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Short-Term Effects of a Revegetation Program on the Orthopteran Diversity in Oak Forests of the Southern Iberian Peninsula

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ABSTRACT. Orthopterans are insects closely linked to vegetation as primary consumers as well as for other biological processes such as oviposition and development. This research aims to assess the effect of a revegetation program that began in 2007 in the compensation area linked to the construction of the Breña II dam on Orthopteran diversity within several different human-created and natural habitats (forest-islands, hedges, and river-copses). We assessed vegetation and orthopteran communities during monthly sampling performed during March through September 2011. For the Orthopterans, two replicates per habitat type were sampled in each of the eight selected sampling plots, providing 48 observations per environment per month. To characterize the structure of communities, diversity, dominance, and evenness were calculated, and posterior comparisons were made using bootstrapping analysis. Additionally, rarefaction curves were obtained. We found large between-habitat differences in plant abundance but smaller differences in diversity. The high degree of vegetational homogeneity likely explains the structural similarity among the Orthopteran communities in the different habitats. Although Caelifera were more abundant and diverse in unmanaged biotopes, Ensifera seem to be favored in revegetated areas. Because accurate management requires documenting diversity at the field scale, work like that presented here should increase the efficiency of future assessments of Orthopteran habitat suitability for diversity conservation.

Key Words: biodiversity, environmental restoration, Orthoptera, Quercus, Mediterranean region

The Habitat Directive (92/43/EEC, 1992) on the conservation of unmanaged habitats and of wild fauna and flora of the European Union establishes that the member states must take all necessary compensatory measures to ensure the overall coherence of Nature 2000 (the European ecological network of conservation areas of biodiversity). Consequently, the Breña II Project, a new dam completed in 2008 in the Guadiato River basin in the southern Iberian Peninsula, involved the implementation of a package of compensatory measures to offset the environmental disturbance caused by the flooding of a nature reserve and the dam infrastructure. As part of these actions, a revegetation program was initiated during 2007, which included different models of environmental restoration (Sandoval and Quiñónez 2007). The term "revegetation" refers to the introduction of native plants into an impoverished environment, regardless of the species selection criteria and the method by which these plants are introduced (Munro et al. 2007). Depending on the intended structural complexity, two basic types of revegetation may be distinguished: "single planting," in which usually only tree species are planted, and "environmental restoration plantations," which are structurally and floristically more diverse and attempt to recreate the predegradation plant communities (McElhinny et al. 2005). The habitat types (=environmental models) created in the Breña II improvement area fall into the "environmental restoration plantation" type of revegetation strategy and include forest-islands, hedges, and river-copses (full information on the overall improvement measures can be found in Sandoval and Quiñónez [2007]).

Measuring the success of management and restoration requires both a site- and taxon-appropriate survey plan and the ability to assess environmental changes in space and time (Hobbs 2003). Identifying important variables for monitoring is a main goal in restoration research. Faunal response to revegetation has been studied from different points of view and in a wide range of animal taxa. Reviews analyzing the effect of revegetation on wildlife in managed environments suggest that for certain communities (such as birds) higher levels of structural complexity are achieved in revegetated areas compared with those have not been improved (Hobbs et al. 2003, Munro et al. 2007). Conversely, vertebrates such as rodents, amphibians, and reptiles appear to benefit less from revegetation in at least the short term (Borsboom et al. 2002, Merritt and Wallis 2004, Kavanagh et al. 2005).

Information regarding the effect of environmental recovery on diverse taxa of invertebrates is scarcer; conclusions are often controversial, require more extended periods of environmental stability to be consistent, or are framed within the overall framework of plant succession. Schnell et al. (2003) report a greater diversity of ants in replanting zones than in nonrevegetated zones, whereas Green and Catterall (1998) found Hymenopteran species composition did not differ as a function of time since disturbance or vegetation management regime. For Orthoptera, it seems that use of the territory directly influences the habitat quality and, indirectly, affects the diversity of these insects (Steck et al. 2007). For instance, intensive grazing in unmanaged ecosystems may reduce food availability, alter microclimates, and disrupt potential oviposition sites for Acrididae (O'Neill et al. 2003). It has also been demonstrated that soil tillage, so often applied in seminatural forest such as pastures, negatively affects both the density and diversity of Orthopterans (Braschler et al. 2009). The close relationship between grasshopper abundance and diversity in grassland habitats and the number and variety of plants (Kemp et al. 1990a) suggests that interventions aimed at the restoring or maintaining vegetation should increase Orthopteran diversity. However, Catterall et al. (2004) found greater grasshopper diversity and abundance on cleared land than in areas reforested with native vegetation or in areas with greater development of the existing vegetation prior to revegetation. Borchard et al. (2013) recently reported that, in central-European heathlands, Orthopteran species from early and midsuccessional stages respond rapidly to vegetation restoration measures. This may be because Orthopterans are closely linked to vegetation as primary consumers (Quinn and Walgenbach 1990, O'Neill et al. 2003) and other biological processes

such as egg laying or nymphs' development (Guido and Gianelle 2001).

Orthopterans are considered excellent bioindicators for use in assessments of ecological change associated with land uses (Armstrong and Van Hensbergen 1997, Samways 1997, Andersen et al. 2001).

The diversity, distribution, and abundance of Orthopteran are variables with ecological significance. Ingrisch and Köhler (1998) and Kebler et al. (2012) found that a first sign of habitat damage could be a decrease in abundance, although this did not necessarily mean the complete and immediate extinction of a species. The high diversity of Orthopterans and their functional importance and responsiveness to environmental disturbances make them a very useful model for assessing ecological succession processes (Andersen 1997). In addition, they have several advantages with respect to other groups (Andersen et al. 2001, Kati et al. 2004, Poniatowski and Fartmann 2008). First, they are abundant and conspicuous insects that are reliable for identification, sampling, and standardizing data (Báldi and Kisbenedek 1997); second, they often constitute the largest fraction of arthropod biomass in grassland ecosystems (Shure and Phillips 1991).

To assess the effectiveness of the revegetation program in the compensatory area of the Breña II dam, we conducted a study assessing the Orthopteran fauna in the different models of restoration plots (forest-islands, hedges, or river-copses).

