Basic and Applied Ecology 7 (2006) 483–495

Available online at www.sciencedirect.com

ScienceDirect

Basic and Applied Ecology

www.elsevier.de/baae

Evaluating the benefits of CAP reforms: Can afforestations restore bird diversity in Mediterranean Spain?

Tomás Santos[a,*], José L. Tellería[a], Mario Díaz[b], Roberto Carbonell[c]

[a]Departamento de Zoología y Antropología Física, Facultad de Biología, Universidad Complutense, E-28040 Madrid, Spain
[b]Departamento de Ciencias Ambientales, Facultad de Ciencias del Medio Ambiente, Universidad de Castilla-La Mancha, E-45071 Toledo, Spain
[c]Servicio de Espacios Naturales, Consejería de Medio Ambiente, Junta de Castilla y León, E-47014 Valladolid, Spain

Received 30 March 2005; accepted 23 November 2005

KEYWORDS
CAP subsidies;
Conservation value;
Farming landscapes;
Forest bird richness;
Holm oak woodlands;
Pine afforestations;
Mediterranean *Sylvia* warblers

Summary

Current Common Agricultural Policy (CAP) subsidies in the Mediterranean region tend to prioritize afforestation on former arable land with oaks rather than pines because pine plantations would maintain lower biological diversities than native forests. Nevertheless, no thorough evaluations of the conservation values of pine plantations as compared to oak remnants have been carried out to date. We analyze the diversity and conservation value of bird assemblages breeding in 200 remnants of Holm oak *Quercus ilex* woodlands and 82 mature (>50-year-old) pine plantations in central Spain, a Mediterranean region mostly devoted to arable farming. Species–area relationships were compared between forest types. The conservation value of bird assemblages was assessed using the "Species of European Conservation Concern" (SPEC) classification of Burfield and van Bommel [(2004). Birds in Europe: Populations estimates, trends and conservation status. Cambridge: BirdLife International]. Overall numbers of bird species maintained by oak and pine archipelagoes were rather similar, but species–area relationships differed between forest types. Intercepts were higher in oak fragments, whereas slopes were steeper in pine plantations. Small oak fragments held more species (mainly Mediterranean *Sylvia* warblers) than plantations, whereas large plantations held more species than large oak remnants. Differences in species–area relationships seemed to be due to differences in vegetation structure, especially understorey shrub cover and tree height and cover. We recorded nine SPECs, all exclusive (6) or near-exclusive (3) to oak woodlands, although such woodlands do not appear to be critical for their conservation. Hence, we conclude that pine afforestations have played a role for

*Corresponding author. Tel.: +34 91 394 49 49; fax: +34 91 394 47.
E-mail address: tsantos@bio.ucm.es (T. Santos).

1439-1791/$ - see front matter
doi:10.1016/j.baae.2005.11.001

maintaining and restoring forest bird communities in the farming landscapes of central Spain. Promoting large and shrubby plantations would enhance their conservation value for breeding birds, together with promoting growth, regeneration and expansion of Holm oak remnants by means of set-aside measures previous or alternative to oak reafforestation. The increasing importance of non-commercial as compared to commercial values of Mediterranean forests would justify subsidizing the proposed policy.

