Variation in winter thrush abundance during the hunting season in southern Europe: the importance of hunting–free reserves

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Abstract

Variation in winter thrush abundance during the hunting season in southern Europe: the importance of hunting–free reserves. We analysed variations in the abundance of the song thrush (*Turdus philomelos*) and the blackbird (*T. merula*) in the hunting season in hunting areas and hunting–free reserves. After controlling for habitat, we found that the abundance of song thrushes (hunted species) was lower in hunting areas than in reserves during the hunting season. This effect was not found for the blackbird (non–hunted species). This finding indicates hunting–free reserves have a positive effect on song thrush conservation. Further research is crucial to determine the traits that should be promoted in this type of reserve in order to improve their efficiency.

Key words: Avian conservation, Hunting, Game birds, Wildlife management

Resumen

Variación de la abundancia del zorzal en invierno durante la temporada de caza en el sur de Europa: la importancia de las reservas. Analizamos las variaciones de la abundancia del zorzal común (*Turdus philomelos*) y el mirlo (*T. merula*) durante la temporada de caza en reservas y en zonas de caza. Una vez controlado el efecto del hábitat, la abundancia del zorzal común (especie cinegética) fue inferior en las zonas de caza que en las reservas. Este efecto no se observó en el mirlo (especie no cinegética). Este resultado indica que las reservas de caza tienen un efecto positivo en la conservación del zorzal común. Es fundamental seguir trabajando para determinar los factores que se deberían promover en este tipo de reservas para mejorar su eficiencia.

Palabras clave: Conservación de aves, Caza, Especies cinegéticas, Gestión de la fauna silvestre

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Introduction

Southern European states are the main wintering areas for large numbers of European breeding birds (Andreotti et al., 1999; Tellería et al., 1999), including species of thrush (Snow and Perrins, 1998; Rivalan et al., 2007). These southern states, therefore, have a high responsibility in the conservation of many European bird populations. Hunting activity and its associated management have obvious effects on the conservation of biodiversity, leading to controversy across several spatial scales and between different sectors of our society (McCulloch et al., 1992; Madsen, 1998a; Sokos et al., 2013; Caro et al., 2014; Hirschfeld and Attard, 2017; Prieto et al., 2019). To increase the protection and conservation of game species it is essential to understand the impact of hunting management tools in order to implement 'good practice' standards (Madsen, 1998a; Guillemain et al., 2002; Hirschfeld and Attard, 2017). Many tools are used to reduce the impact of hunting. Such approaches include bag restriction (Schroeder et al., 2014), shortening hunting periods (Brochet et al., 2009), and the creation of hunting–free reserves. This latter option is considered a chief tools for preservation of game bird populations. The response of wintering birds to these reserves has been investigated especially in waterbirds (e.g. Fox and Madsen, 1997; Madsen 1998b; Guillemain et al., 2002; Brochet et al., 2009; Casazza et al., 2012; Beatty et al., 2014). Most of these studies found a positive effect of reserves on the wintering birds that concentrated in these areas during hunting days (Fox and Madsen 1997; Madsen 1998b; Guillemain et al., 2002; Casazza et al., 2012; Beatty et al., 2014). However, as not all waterfowl species show the same response (Madsen 1998a, 1998b; Guillemain et al., 2002) the role of game reserves may differ even among similar bird species. In the case of land birds, fewer studies have dealt with this issue (Duriez et al., 2005a; Casas et al., 2009; Brøseth and Pedersen, 2010) and we are not aware of any studies in thrushes.

