



RESEARCH ARTICLE - ANTS

Ants (Hymenoptera: Formicidae) as surrogates for epigeic arthropods in Northern Andalusian 'dehesas' (Spain)

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Article History

Edited by

Wesley Dáttilo, Instituto de Ecología A.C., Mexico
 Received 28 November 2019
 Initial acceptance 12 January 2020
 Final acceptance 18 February 2020
 Publication date 30 June 2020

Keywords

Bioindicators; formicidae; pastureland; holm oak; pitfall traps.

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Abstract

The 'dehesas' are important and vast agro-silvo-pastoral systems typical of the Iberian Peninsula that are undergoing a crisis due to their low economic profitability and environmental degradation. Thus, it is necessary to identify effective tools that provide a reliable idea of the status of these ecosystems as a starting point for future measures of conservation. In this study we analyse the possible role of ants as surrogates for epigeic arthropods, a common biodiversity indicator group. A total of 15 farms were sampled throughout Sierra Morena (Andalusia, Spain) with pitfall traps, both for the 'dehesa' habitats themselves and for different microhabitats within the study sites. First, we achieve a complete list of the species of ants of the area. The results indicate that the 'dehesa' habitats were very homogenous for all farms, while microhabitats showed differences in species richness and ant communities' composition compared to the 'dehesas'. To evaluate the role of ants as surrogates, the number of traps occupied by each order of arthropod and by each ant species was compared. We found a high correlation between them what confirm the surrogate character of ants for the rest of arthropods in these ecosystems.

Introduction

The 'dehesas' are agro-silvo-pastoral systems typical of central-western and south-western Iberian Peninsula. They consist of vast extensions of pastures with scattered adult acorn-producing trees that provide shade, shelter and food to livestock (Duque-Lazo & Navarro-Cerrillo, 2017). In general, the ecosystems of 'dehesas' are similar to savannas, due to the low level of soil moisture, little or none litter cover under trees and shrubs, which are usually evergreen, and the high temperatures reached (Leiva & Fernández-Alés, 2003). The 'dehesas' have an anthropogenic origin, deriving from the pre-existing Mediterranean forest through the elimination of the scrub and part of the tree cover, and thus promoting the growth of grass for livestock use (San Miguel, 1994). Although these landscapes come from earlier times,

there is evidence of the use of the word 'dehesa' from the early Middle Ages (Álvarez-Guzmán, 2016). Thenceforth, traditionally they have been dedicated to different activities, mainly extensive livestock rearing (cattle, sheep, goats and/or pigs), but also pasture and grain production, usually for livestock, or hunting use (Klein, 1920; San Miguel, 1994; Martin, 1996). In Spain the 'dehesas' comprise a total of 2,360,700 has (Martin, 1996), principally in the communities of, Extremadura, Castilla La Mancha, Castilla y León and the largest extension in Andalusia with 1,263,143 ha (Costa Pérez et al., 2006). Not only do the 'dehesas' fulfill important roles in the Spanish agriculture but also in the environmental protection. In this respect, they are included in the Natura 2000 network as Ecosystems of Community Interest (Díaz Esteban & Pulido Díaz, 2009; European Directive 92/43/ECC; Marañón et al., 2012), because they are emblematic examples



of Mediterranean managed landscapes with high biodiversity, including threatened species such as the Spanish lynx (*Lynx pardinus* (Temminck, 1827)) (Díaz Esteban & Pulido Díaz, 2009; Álvarez-Guzmán, 2016). They are also considered key cultural landscapes for their aesthetic, historic and ecotourism roles (Marañón et al., 2012; Maldonado et al., 2019). Despite all their values, nowadays the ‘dehesas’ are undergoing a crisis as result of their low economic profitability and of the environmental degradation derived from the loss of traditional management and from other multiple factors, such as the decay of tree canopy or the fragmentation of habitat (Díaz et al., 1997; Plieninger et al., 2004; Escribano et al., 2018). This forces the competent institutions to adopt the necessary measures to keep or restore their status of conservation (Pulido & Picardo, 2010).

As a starting point to set future plans of protection, it is very important to consider what are the best measures to evaluate the current status of the ecosystems. One of the first tasks required when designing conservation strategies is to estimate the biodiversity (Caro, 2010; Laurila-Pant et al., 2015). This knowledge is essential for a proper environmental management. When facing a biodiversity inventory, one of the most outstanding group are the arthropods, because they represent a high percentage in terms of biomass (Bar-On et al., 2018) and they participate in essential functions of the ecosystems (Yang & Gratton, 2014; Schowalter, 2017). But the main problem for the knowledge of this phylum is their megadiversity (Smith et al., 2005). The experts are usually specialized in specific groups, such as orders, and even families or genders; this implies that to study all the arthropod taxa would be an extensive work and the need to involve numerous researchers.

The estimation of species richness is one of the most common ways to measure the biodiversity of an ecosystem (Noss, 1990), but many times this is a complex duty. In recent times, efforts have been made to simplify this task by searching for shortcuts (Moreno et al., 2007). One of the possible methods is the search for groups whose diversity represents other taxa of the community whose study is more complex for any reason, these are the surrogate groups (Moreno et al., 2007; Lewandowski et al., 2010; Lindenmayer & Likens, 2011). The use of surrogates is an important option to mitigate the shortage of biodiversity data and by sampling only one group, instead of the entire community, time and money are saved (Heino et al., 2005). There is no standard agreement to estimate the suitability of a taxon as a surrogate indicator, but efforts have been made for searching certain taxa as surrogates according to their representativeness of the diversity of other groups (see for example Leal et al. (2010)).

Ants (Hymenoptera: Formicidae) display a series of characteristics that make them suitable as a good group of bioindicators (Andersen, 1997; Crist, 2009; Nakamura et al., 2007; Ribas et al., 2012; Underwood & Fisher, 2006; Verdinelli et al., 2017). They present high ecological fidelity

and are functionally important in the ecosystems, participating in multiple relevant functions, such as: the decomposition of organic matter, soil turning, pollination, zoochory, predation of other arthropods or being prey for many other groups of animals (Folgarait, 1998; Philpott & Armbrrecht, 2006; Crist, 2009; Diamé et al., 2017). Moreover, they respond to disturbances in their habitats, regardless of their origin, in a predictable, quick and generally in a linear way (Philpott et al., 2009). They are abundant and well distributed throughout the planet, being in all continents and ecosystems except in Antarctica (Hölldobler & Wilson, 1990). Finally, there is a good knowledge of their taxonomy and they are easily found in the field and sampled (Agosti et al., 2000).

All these factors may lead to conclude that ants are a group susceptible of being surrogate for other taxa. At this respect, some studies have shown that ants, either by their own or together with other groups of fauna or flora, can be considered as surrogates for plants (Gadagkar et al., 1993; Pfeiffer et al., 2003) or for a set of taxa, either just of invertebrates (Sauberer et al., 2004; Nakamura et al., 2007), or also including vertebrates and/or plants (Sauberer et al., 2004; Majer et al., 2007; Leal et al., 2010). However, other studies reported negative results (Osborn et al., 1999; Allen et al., 2001; Dauber et al., 2003; Sackmann et al., 2006; Bennett et al., 2009; Uys et al., 2010; Landeiro et al., 2012; Pérez-Fuertes et al., 2016; Gibb et al., 2017; Hanford et al., 2017; Barton et al., 2019). These contradictory results are not rare. The accuracy of these shortcuts depend on different factors, which are: the studied taxa, the scale of the study and, being the case, the environmental information used (Moreno et al., 2007). Indeed, everything indicates that ants may act as surrogates or not, depending on the group with which they are being related, as well as on the type of habitat. For example, tropical zones are megadiverse, which probably make difficult the task of describing completely their communities. Therefore, individualized studies for every zone and taxon are required if we propose the search of a shortcut for assessing species diversity (Moreno et al., 2007).

