decades of the 20th century, agriculture and stockbreeding were abandoned and Cabañeros was established as a protected area in 1988. Currently, large areas of grassland, called "raña", are maintained by a large population of wild herbivores, mainly red deer, *Cervus elaphus* L., 1758, but also wild boar, *Sus scrofa* L., 1758, and roe deer, *Capreolus capreolus* (L., 1758) (Vaquero 1997). These species are at very high density, not only because of the scarcity of predators, but also because some private farms in the park have hunting businesses (Guzmán 1997). Although the abandonment of traditional activities is promoting a higher density of the vegetal cover (Velasco 1978), mainly in scrublands, the overpopulation of herbivores is modifying the vegetation dynamics in Cabañeros; for instance, forests are not regenerating because wild herbivores feed on the young trees. Scrublands are becoming a close, dense mass of tall vegetation encroaching on clearings within them. This encroachment is related to the abandonment of traditional activities such as charcoal production and the extraction of cork from cork oaks, *Quercus suber* L.

To assess the impact of changes in land use on biodiversity, especially in regions with a long history of human activity, studies at the landscape scale (mesoscale) have been suggested as the most appropriate (Halffter 1998; Haber 2004). The environmental heterogeneity of an area affects ecological patterns and processes in landscapes (Wiens et al. 1993; Welsh et al. 2005; Farina 2006; Gaucherel 2007); for instance, the diversity of a patch is affected by the diversity in surrounding areas (Shmida and Wilson 1985; Cook et al. 2002; Steffan-Dewenter 2003; Murphy and Lovett-Doust 2004; Devictor and Jiguet 2006). Species richness is a measure of diversity that underlies many ecological models and conservation strategies (Gotelli and Colwell 2001). The quantification of the three components of species diversity—alpha, beta and gamma (Whittaker 1972)—has been used in studies at different scales (e.g. Wagner et al. 2000; Gabriel et al. 2006). At the landscape scale, diversity of the total number of communities composing a landscape (gamma) may be high because of either high species richness in a particular community within the landscape (alpha) or a high turnover of species among communities within the landscape (beta) (Whittaker 1972). In addition, the impact of human-derived changes to the landscape may increase or decrease these components depending on the biological group in question (Pineda et al. 2005).

Hoverflies (Diptera: Syrphidae) have been the focus of several studies of ecology and biodiversity in Europe (e.g. Haslett 1994, 1997; Gittings et al. 2006; Ouin et al. 2006; Schweiger et al. 2007). Hoverflies are adept fliers that interact directly with vegetation because of their dietary requirements (Haslett 1997). Adult hoverflies feed on pollen and nectar but the larvae feed on other insects (predators), on vegetal tissues (phytophagous) or on microorganisms and detritus (saprophagous) (Thompson and Rotheray 1998). The use of hoverflies as bioindicators of landscape has been proposed (Burgio and Sommaggio 2007; Billeter et al. 2008), and they are a focus of conservation in the European continent (e.g. Rotheray et al. 2001; Marcos-García 2006). Although greater patch area, connectivity, and habitat heterogeneity have positive effects on hoverfly richness (Ouin et al. 2006), it has been found that patch size and shape have little effect on the spatial distribution of hoverflies, probably because of the high mobility of adults (Sutherland et al. 2001). Hoverflies include generalist and cosmopolitan species, which can be very abundant, but also species that are represented by a very low number of individuals because of spatial or temporal restrictions or habitat specialisation (Owen and Gilbert 1989). It has been shown that most species of Syrphidae live in woodlands, where they form highly diverse and species rich communities, including rare species which are often saproxylic (Speight 1989; Rotheray et al. 2001). Although open spaces within woodlands promote hoverfly diversity (Fayt et al. 2006; Gittings et al. 2006) only a few species have successfully colonised open and anthropogenic habitats (Branquart and Hemptinne 2000).

In this study, we assessed how vegetation type in the wider landscape affects the diversity of hoverflies within three different landscapes. Each landscape included three vegetation types (grassland, scrubland, and woodland), and was characterised by a different dominant vegetation type. We hypothesised that the dominant vegetation type in a landscape influences the hoverfly diversity of sites due to the effect of the areas around sites (Shmida and Wilson 1985; Ås 1999). On the basis of the preference for woodland that is exhibited by many hoverfly species, including rare and threatened species and the positive effect of open areas in woodlands on species diversity, we expected that mean alpha diversity would be high and beta diversity would be low in the woodland-dominated landscape in Cabañeros. All the prospected woodlands in Cabañeros comprise mostly trees that provide microhabitats for saproxylic hoverfly larvae but also other more restricted environmental elements such as bodies of water (temporary or permanent) and/or invasive areas of scrub and grassy clearings (Vaquero 1997). In contrast, we expected that in the scrubland-dominated landscape, mean alpha and beta diversity would be low; Cabañeros scrubland is a closed, dense, extensive and homogeneous vegetation type with few trees, and it is progressively increasing because of the abandonment of past human activities (see above). The invasion of grassy areas by scrubs has a negative effect on Mediterranean biodiversity (Verdú et al. 2000; Zamora et al. 2006) and adequate management of affected areas such as Cabañeros should avoid this invasive process. An intermediate situation was expected for the grassland-dominated landscape; Cabañeros grassland is a very open vegetation type with some water courses and scattered trees providing microhabitats for a broader range of hoverflies than that found in the scrubland, but narrower than that in the woodland. The maintenance of the grasslands in Cabañeros is guaranteed because of the grazing action of wild herbivores, but the encroachment of the scrublands is not stopped because wild herbivores do not feed on scrub (Guzmán 1997).