Zusammenfassung

Derzeitige CAP-Fördermittel (Common Agricultural Policy) im mediterranen Raum setzen ihre Prioritäten bei der Aufforstung von ehemalig bebautem Land auf Eichen anstelle von Kiefern weil Kieferanpflanzungen eine geringere biologische Diversität erhalten als bodenständige Wälder. Dennoch wurden bis heute keine sorgfältigen Evaluationen des Naturschutzwertes von Kieferanpflanzungen im Vergleich zu Eichenrestbeständen durchgeführt. Wir analysierten die Diversität und den Naturschutzwert von Vogelgesellschaften, die in 200 Restbeständen von Wäldern der Steineiche *Quercus ilex* und in 82 reifen ($>$50 Jahre alten) Kiefernanpflanzungen in Zentralspanien brüten, einer mediterranen Region, die hauptsächlich von Ackerbau geprägt ist. Die Art-Areal-Beziehungen wurden zwischen den Waldtypen verglichen. Der Naturschutzwert der Vogelgesellschaften wurde unter Verwendung der „Species of European Conservation Concern'' (SPEC) Klassifikation nach [Burfield and van Bommel (2004) Birds in Europe: Populations estimates, trends and conservation status. Cambridge: BirdLife International] abgeschätzt. Die gesamte Anzahl der Vogelarten, die sich in den Eichen- und Kiefernarchipelen hielten, war ziemlich ähnlich, aber die Art-Areal-Beziehung zwischen den Waldtypen unterschied sich. Die Achsenabschnitte lagen in den Eichenfragmenten höher, während die Steigungen in den Kieferanpflanzungen steiler waren. Kleine Eichenfragmente enthielten mehr Arten (vor allem die mediterranen *Sylvia*-Laubsänger) als die Anpflanzungen, während große Anpflanzungen mehr Arten enthielten als große Eichenrestbestände. Die Unterschiede schienen sich auf Unterschiede in der Vegetationsstruktur zurückführen zu lassen, besonders auf die Unterholzdeckung und die Baumhöhe und –deckung. Wir erfassten neun SPECs, alle ausschließlich (6) oder nahezu ausschließlich (3) in Eichenwäldern, auch wenn diese Wälder für ihren Schutz nicht entscheidend zu sein scheinen. Deshalb haben wir den Schluss gezogen, dass die Aufforstungen mit Kiefern eine Rolle bei der Erhaltung und Wiederherstellung der Waldvogelgesellschaften in den Agrarländern Zentralspaniens gespielt haben. Die Förderung von großen und buschigen Anpflanzungen würde zusammen mit der Förderung des Wachstums, der Regeneration und der Ausdehnung der Steineichenrestbestände durch Maßnahmen der Flächenstilllegung, vor der oder alternativ zur Aufforstung mit Eichen, den Naturschutzwert für brütende Vögel verbessern. Die zunehmende Wichtigkeit des nichtkommerziellen im Vergleich zum kommerziellen Wert der mediterranen Wälder würde die Förderung der vorgeschlagenen Politik rechtfertigen.

Introduction

Afforestation programs have been widely used in most European countries as a way of recovering the woodlands lost by the expansion of agricultural and grazing uses, especially in the Mediterranean region, where climatic conditions prevent rapid forest growth (Shochat, Abramsky, & Pinshow, 2001; Tellería, 1992). Public funding to afforestations until they reach economic viability has been justified on the basis of its potential role for restoring ancient forest habitats and their associated communities (Brotons & Herrando, 2001; Díaz, Carbonell, Santos, & Tellería, 1998; López & Moro, 1997; Tellería, 1992). Nevertheless, the fact that the main final use of afforestations is timber production has lead to inconsistencies with this environmental goal (Shochat et al., 2001; Tellería, 1992). So, in Spain pines (*Pinus* spp.) are preferred to the slow-growing native trees (mainly oaks; *Quercus* spp.) because of their faster growing rates and their straightforward establishment, and plantation and management techniques are mostly directed to increase stand density and reduce plant competition with the growing trees (Barbero, González, & Catalán, 1994).

Pine plantations are expected to maintain less bird species than the original forests they are supposed to replace, since diversity of forest communities is closely linked to floristic composition and habitat heterogeneity (see Wiens, 1989, for forest birds). In fact, several papers have shown that exotic plantations maintain lower bird diversities than native woodlands (Berg, 1997; Rodewald & Abrams, 2002; Shochat et al., 2001) and that loss of structural heterogeneity and plant diversity in plantations reduces the richness of bird species (Díaz et al., 1998; Shochat et al., 2001). Hence, a simple way to increase the conservation value of afforestations is the implementation of specific management measures favouring the restoration of native assemblages (see Brotons & Herrando, 2001; Díaz et al., 1998; López & Moro, 1997; Shochat et al., 2001, for birds in pine afforestations in the Mediterranean region). In the European Union, higher subsidies to reafforestation with native oaks than to pine plantations on former arable land under the measures accompanying the Common Agricultural Policy (CAP) reform (Regulation EEC no. 2080/92) is a clear step in this direction, although no evaluations of the environmental benefits of oak as compared to pine plantations are yet available (Pulido, Campos, & Montero, 2002).