The hunting of migrant thrushes (*Turdus* spp.) remains a popular practice in Mediterranean countries where important stopover/wintering areas for these species are located (e.g., Santos and Tellería, 1985). Overall, ca. 15 millions of thrushes are hunted yearly in Europe (Myrberget, 1990), mostly in France (around 2 million; Aubry et al., 2016), Italy (around 7 million; Andreotti et al., 2010) and Spain (more than 4 million; Hirschfeld and Attard, 2017). The effects of hunting on thrush population dynamics is poorly known (Sokos et al., 2013). Some studies have shown that the annual adult survival rates of song thrushes were correlated with hunting pressure (Aebischer et al., 1999), while others failed to demonstrate such an effect (Payevsky and Vysotsky, 2003).

In this context, it is unclear whether management tools such as the creation of hunting–free reserves within the winter quarters may benefit the conservation status of thrushes. To address this issue, we analysed abundance patterns in hunting areas and hunting–free reserves of two phylogenetically related thrush species, the song thrush (*T. philomelos*) and the blackbird (*T. merula*) in a main wintering area of northern Spain. Both species share post–nuptial migratory phenology, that finishes in November (Aparicio, 2016; Purroy and Purroy, 2016), winter space use (Aparicio, 2016; Purroy and Purroy, 2016), and diet (Guitián, 1985; Soler et al., 1991; Paralikidis et al., 2009). However, the song thrush is a game bird within this region, while the blackbird is not. Therefore, we hypothesized that, after controlling for factors such as habitat type, if birds react to the existence of the reserve, abundance should contrast between areas, especially in the game species.

Methods

The study was carried out in the Alhama River basin, Navarra, northern Spain (42º 3' 17.80'' N to 42º 10' 55.70'' N, 1º 53' 39.79'' W to 1º 43' 53.09'' W; fig. 1). The river bank consists of riparian forest comprising mainly ash trees *Fraxinus* spp., poplars *Populus alba*, tamarisks *Tamarix* spp., and shrubs such as brambles *Rubus* spp., rosebush *Rosa* spp. and hawthorns *Crataegus monogyna*. Contiguous to this forest and more distant from the river there is an agricultural matrix comprising olive trees, fruit orchard, vineyards, cereal crops and small farms. The zone is thus an attractive stopover and wintering site for many migratory passerines, further promoted by the zone's location near the Pyrenees and the southeastern edge of the Bay of Biscay (Galarza and Tellería, 2003; Carrascal and Díaz, 2006).

All thrush species except the blackbird and the ring ouzel (*T. torquatus*) can be hunted in Navarra from November to January, usually from fixed hunting sites. Overall, ca. 30,000 thrushes are shot every year in Navarra (source: Government of Navarra), which means relatively high hunting pressure in a region of ca. 10,400 km^2. Navarra's hunting laws require the creation of hunting–free reserves. These areas must cover at least 12 % of the entire hunting surface within each regional county, and they must also comprise those habitats which have a higher importance for the conservation of game species. Reserves created for thrush species usually include olive groves and vineyards or riparian forests, which correspond to habitats selected by these species in winter (Soler et al., 1988).

In this study we carried out censuses immediately prior to the hunting season in 2018 (in October) and one month after the hunting season began (in December). The point–count method (Blondel et al., 1970) was used. This method consists of counting all the birds seen or heard within a radius of 25 m from the observer's position over a period of 10 min. We considered this radius in order to elude the possible negative effect of vegetation structure on bird detectability (Pacifici et al., 2008). To avoid observer–associated bias we performed all censuses were performed by the same observer (DV) (Diefenbach et al., 2003), from +1 h from dawn to –1 h before dusk. Twilight periods were then avoided to exclude the variations in birds' song activity throughout the day (Robbins, 1981) and the influence