The objectives were: (1) to assess how important the influence of the dominant vegetation type and the vegetation type of sites are on hoverfly diversity at the landscape scale; (2) to assess whether the scrublands are contributing to the loss of hoverfly diversity in a typical Mediterranean ecosystem, protected nowadays but with a long history of human use. To achieve these goals, our specific objectives were to compare hoverfly species diversity among sampling sites by prospecting all three diversity levels: alpha, beta and gamma.

Materials and methods

Study area

The area of study was the Cabañeros National Park (39° 23′ 47″ N, 4° 29′ 14′ W), Central Spain. The park consists of 40,856 ha of typical Mediterranean ecosystem, with altitude ranging from 600 to 1488 m. The average annual temperature varies from 12.9°C in the north to 15.6°C in the south, the monthly average temperature fluctuates from 3.9°C (December) to 23.8°C (July); extreme temperatures of more than 40°C in summer and less than −12°C in winter are possible (Vaquero 1997). The average annual precipitation is 586.4 mm, and the wettest and driest months are January and July/August, respectively; the park has annual precipitation averages of 750 mm in the north and 500 mm in the south (Vaquero 1997).

A detailed vector map of vegetation that had been produced by staff of the national park was simplified by Numa et al. (2009) to show the most widely distributed structural categories of vegetation in the park: grassland ("raña"), scrubland, and woodland (Fig. 1). Using this simplified landscape analysis, we considered three 10 × 10 km plots as different landscapes (Fig. 1). Each landscape comprised all three vegetation types but was dominated by only one of them (Table 1), comprising a grassland-dominated landscape (G-landscape), a scrubland-dominated landscape (S-landscape), and a woodland-dominated landscape (W-landscape; Fig. 1). The G-landscape was at the lowest altitudinal level, although some mountains with an average altitude of 1,000 m were present in the north of the plot. The grassland of the G-landscape was mostly represented by a vast plain covered by grassland plants (e.g. Cariofilaceae, Compositae, Gramineae, Leguminosae, and Liliaceae) with scattered trees (*Quercus ilex* L. subsp. *rotundifolia* (Lam.) Tab. Morais, *Q. faginea* Lam., and *Q. suber*; Vaquero 1997). The S-landscape comprised low and moderately flat areas as well as hillier areas with valleys. The scrubland of the S-landscape had medium- to large-sized scrubs (1–1.75 m tall), where the most common species were *Arbutus unedo* L., *Cistus ladanifer* L., *Erica arborea* L., *E. umbellata* L., *Phillyrea angustifolia* L., *Rosmarinus officinalis* L. and some scattered *Pyrus* and *Quercus* trees (Vaquero 1997). The W-landscape was the highest and hilliest of the three landscapes. The woodland of the W-landscape comprised deciduous (*Q. faginea* and *Q. pyrenaica* Willd.) and/or evergreen species (*Q. ilex* subsp. *rotundifolia* and *Q. suber*). Until the extensive clearings of the 20th century the current vast grassy plain of the G-landscape was mainly covered by native scrubland. Today woodlands occur along the valleys and bordering water courses in both the G- and S-landscapes.

Hoverfly sampling

We established two sampling sites as replicates within each of the three vegetation types within each landscape, to give a total of six sampling sites within each landscape and 18 sampling sites overall (Fig. 1). Sampling sites were located in different patches of a given vegetation type. Sites were denoted by a code of two letters and a number: first, an upper-case letter indicated the dominant vegetation type (grassland = G; scrubland = S; woodland = W) in the corresponding landscape; second, a lower-case letter indicated the