The spatial arrangement of afforestations within landscapes is a key factor for maintaining diverse forest bird communities (Berg, 1997; Brotons & Herrando, 2001; Díaz et al., 1998; Shochat et al., 2001). In Spain, extensive pine plantations were mostly established on deforested hilly grounds and on soil devoted previously to subsistence farming (ICONA, 1979; Tellería, 1992). Yet, many plantations were made in flatland farming landscapes of central Spain, mainly on former cereal patches, giving rise to an "inverse" fragmentation process of the earlier forest habitat: the appearance and increase of forest fragments within a farmed matrix (Santos & Tellería, 1998). Bird species richness and the incidence of most species in mature (more than 50-year-old) pine afforestations depend mostly on the size of plantations, with secondary effects of distance from large forest patches, vegetation structure and geographical location in a north-south gradient (Díaz et al., 1998). Since most plantations are very small (less than 2 ha in size) due to the size distribution of landholdings, we concluded that pine afforestations seem to have little potential of maintaining rich forest bird communities in Mediterranean Spain (Díaz et al., 1998). On the other hand, native Holm oak *Quercus ilex* forest remnants still comprise a small fraction of these flatland landscapes, in spite of its fragmented distribution and poor development due to a variety of human impacts (Santos, Tellería, & Carbonell, 2002). Richness and incidence of birds in these native forests also depend on the size of remnants and on geographical location in a north-south gradient, with smaller effects of isolation and vegetation structure (Santos et al., 2002).

Explicit comparisons of the effect of size of pine afforestations and native forest remnants on bird communities would help to anticipate the environmental benefits of reafforestation policies covered by the reformed CAP. Specifically, the main aims of this study are to test (1) whether species–area relationships differ between remnants of the original forest vegetation (Holm oak archipelagoes) and pine plantations in Mediterranean Spain and (2) if there are differences in the conservation value of bird species exclusive to oak and pine bird assemblages. On these grounds, specific recommendations could be made regarding the features of reafforestations that would yield the largest conservation values, both in terms of diversity and in terms of species of European concern (Burfield & van Bommel, 2004).

Materials and methods

Study area

Landscape, vegetation and climatic traits of the study areas are thoroughly described in Díaz et al. (1998), Santos and Tellería (1998) and Santos et al. (2002), and summarized in Table 1. All oak forests and pine plantations studied are located in the plateaux of central Spain, an extensive treeless agricultural region characterized by a continental Mediterranean climate. The east–west mountain range of the Sistema Central divides the region in two areas (north and south) well differentiated by their climate and forest growth conditions (Rivas-Martínez, 1981). The northern plateau is cooler and wetter and shows more diverse and developed woodlands than the southern one (Santos et al., 2002). These differences are associated to richer bird pools and more diverse avian communities in northern woodlands (Díaz et al., 1998; Santos et al., 2002).

Sampling

The study was carried out in 200 Holm oak fragments and 82 pine plantations more than 50 years old, which covered a wide range of sizes,

Table 1. Landscape and vegetation traits of the study areas

Woodland type	Plateau	*n*	Elevation (m a.s.l.)	Regional cover of forest (%)	Tree height (m)	Tree cover (%)	Shrub cover (%)
Holm oak	Northern	122	750–900	7.6	4.7	47.9	18.5
fragments	Southern	78	700–800	7.2	5	37.4	18.3
Pine	Northern	37	770–790	<15	8.9	80.6	1.1
plantations	Southern	45	710–730	<15	8.1	75.7	13.5

N is the sample size for either fragments (Holm oak *Quercus ilex* forests) or plantations (mostly *Pinus pinaster*, with *P. pinea* in the northern plateau and *P. halepensis* in the southern plateau as secondary species). Mean vegetation values were taken from Tellería and Santos (1999) for oak fragments and from Díaz et al. (1998) for pine plantations. Details on methods for estimating tree height, tree cover and shrub cover can be found in Santos et al. (2002).