The main goal of this study was to determine the effect of the environmental improvement on Orthopteran communities. We addressed the following questions:

1. Are the species richness and abundance of grasshoppers, bush crickets, and crickets significantly different in the restored versus nonrestored plots?
2. If so, are these differences according to the type of environmental model applied?
3. What is the result of an initial assessment of the environmental restoration program based on the changes noticed in the Orthoptera fauna?

Materials and Methods

Study Area. This research was carried out in the area surrounding the Breña II dam next to the Natural Park of Sierra de Hornachuelos (Córdoba Southern Iberian Peninsula). The overall study area is included in the area of environmental improvement linked to the Breña II Recovery Program, and it comprises a total of eight restoration plots (Table 1; Fig. 1).

The climate in the area is typically Mediterranean, with annual rainfall ranging between 500 and 800 mm and mean annual temperatures of ~17°C. The summers are relatively warm (mean ≈24°C), and the winters are temperate, with mean temperatures ranging between 6 and 10°C (Gallardo et al. 2010).

The human population density in this area is low, and forestry is the main natural resource. The landscape's relief shows a moderate altitude ranging from 250 to 725 m (Blanco 2006, Pinilla 2006). Lithologically, Palaeozoic metamorphic rocks predominate, particularly quartzite, slates, or semiacidic intrusive rocks. Sandy or clayey substrates can also be found. The soils are chemically and physically homogeneous and contain high levels of organic material and carbon (Pinilla 2006).

The landscape is dominated by Mediterranean mixed sclerophyllous forests that sit on the thermo and meso-Mediterranean belts (Cárdenas and Bach 1989). The vegetation in the area belongs to the *Duriilignosa* formation, represented in the Iberian Peninsula by the *Quercetea ilicis* type. This vegetation is constituted by evergreen trees and phanerophyte communities dominated by shrubs and bushes. These sclerophyllous forests are characterized by the predominance of holm oaks (*Quercus ilex* L.) and cork oaks (*Quercus suber* L.). There are middle-aged trees ranging between 65- and 100-year old with a mean density of ~45 trees per hectare (Cárdenas and Gallardo 2012).

Sampling Area. Field sampling was carried out in the different environmental models established in the Restoration Program. These models are defined as follows (De Andrés et al. 2003): "forest-islands," or patches of woody vegetation recreating the original forest; "hedges," or aligned group of trees, shrubs, and herbaceous species interconnecting the relict forest patches; and "copses," or mixed formations of deciduous trees and shrubs that develop on the riverbanks. To make comparisons, two types of nonrestored environments were also considered: "unmanaged forests," which were used for comparisons with forest-islands and hedges, and "unmanaged copses," which were used for comparison with restored copses. Each restored environmental model has a specific composition of plants, which were selected according to the intended ecological characteristics (Aguas de la Cuenca del Guadalquivir S.L. (AQUAVIR) 2000). Planting was done during spring 2007. Data on the annual growth rate of shrub species planted at the restoration plots are available in Villar et al. (2004). This information indicates that sufficient time has elapsed to generate structural changes in vegetation.

Autochthonous plant species remaining at the nonrestored areas ("unmanaged forests" and "unmanaged copses") are recorded in Blanco (2006) and Torres and Ruiz (2009).

Sampling Methods. To characterize the Orthopteran communities, preliminary sampling was performed in 2010. According to these first results (Cárdenas et al. 2010), plots P1, P2, P3, and P4 were selected for sampling forest-islands and hedges, and plots P5, P6, P7, and P8 were selected for sampling restored versus unmanaged copses (Fig. 1). The criteria for selecting plots were the highest values of richness and abundance of species observed in the aforementioned preliminary sampling.

Table 1. Nomination, code, locality, UTM coordinates, and extension for the sampling plots

| Nomination	Code	Locality	UTM coordinates	Extension (ha)
Los Baldíos	P1	Córdoba	30S0335094 4200826	223
Las Tonadas	P2	Villaviciosa de Córdoba	30S0323721 4210546	169
La Morilla	P3	Villaviciosa de Córdoba	30S 0333645 4213198	149
Umbría de las Perchas	P4	Córdoba	30S0326572 4196606	273
Las Mesas	P5	Córdoba-Almodóvar del Río	30S0323505 4198113	171
Cerro del Trigo	P6	Almodóvar-Villaviciosa de Córdoba	30S0322844 4198706	59
Los Lagares	P7	Almodóvar del Río	30S0317794 4198726	112
Mezquitillas	P8	Villaviciosa de Córdoba	30S0318183 4202660	125
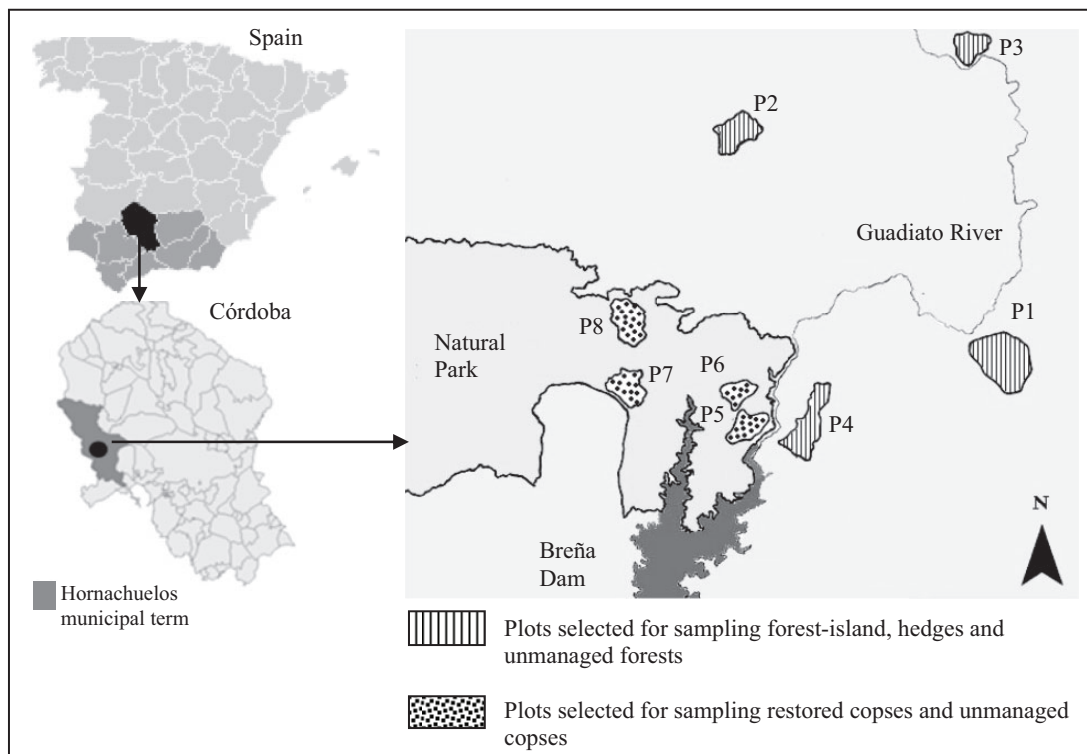
Fig. 1. Location of the sampling plots in the research area.