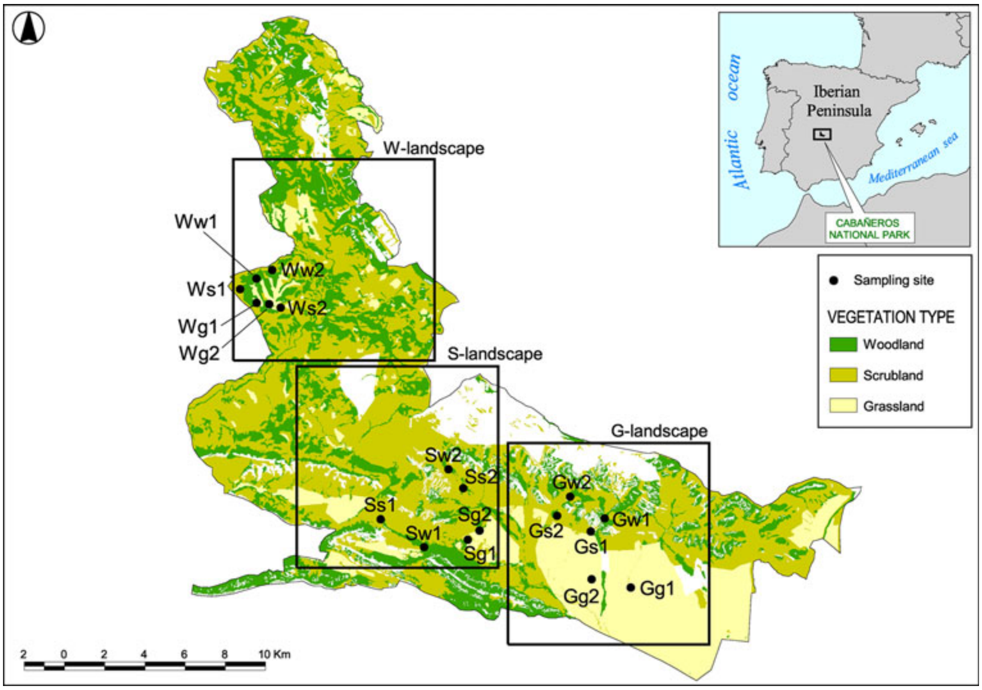

Fig. 1 Map showing the distribution of the 18 sampling sites within the three defined 10 × 10 km plots at Cabañeros National Park, Spain. Sites were denoted by a code of two letters and a number: first, an upper-case letter indicated the dominant vegetation type (*G* grassland, *S* scrubland, *W* woodland) in the corresponding landscape; second, a lower-case letter indicated the vegetation type of the site (*g* grassland, *s* scrubland, *w* woodland); and third, a number indicated the replicate. Each plot is designated with the following abbreviations: *W-landscape* woodland-dominated landscape, *S-landscape* scrubland-dominated landscape, *G-landscape* grassland-dominated landscape. In blank, stony areas and pine plantations, both considered in the calculations of the parameters provided in Table 1 but not in the diversity analyses

Table 1 General characteristics of each of 10 × 10 km landscapes defined in Cabañeros National Park, Spain

	Vegetation type	G-landscape	S-landscape	W-landscape
Area%				
	Grassland	40	8.8	7.7
	Scrubland	31.1	54.4	45.6
	Woodland	28.8	36.8	46.6
Mean patch size (Ha)				
	Grassland	105.9	12.8	11.9
	Scrubland	16.5	31.9	29.2
	Woodland	11	14	44.5

G-landscape grassland-dominated landscape, *S-landscape* scrubland-dominated landscape, *W-landscape* woodland-dominated landscape. [Table adapted from Numa et al. (2009)]

vegetation type of the site (grassland = g; scrubland = s; woodland = w); and third, a number indicated the replicate. For example, Gw1 and Gw2 were the two sampling sites in the woodlands within the G-landscape (Fig. 1).

Following a standardised sampling protocol (Pollard and Yates 1993), we used an entomological hand net to collect adult hoverflies. We performed sampling for 2 h (effort unit) each month at each of the 18 sampling sites for 13 months from May to November 2004 and from April to September 2005, which corresponded to a total of 468 h. The sampling period in both years was chosen to avoid the period of winter inactivity that is typical of Mediterranean insect communities (Lumaret and Kirk 1991). Species were identified using keys and descriptions listed in Ricarte and Marcos-García (2008). Taxonomic nomenclature follows Speight (2010). A detailed table of species and abundances for each vegetation type in each landscape is provided in Appendix, where the larval trophic habits of each species are specified following Speight (2010).

Data analysis

We measured all three levels of diversity, alpha, beta and gamma. The spatial scale used refers to the three diversity levels as follows: alpha = sampled site/vegetation-type; beta = turnover among vegetation types; gamma = landscape.

Alpha diversity

To assess the completeness of the inventory of each sampling site, we compared the observed species richness with the expected species richness according to the Chao 1 estimator computed using EstimateS 8.0® (Colwell 2006). Inventory completeness for each site was calculated as the number of species observed as a proportion of the total number of species predicted by the estimator (Colwell and Coddington 1994). To assess the effect of the dominant vegetation type on alpha diversity we compared the species richness of the six sampling sites for each vegetation type in the three landscapes, to determine whether the number of species present depended on the landscape where they were located. For this comparison of species richness, we used the Mao-Tau function (Gotelli and Colwell 2001), which employs the same basic algorithm as that in the rarefaction (Colwell 2006); we also calculated 95% confidence intervals of the Mao-Tau function.

Beta diversity

Non-metric multidimensional scaling (NMDS) was used to analyse the general pattern of dissimilarity in species composition and relative abundance among the 18 sampling sites using Bray-Curtis dissimilarity values between the sampling sites. The NMDS was performed using PRIMER software (Clarke and Gorley 2001).

Gamma diversity

To determine whether species richness within sampling sites (alpha diversity) or species turnover (beta diversity) are the principal drivers of the diversity in each landscape (gamma diversity), we performed the procedure described by Lande (1996) which additively partitions the total gamma diversity of a landscape by analysing the contribution of each alpha and beta component (Loreau 2000; Veech et al. 2002). In following the procedure, only species richness was considered and an equal weight was assigned to each of the three vegetation type conditions.