Within this context, given the importance of the ‘dehesas’ in the Iberian Peninsula and the scarcity of information on their conservation status, we wished to test the hypothesis of whether ants could be a good subrogated group for arthropods in these ecosystems. For this purpose, we developed this work with the following specific objectives: first, to carry out an exhaustive work for the knowledge of the ‘dehesa’ ants communities, then to evaluate the role of ants as a subrogated group for the rest of arthropods.

Material and Methods

Study site

This study was carried out in 15 ‘dehesas’ along Andalusia (Southern Spain), in the years 2016 and 2017 (Table 1). The distances between more remote farms is 300

km (Figure 1A and Table 1). The study ‘dehesa’ ecosystems are dedicated to livestock, hunting and agriculture. The predominant arboreal species is *Quercus ilex* L. subsp. *ballota* (Desf.) Samp. In some ‘dehesas’ *Olea europaea* var. *sylvestris* Brot. and *Quercus suber* L. may appear occasionally, and with less frequency and scattered distribution: *Ceratonia siliqua* L., *Pyrus bourgaeana* Decne., *Pinus pinea* L., *Prunus dulcis* (Mill.) D.A. Webb, *Quercus faginea* Lam. and *Quercus pyrenaica* Willd. Depending on the ‘dehesa’, the herbaceous

layer may include either natural or improved pastures, or monospecific crops (such as wheat, oats, barley, vetch or pea). Regarding the livestock species, swines and bovines predominate. Ovines, caprines, equines and even beehives are also present. The shrub layer, when present, forms patches in steep zones and low exploitation value areas, either for livestock or agriculture. The main species are *Cistus* sp., *Quercus coccifera* L., *Thymus* spp. *Nerium oleander* L., *Pistacia lentiscus* L., *Retama sphaerocarpa* (L.) Boiss.

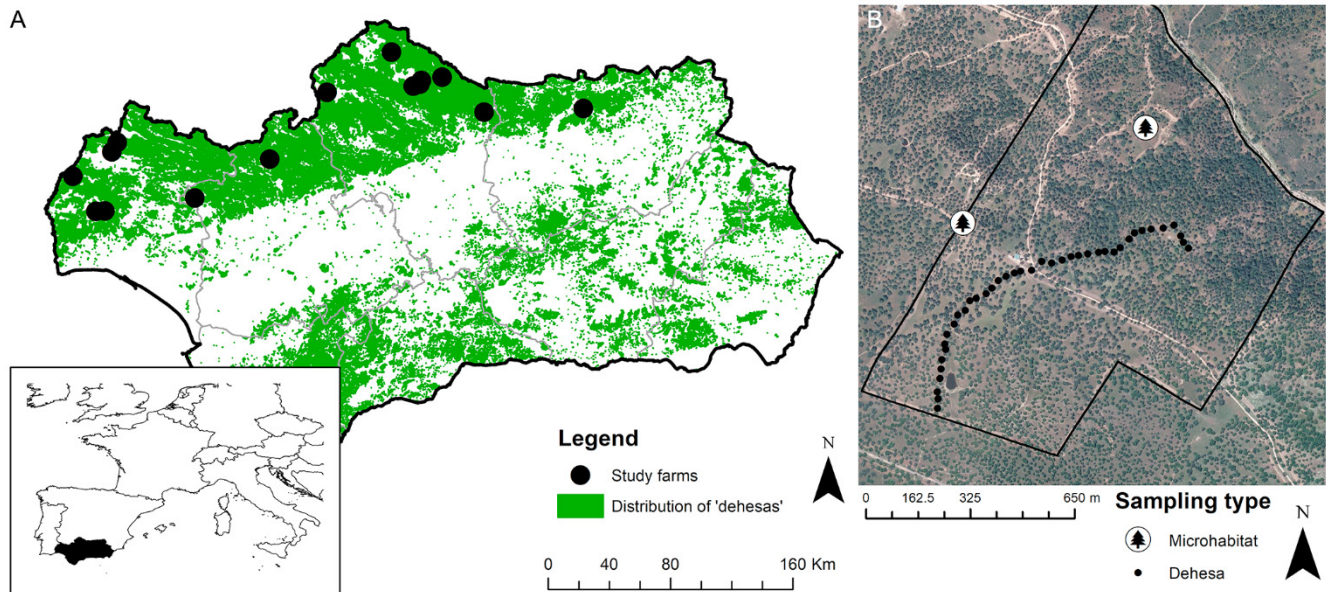


Fig 1. A) Location map of the study ‘dehesas’ (layer of distribution of ‘dehesas’ from REDIAM: Andalusian Environmental Information Network). B) Pitfall traps position according to both sampling types (‘dehesas’ and microhabitats) for the farm AS05, as an example.

Most of the surface of the study farms shows the typical ‘dehesa’ landscape, but there are also small unmanaged areas (microhabitats) with different characteristics. They include: the dense scrub patches describe above; ponds; streams’ riparian forests; and vegetation zones adjacent to traditional stone walls (Table 1). Moreno et al. (2007) found that despite occupying a low proportion of the farm area, these microhabitats (called by them marginal habitats) contribute largely to the biodiversity of these ecosystems. Therefore, for the study we differentiated two types of sampling areas: the landscape of the ‘dehesa’ itself (DE) and the microhabitats (MH).

Climatology

The predominant climate is the Mediterranean, specifically Mediterranean mountain climate (Junta de Andalucía, 2019), with hot summers and cold winters (annual average range between 11 and 18 °C). The accumulated rains range between 600 and 1,200 mm, with maximums in autumn and winter and minimums in summer (Gómez-Zotano et al., 2015).

Experimental design

Sampling was performed with pitfall traps. These constitute an easy and effective sampling system for

communities of epigeic arthropods and they are recommended as part of a standard protocol for measuring biodiversity (Agosti et al., 2000; Gotelli & Colwell, 2001; Prasifka et al., 2007; Sheikh et al., 2018). Traps consisted of 150-ml translucent plastic cups (upper \varnothing 5.7 cm, base \varnothing 5 cm, depth 7.3 cm. REF 409702, DELTALAB SL). They were set at ground level, flushed with the soil surface, and placed in the field for 48 h. Traps were filled with a killing agent consisting of 30-35 ml of water with 1% of detergent, to break the surface tension of water and prevent the escape of little individuals. We did not employ any bait so the traps were suitable to calculate both presence of species and their relative abundance (Wang et al., 2001).

Collected specimens were separated into two groups: Formicidae and other arthropods. Ants were identified to species level. Their abundance was quantified by counting the number of workers per each trap, as well as by the number of traps occupied by each species (Gotelli et al., 2011). In the case of the rest of the arthropods, they were identified to order level (except for subclass Acari) and the number of traps occupied by each one was recorded. Numerous studies have shown that the estimation of the diversity of arthropods at the taxonomic level of order can be a very useful tool for the

Table 1. Codes for the study farms, their location, surface and years of sampling. Five farms were full sampled in 2017 (‘dehesas’ and microhabitats). In the rest of the farms the ‘dehesas’ were sampled in 2016 and the microhabitats in 2017. Means of annual temperatures (MEAN T.) and means of accumulated rainfall (MEAN ACU. RAIN.) are included. Microhabitats column shows the different types of microhabitats sampled per farm: zones adjacent to traditional stone walls (SW); scrub patches (SP); vegetation of temporary water courses (VW); ponds (P); riparian forest (RF).