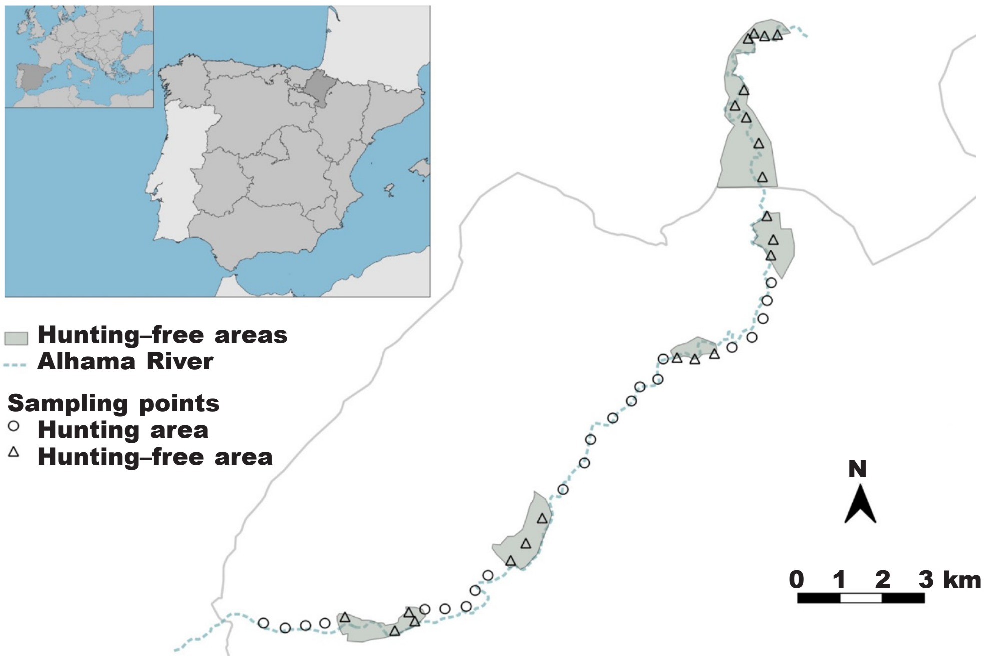

Fig. 1. Map showing the location of the study area, the distribution of the reserves, and bird count points.

Fig. 1. Mapa de la localización de la zona de estudio en el que se muestra la distribución de las reservas y los puntos de conteo de las aves.

of typical movements from or to communal roosting places –rather common within the region– which would result in an over–estimation of bird counts. We considered a total of 44 sampling points, with 22 being situated in hunting–free reserves, and 22 in hunting areas (fig. 1). All sampling points were situated at a distance of > 500 m from each other in order to reduce double–counts (Sutherland et al., 2004).

Within each sampling point, the habitat was characterized within a radius of 25 m by calculating the proportion of nine habitats: riparian native forest, tamarisk, olive grove, fruit trees, orchard, vineyard, shrub, cereal crop, and farms (building) using GIS tools. The initial distribution of habitats was obtained from the Spanish map of crops and territory use, provided by the Ministry of Agriculture (source: www.mapa.gob.es) for 2000–2010. During the sampling work, the data provided by such maps were updated.

Statistical analyses

Many habitat–related variables correlated with each other as they were calculated as a percentage over the area comprised within a buffer of 25 m–radius around each sampling unit (point–count) (for details see appendix 1). Thus, before starting to build a model we conducted a principal component analysis (PCA) on such variables in order to obtain a number of new variables that summarized habitat structure. The PCA provided five components (hereafter, PC1 to PC5) with an eigenvalue > 1, which, overall, accounted for 77.6 % of the variance (table 1): the PC1 correlated positively with natural habitat (riparian forest, shrubs) and negatively with croplands (groves, vineyards, and similar); PC2 correlated negatively with shrubs, cereals and olive groves; PC3 correlated positively with the presence of tamarisk and negatively with orchards and farms; PC4 correlated positively with vineyards and, to a lesser extent, with cereals; and PC5 correlated negatively with farms.