Following the criteria of Sänger (1977) and Poniatowski and Fartmann (2008), the environmental models and the unmanaged areas had, insofar as possible, equivalent surfaces and homogenous vegetal structure. There was a between-plot separation distance >10 m to avoid edge effects (Picaud and Petit 2007).

To make comparisons we have two nonrestored environments (more natural areas, where no revegetation has been made): "unmanaged copses" and "unmanaged forest." These are used for comparisons with forest islands and hedges, habitats which represent the environment prior to revegetation. This implies five habitat types: three restored and two unmanaged or nonrestored. So, we have three sets of paired environments (forest islands-unmanaged forest; restored hedges-unmanaged forest; and restored copses-unrestored copses) in which vegetation and Orthopterans can be compared.

Two replicates of each environmental type (forest island, copses, hedges, unmanaged copses, and unmanaged forest) were sampled in each of the above-selected research plots. We sampled a total of eight replicates per of each environmental type.

Vegetation sampling was performed in early spring (March 2011) following the linear transect procedure (González-Uribe and Sánchez-Pérez 2004) that involved recording the species identity and abundance of all plants intercepted along a linear path.

In forest-island habitats, we sampled a mean of six transects per replicate. Forest-island plot size ($\bar{x} \pm$ SD) was $41.5 \pm 14.3 \times 40 \pm 12.7$ m. The total forest-island sampling effort (41.5 m by 6 transects by 8 replicates) involved 1,992 m of linear path.

In restored copses, we sampled a mean of seven transects per replicate. Restored copses plot size ($\bar{x} \pm$ SD) was $53 \pm 11.3 \times 22 \pm 5.5$ m. The total copses sampling effort (22 by 7 by 8) involved 1,232 m of linear path.

In restored hedges, we sampled a mean of 10 transects per replicate. Restored hedges plot size ($\bar{x} \pm$ SD) was $72 \pm 18.3 \times 9 \pm 1.3$ m. The total restored hedges sampling effort (9 by 10 by 8) involved 720 m of linear path.

In unmanaged forest plots, we sampled a mean of five transects per replicate. Unmanaged forest plot size ($\bar{x} \pm$ SD) was $33.5 \pm 5.6 \times 34.5 \pm 8.6$ m. The total unmanaged forest sampling effort (34.5 by 5 by 8) involved 1,380 m of linear path.

In unmanaged copses, we sampled a mean of six transects per replicate. The mean unmanaged copses' plot size ($\bar{x} \pm$ SD) was $44 \pm 0.7 \times 29 \pm 1.8$ m. The total unmanaged copses sampling effort (29 by 6 by 8) involved 1,392 m of linear path.

For each environmental type, the individuals of each species intercepted along all the paths were summed, and density of each species was estimated as the mean number of individuals recorded per meter of sampled transect. Only the tree and shrub layers were recorded, forbs and grasses were not considered because this layer was not involved on the revegetation program.

Orthopteran sampling was carried out monthly between April and September in 2011, coinciding with the most suitable period for the activity of these insects (Bellmann and Luquet 1995). At each of the aforementioned plots and inside each environmental type (forest-island, hedge, copse, unmanaged forest, and unmanaged copse), six surveys were performed during the sampling period, providing a total of 48 observations per environmental type (6 sampling by 4 plots by 2 replicates).

Linear transects (Gardiner and Hill 2006) with zig-zag paths over a time of 30 min per environmental type each sampling day were carried out. Direct manual capture and sweep nets were used to catch the insects. The specimens were identified in the field, censused, and released. Sex and maturation stage (nymph or imago) were also noted before being released. Species that could not be identified in the field were collected, preserved in 70% ethanol, and transported to the laboratory for classification.

Data Analysis. To test differences in plant densities related to different environmental types, one-way analysis of variance was applied, provided that the data met the assumptions of normality. Otherwise, the Kruskal–Wallis nonparametric test was used. The Shapiro–Wilk and

Levene tests were applied to assess the normality and homogeneity of variances (Levene 1960).

The independent sample *t*-test was used to check differences in average species diversity and abundance of Orthoptera both linked to different environmental types. If a Shapiro–Wilk test found that the normality assumptions were not satisfied, we used the equivalent nonparametric Mann Whitney *U*/Wilcoxon ranked sum test (Zar 1984).

In accordance with Franco (1985) and Magurran (2004), the measures of species diversity were grouped into three categories: richness or number of species, indices based on the proportional abundance of species, and species-abundance models.

Because the study was performed at a local scale, the indices based on proportional abundance of species were deemed appropriate and useful for comparing the different sites (Báldi and Kisbenedek 1997). Thus, to characterize the structure of vegetation and Orthopteran communities, the commonly used indices of diversity, dominance, and evenness were calculated (Ludwing and Reynolds 1988, Southwood 1991). Differences in the indices were tested by resampling (bootstrapping for inferential statistics; Rochowicz 2011).

Rarefaction curves were also obtained to estimate the number of species expected for similar sampling size in each environment. The rarefaction function integrates data on each species' commonness or rarity in a given area (Koellner et al. 2004).

Statistical tests were conducted using SPSS statistical software (SPSS Inc. 20.0 2011) with α value of 0.05. Ecological indices were calculated and compared using the Past Paleontological software package (Hammer et al. 2001).

Vegetation nomenclature follows Valdés et al. (1987) and Greuter et al. (2000). Eades et al. (2013) was followed for the classification of Orthoptera.