CODE	FARM	LOCATION	PROVINCE	COORDINATES	SURFACE (Ha)	YEARS	MEAN T. (°C)	MEAN ACU. RAIN. (mm)	MICRO-HÁBITATS
AP05	La Juanita	Alosno	Huelva	37.555913°, -7.082250°	191.14	17	17.81	639.01	SP, RF
AP06	Paymoguillo	Paymogo	Huelva	37.751957°, -7.330534°	109.4	16-17	17.00	671.00	VW, SP
AS02	El Palomar de la Morra	Pozoblanco	Córdoba	38.348277°, -4.819241°	96.69	17	15.70	544.00	SW, VW
AS05	Lote de los Pérez	Cazalla de la Sierra	Sevilla	37.896797°, -5.875586°	107.7	16-17	16.27	661.59	SW, SP
AS06	Las Morrillas	Pozoblanco	Córdoba	38.360550°, -4.771079°	157.42	17	15.62	538.96	SW, P
CO01	Las Ánimas	Aroche	Huelva	37.962325°, -7.012390°	77.62	16-17	15.96	779.81	SW, VW
CO05	Monterrey y Carretero	Aroche	Huelva	37.905465°, -7.049219°	119.59	17	16.34	773.34	SW, VW
CO08	Quebradahonda	Castillo de las Guardas	Sevilla	37.653961°, -6.421098°	114.14	16-17	17.08	782.91	SW, VW
CO12	Majada del Indio	El Viso	Córdoba	38.545339°, -4.986985°	123.94	16-17	16.05	527.79	VW, SP
EN04	Encinarejo	Alosno	Huelva	37.551441°, -7.148217°	281.33	16-17	17.65	617.79	SW, VW
FA01	Las Hazas	Villanueva de Córdoba	Córdoba	38.403720°, -4.603515°	459.09	16-17	15.35	594.48	SW, VW
FA05	La Panadera	Pozoblanco	Córdoba	38.381964°, -4.758971°	83.6	17	15.70	553.00	SW, SW
FA11	Santa Clotilde	Cardeña	Córdoba	38.202356°, -4.286978°	292	16-17	15.48	887.26	VW, SP
UP23	Oropesa	Fuente Ovejuna	Córdoba	38.299421°, -5.463488°	110.69	16-17	15.73	564.71	SW, VW
UP24	Las Caras	Vilches	Jaén	38.229856°, -3.544986°	592.61	16-17	16.76	574.51	SW, SP

evaluation of the conservation status of different ecosystems (Wettstein & Schmid, 1999; Cecil et al., 2019; Holmquist & Schmidt-Gengenbach, 2019).

According to Crist and Wiens (1996), ants may display different distribution patterns even at small scales. To avoid this possible effect and to cover the largest possible area of the study sites, we placed a single linear transect of 1600 m that crossed the larger axis of every farm (Figure 1B). These transects consisted of 40 pitfall traps, separated one trap from the next by 40 m. If there were fences with pigs (*Sus scrofa domestica* L.) they were avoided, since they often unearth and destroy the traps.

As regards microhabitats (MH), we sampled two of them per farm. Depending on the study site the MH type could vary (see in Table 1). As the area of the MH is much smaller than the one of the DE, the method to set the pitfall traps necessarily had to be different. Accordingly, we set one transect per MH, with 10 pitfall traps separated each trap from the next 2 m. This methodology has been used by our research group in numerous occasions with proven effectiveness. Therefore, in total per farm and for both sampling modalities, 60 pitfall traps were placed (40 in DE and 10+10 in MH).

The sampling was conducted during the springs and summers (May-June) of 2016 and 2017, being this the peak period of activity for most ant species of these latitudes (Cros et al., 1997; Carpintero et al., 2007). In Table 1 it is specified the timing of sampling for every farm.

Statistical analysis

Adequation of the sampling systems - Ants

The adequation of sampling effort and methodology was tested with ants’ rarefaction curves (Mao’s tau) based on abundance of workers per species and trap. The sample coverage was calculated for each sampling area (Chao & Chiu, 2016; function “Diversity”; package “SpadeR”), this index measures “[...] the proportion of the total individuals in a community that belong to the species represented in the sample”, this is a measure of sample completeness (Chao & Jost, 2012).

Comparison of the fauna of DE and MH and evaluation of farms homogeneity - *Ants*

In order to compare the richness and composition of species (relative abundance of each species) of ants of the ‘dehesas’ versus microhabitats, the traps were divided into three categories: first comprised the twenty traps of the

microhabitats (MH); for the second and third categories we selected the first and second twenty traps of the ‘dehesas’ (D1 and D2 respectively), this is to have a balance design (20 traps per category). Besides, comparing the first and second twenty traps of DE it was also possible to test the homogeneity of these ecosystems. With these groups of traps (D1, D2 and MH) we performed two statistical analyzes: a factorial One-way ANOVA analysis was performed with the number of ant species per trap to compare the richness between sampling categories (D1, D2 and MH); to compare the structure of ant assemblages (relative abundance of the species) per sampling category (D1, D2 and MH), a One-way PERMANOVA analysis was performed with the matrix of the number of traps occupied by each species. Bray-Curtis distances were applied, with 9999 permutations for the calculation of the similarity matrix.

Surrogacy study

The role of ants as surrogates for arthropods was analysed by means of a PLSR (Partial Least Squares Regression) analysis. This allows to establish a linear regression between two matrices of variables (predicted vs. observed variables) of unequal size (ants vs. arthropods), which are projected to a new space (Abdi, 2010). The analysis was performed with the number of traps occupied by each species of ants versus number of traps occupied by

the different groups of arthropods. The use of the covariance matrix and of the correlation matrix were evaluated to perform this analysis, using the system that explained more variance.

Spatial autocorrelation

In order to account for spatial autocorrelation we carried out a Mantel test (Mantel, 1967), which calculated the conditional correlation of two matrices of the same rank (diversity of ants and of arthropods) eliminating the effect of a third one (geographical location of the study sites) (Smouse et al., 1986).

These analyses were performed with the statistical packages PAST 3.20 (Hammer et al., 2001), R v 3.6.2 (R Core Team, 2019) and STATISTICA v 8.0 (StatSoft Inc., 2001).

Results

Formicidae study

Ants species

The species rarefaction curves (Figure 2) for each site reached or approached their asymptote. These results are reinforced by the sample coverage analysis (Table 3), that shows values close to 1, what confirms that all or most of the species of the sampling areas have been recorded.