To test for the effect of hunting on bird counts we used generalized linear mixed models with bird counts as an object (dependent) variable, and the following explanatory variables: hunting regimen (hunting vs. reserve), period (October vs. December), a hunting regimen–period interaction, PC1 to PC5, and the sampling site, as random factor. Our target thrushes here were the song thrush and the blackbird. In this GLMM we considered a distribution of Poisson errors, with a log–linear link function. Using the 'lmer' package in R (Kuznetsova et al., 2017), we conducted a global starting model which included all the explanatory terms listed above (R notation): counts~hunting regimen × period + PC1 + PC2 + PC3 + PC4 + PC5 + (1|sampling site).

Between alternative models, those with lower Akaike (AIC) values were considered to have a better fit with the data (Akaike, 2011). A model would have a better fit with the data if it had an Akaike value lower than 2 as compared to a second model (Burnham and Anderson, 1998). We used the 'dredge' function for the model selection procedure (Barton, 2014). This

Table 1. Factor loadings obtained from a principal component analysis on nine habitat–related variables. We show here only those components that had eigenvalues > 1 (PC1 to PC5).

Tabla 1. Cargas factoriales obtenidas a partir de un análisis de componentes principales en nueve variables relacionadas con el hábitat. Solo se muestran los componentes con valores propios > 1 (de PC1 a PC5).

Factor	PC1	PC2	PC3	PC4	PC5
Riparian forest	+0.58	+0.28	+0.20	–0.06	+0.13
Tamarisk	–0.01	+0.17	+0.59	–0.17	–0.06
Olive grove	–0.45	–0.35	+0.13	–0.26	–0.23
Fruit trees	–0.52	+0.25	+0.06	–0.35	+0.21
Vineyard	–0.24	+0.11	+0.11	+0.84	–0.05
Orchards	–0.16	+0.21	–0.62	–0.01	+0.42
Farms	+0.13	+0.13	–0.42	–0.13	–0.79
Shrubs	+0.32	–0.50	–0.12	–0.16	+0.28
Cereal crop	–0.01	–0.62	+0.01	+0.18	–0.01
Eigenvalue	1.91	1.56	1.36	1.15	1.01
Variance (%)	21.2	17.3	15.0	12.8	11.3

function runs all the possible combinations starting from the global model described above, and then list all of them according to their AIC value. Model averaging was carried out when more than a single model fitted the data equally well; for this the function we used 'model.avg' on the 'dredge' object.

We tested for the potential existence of spatial autocorrelation between bird counts and the location of our sampling points. With this goal we used the function 'correlogram' in R, built using a distance matrix for the sampling points and the residual values of the model which best fitted the data. Overall,

Table 2. List of the top–ranked models (those differing in less than 2 AICc units as compared to the first one) used to test for the effect of hunting–free reserves on song thrush and blackbird abundance. For comparison, we also included null models.

Tabla 2. Lista de los modelos mejor clasificados (los que difieren en menos de 2 unidades de AICc respecto del primero) utilizados para determinar el efecto de las reservas en la abundancia del zorzal y el mirlo. A efectos de comparación, también se han incluido modelos nulos.

	AICc	ΔAICc	AICc weight
Song thrush			
Model 1: PC3 + PC5 + hunt × peri	572.8	0.0	0.230
Model 2: PC3 + PC4 + PC5 + hunt × peri	574.1	1.3	0.117
Model 3: PC5 + hunt × peri	574.8	2.0	0.085
Null	722.6	149.8	0.000
Blackbird			
Model 1: PC1 + PC3	259.0	0.0	0.094
Model 2: PC3	259.1	0.1	0.089
Model 3: PC1 + PC3 + peri	260.6	1.6	0.042
Model 4: PC1 + PC3 + PC4	260.7	1.7	0.040
Model 5: PC3 + peri	260.7	1.7	0.040
Model 6: PC3 + PC4	260.9	1.9	0.036
Model 7: PC3 + hunt	261.0	2.0	0.034
Null	266.2	7.2	0.000