Results

Vegetation Analysis. From the sampling of the eight restoration plots, a total of 32 plant species belonging to the tree and shrub layers were recorded. For each environmental type, the mean density of each of the recorded species was calculated. After testing the criteria of normality and homogeneity, the overall data obtained (Appendix 1) were subjected to a Kruskal–Wallis statistical test, which revealed significant differences in plant cover between restored environments (forest-islands, hedges, and copses) and their respective controls (unmanaged forests and unmanaged copses) ($P = 0.017$, $H_{0.05,8.8} = 5.805$). Comparing the mean plant density (trees and shrubs) from each environment graphically using box plots (Fig. 2), the highest density was recorded in unmanaged forests, whereas restored copses had more impoverished plant cover.

Structural differences in trees and shrub layers were assessed by diversity, dominance, and evenness indices (Table 2). The paired comparison of these indices for improved habitats versus unmanaged environments (i.e., forest-islands vs. unmanaged forests) indicates that in unmanaged forests and hedges differed in their Shannon diversity (boot $P = 0.015$) but only marginally in their Simpson diversity (boot $P = 0.087$) and not in their species evenness. Plant communities in the other habitats did not differ significantly in evenness or either of the diversity indices.

Orthoptera Analyses. The Orthopteran species recorded are displayed in Appendix 2. Across all the different environmental types, a total of 13,066 specimens (1,666 Ensifera and 11,400 Caelifera) from 25 species (12 Ensifera and 13 Caelifera) were counted.

A detailed analysis of the species recorded at each sampling plot throughout the overall sampling period shows that the same species are common and abundant in all the environmental types. *Phaneroptera nana* Fieber, 1953 and *Tessellana tessellata* (Charpentier, 1825) were the Ensifera predominant in the forest-islands, hedges, copses, and unmanaged zones. Similarly, *Pezottetix giornae* (Rossi, 1794) and *Dociostaurus jagoi* Soltani, 1978 were the Caelifera prevailing in any type of prospected habitat.
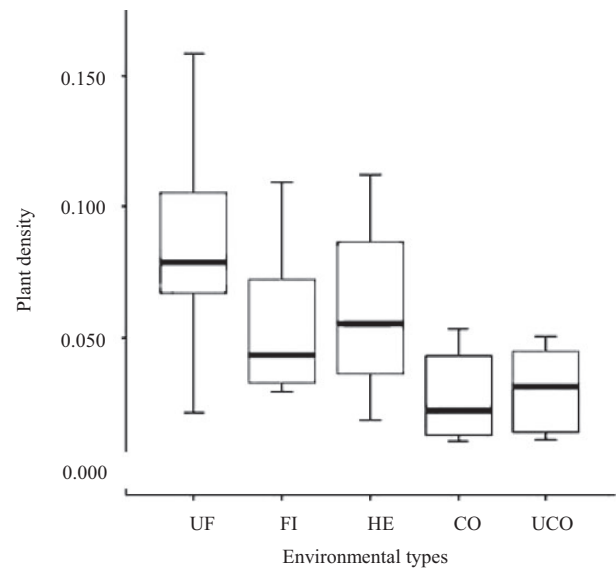
Fig. 2. Box plots representative of plant density (individuals per meter) for tree and shrub layers in the different environmental types; the boundaries of the box indicate the 25th and 75th percentiles; whiskers indicate the 90th and 10th percentiles. The horizontal line in the box corresponds to the median value. UF, unmanaged forest; FI, forest-island; HE, hedge; CO, copse; UCO, unmanaged copse.

Table 2. Richness, abundance, diversity (Shannon–Wiener *H'*), dominance (Simpson), and evenness indices for plant communities

Indices	Habitat types				
	UF	FI	HE	CO	UCO
Richness	20	21	20	22	22
Abundance	953	828	326	279	269
Shannon	2.21	2.27	2.37	2.47	2.49
Simpson	0.85	0.86	0.87	0.87	0.87
Evenness	0.45	0.46	0.54	0.54	0.55

UF, unmanaged forest; FI, forest-island; HE, hedge; CO, copse; UCO, unmanaged copse.

However, one can recognize certain species restricted to a specific type of environment. *Gryllus bimaculatus* De Geer, 1773 and *Pteronemobius lineolatus* (Brullé, 1835) were exclusively recorded in the hedges, whereas *Uvarovitettix nodulosus* (Fieber, 1853) and *Xya variegata* (Latreille, 1809) exclusively colonized the unmanaged forests.

There were no significant differences in the relative abundance of species in recovered versus unmanaged areas ($t_{14} = -0.942$, $P = 0.362$ for comparison between unmanaged forests and forest-islands; $t_{14} = 0.587$, $P = 0.567$ for unmanaged forests vs. hedges; and $t_{14} = 1.064$, $P = 0.305$ for unmanaged copses vs. revegetated copses).

The same results were obtained when the abundance of Caelifera was independently analyzed ($t_{14} = -0.598$, $P = 0.559$ for comparison between unmanaged forests and forest-islands; $t_{14} = 1.137$, $P = 0.275$ for unmanaged forests vs. hedges; and $t_{14} = 0.898$, $P = 0.384$ for unmanaged copses vs. revegetated copses).

When Ensifera were analyzed alone, however, a significant difference was found in the number of specimens colonizing unmanaged zones versus restored hedges (*U* Mann–Whitney test, $Z = -2.155$, $P = 0.031$), with higher relative abundance in the latter environmental. No differences were observed, however, in abundance recorded in

unmanaged forests versus forest-islands ($Z = -1.785$, $P = 0.074$) or in unmanaged copses versus revegetated copses ($t_{14} = 1.759$, $P = 0.116$).

Statistical analyses of species richness found differences in the number of Ensifera species colonizing forest-islands ($t_{14} = 3.669$, $P = 0.003$) and restored hedges ($t_{14} = 3.789$, $P = 0.002$) relative to the number of species inventoried in their respective control plots (unmanaged forest).

In addition to the quantitative analysis in terms of abundance and number of species, a structural study of the different Orthopteran communities was made, for which diversity, dominance, and evenness indices were calculated. These parameters were obtained for the whole community and for Ensifera and Caelifera independently. The results are given in Table 3.

The most diverse assemblages of Ensifera occurred in restored copses, whereas most diverse assemblages of Caelifera were found in unmanaged forests. The whole community shows the same tendency as the grasshopper populations, with highest diversity in unmanaged forests. In contrast, the dominance index indicated that the forest-islands had the largest populations of abundant species. The copses and unmanaged copses harbored the most balanced communities in terms of evenness.