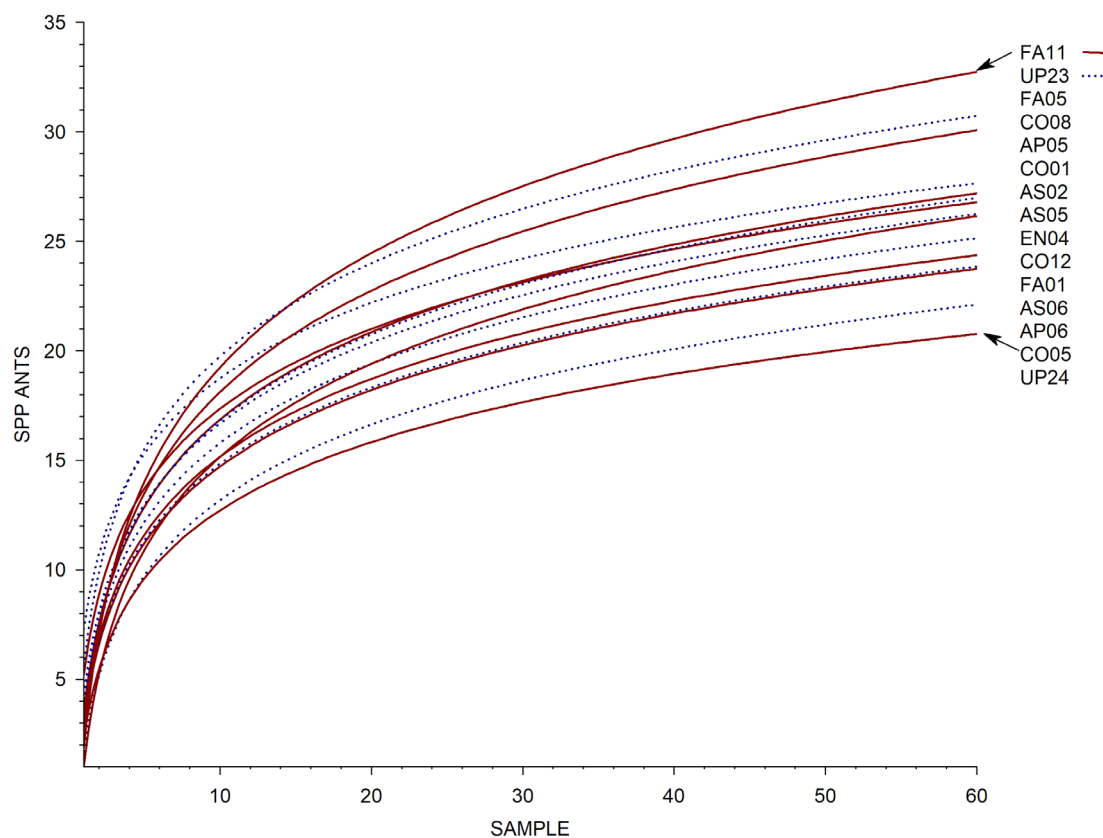


Fig 2. Rarefaction curves of ants for each farm, with the abundance of workers per species and trap. The curves are alternatively drawn in solid and dotted lines for a better differentiation.

Table 2. List of the species of ants and their abbreviations. Number of farms where each species was located (NF); total number of traps occupied by each species (NT); total abundance of workers-individuals (NW).

SPECIES	AUTHOR, YEAR	ABBREV	NF	NT	NW
<i>Aphaenogaster dulcinea</i>	Emery, 1924	APHDUL	11	45	79
<i>Aphaenogaster gibbosa</i>	(Latreille, 1798)	APHGIB	15	139	322
<i>Aphaenogaster iberica</i>	Emery, 1908	APHIBE	14	288	1246
<i>Aphaenogaster senilis</i>	Mayr, 1853	APHSEN	9	170	865
<i>Camponotus cruentatus</i>	(Latreille, 1802)	CAMCRU	11	186	1119
<i>Camponotus fallax</i>	(Nylander, 1856)	CAMFAL	4	7	7
<i>Camponotus figaro</i>	Collingwood & Yarrow, 1969	CAMFIG	1	2	2
<i>Camponotus foreli</i>	Emery, 1881	CAMFOR	8	16	76
<i>Camponotus lateralis</i>	(Olivier, 1792)	CAMLAT	5	10	37
<i>Camponotus micans</i>	(Nylander, 1856)	CAMMIC	4	8	9
<i>Camponotus pilicornis</i>	(Roger, 1859)	CAMPIL	14	50	94
<i>Camponotus sylvaticus</i>	(Olivier, 1792)	CAMSYL	7	12	22
<i>Cataglyphis hispanica</i>	(Emery, 1906)	CATHIS	15	612	2704
<i>Cataglyphis iberica</i>	(Emery, 1906)	CATIBE	9	60	234
<i>Cataglyphis rosenhaueri</i>	Santschi, 1925	CATROS	7	34	107
<i>Colobopsis truncata</i>	(Spinola, 1808)	COLTRU	2	4	4
<i>Crematogaster auberti</i>	Emery, 1869	CREAUB	11	49	191
<i>Crematogaster scutellaris</i>	(Olivier, 1792)	CRESCU	14	77	511
<i>Crematogaster sordidula</i>	(Nylander, 1849)	CRESOR	3	4	44
<i>Formica cunicularia</i>	Latreille, 1798	FORCUN	3	5	15
<i>Formica gerardi</i>	(Bondroit, 1917)	FORGER	2	2	5
<i>Gonomma baeticum</i>	Reyes & Rodriguez, 1987	GONBAE	8	18	34
<i>Gonomma hispanicum</i>	(André, 1883)	GONHIS	13	46	139
<i>Gonomma kugleri</i>	Espadaler, 1986	GONKUG	3	5	5
<i>Iberoformica subrufa</i>	(Roger, 1859)	IBESUB	15	448	6858
<i>Lasius grandis</i>	Forel, 1909	LASGRA	4	20	478
<i>Lasius lasioides</i>	(Emery, 1869)	LASLAS	13	45	97
<i>Messor barbarus</i>	(Linnaeus, 1767)	MESBAR	15	484	6010
<i>Messor bouvieri</i>	Bondroit, 1918	MESBOU	5	20	120
<i>Messor celiae</i>	Reyes, 1985	MESCEL	3	7	19
<i>Messor hispanicus</i>	Santschi, 1919	MESHIS	8	23	65
<i>Messor lusitanicus</i>	Tinaut, 1985	MESLUS	1	2	2
<i>Myrmica aloba</i>	Forel, 1909	MYRALO	1	2	8
<i>Oxyopomyrmex saulcyi</i>	Emery, 1889	OXYSAU	13	36	181
<i>Pheidole pallidula</i>	(Nylander, 1849)	PHEPAL	9	38	627
<i>Plagiolepis pygmaea</i>	(Latreille, 1798)	PLAPYG	15	192	667
<i>Plagiolepis schimitzii</i>	(Latreille, 1798)	PLASCH	11	56	135
<i>Proformica ferrerii</i>	Bondroit, 1918	PROFER	1	5	25
<i>Solenopsis spp.</i>		SOLSPP	14	56	105
<i>Tapinoma madeirense</i>	Forel, 1895	TAPMAD	1	1	2
<i>Tapinoma nigerrimum cf.</i>	(Nylander, 1856)	TAPNIG	15	340	5227
<i>Temnothorax alfacarensis</i>	Tinaut, <i>in littere.</i>	TEMALF	1	2	4
<i>Temnothorax angustulus</i>	(Nylander, 1856)	TEMANG	2	3	3
<i>Temnothorax racovitzai</i>	(Bondroit, 1918)	TEMRAC	11	39	141
<i>Temnothorax recedens</i>	(Nylander, 1856)	TEMREC	3	8	12
<i>Temnothorax tyndalei</i>	(Nylander, 1856)	TEMTYN	2	5	8
<i>Tetramorium caespitum cf.</i>	(Linnaeus, 1758)	TETCAE	5	11	34
<i>Tetramorium forte</i>	Forel, 1904	TETFOR	15	262	2364
<i>Tetramorium semilaeve</i>	André, 1883	TETSEM	15	310	1904

A total of 32,820 workers from 49 different species, belonging to 19 genera, were captured. There was an average of 26 species per farm (21-33 species) (SM1). The following species were in every farm and with high abundance: *Aphaenogaster gibbosa* (Latreille, 1798), *Cataglyphis hispanica*

(Emery, 1906), *Iberoformica subrufa* (Roger, 1859), *Messor barbarus* (Linnaeus, 1767), *Plagiolepis pygmaea* (Latreille, 1798), *Tapinoma nigerrimum* cf. (Nylander, 1856), *Tetramorium forte* Forel, 1904 and *Tetramorium semilaeve* André, 1883 (Table 2 and 3, and SM1).

Table 3. Species of ants per farm: Total number of species (SPP.TO); Sample coverage (SC); species in 'dehesas' (SPP.DE); species in microhabitats (SPP.MH); common species in both types of habitats (SPP.CO); species exclusive to microhabitats (SPP.EX.MH).