Results

We found 72 hoverfly species from a total of 2361 individuals, 69% of which belonged to four species: *Sphaerophoria scripta*, *Eristalis similis*, *Eristalis tenax*, and *Melanostoma mellinum*, in decreasing order of abundance (Appendix). *Sphaerophoria scripta* and *M. mellinum* are cosmopolitan species associated with grassy vegetation (Speight 2010), but we collected high quantities of individuals of *S. scripta* and most of the individuals of *M. mellinum* in the woodland sites within all three landscapes. *Eristalis tenax* is a widespread generalist species but *E. similis* is associated with woodlands in the Mediterranean region (Speight 2010); the stated preference of *E. similis* was not followed in this sampling (Appendix).

We found 30 rare species, represented by one or two individuals among the total number of collected specimens (Appendix). A third of these rare species have saproxylic larvae and are associated with woody environments (Speight 2010); all of them were collected in woodland sites, except for *C. aurea* (Appendix). Almost a third of the 30 rare species have predator larvae and seven species have phytophagous larvae living in bulbs and tubers of geophytes (Appendix). For all three landscapes, woodlands always exhibited the highest numbers of species and individuals, followed by grasslands, and scrublands (Appendix).

Alpha diversity

Species inventories at the different sampling sites were 42.3–96.6% complete, although nine of the 18 sampling sites were more than 70% complete according to the Chao 1 estimator (Table 2). At 12 sites, 40% of the species, or more, were singletons (Table 2). There was no bias in the completeness with respect to vegetation type or landscape, because the six inventories that were the least complete were obtained from all vegetation types and landscapes (Table 2).

In general, we found that species richness curves for sampling sites of the same vegetation type overlapped, even when they were located in landscapes with different dominant vegetation types (Fig. 2). For grasslands and scrublands (Fig. 2a, b), 95% confidence intervals of Mao-Tau function curves overlapped at the level of the sampling site with the lowest abundance. For woodlands (Fig. 2c), there was only a slight separation of confidence intervals between Gw1 and Ww1.

Beta diversity

The NMDS shows an evident grouping of sites according to vegetation type and not to the landscapes where they were located (Fig. 3). The three well-defined groups of sites can be explained by the presence of species sensitive to particular vegetation conditions, while the influence of a landscape with a dominant vegetation type is not evident for hoverflies at the scale of this study (NMDS Stress: 0.11).

Gamma diversity

The highest level of gamma diversity was calculated for the G-landscape, followed by the S-landscape, and the W-landscape (Table 3). Furthermore, regardless of the dominant vegetation type, the gamma diversity of the three landscapes was affected mainly by beta diversity (61–64%), whereas the mean alpha diversity of the

Table 2 Observed species richness (S_{obs}) of hoverflies and number of species represented by one (singletons) or two individuals (doubletons)

Site	S_{obs}	Singletons	Doubletons	Chao 1	Completeness (%)
Grasslands					
Gg1	16	7	3	21.25	75.3
Gg2	13	5	0	23	56.5
Sg1	11	6	0	26	42.3
Sg2	10	4	1	13	76.9
Wg1	12	6	1	19.5	61.5
Wg2	12	3	4	12.6	95.2
Scrublands					
Gs1	11	4	2	13	84.6
Gs2	14	7	1	24.5	57.1
Ss1	12	5	1	17	70.6
Ss2	8	5	0	18	44.4
Ws1	12	7	1	22.5	53.3
Ws2	10	3	0	13	76.9
Woodlands					
Gw1	33	14	5	48.17	68.5
Gw2	27	11	4	38	71.1
Sw1	18	10	1	40.5	44.4
Sw2	28	4	5	29	96.6
Ww1	16	4	1	19	84.2
Ww2	30	14	6	43	69.7

Inventory completeness is observed richness as a percentage of total expected richness according to the Chao 1 estimator, for each sampling site at Cabañeros National Park, Spain

sampling sites contributed only 35–39% to the gamma diversity (Table 3). Thus, species turnover (beta diversity) is the principal driver of the diversity in each landscape (gamma diversity); this is related to the importance of the vegetation mosaic in this Mediterranean ecosystem.

Discussion

Our results for the alpha, beta, and gamma diversity show that the dominant vegetation type defining a landscape does not have an effect on hoverfly diversity in Cabañeros at the landscape scale, but that the vegetation type of sites clearly affects the richness and composition of species. The following results confirm the weak effects of the dominant vegetation type: (1) richness of sampling sites of the same vegetation type did not differ even when they were located in a landscape with a different dominant vegetation type; (2) sampling sites were grouped according to their vegetation type but not according to their location in a particular landscape in terms of the dissimilarity of the range of collected hoverflies; and (3) the relative contribution of alpha and beta components to the gamma diversity was very similar among landscapes, in spite of their different dominant vegetation types. These results from this Mediterranean landscape support the close interaction of hoverflies with the vegetation, as described by Haslett (1997). In addition, the results show that a mosaic of vegetation types is necessary—and must be preserved and adequately managed—for hoverflies, as this mosaic provides all the resources required for the development of the different stages in the life cycle of a hoverfly species; for instance, *Ferdinandea aurea* is a species with saproxylic larvae that develop in trees, but it was collected in the grasslands of the W-landscape (Appendix) because grasslands provide an abundant number of flowers for adult feeding.