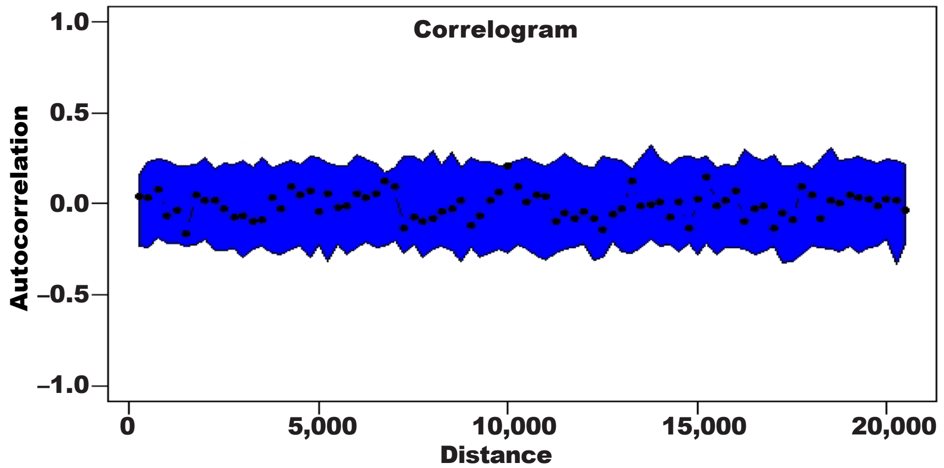

Fig. 2. Graphic representation of the spatial autocorrelation between bird count points.

Fig. 2. Representación gráfica de la autocorrelación espacial entre los puntos de conteo de las aves.

autocorrelation values for all distances were low, indicating a lack of spatial autocorrelation (fig. 2).

All statistical analyses were done in R (R Core Team, 2014).

Results

We counted a total of 718 song thrushes in hunting–free areas (refuges) and the hunting area (495 and

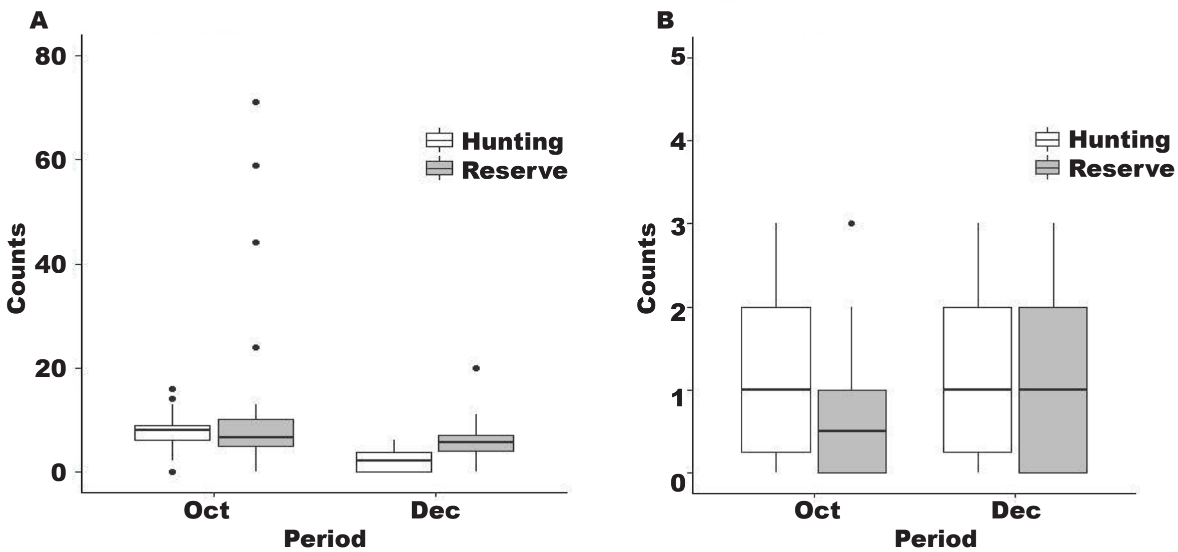

Fig. 3. Number of song thrushes (A) and blackbirds (B) detected in hunting and hunting–free reserve sampling sites before (Oct) or during (Dec) the hunting season. Horizontal lines denote median values; boxes extend from 25th to 75th percentile and the whiskers extend up to 1.5 times the interquartile range. Any data beyond that distance are represented individually as points ('outliers').