FARM	SPP. TO	SC	SPP. DE	SPP. MH	SPP. CO	SPP. EX. MH	% SPP. CO	% SPP. EX. MH
AP05	27	0.996	26	18	17	1	62.96	3.70
AP06	23	0.998	20	18	15	3	65.22	13.04
AS02	27	1	23	20	16	4	59.26	14.81
AS05	26	0.999	22	16	12	4	46.15	15.38
AS06	24	0.999	23	12	11	1	45.83	4.17
CO01	27	0.999	26	20	19	1	70.37	3.70
CO05	22	0.999	18	16	12	4	54.55	18.18
CO08	26	0.999	24	16	14	2	53.85	7.69
CO12	25	0.998	22	21	18	3	72.00	12.00
EN04	25	0.997	25	14	14	0	53.85	0.00
FA01	24	0.998	22	14	12	2	50.00	8.33
FA05	31	0.999	28	18	15	3	48.39	9.68
FA11	33	0.997	26	24	17	7	51.52	21.21
UP23	30	0.999	27	22	19	3	63.33	10.00
UP24	21	0.998	21	11	11	0	52.38	0.00
MINIMUM	21	0.996	18	11	11	0	45.83	0.00
AVERAGE	26.1	100	23.5	17.3	14.8	2.5	56.79	9.46
MAXIMUM	33	1.000	28	24	19	7	72.00	21.21

Comparison of the fauna of DE and MH and evaluation of farms homogeneity - Ants

The results of the ANOVA showed that there were significant differences in the number of ant species per trap according to the different category of traps: first or second twenty traps of the 'dehesas' transects (D1 and D2), and twenty traps of microhabitats transects (MH) ($F=8.990$, $p<0.0001$). The post-hoc Tukey test HSD group to group delved into these results and revealed that D1 and D2 did not show significant differences ($p=0.2978$), while MH registered a significant higher capture rate than the other groups (D1 vs. MH $p=0.0188$ y D2 vs. MH $p<0.0001$). The sampling in microhabitats added 0-7 more species of ants to the list per farm, which accounted for 9.46 % of the species (Table 3). Moreover, there are three species that were only found in microhabitats: *Messor lusitanicus* Tinaut, 1985, *Myrmica aloba* Forel, 1909 and *Tapinoma madeirense* Forel, 1895.

A one-way PERMANOVA showed that there were significant differences between D1, D2 and MH ant assemblages' structure, according to the matrices of the number of traps occupied by each species (PERMANOVA one-way $F = 5.1230$, $p = 0.0001$). Again, a comparison by pairs revealed that the differences were due to MH (D1 vs. D2 $p = 0.5803$, D1 vs. MH $p = 0.0001$, D2 vs. MH $p = 0.0001$)

Other arthropods

Non-formicid arthropods comprised 34 groups, with an average of 21 per farm (17-25 groups). All the groups and the abbreviations used in the figures are shown in SM2. The most abundant groups, and found in every farm, were: Acari, Araneae, Coleoptera, Diptera, Entomobriomorpha, Hemiptera, Hymenoptera, Orthoptera and Symphyleona.

Comparison ants vs. other arthropods

Surrogacy study

A PLSR analysis performed with the matrix of covariance (first axis explained 66.38% of the variance) showed that the diversity of ants vs. other arthropods was highly and significantly correlated ($r^2=0.8134$; $P < 0.0001$) (Figure 3).

Study of the spatial autocorrelation: Mantel Test.

The results of the test with the matrices of the number of traps occupied by ant species and by the arthropod groups was of $r= 0.45$ ($p=0.0012$; for 9999 permutations). Then, the test analysed their correlation eliminating the possible effect of farms distance, with similar results ($r = 0.44$ $p=0.0011$, for 9999 permutations). Therefore, the location of farms did not have any effect on ants and arthropods correlation.

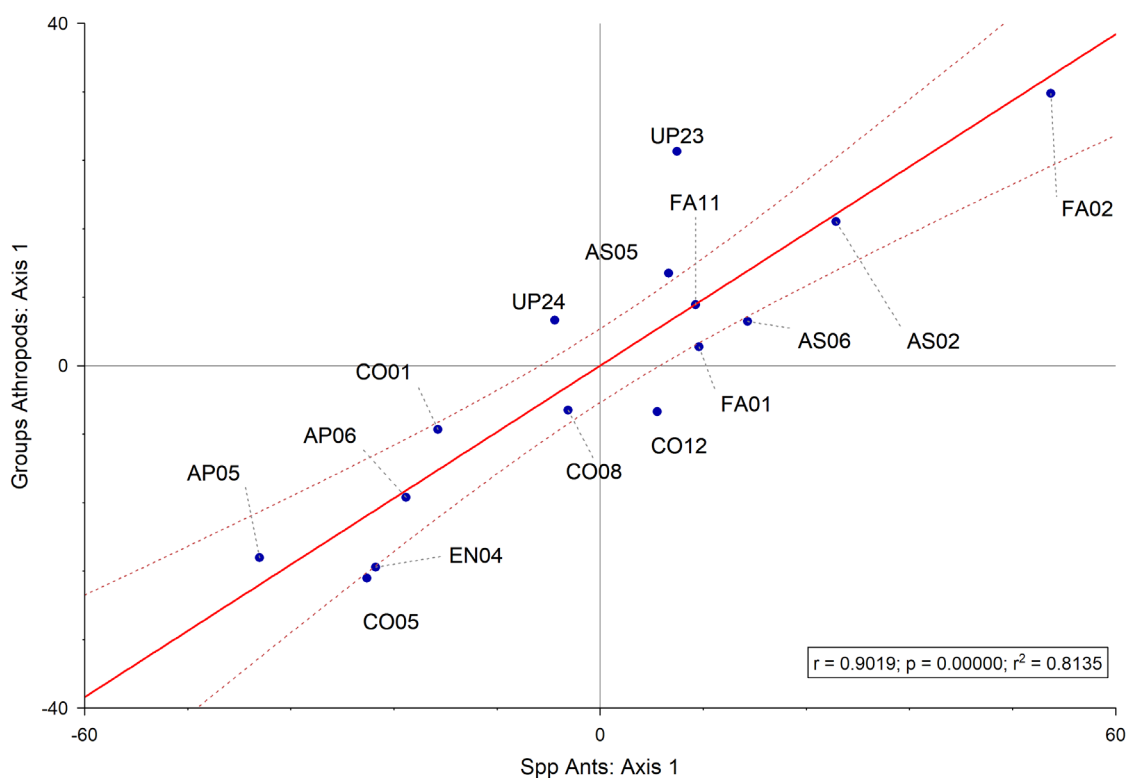


Fig 3. Correlation PLS of scores for axis 1 for the block of arthropod groups versus ants.

Discussion

The present study supports the hypothesis that ants can act as a surrogate group for general epigeic arthropods diversity in ‘dehesa’ ecosystems.

According to the ant fauna, the studied ‘dehesas’ constitutes a homogeneous habitat, with a group of common and very abundant species in all the sites sampled throughout Sierra Morena. These are mainly species adapted to open and warm ecosystems, such as *M. barbarus* and *C. hispanica*, or to open areas with the presence of a dispersed tree stratum (*I. subrufa*). Some species are adapted to live in the litter layer, for example *P. pygmaea*. Finally, there are a large group of generalist species, such as *A. gibbosa*, *T. nigerrimum* cf., *T. forte* and *T. semilaeve* (Roig & Espadaler, 2010).

The fact that not only did we study the typical ‘dehesa’ habitat, but also the different microhabitats of the farms completed the information about the sites. These samplings increased the number of species, even with the appearance of species exclusively found in microhabitats. Actually, the composition of ant assemblages in microhabitats showed significant differences with those of the typical habitat of ‘dehesas’. These results confirm what has already been said by numerous authors that small variations in the structure of habitats will bring modifications in the composition of ant species (Menke & Vachter, 2014; Vasconcelos et al., 2014). Thus, in the microhabitats there were found some species typical of shaded and/or humid areas, such as *M. aloba* or *M. lusitanicus*.