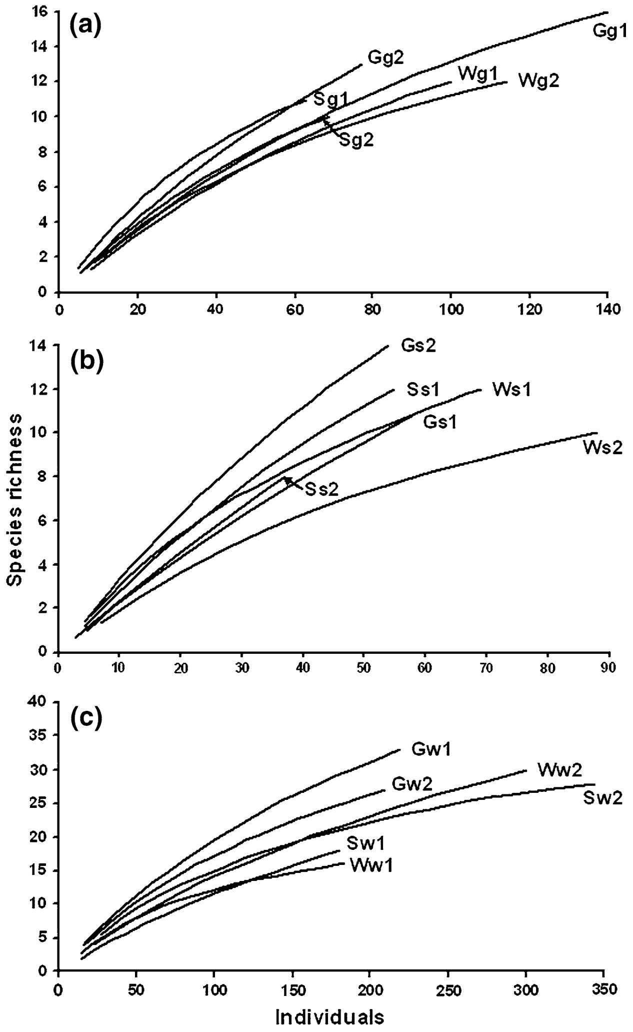

Fig. 2 Mao-Tau function curves for hoverfly ensembles in the three vegetation types studied in Cabañeros National Park, Spain. **a** grassland, **b** scrubland, **c** woodland

All hoverfly species depend on flowers when adults and most on particular vegetation types when larvae: species with saproxylic larvae on trees; species with phytophagous

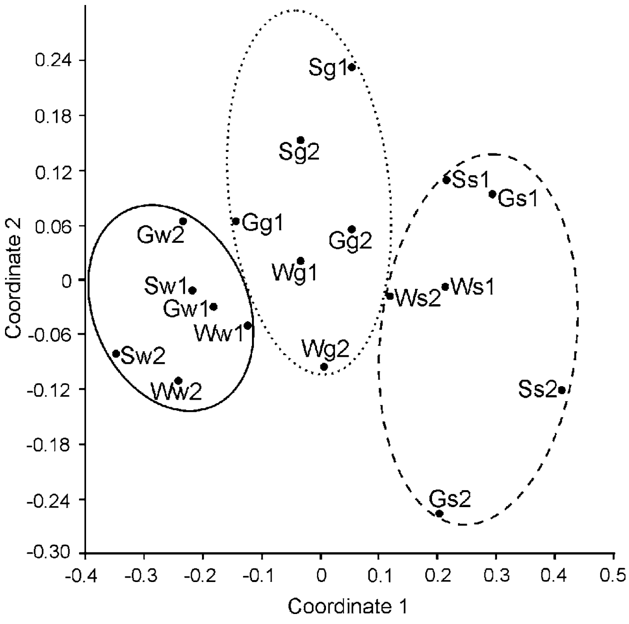

Fig. 3 Spatial distribution of sampling sites in an NMDS based on hoverflies dissimilarity. *Ovals* indicate the different groups of sampling sites: *solid line* includes woodland sites, *dotted line* includes grassland sites, and *dashed line* includes scrubland sites

larvae on specific plants; and species with predatory larvae on the plants where their prey—usually aphids—feed. Those larvae not associated with specific vegetation structures are saprophagous in aquatic media. However, the adult hoverflies' need to search for flowers and their high mobility contribute to some species being collected from sites of vegetation types which cannot provide them with the resources required for breeding; for instance, *Spilomyia digitata*, a saproxylic species, was collected as a tourist species feeding on flowers from grassland sites of the S-landscape (Appendix).

Hoverfly larvae are affected strongly by specific environmental conditions and are more closely related to specific vegetation types than the adults. Protocols to sample larvae of a whole community of hoverflies are not developed sufficiently, and the early stages of several species and their microhabitats remain unknown (Speight 2010). The absence of such protocols currently precludes the use of larvae in studies of diversity.

The W-landscape had the lowest mean beta, as initially hypothesised, but its mean alpha was between the values found for both G-landscape and S-landscape. The S-landscape had the lowest mean alpha but an intermediate mean beta. Contrary to our initial suppositions, the G-landscape had the highest mean alpha and mean beta. Among the woodland sites of the three landscapes, we collected the lowest number of species in the W-landscape, contrary to the expected tendency; but we collected the highest number of species in the woodlands of the G-landscape (Appendix).