Fig. 3. Número de zorzales (A) y mirlos (B) detectados en sitios de muestreo ubicados en zonas de caza y en reservas antes (Oct, octubre) del inicio de la temporada de caza o durante (Dec, diciembre) la misma. Las líneas horizontales denotan los valores medianos, las cajas se extienden desde el percentil 25 hasta el 75 y los bigotes, hasta 1,5 veces el intervalo intercuartílico. Los datos situados más allá de esa distancia se representan individualmente como puntos (valores atípicos).

Table 3. Beta–parameter estimates of the averaged best–ranked models used to test for the effect of several factors on counts of song thrushes or blackbirds: [1] reference values (*Beta* = 0): hunt, hunting regimen; period, December.

Tabla 3. Estimaciones del parámetro beta de los modelos mejor clasificados de media y que se emplearon para determinar el efecto de varios factores en los conteos de zorzales y mirlos: [1] valores de referencia (Beta = 0); hunt, régimen de caza; período, diciembre.

Factor	Beta	SE (Beta)	*P*
Song thrush			
Hunt[1]:reserve	+0.95	0.25	< 0.001
Period[1]:Oct.	+1.24	0.16	< 0.001
Hunt–Period	–0.45	0.19	0.018
PC3	+0.20	0.09	0.035
PC4	–0.03	0.07	0.651
PC5	+0.34	0.11	0.002
Blackbird			
Period[1]:Oct.	–0.04	0.12	0.755
PC1	–0.07	0.09	0.454
PC3	+0.34	0.11	0.004
PC4	+0.02	0.06	0.781

223 individuals, respectively), and 106 blackbirds (47 and 59 individuals, respectively).

After our selection procedure, we obtained two models that fitted the song thrush data equally well (table 2). Both models included an effect of the hunting regime, period, the interaction between these two factors, and the PC3 and PC5. The second model also included an effect of PC4, although the parameter estimates of the averaged model showed that this effect was non–significant (table 3). Overall, song thrushes were more abundant in reserves in October than in December, in places richer in riparian habitats (riparian forest, tamarisk; PC3), and in orchards, shrubs and fruit crops and with a low proportion of farms (PC5). However, the hunting regime–period interaction showed that the number of song thrushes remained almost constant between periods in hunting–free reserve areas, and that abundance in both types of areas was similar in October (that is before the hunting season), whereas abundance was found to decrease notably in December in hunting areas (fig. 3).

For blackbirds, we obtained six models that fitted the data equally well. All of these models included an effect of PC3 on bird counts, with some models also including an effect of PC1 (3 models), PC4 (2 models), and period (2 models) (table 2). Interestingly, none of these models showed an effect of hunting regime on bird counts. The averaged model included a significant, positive effect of PC3 on bird counts (table 3), showing that blackbirds tended to be more abundant at survey points richer in riparian forest or tamarisk.