Regarding the use of ants as a surrogate group for epigeic arthropods in ‘dehesas’ of northern Andalusia, in our work we have verified how both groups have a high correlation. At this respect, Leal et al. (2010) proposed a benchmark for assessing if a surrogate group provides a reliable prediction of other groups. They consider a surrogate “reasonable” if it explains > 60% of total species richness, “good” if it explains > 70% and “excellent” if it explains > 80%. In our case, the value of correlation of ants’ diversity and arthropods is of $r^2 = 0.8134$, therefore we may consider that the ants of the ‘dehesas’ of Andalusia reflects to a large extent the community of epigeic arthropods. We found similar results in Nakamura et al. (2007), where a strong relationship between ant species and orders of insects from forests and subtropical grasslands of eastern Australia is found. Other authors also obtain positive values of surrogacy using different taxa. For example, Biaggini et al. (2007) studied in an area of similar climate to ours the possible role of the diversity of species of the family Carabidae (Coleoptera) as surrogate for other insect orders obtaining a significant correlation of more than 90% ($p \ll 0.05$). Guan et al. (2018), analysed the status of the species of gastropods as surrogates for the invertebrate orders of the lakes of China and obtained also a high correspondence ($r = 0.66$, $p \ll 0.05$).

These studies contribute to highlight the use of surrogate groups, at least in particular circumstances; we already established in the introduction the need to be wary and consider that there are multiple factors that may influence the validity of the surrogate groups (Moreno et al., 2007). And

also, it would depend on one's objectives, considering always the balance between the level of accuracy desired and the need to reduce the burden of addressing the study. In other words, as Wiens et al. (2008) suggest, we need to take into account how good is good enough. In the case of studies related to conservation and management of ecosystems or species, many times we need to achieve results in an effective, quick way. The use of surrogate groups may be especially helpful in this context. With our study we conclude that just with the study of ants, a single group with a good taxonomic resolution in the Iberian Peninsula, the situation of the epigeic arthropod community can be extrapolated, and therefore they could be used as a tool that help to evaluate the state of conservation of the 'dehesa' ecosystem.

Acknowledgements

This work has been financed by Life BioDehesa project (LIFE11 / BIO / ES / 000726) 'Dehesa ecosystems: Development of policies and tools for the management and conservation of biodiversity', European Union. We are very grateful to Jose Emilio Guerrero-Ginel (University of Cordoba) for the coordination of this project. Thank you to Sergio Andicoberry de los Reyes (University of Cordoba) and Alma M^a García Moreno, Belen Caño Vergara, Pedro J. Gómez Giráldez (IFAPA Córdoba, Junta de Andalucía) for their participation in the field work. The revision of the manuscript by Alba Jiménez-Guirval is gratefully acknowledged.

References

Abdi, H. (2010). Partial least squares regression and projection on latent structure regression (PLS Regression). *Wiley Interdisciplinary Reviews: Computational Statistics* 2: 97–106. doi: 10.1002/wics.51

Agosti, D., Majer, J.D., Alonso, L.E., & Schultz, T.R. (2000). *Ants: standard methods for measuring and monitoring biodiversity*. Smithsonian Institution Press: Washington, 280 p.

Allen, C.R., Pearlstine, L.G., Wojcik, D.P., & Kitchens, W.M. (2001). The spatial distribution of diversity between disparate taxa: Spatial correspondence between mammals and ants across South Florida, USA. *Landscape Ecology* 16: 453–464. doi: 10.1023/A:1017998124698

Álvarez-Guzmán, J.R. (2016). The image of a tamed landscape: *dehesa* through History in Spain. *Culture and History Digital Journal* 5: 1–17. doi: 10.3989/chdj.2016.003

Andersen, A. (1997). Using ants as bioindicators: multiscale issues in ant community ecology. *Conservation Ecology* 1: 1–10. doi: 10.5751/ES-00014-010108

Bar-On, Y.M., Phillips, R., & Milo, R. (2018). The biomass distribution on Earth. *Proceedings of the National Academy of Sciences* 115: 6506–6511. doi: 10.1073/pnas.1711842115

Barton, P.S., Evans, M.J., Sato, C.F., O'Loughlin, L.S., Foster, C.N., Florance, D., & Lindenmayer, D.B. (2019). Higher-taxon and functional group responses of ant and bird assemblages to livestock grazing: a test of an explicit surrogate concept. *Ecological Indicators* 96: 458–465. doi: 10.1016/j.ecolind.2018.09.026

Bennett, J.M., Kutt, A.S., Johnson, C.N., & Robson, S.K.A. (2009). Ants as indicators for vertebrate fauna at a local scale: an assessment of cross-taxa surrogacy in a disturbed matrix. *Biodiversity and Conservation* 18: 3407–3419. doi: 10.1007/s10531-009-9650-2

Biaggini, M., Consorti, R., Dapporto, L., Dellacasa, M., Paggetti, E., & Corti, C. (2007). The taxonomic level order as a possible tool for rapid assessment of arthropod diversity in agricultural landscapes. *Agriculture, Ecosystems & Environment* 122: 183–191. doi: 10.1016/j.agee.2006.12.032

Caro, T. (2010). *Conservation by proxy: indicator, umbrella, keystone, flagship and other surrogate species*, 1st edn. Island Press: Washington, 374 p.

Carpintero, S., Retana, J., Cerdá, X., Reyes-López, J.L., & Arias de Reyna, L. (2007). Exploitative strategies of the invasive Argentine ant (*Linepithema humile*) and native ant species in a southern Spanish pine forest. *Environmental entomology* 36: 1100–1111. doi: http://dx.doi.org/10.1603/0046-225X(2007)36[1100:ESOTIA]2.0.CO;2

Cecil, E.M., Spasojevic, M.J., & Cushman, J.H. (2019). Cascading effects of mammalian herbivores on ground-dwelling arthropods: variable responses across arthropod groups, habitats and years. *Journal of Animal Ecology* 88: 1319–1331. doi: 10.1111/1365-2656.13033

Chao, A. & Chiu, C.-H. (2016). Species richness: estimation and comparison. *Wiley StatsRef: Statistics Reference Online* 1–26. doi: 10.1002/9781118445112.stat03432.pub2

Chao, A. & Jost, L. (2012). Coverage-based rarefaction and extrapolation: standardizing samples by completeness rather than size. *Ecology* 93: 2533–2547. doi: 10.1890/11-1952.1

Costa Pérez, J.C., Martín Vicente, Á., Fernández Alés, R., & Estirado Oliet, M. (2006). *Dehesas de Andalucía: caracterización ambiental*, 1st edn. Consejería de Medio Ambiente y Ordenación del Territorio. Junta de Andalucía: Sevilla, 294 p.