To verify these initial observations, a comparative analysis was performed (Table 4). This analysis revealed statistically significant differences between most of the restored habitats and their respective unmanaged replicates in terms of abundance, dominance, and diversity. This was true for both the Orthoptera community as a whole and for Ensifera alone. Only the Caelifera showed clear differences in terms of community structure for all the environments compared.

In addition, the rarefaction curves displayed in Fig. 2 indicate that the unmanaged forests may harbor higher populations of Caelifera than the forest-islands and hedges. Conversely, species of this group of insects were less common in both restored and unmanaged copses (Fig. 3a).

The rarefaction curves for Ensifera reveal that hedges, forest-islands, and copses showed higher trend to have specific diversification than the unimproved environments (unmanaged forests and unmanaged copses; Fig. 3b). The trends are unclear for the whole community (Fig. 3c).

Discussion

In recent decades, research has been performed to determine whether variation in Orthopteran abundance and/or diversity could be explained by differences in plant species richness and diversity. Branson (2011) found that in ecosystems bearing a relatively low number of plant species, grasshopper diversity and abundance were not significantly correlated with plant species richness. Conversely, in more structured vegetal formations, the Orthopteran communities (particularly the grasshoppers) are facultative associations of species on which vegetation works as an environmental filter, controlling the spatiotemporal dynamic of these communities (Kemp et al. 1990a,b; Kemp 1992a,b; Cigliano et al. 1995; Lockwood and Schell 1995; Schell and Lockwood 1997). In addition, Kemp et al. (2002) noted the importance of the vegetation type and the specific local physiognomies in structuring grasshopper populations at a local scale.

Bearing in mind the preceding information, the first issues analyzed on this research dealt with compositional and structural aspects of vegetation. As grasses and forbs have not been replanted and most native species are annuals that rapidly regenerate, the analysis of the vegetation has been limited to woody plants (trees and shrubs).

We found significant differences in plant cover (of trees and shrubs) between improved and unmanaged environments, with the greatest densities in unmanaged forests and the lowest in the restored copses. From a structural point of view, the analysis only found differences in the Shannon's diversity of hedges relative to their less-diverse control plots. This implies that vegetation changes linked to the revegetation program have primarily affected plant abundance rather than richness, which suggests that the revegetation program has not been as successful as expected.

From a conservation perspective, both community structural parameters and the presence of singular species are important (Báldi and Kisbenedek 1997). The Orthopteran communities of unmanaged areas generally possess lower species diversity than more managed sites, but there are often more steno-topic species restricted to undisturbed habitats (Fartmann et al. 2008).

After 4 years of environmental improvement, the most abundant species in the restored areas are ubiquitous and generalist elements, next to the r-strategists (sensu Price et al. 2011). These species have

Table 3. Richness, abundance, diversity (Shannon–Wiener H′), dominance (Simpson), and evenness indices for Orthoptera, Ensifera, and Caelifera

Indices	Orthoptera					Ensifera					Caelifera				
	UF	FI	HE	CO	UCO	UF	FI	HE	CO	UCO	UF	FI	HE	CO	UCO
Richness	21	19	21	17	11	8	10	12	9	5	13	9	9	8	6
Abundance	3,307	4,105	2,879	1,563	1,212	377	543	566	128	52	2,930	3,562	2,313	1,435	1,160
Shannon H′	1.8	1.38	1.57	1.45	1.4	1.29	1.09	1.31	1.43	1.02	1.47	0.98	1.02	1.15	1.24
Simpson	0.76	0.6	0.65	0.67	0.67	0.63	0.48	0.56	0.64	0.52	0.7	0.48	0.48	0.61	0.64
Evenness	0.29	0.21	0.23	0.25	0.37	0.45	0.3	0.31	0.46	0.55	0.33	0.3	0.31	0.4	0.57

Table 4. Significance (Boot P) of richness, abundance, and diversity comparison between restored and nonrestored environments for Orthoptera, Ensifera, and Caelifera

Significance (Boot P)	Orthoptera			Ensifera			Caelifera		
	UF-FI	UF-HE	CO-UCO	UF-FI	UF-HE	CO-UCO	UF-FI	UF-HE	CO-UCO
Richness	0.252	1	0	0.092	0.017	0.031	0	0	0
Abundance	0	0	0	0	0	0	0	0	0
Shannon	0	0	0.258	0.01	0.822	0.035	0	0	0.018
Simpson	0	0	0.813	0	0.025	0.084	0	0	0.006
Evenness	0	0.004	0	0	0.017	0.427	0.072	0.287	0

UF, unmanaged forest; FI, forest-island; HE, hedge; CO, copse; UCO, unmanaged copse.
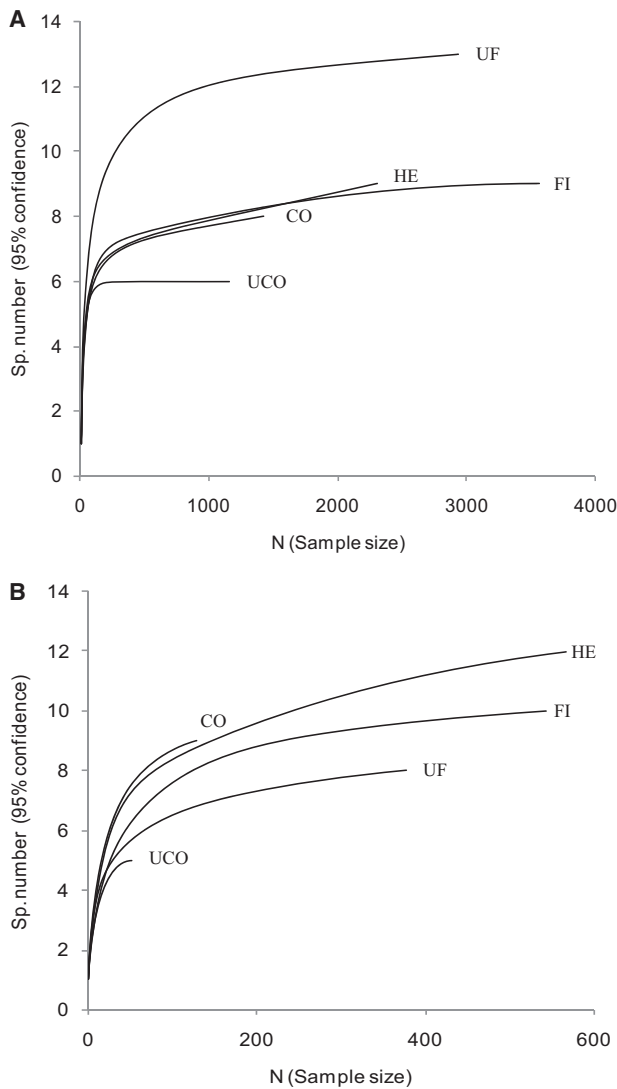
Fig. 3. Rarefaction curves obtained for each environmental type (UF, unmanaged forest; FI, forest-island; HE, hedge; CO, copse; UCO, unmanaged copse). (a) Caelifera; (b) Ensifera; and (c) Orthoptera. *N*, number of specimens.