The highest species richness of hoverflies occurred in woodland sites in Cabañeros. Woodland sites provide a broad range of resources including high diversity and an abundance of microhabitats. The Cabañeros woodlands comprise not only over-mature trees that provide microhabitats for saproxylic hoverfly larvae—of rare species in many cases and therefore of key relevance in a checklist of the fauna of a study area (e.g., *Myolepta dubia*, *Sphiximorpha binominata*; Appendix)—but also bodies of water (temporary or permanent) that explain the presence of hoverflies such as *Chalcosyrphus nemorum*. Woodland sites in Cabañeros also usually include small invasive areas of scrub and small grassy clearings. These clearings provide ideal sheltered sites for thermoregulation by some species (for instance, *Merodon* spp; Hurkmans 1985; Hurkmans 1993). They also contain a number of grassy plants, which are frequently hosts of aphids (such as used by *Paragus* spp; Appendix), and geophytes, which are the dominant herbaceous plants in the grassy clearings of the Cabañeros woodlands (Fernández-González and Pérez-Badía 2004). The geophytes *Asphodeus* spp and *Urginea maritima* found in Cabañeros support the phytophagous larvae of species of *Eumerus* and *Merodon* (Ricarte et al. 2008); almost 100% of the collected *Eumerus* and *Merodon* species were captured in woodland sites (Appendix). Our conclusions are in accordance with the results of Gittings et al. (2006), who highlighted the importance of open areas for the maintenance of hoverfly biodiversity in woodland habitats. In fact, the presence of grassy clearings in

Table 3 Lande's (1996) gamma diversities and proportional contributions (%) of alpha and beta diversities to the total diversity for each landscape in Cabañeros National Park, Spain

	Observed gamma	Mean alpha	Mean beta gamma	Lande's	Alpha (%)	Beta (%)
G-landscape	50	19	31	50	38	62
S-landscape	41	14.5	26.5	41	35.37	64.63
W-landscape	39	15.33	23.67	39	39.31	60.69

G-landscape grassland-dominated landscape, *S-landscape* scrubland-dominated landscape, *W-landscape* woodland-dominated landscape

Cabañeros woodlands may explain the finding of hoverfly species typically associated with grasslands such as *Cheilosia latifrons*, *S. scripta* or *M. mellinum*.

In addition, woodlands of the G-landscape may act as refuges for hoverflies when harsh environmental conditions (wind and rain, extreme temperatures and sun irradiation, water scarcity or predators, etc.) affect more open habitats strongly. This effect, which increased the density of species in woodland sites of the G-landscape, was observed by the authors during fieldwork in the summer, when most of the species were found in the woodlands, and especially in 2005, when weather conditions were more extreme than in 2004 (Ministerio de Medio Ambiente de España 2006). Dense colonies of insects, which included hoverflies, were observed in the shade and near bodies of water in the woodlands in summer. Thus these remnant woodlands provide short term refuges for hoverflies in situations of adverse conditions elsewhere. Consequently, these woodlands become of special interest for conservation purposes. They should be managed in an appropriate way, as wild herbivores such as red deer contribute to make tree regeneration difficult.

We suggest extending the fencing of parts of the woodland remnants of the G- and S-landscape to avoid grazing by wild herbivores and allow tree regeneration. Sampling sites Gw2 and Sw2 have each had a parcel fenced since 2006. Park managers tried to group the oldest trees and some grassy clearings to be re-colonised by new trees within each parcel. Individual parcels have an extension up to half the size of the area currently covered by trees at each sampling site. These actions favour the regeneration of oak and ash trees, as dense masses of small trees occur inside the fenced parcels now (pers. obs.). Tree regeneration will ensure the maintenance of the forests and their associated community, which includes hoverflies, in the mid/long term. Following the same protocol as described previously, we suggest fencing a part of sampling site Gw1, which should be a priority site because of the presence of rare hoverfly species such as the saproxylic *Sphiximorpha binominata* (Appendix; Ricarte and Marcos-García 2010), and *Mallota dusmeti* Andreu 1926, catalogued as Vulnerable in Spain (Marcos-García 2006; Ricarte et al. 2007).

Our results also show that beta diversity is the main contributor to gamma diversity of each landscape in Cabañeros. This is related to the presence of a vegetation mosaic that is typical of Mediterranean ecosystems (Bignal and McCracken 2000). Each vegetation type contributes to the gamma diversity with unique species (Appendix), often adapted to the conditions found in each vegetation type; for instance, 25 species were unique to the woodlands of the G-landscape, where the observed richness was 50 species (Appendix). However, a high species turnover among the sampling sites within landscapes caused a marked difference between mean alpha and the observed gamma. These results were in accordance with a previous finding for agricultural landscapes, where the level of gamma diversity of different arthropod taxa, including hoverflies, is due primarily to beta diversity between local communities (Hendrickx et al. 2007). Although Cabañeros is a protected area, it was used for agriculture relatively recently (Vaquero 1997), and shares a vegetation mosaic with agricultural landscapes, where hedgerows with native flora may alternate with cropped parcels. We confirm that the analysis of alpha diversity alone may be insufficient to draw accurate conclusions when landscape diversity is assessed, and the analysis of beta diversity constitutes a crucial tool for conservation assessment (e.g. Verdú et al. 2007), even more so in Mediterranean ecosystems with a typical vegetation mosaic.