Crist, T.O. (2009). Biodiversity, species interactions, and functional roles of ants (Hymenoptera: Formicidae) in fragmented landscapes: a review. *Myrmecological News* 12: 3–13

Crist, T.O. & Wiens, J.A. (1996). The distribution of ant colonies in a semiarid landscape: implications for community and ecosystem processes. *Oikos* 76: 301. doi: 10.2307/3546202

Cros, S., Cerdá, X., & Retana, J. (1997). Spatial and temporal variations in the activity patterns of Mediterranean ant communities. *Ecoscience* 4: 269–278. doi: 10.1080/11956860.1997.11682405

- Dauber, J., Hirsch, M., Simmering, D., Waldhardt, R., Otte, A., & Wolters, V. (2003). Landscape structure as an indicator of biodiversity: matrix effects on species richness. *Agriculture, Ecosystems & Environment*, 98: 321–329. doi: 10.1016/S0167-8809(03)00092-6
- Diamé, L., Rey, J.-Y., Vayssières, J.-F., Grechi, I., Chailleux, A., & Diarra, K. (2017). Ants: major functional elements in fruit agro-ecosystems and biological control agents. *Sustainability*, 10: 23. doi: 10.3390/su10010023
- Díaz Esteban, M. & Pulido Díaz, F.J. (2009). *Dehesas perennifolias de Quercus spp.*, 1st edn. Ministerio de Medio Ambiente, y Medio Rural y Marino.: Madrid, 169 p.
- Díaz, M., Campos, P., & Pulido, F.J. (1997). The Spanish *dehesas*: a diversity in land-use and wildlife. In: Pain D, Pienkowski M (eds) *Farming and birds in Europe: The Common Agricultural Policy and its implications for bird conservation*. Academic Press: London, pp 178–209
- Duque-Lazo, J. & Navarro-Cerrillo, R.M. (2017). What to save, the host or the pest? The spatial distribution of xylophage insects within the Mediterranean oak woodlands of Southwestern Spain. *Forest Ecology and Management* 392: 90–104. doi: 10.1016/j.foreco.2017.02.047
- Escribano, M., Díaz-Caro, C., & Mesias, F.J. (2018). A participative approach to develop sustainability indicators for dehesa agroforestry farms. *Science of The Total Environment* 640–641: 89–97. doi: 10.1016/j.scitotenv.2018.05.297
- European Council (1992). Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. *Official Journal of the European Union* 206: 7–50
- Folgarait, P. (1998). Ant biodiversity to ecosystem functioning: a review. *Biodiversity And Conservation* 7: 1121–1244. doi: 10.1023/A:1008891901953
- Gadagkar, R., Nair, P., Chandrashekhara, K., & Bhat, D.M. (1993). Ant species richness and diversity in some selected localities of Western Ghats. *Hexapoda* 5: 79–94
- Gibb, H., Dunn, R.R., Sanders, N.J., et al. (2017). A global database of ant species abundances. *Ecology* 98: 883–884. doi: 10.1002/ecy.1682
- Gómez-Zotano, J., Alcántara-Manzanares, J., Olmedo-Cobo, J.A., & Martínez-Ibarra, E. (2015). La sistematización del clima mediterráneo: identificación, clasificación y caracterización climática de Andalucía (España). *Revista de geografía Norte Grande* 61: 161–180. doi: 10.4067/S0718-34022015000200009
- Gotelli, N.J. & Colwell, R.K. (2001). Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters* 4: 379–391. doi: 10.1046/j.1461-0248.2001.00230.x
- Gotelli, N.J., Ellison, A.M., Dunn, R.R. & Sanders, N.J. (2011). Counting ants (Hymenoptera: Formicidae): biodiversity sampling and statistical analysis for myrmecologists. *Myrmecological News* 15: 13–19
- Guan, Q., Liu, J., Batzer, D.P., Lu, X., & Wu, H. (2018). Snails (Mollusca: Gastropoda) as potential surrogates of overall aquatic invertebrate assemblage in wetlands of Northeastern China. *Ecological Indicators* 90: 193–200. doi: 10.1016/j.ecolind.2018.01.069
- Hammer, Ø., Harper, D.A.T., & Ryan, P.D. (2001). PAST: paleontological statistics software package for education and data analysis. *Palaeontologia Electronica* 4: 1–9. doi: 10.1016/j.bcp.2008.05.025
- Hanford, J.K., Crowther, M.S., & Hochuli, D.F. (2017). Effectiveness of vegetation-based biodiversity offset metrics as surrogates for ants. *Conservation Biology* 31: 161–171. doi: 10.1111/cobi.12794
- Heino, J., Paavola, R., Virtanen, R., & Muotka, T. (2005). Searching for biodiversity indicators in running waters: do bryophytes, macroinvertebrates, and fish show congruent diversity patterns? *Biodiversity and Conservation* 14: 415–428. doi: 10.1007/s10531-004-6064-z
- Hölldobler, B. & Wilson, E.O. (1990). *The Ants*. Belknap Press: Cambridge, 746 p.
- Holmquist, J.G. & Schmidt-Gengenbach, J. (2019). Arthropod assemblages in a montane wetland complex: influences of adjoining lotic and lentic habitat and temporal variability. *Wetlands* 1–13. doi: 10.1007/s13157-019-01175-6
- Junta de Andalucía (2019). Rediam (Red de información ambiental de Andalucía). <http://www.juntadeandalucia.es/medioambiente/site/portalweb/menuitem.7e1cf46ddf59bb227a9e205510e1ca/>. Accessed 10 Sep 2019
- Klein, J. (1920). The Mesta: a study in Spanish economic history, 1273-1836. *The Hispanic American Historical Review* 5: 255. doi: 10.2307/2506028
- Landeiro, V.L., Bini, L.M., Costa, F.R.C., Franklin, E., Nogueira, A., de Souza, J.L.P., Moraes, J., & Magnusson, W.E. (2012). How far can we go in simplifying biomonitoring assessments? An integrated analysis of taxonomic surrogacy, taxonomic sufficiency and numerical resolution in a megadiverse region. *Ecological Indicators* 23: 366–373. doi: 10.1016/j.ecolind.2012.04.023
- Laurila-Pant, M., Lehtikoinen, A., Uusitalo, L., & Venesjärvi, R. (2015). How to value biodiversity in environmental management? *Ecological Indicators* 55: 1–11. doi: 10.1016/j.ecolind.2015.02.034
- Leal, I.R., Bieber, A.G.D., Tabarelli, M., & Andersen, A.N. (2010). Biodiversity surrogacy: indicator taxa as predictors of total species richness in Brazilian Atlantic forest and