low diagnostic value in the assessment of the recovery progress. Species exclusive to each type of environment would be more indicative because their presence could be due to the new environmental conditions. Studies addressing habitat selection in Orthopterans (Ingrisch and Köhler 1998) concluded that it involves a complex relationship of factors, among which vegetation structure is highlighted because vegetation affects key factors for survival, such as food or the suitability of oviposition sites (Poniatowski and Fartmann 2008). Thus, *G. bimaculatus* and *Pt. lineolatus* have been exclusively recorded in hedges. Although both of these crickets are hygrophilous species, the former is linked to stony soils with low plant coverage, whereas *Pt. lineolatus* mostly colonizes wetlands with abundant vegetation (Llucià-Pomares 2002). More indicative is the exclusive presence of *U. nodulosus*, the only Tetrigidae colonizing nonrevegetated areas. This species is hygrophilous (Badih and Pascual 1998) and is confined to wet meadows possessing a diverse mixture of reeds and other riparian vegetation. This factor explains its exclusive presence in unmanaged areas of "The Morilla" sampling plot, where it seems to be closely linked to *Scirpus holoschoenus* L., a plant species that is also exclusive to this environment. Both constitute a binomial that characterizes the biological

system of this area. Another species exclusively located in the nonmanaged areas is Tridactylidae *X. variegata*, a species commonly found in riparian vegetation growing between sand bars and the water's edge (Bellmann and Luquet 1995, Llucià-Pomares 2002).

Broadening this analysis of the "exclusive presence of Orthoptera species/plant composition" to the other environments, it is worth noting the singularity of the restored areas, not only from the viewpoint of fauna but even in the plant components. Given the close relationship between orthopterans and vegetation, the lack of orthopteran diversity likely stems from the low environmental diversification produced by the revegetation program. Several factors could be at play here. On one hand, the time period since revegetation may be insufficient to produce larger differences. In this regard, Bonnet et al. (1997) indicated that Orthopterans closely follow their local plant communities, with progressive adjustment over time. On the other hand, deficiencies in the implementation of the revegetation program, such as inadequate time for replanting, lack of irrigation, and failure of fences allowing free access to livestock within revegetated enclosures, have been observed. All these circumstances could have slowed or interrupted the progress of succession.

In this respect, it is necessary to consider that a succession is a structural change in the species composition of an ecological community over time. The disturbance inherent in creating the revegetation areas may, for instance, have itself been harmful to the organisms living in the area (Picaud and Petit 2007). Nevertheless, the importance of keeping a habitat mosaic for Orthoptera conservation has recently been shown by Schirmel et al. (2010), who concluded that extensive homogeneous and undisturbed stands of dwarf-shrub heath are not optimum habitats for many Orthoptera and that species conservation requires heterogeneous habitats. In a broad sense, complex, structured, and diversified landscapes yield the greatest diversity of Orthopteran communities (Tscharntke et al. 2002, Tews et al. 2004). Conversely, another recolonizing process has been described in which the initial settlement of highly competitive species prevents colonization by later arriving species and slows succession (Majer 1989). Results from undisturbed habitats show that plant and grasshopper species composition changes over environmental gradients, suggesting that habitat type influences both species presence and relative abundance (Kemp et al. 1990a). Thus, initial quantitative imbalances would yield, over time, variation in the abundance or density of key species, which could have a clearer diagnostic value.

For this reason, we analyzed the population indices. These parameters are related to the demographic component of the species and the structural dimension of the communities in which they are integrated. Based on the first parameter (abundance of species), the significant differences uniquely refer back to variations in the distribution of the populations of the most ubiquitous species such as *T. tessellata* and *P. giornae*, species whose population sizes are noticeably larger in the forest-islands and hedges than in their respective control habitats. Both are pioneer species, something that may explain their massive presence on these restored environments.

As for the structural component, it can be stated that the parameters that characterize the community associated with each biotope are fairly balanced with each other. This is especially true in terms of species richness, although the population sizes of some species are remarkably high in unmanaged forests. The values of diversity and evenness confirm that at the current stage of succession, the Orthopteran fauna from the unmanaged areas is still the best structured.

When examining how the structure of the different Orthoptera assemblages differs among the environmental types analyzed, the rarefaction curves indicate that the unmanaged forests constitute a more suitable habitat for Caelifera than the forest-islands and hedges. Conversely, copses (restored or not) seem to be less favorable for the establishment of grasshoppers. The rarefaction graphs obtained for the Ensifera are very different, with the forest-islands, hedges, and copses

more propitious than the unmanaged environments (unmanaged forests and unmanaged copses).

Our results agree with those of Bieringer and Zulka (2003) and Marini et al. (2009) in the sense that the presence of shrubs is particularly detrimental to Caelifera, whereas Ensifera seems to be less affected.