Discussion

Our results show that hunting activity in Navarra had an important effect on the abundance of song thrushes that over–wintered in an area comprising a mosaic of riparian forest and agricultural landscapes. Once the hunting season had started, song thrush abundance decreased in hunting areas but not in hunting–free reserves. Such effects were not found for the non–hunted thrush, the blackbird. The decrease in song thrushes in hunting sites could respond to several causes, either complementary or alternative: first, mortality by hunting in hunting areas could result in lower bird numbers across the season (e.g., Duriez et al., 2005a; Prieto et al., 2019); second, hunting disturbance to song thrushes may promote the abandonment of places where they were hunted in favor of those where hunting was not allowed (hunting–free reserve places) (e.g., Evans and Day, 2002; Bechet et al., 2004; Brochet et al., 2009; Casas et al., 2009; Casazza et al., 2012; Garaita and Arizaga, 2015). However, data in figure 2 show that the number of song thrushes did not increase in the hunting–free reserve sites in December, thus not supporting the idea that song thrushes displaced from hunting to hunting–free areas across the season. Rather, our results may be more compatible with the idea that higher mortality causes a population decline in hunting places, but our data are insufficient to demonstrate this.Disturbed song thrushes may also leave the region and move to other areas outside the geographical range considered in this work. To confirm the exact mechanism driving abundance changes, it would be necessary to conduct studies based on individual marking (Salewski et al., 2007) or tracking e.g. using radio–telemetry (Bechet et al., 2004; Duriez et al., 2005b; Brøseth and Pedersen, 2010; Beatty et al., 2014).

Despite these shortcomings, the use of a hunting–free reserve is a critical management tool to contribute to the conservation of song thrush populations within the region, since song–thrushes did not decrease in numbers in hunting–free reserves in December, when the hunting season was about to end. A second question for future research will be to determine whether the current surface area of hunting–free reserves is sufficient to compensate for the apparently lower survival rates in hunting estates (an additional aspect to be investigated). Among other factors, the design of hunting–free reserves should consider all the ecological requirements of target species, and primary foraging and roosting places (e.g., Guillemain et al., 2002; Beatty et al., 2014).

In conclusion, we found hunting–free reserves had a positive impact on song thrush abundance at a local scale. The detected decline in abundance in hunting areas, however, could be due to various causes needing further research to discriminate between the factors potentially shaping this decline (mortality or emigration). Further studies will also help to determine which traits (e.g. size, habitat, connectivity with protected areas) should be promoted in this type of reserve (Fox and Madsen, 1997; Madsen, 1998a; Brochet et al., 2009) in order to improve their role as a refuge for thrushes and other game species.

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Appendix 1. Correlation matrix of habitat–related variables: upper triangle, Pearson *r* values; lower triangle, *P*–values (significance, $P < 0.05$): RIPA, riparian forest; TAMA, tamarisk; OLIV, olive grove; FRUI, fruit trees; VINE, vineyard; ORCH, orchards; FARM, farms; SHRU, shrubs; CERE, cereal crop.

Apéndice 1. Matriz de correlación de las variables relacionadas con el hábitat: triángulo superior, valores de la r de Pearson; triángulo inferior, valores de P (significación, P < 0,05): RIPA, bosque ribereño; TAMA, tamarisco; OLIV, olivar; FRUI, árboles frutales; VINE, viñedos; ORCH, huertos; FARM, explotaciones agrícolas; SHRU, arbustos; CERE, cultivo de cereales.

	RIPA	TAMA	OLIV	FRUI	VINE	ORCH	FARM	SHRU	CERE
RIPA		+0.00	–0.50	–0.39	–0.30	–0.25	–0.08	–0.02	–0.26
TAMA	0.997		–0.08	+0.08	–0.04	–0.25	–0.13	–0.12	–0.09
OLIV	< 0.001	0.442		+0.22	–0.08	–0.18	–0.15	–0.11	+0.13
FRUI	< 0.001	0.413	0.034		–0.05	+0.15	–0.18	–0.35	–0.22
VINE	0.003	0.674	0.438	0.608		–0.05	–0.17	–0.30	–0.03
ORCH	0.014	0.014	0.077	0.158	0.636		+0.03	–0.09	–0.13
FARM	0.424	0.214	0.153	0.081	0.105	0.765		–0.09	–0.12
SHRU	0.8112	0.234	0.286	< 0.001	0.003	0.361	0.365		+0.22
CERE	0.012	0.373	0.192	0.034	0.765	0.195	0.241	0.031	