- Caatinga. *Biodiversity and Conservation* 19: 3347–3360. doi: 10.1007/s10531-010-9896-8
- Leiva, M.J. & Fernández-Alés, R. (2003). Post-dispersive losses of acorns from Mediterranean savannah-like forests and shrublands. *Forest Ecology and Management* 176: 265–271. doi: 10.1016/S0378-1127(02)00294-3
- Lewandowski, A.S., Noss, R.F., & Parsons, D.R. (2010). The effectiveness of surrogate taxa for the representation of biodiversity. *Conservation Biology* 24: 1367–1377. doi: 10.1111/j.1523-1739.2010.01513.x
- Lindenmayer, D.B. & Likens, G.E. (2011). Direct measurement versus surrogate indicator species for evaluating environmental change and biodiversity loss. *Ecosystems* 14: 47–59. doi: 10.1007/s10021-010-9394-6
- Majer, J.D., Orabi, G., Boisevac, L., Bisevac, L., & Byievac, L. (2007). Ants (Hymenoptera: Formicidae) pass the bioindicator scorecard. *Myrmecological News* 10: 69–76
- Maldonado, A.D., Ramos-López, D., & Aguilera, P.A. (2019). The role of cultural landscapes in the delivery of provisioning ecosystem services in protected areas. *Sustainability (Switzerland)* 11: 1–18. doi: 10.3390/su11092471
- Mantel, N. (1967). The detection of disease clustering and a generalized regression approach. *Cancer Research* 27: 209–220
- Marañón, T., Ibáñez, B., Anaya-Romero, M., Muñoz-Rojas, M., & Pérez-Ramos, I.M. (2012). Oak trees and woodlands providing ecosystem services in Southern Spain. In: *Trees beyond the wood*. Sheffield, pp 369–378
- Martin, M. (1996). La dehesa. *Agricultura* 762: 44–49
- Menke, S.B. & Vachter, N. (2014). A Comparison of the effectiveness of pitfall traps and winkler litter samples for characterization of terrestrial ant (Formicidae) communities in temperate savannas. *The Great Lakes Entomologist* 47: 149–165
- Moreno, C.E., Rojas, G.S., Pineda, E., & Escobar, F. (2007). Shortcuts for biodiversity evaluation: a review of terminology and recommendations for the use of target groups, bioindicators and surrogates. *International Journal of Environment and Health* 1: 71. doi: 10.1504/IJENVH.2007.012225
- Nakamura, A., Catterall, C.P., House, A.P.N.N., Kitching, R.L., & Burwell, C.J. (2007). The use of ants and other soil and litter arthropods as bio-indicators of the impacts of rainforest clearing and subsequent land use. *Journal of Insect Conservation* 11: 177–186. doi: 10.1007/s10841-006-9034-9
- Noss, R.F. (1990). Indicators for monitoring biodiversity: a hierarchical approach. *Conservation Biology* 4: 355–364. doi: 10.1111/j.1523-1739.1990.tb00309.x
- Osborn, F., Goitia, W., Cabrera, M., & Jaffé, K. (1999). Ants, plants and butterflies as diversity indicators: comparisons between strata at six forest sites in Venezuela. *Studies on Neotropical Fauna and Environment* 34: 59–64. doi: 10.1076/snfe.34.3.59.8900
- Pérez-Fuertes, O., García-Tejero, S., Pérez Hidalgo, N., Mateo-Tomás, P., Cuesta-Segura, A.D., & P. Olea, P. (2016). Testing the effectiveness of surrogates for assessing biological diversity of arthropods in cereal agricultural landscapes. *Ecological Indicators* 67: 297–305. doi: 10.1016/j.ecolind.2016.02.041
- Pfeiffer, M., Chimedregzen, L., & Ulykpan, K. (2003). Community organization and species richness of ants (Hymenoptera/Formicidae) in Mongolia along an ecological gradient from steppe to Gobi desert. *Journal of Biogeography* 30: 1921–1935. doi: 10.1046/j.0305-0270.2003.00977.x
- Philpott, S.M. & Armbrrecht, I. (2006). Biodiversity in tropical agroforests and the ecological role of ants and ant diversity in predatory function. *Ecological Entomology* 31: 369–377. doi: 10.1111/j.1365-2311.2006.00793.x
- Philpott, S.M., Perfecto, I., Armbrrecht, I., & Parr, C.L. (2009). Ant diversity and function in disturbed and changing habitats. In: Lach L, Parr CL, Abbott KL (eds) *Ant Ecology*. Oxford University Press: Oxford, pp 137–156
- Plieninger, T., Pulido, F.J., & Schaich, H. (2004). Effects of land-use and landscape structure on holm oak recruitment and regeneration at farm level in *Quercus ilex* L. dehesas. *Journal of Arid Environments* 57: 345–364. doi: 10.1016/S0140-1963(03)00103-4
- Prasifka, J.R., Lopez, M.D., Hellmich, R.L., Lewis, L.C., & Dively, G.P. (2007). Comparison of pitfall traps and litter bags for sampling ground-dwelling arthropods. *Journal of Applied Entomology* 131: 115–120. doi: 10.1111/j.1439-0418.2006.01141.x
- Pulido, F. & Picardo, Á. (2010). Libro verde de la dehesa. Documento para el debate hacia una estrategia ibérica de gestión.
- R Core Team (2019). R: a language and environment for statistical computing. *R Found. Stat. Comput.* 1:1–2630
- Ribas, C.R., Campos, R.B.F., Schmidt, F.A., & Solar, R.R.C. (2012). Ants as indicators in Brazil: a review with suggestions to improve the use of ants in environmental monitoring programs. *Psyche: A Journal of Entomology* 2012: 1–23. doi: 10.1155/2012/636749
- Roig, X. & Espadaler, X. (2010). Propuesta de grupos funcionales de hormigas para la Península Ibérica y Baleares, y su uso como bioindicadores [Proposal of functional groups of ants for the Iberian Peninsula and Balearic Islands, and their use as bioindicators]. *Iberomyrmex* 2: 28–29
- Sackmann, P., Ruggiero, A., Kun, M., & Farji-Brener, A.G. (2006). Efficiency of a rapid assessment of the diversity of ground beetles and ants, in natural and disturbed habitats of the Nahuel Huapi region (NW Patagonia, Argentina).

- Biodiversity & Conservation 15: 2061–2084. doi: 10.1007/s10531-005-2931-5
- San Miguel, A. (1994). *La dehesa española. Origen, tipología, características y gestión.*, 1st edn. Madrid,
- Sauberer, N., Zulka, K.P., Abensperg-Traun, M., Berg, H.M., Bieringer, G., Milasowszky, N., Moser, D., Plutzer, C., Pollheimer, M., Storch, C., Tröstl, R., Zechmeister, H., & Grabherr, G. (2004). Surrogate taxa for biodiversity in agricultural landscapes of eastern Austria. *Biological Conservation* 117: 181–190. doi: 10.1016/S0006-3207(03)00291-X
- Schowalter, T. (2017). Arthropod diversity and functional importance in old-growth forests of north america. *Forests* 8: 97. doi: 10.3390/f8040097
- Sheikh, A.H., Ganaie, G.A., Thomas, M., Bhandari, R., & Rather, Y.A. (2018). Ant pitfall trap sampling: an overview. *Journal of Entomological Research* 42: 421–436. doi: 10.5958/0974-4576.2018.00072.5
- Smith, M.A., Fisher, B.L., & Hebert, P.D.N. (2005). DNA barcoding for effective biodiversity assessment of a hyperdiverse arthropod group: the ants of Madagascar. *Philosophical Transactions of the Royal Society B: Biological Sciences* 360: 1825–1834. doi: 10.1098/rstb.2005.1714
- Smouse, P.E., Long, J.C., & Sokal, R.R. (1986). Multiple regression and correlation Mantel test of matrix correspondence. *Systematic Zoology* 35: 627–632
- StatSoft Inc. (2001). *Statistica*
- Uys, C., Hamer, M., & Slotow, R. (2010). Step process for selecting and testing surrogates and indicators of afrotemperate forest invertebrate diversity. *PLoS ONE* 5: e9100. doi: 10.1371/journal.pone.0009100
- Vasconcelos, H.L., Frizzo, T.L.M., Pacheco, R., Maravalhas, J.B., Camacho, G.P., Carvalho, K.S., Koch, E.B.A., & Pujol-Luz, J.R. (2014). Evaluating sampling sufficiency and the use of surrogates for assessing ant diversity in a Neotropical biodiversity hotspot. *Ecological Indicators* 46: 286–292. doi: 10.1016/j.ecolind.2014.06.036
- Verdinelli, M., Yakhlef, S., Cossu, C., Pilia, O., & Mannu, R. (2017). Variability of ant community composition in cork oak woodlands across the Mediterranean region: implications for forest management. *iForest - Biogeosciences and Forestry*, 10: 707–714. doi: 10.3832/ifor2321-010
- Wang, C., Strazanac, J., & Butler, L. (2001). A comparison of pitfall traps with bait traps for studying leaf litter ant communities. *Journal of Economic Entomology* 94: 761–765. doi: 10.1603/0022-0493-94.3.761
- Wettstein, W. & Schmid, B. (1999). Conservation of arthropod diversity in montane wetlands: effect of altitude, habitat quality and habitat fragmentation on butterflies and grasshoppers. *Journal of Applied Ecology* 36: 363–373. doi: 10.1046/j.1365-2664.1999.00404.x
- Wiens, J.A., Hayward, G.D., Holthausen, R.S., & Wisdom, M.J. (2008). Using surrogate species and groups for conservation planning and management. *BioScience* 58: 241–252. doi: 10.1641/B580310
- Yang, L.H. & Gratton, C. (2014). Insects as drivers of ecosystem processes. *Current Opinion in Insect Science* 2: 26–32. doi: 10.1016/j.cois.2014.06.004