Acknowledgments

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Appendix 1

Appendix 1. Mean number of individuals of each vegetal species per meter of transect (tree and shrub layers)

A	Forest-islands (FI)							
	FI1(P1)	FI2(P1)	FI3(P2)	FI4(P2)	FI5(P3)	FI6(P3)	FI7(P4)	FI8(P4)
Tree layer								
Pinus pinea L.	0.005	—	—	—	—	—	—	—
Q. ilex L.	—	0.005	—	—	0.015	0.003	0.006	0.007
Shrub layer								
Arbutus unedo L.	0.011	0.031	0.012	—	0.015	0.003	—	—
Asparagus sp. L.	—	0.005	—	—	—	—	—	—
Cistus albidus L.	0.011	—	0.059	0.139	0.066	0.003	0.006	—
Cistus crispus L.	0.077	0.175	—	—	0.388	0.178	—	—
Cistus ladanifer L.	0.120	0.098	0.148	0.018	0.097	—	—	0.014
Cistus monspeliensis L.	0.011	0.010	0.703	0.048	—	—	—	0.402
Cistus salvifolius L.	0.033	0.258	0.012	0.012	0.005	0.046	—	—
Cytisus scoparius (L.)	—	0.005	—	—	0.015	—	—	—
Daphne gnidium L.	0.005	0.026	—	—	—	—	—	—
Genista cinerea (Villar)	—	0.005	—	0.006	0.015	—	0.006	—
G. hirsuta Vahl	0.011	—	—	—	—	—	—	—
Lavandula stoechas L.	0.202	0.052	—	0.006	—	—	—	—
Phlomis purpurea L.	0.005	—	—	0.024	0.015	—	—	—
Phyllirea angustifolia L.	0.005	0.005	0.018	—	—	—	—	—
Pistacia lentiscus L.	—	0.015	0.018	—	0.026	0.006	0.006	—
Quercus coccifera L.	0.044	0.093	—	0.012	0.046	0.058	0.150	0.020
Rhamnus alaternus L.	0.005	0.010	0.006	—	0.020	—	—	—
Rosmarinus officinalis L.	0.016	0.041	0.006	0.018	0.010	0.015	0.006	0.007
Rubus ulmifolius Schott	—	0.010	—	—	0.026	0.003	—	—

B	Hedges (HE)							
	HE1(P1)	HE2(P1)	HE3(P2)	HE4(P2)	HE5(P3)	HE6(P3)	HE7(P4)	HE8(P4)
Tree layer								
Olea europaea L.	—	—	—	—	0.030	—	—	—
Q. ilex L.	—	—	0.052	0.022	—	—	0.011	—
Q. suber L.	0.031	—	—	—	—	—	—	0.012
Shrub layer								
A. unedo L.	0.031	—	—	—	0.020	0.015	—	—
Asparagus sp. L.	—	—	—	—	—	0.015	—	—
C. albidus L.	0.021	0.018	0.069	0.109	0.030	0.030	—	0.012
C. crispus L.	0.062	0.125	—	—	0.020	—	—	—
C. ladanifer L.	0.062	0.089	0.017	0.022	0.051	—	—	—
C. monspeliensis L.	0.021	0.009	0.138	0.565	—	—	0.089	0.313
C. salvifolius L.	0.442	0.116	0.017	0.022	0.030	0.022	—	—
Crataegus monogyna Jacques	—	—	—	—	—	0.015	—	—
G. cinerea (Villar)	0.010	0.009	—	—	0.010	0.007	—	—
G. hirsuta Vahl	—	0.018	0.017	—	—	—	—	—
L. stoechas L.	—	0.009	0.017	—	—	—	—	—
Phl. purpurea L.	—	—	—	—	0.030	0.030	—	—
Phy. angustifolia L.	—	0.009	—	—	—	0.007	—	—
Pis. lentiscus L.	0.010	0.009	—	—	0.030	0.030	—	—
Q. coccifera L.	0.185	0.143	0.035	0.022	0.010	0.044	0.100	—
Rhamnus lycioides L.	—	0.009	—	0.022	0.051	0.007	—	—
R. officinalis L.	0.041	0.018	0.035	—	0.041	0.007	—	0.024

C	Copses (CO)							
	CO1(P5)	CO2(P5)	CO3(P6)	CO4(P6)	CO5(P7)	CO6(P7)	CO7(P8)	CO8(P8)
Tree layer								
Q. ilex L.	—	—	—	—	—	—	0.006	—
Q. suber L.	0.014	—	—	0.007	—	—	—	—
Shrub layer								
A. unedo L.	0.007	0.014	—	0.007	—	—	—	—
C. albidus L.	—	0.014	0.031	0.013	0.071	0.102	0.022	0.013
C. crispus L.	0.070	—	—	—	0.008	—	0.006	—
C. ladanifer L.	0.014	—	—	—	0.056	0.014	—	0.007
C. salvifolius L.	0.070	—	—	—	—	—	—	0.020
Cr. monogyna Jacques	0.014	0.021	0.023	0.007	0.016	0.007	0.006	—
D. gnidium L.	0.007	—	—	—	—	—	—	—
Genista hirsuta Vahl	—	—	—	—	0.008	0.007	—	—
L. stoechas L.	—	—	—	—	0.016	0.020	—	—
Lonicera. implexa Aiton	0.007	0.014	—	—	—	—	—	—
Myrtus communis L.	0.021	0.007	—	—	—	—	0.006	—
Nerium oleander L.	0.007	—	—	—	—	0.007	—	—

(*Continued*)

Appendix 1. (Continued)

C

	Copses (CO)							
	CO1(P5)	CO2(P5)	CO3(P6)	CO4(P6)	CO5(P7)	CO6(P7)	CO7(P8)	CO8(P8)
O. europea L.	0.049	0.029	0.015	—	0.016	0.041	0.006	0.007
Phillyrea latifolia L.	—	0.014	0.008	—	—	—	0.006	—
Phl. purpurea L.	—	—	0.214	—	0.127	0.259	—	—
Pis. lentiscus L.	0.042	0.007	—	0.027	0.008	0.027	—	0.007
Pyrus bourgaeana Decaisne	0.007	—	—	—	—	—	—	—
Q. coccifera L.	0.014	0.007	0.008	—	0.040	0.007	—	—
Ru. ulmifolius Schott	0.007	0.014	—	—	—	—	—	0.013
Scirpus holoschoenus L.	—	—	—	—	—	—	0.117	—