Although beta diversity is strongly scale-dependent (Rooney et al. 2007), the high values of beta diversity obtained in this study indicate a need to preserve vegetation mosaics rather than isolated vegetation patches in the landscape to promote species diversity. One of the main threats to the Mediterranean mosaic landscape is the spread of certain vegetation types due to changes in traditional management practices, as occurs with scrubland in Cabañeros caused by abandonment of activities such as agriculture and charcoal production (Vaquero 1997). The encroachment of the scrubland in Cabañeros corresponds to the spread of the vegetation type that supports the fewest species (only 8–12 species were collected at each scrubland site). This is almost impenetrable in some areas and encroachment by this reduces the frequency of patches of grassy clearings which can promote insect biodiversity (Verdú et al. 2000; Zamora et al. 2006). The dominant scrubland in the S-landcape is becoming progressively more closed and this seems to be reflected by alpha diversity, as, for instance, Ss2 has the lowest species richness among the sampling sites when considered as a whole. Grassy clearings within scrubland sites are becoming reduced and the limited hoverfly community of the scrubland (only three species were unique to scrublands from all three landscapes) is itself being negatively affected.

We suggest enriching the diversity of the S-landscape by incorporating grassland patches in the vast mass of scrubland occupying nearly 55% of the total area of the S-landscape. Our assumption about the positive effect of grassy clearings in the scrubland is based on two facts. Firstly, Gittings et al. (2006) showed that open spaces contribute to the maintenance of hoverfly biodiversity and, secondly, our results show that some hoverfly species (7) that occur in the grasslands of the S-landscape do not occur in the scrublands (see Appendix), therefore the existence of new open spaces in the scrubland mass may attract new hoverfly species to this environment. The opening of clearings in the scrubland mass of the S-landscape will also contribute to increase the turnover of species (beta diversity) within this particularly poor environment. Therefore we suggest that the traditional activities of this landscape, such as the extraction of cork from cork oaks under controlled conditions should be reintroduced. The process of extraction of cork, which was frequent in the past in Cabañeros but prohibited in the public part of the park today, produces small clearings around the trees that are expected to be self-managed, since the periodic activity of extraction and wild herbivores will make the rapid regeneration of scrub difficult.

The vast plain of grassland (“raña”) within the G-landscape is not under a serious threat of colonisation by scrub because of the high intensity of grazing by the large wild herbivores populations (mainly red deer). The same factor that negatively affects the woodland remnants in the G- and S-landscape stops the colonisation by scrubs in the “raña”. Only slow colonisation by scrubs is taking place in narrow bands at the borders of the “raña”, near the mountains.

We conclude that special attention should be paid to the preservation of the vegetation mosaic that is typical of Mediterranean landscapes by favouring the regeneration of the woodlands of G- and S-landscape and by preventing the encroachment by scrubland due to changes in traditional management practices. Woodlands of G- and S-landscape are of particular importance within this mosaic pattern because of the number of species that they support, some of which are very rare (Appendix). To conserve the biodiversity and ecological functions of hoverflies in Mediterranean ecosystem, we should focus on these “high-quality” habitats, which may also operate as sources of dispersal for individuals of rare species in ecological restoration, as noted by other authors (Kohler et al. 2008).

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Appendix

See Table 4.

Table 4 Number of individuals of hoverfly species collected at sites with different vegetation types (*g* grasslands, *s* scrublands, *w* woodlands), imbedded in three landscapes with a particular dominant vegetation type (*G* grassland, *S* scrubland, *W* woodland) at Cabañeros National Park, Spain

Landscape	G-landscape			S-landscape			W-landscape			Total
Dominant vegetation type at the landscape	G			S			W			
Vegetation type at the sampling site	g	s	w	g	s	w	g	s	w	
Brachyopa insensilis Collin, 1939^{sx}			1							1
Brachypalpoides lentus (Meigen, 1822)^{sx}			1			1				2
Callicera aurata (Rossi, 1790)^{sx}								1		1
Ceriana vespiformis (Latreille, 1804)^{sx}		1	1							2
Chalcosyrphus nemorum (Fabricius, 1805)^{sx}						3				3
Chamaesyrphus lusitanicus Mik, 1898^{ph}		1								1
Cheilosia latifrons (Zetterstedt, 1843)^{ph}			2							2
Chrysogaster basalis (Loew, 1857)^{s}			1							1
Chrysotoxum bicinctum (L., 1758)^{pr}						3				3
Chrysotoxum intermedium Meigen, 1822^{pr}	1		1		2	5		2	2	13
Chrysotoxum latifasciatum Becker, 1921^{pr}						10				10
Chrysotoxum octomaculatum Curtis, 1837^{pr}			1	1		10			1	13

Table 4 continued

Landscape	G-landscape			S-landscape			W-landscape			Total
Dominant vegetation type at the landscape	G			S			W			
Vegetation type at the sampling site	g	s	w	g	s	w	g	s	w	
Chrysotoxum vernale Loew, 1841^{pr}			1							1
Criorhina pachymera Egger, 1858^{sx}			1			5				6
Dasysyrphus albostriatus (Fallén, 1817)^{pr}							4			4
Epistrophe eligans (Harris, 1780)^{pr}							1		1	2
Epistrophe nitidicollis (Meigen, 1822)^{pr}			1							1
Episyrphus balteatus (de Geer, 1776)^{pr}		12	14		1	41		7	8	83
Eristalinus aeneus (Scopoli, 1763)^{s}								1		1
Eristalinus megacephalus (Rossi, 1794)^{s}	1									1
Eristalinus taeniops (Wiedemann, 1818)^{s}	3		1							4
Eristalis arbustorum (L., 1758)^{s}			1			1			1	3
Eristalis similis (Fallén, 1817)^{s}	9	21	7		32	1	20	54	18	162
Eristalis tenax (L., 1758)^{s}	7	18	15	1	7	14	38	13	34	147
Eumerus barbarus (Coquebert, 1804)^{ph}							1			1
Eumerus caballeroi Gil-Collado, 1929^{ph}									1	1
Eumerus olivaceus Loew, 1848^{ph}			3	3		5		1	5	17
Eumerus pulchellus Loew, 1848^{ph}	9	6	6	11	1	4	3	1	11	52
Eumerus pusillus Loew, 1848^{ph}	14	7	16	19		1				57
Eumerus sabulonum (Fallén, 1817)^{ph}			1							1
Eumerus strigatus (Fallén, 1817)^{ph}				1		2				3
Eumerus subornatus Claussen, 1989^{ph}	4	18	11	2	6	5	2	4	17	69
Eupeodes corollae (Fabricius, 1794)^{pr}	15	3	17	6	2	5	12	9	14	83
Eupeodes lucasi (Marcos-García and Laska, 1983)^{pr}							1			1
Eupeodes luniger (Meigen, 1822)^{pr}									1	1
Eupeodes nuba (Wiedemann, 1830)^{pr}						3				3
Ferdinandea aurea Rondani, 1844^{sx}	1		4			1	3			9
Ferdinandea cuprea (Scopoli, 1763)^{sx}			1							1
Ferdinandea fumipennis Kassebeer, 1999^{sx}			4							4
Helophilus trivittatus (Fabricius, 1805)^{s}			1				1			2
Mallota cimbiciformis (Fallén, 1817)^{sx}						1		1		2
Melanostoma mellinum (L., 1758)^{pr}	6	1	42		9	38			25	121
Melanostoma scalare (Fabricius, 1794)^{pr}		1	13	1	4	3			3	25
Meliscaeva auricollis (Meigen, 1822)^{pr}			2		1	3	2		1	9
Merodon avidus (Rossi, 1790)^{ph}			3			5			1	9

Table 4 continued

Landscape	G-landscape			S-landscape			W-landscape			Total
Dominant vegetation type at the landscape	G			S			W			
Vegetation type at the sampling site	g	s	w	g	s	w	g	s	w	
Merodon funestus (Fabricius, 1794)^{ph}									1	1
Merodon geniculatus Strobl, 1909^{ph}	1								1	2
Merodon luteihumerus Marcos-García, Vujić and Mengual, 2007^{ph}	6	2	6	4						18
Merodon obscuritarsis Strobl in Czerny and Strobl, 1909^{ph}			1					1	3	5
Merodon confusus Marcos-García et al. *in prep*^{ph}			3			4				7
Milesia semiluctifera (Villers, 1789)^{sx}		1	1						1	3
Myathropa florea (L., 1758)^{s}			5			7	1		3	16
Myolepta dubia (Fabricius, 1805)^{sx}									1	1
Orthonevra frontalis (Loew, 1843)^{s}			3							3
Paragus bicolor (Fabricius, 1794)^{pr}						3				3
Paragus quadrifasciatus Meigen, 1822^{pr}	1									1
Paragus tibialis (Fallén, 1817)^{pr}		1		1	5		1	3	2	13
Paragus vandergooti Marcos-García, 1986^{pr}				2	1					3
Scaeva dignota (Rondani, 1857)^{pr}	4	1			1	1	3		3	13
Scaeva mecogramma (Bigot, 1860)^{pr}						1				1
Scaeva pyrastri (L., 1758)^{pr}	2						1			3
Sphaerophoria scripta (L., 1758)^{pr}	130	17	197	78	20	296	118	58	286	1200
Sphiximorpha binominata (Verrall, 1901)^{sx}			1							1
Spilomyia digitata (Rondani, 1865)^{sx}			1	1						2
Syritta pipiens (L., 1758)^{s}	2	1	3			3		1	2	12
Syrphus vitripennis Meigen, 1822^{pr}						3	2		3	8
Volucella elegans Loew, 1862^{pr}	1	1	13	1		16			28	60
Volucella inanis (L., 1758)^{pr}			4			1			6	11
Xanthandrus comtus (Harris, 1780)^{pr}						1				1
Xanthogramma marginale (Loew, 1854)^{pr}			2							2
Xanthogramma pedissequum (Harris, 1780)^{pr}			15			17				32
Xylota segnis (L., 1758)^{sx}						1				1
Total abundance	217	113	428	132	92	524	214	157	484	2361
Total species	19	18	43	15	14	37	18	15	30	72

Following the species name indication of the larval feeding habit: *pr* predator, *ph* phytophagous, *sx* saproxylic, *s* saprophagous other than saproxylic

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