D

	Unmanaged forests (UF)							
	UF1(P1)	UF2(P1)	UF3(P2)	UF4(P2)	UF5(P3)	UF6(P3)	UF7(P4)	UF8(P4)
Tree layer								
O. europaea L.	—	—	—	—	—	0.006	—	—
Pin. pinea L.	0.006	—	—	—	—	—	—	—
Q. ilex L.	0.006	0.020	0.024	0.013	0.022	0.039	0.022	0.048
Shrub layer								
A. unedo L.	0.019	0.007	—	—	—	—	—	—
Asparagus sp. L.	—	—	0.008	—	—	—	—	—
C. albidus L.	0.012	0.027	0.331	0.127	—	0.050	—	—
C. crispus L.	0.075	0.067	—	—	—	0.267	—	—
C. ladanifer L.	0.019	0.093	0.137	0.033	—	0.361	0.170	—
C. monspeliensis L.	0.012	0.540	0.186	0.060	—	—	0.526	0.158
C. salvifolius L.	0.530	0.047	0.250	0.067	—	—	0.004	—
D. gnidium L.	—	0.007	0.016	—	—	—	—	0.004
G. hirsuta Vahl	—	0.007	0.137	—	—	—	0.070	—
L. stoechas L.	0.031	0.093	0.097	—	—	—	—	0.079
Phl. purpurea L.	0.012	0.047	—	0.040	—	—	—	—
Pis. lentiscus L.	—	0.007	—	—	—	0.006	—	—
Q. coccifera L.	0.100	0.040	0.040	0.127	—	0.050	—	—
R. officinalis L.	—	0.087	—	—	—	0.011	—	—
Ru. ulmifolius Schott	—	—	—	—	—	—	—	0.004
S. holoschoenus L.	—	—	—	—	—	—	—	0.114
Thymus mastichina (L.)	—	—	—	—	—	0.006	—	—

E

	Unmanaged copses (UCO)							
	UCO1(P5)	UCO2(P5)	UCO3(P6)	UCO4(P6)	UCO5(P7)	UCO6(P7)	UCO7(P8)	UCO8(P8)
Tree layer								
O. europaea L.	—	—	—	—	0.067	—	—	—
Quercus faginea Lam.	—	—	—	—	—	—	0.011	—
Q. ilex L.	0.007	0.006	—	0.017	0.006	—	0.011	—
Q. suber L.	0.007	0.017	—	—	—	—	—	0.039
Shrub layer								
A. unedo L.	—	—	—	—	—	—	0.006	—
C. albidus L.	0.027	—	—	0.006	0.044	0.033	0.039	—
C. crispus L.	0.020	0.011	—	—	—	—	0.006	—
C. ladanifer L.	—	—	—	—	0.044	0.013	0.017	—
C. monspeliensis L.	—	—	—	—	—	—	0.011	—
C. salvifolius L.	0.113	—	—	—	—	—	0.022	—
Cr. monogyna Jacques	—	—	—	—	—	—	0.006	—
D. gnidium L.	—	—	—	—	—	—	0.011	—
G. hirsuta Vahl	—	—	—	—	—	0.033	—	—
L. stoechas L.	—	—	—	—	0.144	0.007	0.022	—
M. communis L.	0.013	—	—	—	0.006	—	—	—
N. oleander L.	—	—	—	—	0.044	—	—	—
O. europaea L.	0.020	0.022	—	—	0.006	0.007	—	—
Phl. purpurea L.	—	—	0.161	0.017	0.017	0.273	—	—
Pis. lentiscus L.	—	0.011	—	0.006	0.011	0.013	0.017	—
Py. bourgaeana Decaisne	—	0.006	—	—	—	—	—	—
Q. coccifera L.	—	—	—	—	0.006	0.027	—	—
Ru. ulmifolius Schott	—	—	—	—	—	—	0.033	—
S. holoschoenus L.	—	—	—	—	0.006	—	0.022	—
Smilax aspera L.	—	—	—	—	—	—	0.017	—

(A) Forest-islands (FI); (B) hedges (HE); (C) copses (CO); (D) unmanaged forests (UF); (E) unmanaged copses (UCO); *n* (1–8): replicate number.

Appendix 2. Abundance (number of individuals) of Orthopteran in different environmental types considering all the sampling plots

Orthoptera species	Environmental types				
	UF	FI	HE	CO	UCO
Suborder Ensifera					
Gryllus (*Gryllus*) *bimaculatus* De Geer, 1773	0	0	1	0	0
Sciobia lusitanica (Rambur, 1838)	0	1	2	2	0
Pt. (*Stilbonemobius*) *lineolatus* (Brullé, 1835)	0	0	1	0	0
Oecanthus pellucens (Scopoli, 1763)	67	56	56	8	0
Steropleurus andalusius (Rambur, 1838)	8	6	21	2	0
Phaneroptera (*Phaneroptera*) *nana* Fieber, 1853	57	39	40	25	9
Tylopsis lilifolia Fieber, 1853	0	30	17	5	2
Platycleis sabulosa Azam, 1901	25	9	34	0	0
T. tessellata (Charpentier, 1825)	211	386	368	71	35
Pterolepis spoliata Rambur, 1838	2	6	2	9	4
Tettigonia viridissima Linnaeus, 1758	6	8	21	5	2
Thyreonotus bidens (Bolívar, 1887)	1	2	3	1	0
Suborder Caelifera					
Calliptamus barbarus (Costa, 1836)	56	77	36	44	34
P. giornae (Rossi, 1794)	1,300	2,467	1,618	703	570
Dociostaurus (*Kazakia*) *jagoi* Soltani, 1978	643	722	370	555	380
Chorthippus (*Glyptobothrus*) *apicalis* (Herrich-Schäffer, 1840)	11	28	10	6	19
C. (*Glyptobothrus*) *vagans* (Eversmann, 1848)	71	71	116	75	42
Omocestus (*Omocestus*) *panteli* (Bolívar, 1887)	22	61	92	15	0
Acrotylus patruelis (Herrich-Schäffer, 1838)	82	2	0	0	0
Locusta migratoria (Linnaeus, 1758)	1	0	1	1	0
Oedipoda caerulescens (Linnaeus, 1758)	676	132	69	36	115
Sphingonotus (*Sphingonotus*) *lluciapomaresi* (Défaut, 2005)	11	2	0	0	0
Ocnerodes prosternalis Bolívar, 1912	4	0	1	0	0
Uvarovitettix nodulosus (Fieber, 1853)	6	0	0	0	0
Xya variegata (Latreille, 1809)	47	0	0	0	0

UF, unmanaged forest; FI, forest-island; HE, hedge; CO, copse; UCO, unmanaged copse.