from tiny tracts (<0.1 ha) to woodlands extending over 100 ha. An additional sample of 18 larger forests was considered for some analyses, although they were excluded from the main comparisons due to their disparity in sizes (from 107 to 2450 ha for 12 oak woodlands, and 137–6775 ha for six plantations) that could have affected strongly species–area functions. Bird and forest variables were measured during the spring and summer of 1994 (for details on methods and rationale, see Díaz et al., 1998 and Santos et al., 2002). Presence–absence of breeding bird species was established by applying a sampling effort logarithmically proportional to forest size. No attempt was made to estimate densities because of the problems associated with density estimates in very small areas. Species with large territory sizes (raptors, owls and some other non-passerines) were excluded, since most fragments and plantations were too small to maintain even a single breeding pair. The 60 bird species found were classified into three ecological groups according to their dependence on forest habitats for nesting and feeding: ubiquitous (U), species that can also forage and nest outside forest patches, in isolated trees, shrubs in set-asides, etc.; forest generalists (F), forest breeders that usually forage on the ground, frequently in the nearby farming matrix; and forest specialists (FF), restricted to wooded patches both for nesting and foraging (Díaz et al., 1998; Santos et al., 2002).

Analyses

Differences in species–area relationships between woodland types (Holm oak fragments vs. pine plantations) were examined by means of regression analyses for each plateau (northern and southern). Regressions of bird species richness (*S*) on patch size were estimated for all bird species (ST) and for each bird group (SU, SF and SFF). Slopes and *Y*-intercepts of regression models were compared between woodland types for each bird group and plateau (four groups $\times$ two plateaux = eight comparisons) by means of ANCOVAs with forest area (log-transformed) as the covariate, woodland type as the factor and species richness as the dependent variable. Differences in slopes between fragments and plantations were tested by means of the test of parallelism (factor $\times$ covariate interaction; StatSoft, 1999). If the interaction was not significant, significant factor effects indicated differences in *Y*-intercepts ((i.e., average number of bird species for woodlots of 1 ha, as $\log_{10}(1) = 0$; Sokal & Rohlf, 1981). Significant factor $\times$ covariate interactions would indicate differences in slopes, *Y*-intercepts, or both. Hence, if the interaction was significant, we analyzed whether there were significant differences in *Y*-intercepts by means of a specific test (Zar, 1984).

Conservation values of bird assemblages were established according to European criteria, using the Species of European Conservation Concern (SPEC) classification of Burfield and van Bommel (2004), and were compared between oak fragments and pine plantations. For this comparison, we only considered birds exclusive to one woodland type and that were present in more than 7% of the forests sampled (hereafter, "frequent-exclusive" species). These species might potentially maintain stable populations through a wide range of habitat patches of one woodland type although they did not occupy the other. Additionally, we also considered some bird species "near-exclusive" to a woodland type, that were widespread (>7% patches occupied) in one type but also entered a few patches (<7%) of the other.

Results

Species–area relationships

All species–area regression models were significant ($P<0.001$) and explained from 40% (SF in the oak fragments of southern plateau) to 80% (ST in the oak fragments of northern plateau) of the variance in bird species richness. Models showed two remarkable features according to forest type: (1) intercepts were higher in Holm oak fragments than in pine plantations, with the only exception of SU in the northern plateau; and (2) slopes were steeper for pine plantations, with the only exception of SFF in the southern plateau (Table 2, Fig. 1). Hence, bird assemblages tended to be more diverse in oak fragments than in pine plantations for small habitat patches (1–2 ha), but the reverse was true for large patches (50–100 ha).

Variation in the richness of ubiquitous and forest birds with patch size

Relative species richness of the two main bird groups (ubiquitous and forest species) varied consistently with patch size. The accumulated proportion of ubiquitous species decreased with patch size whereas that of forest species increased, with the only exception of southern oak woodlands, in which no trend was evident (Fig. 2). The increased proportion of forest birds tended to be due to increased proportions of forest specialists in the case of pine plantations and by forest generalists in oak woodlands. These trends were extended towards the larger woodlands and plantations (over 100 ha), where the highest proportions of forest birds and the lowest proportions of ubiquitous species were recorded (Fig. 2).

Species pools and conservation value

Overall, 60 bird species were recorded (Table 3). Distribution of species among ecological groups did not differ significantly between oak and pine forests (Table 4). Fifty-five species were recorded in the oak fragments and only 43 in the plantations, although this difference is attributable to the larger number of oak fragments sampled (see Methods and species–area relationships). In fact, the bird assemblages of large forests ($>$100 ha; see

Table 2. *Y*-intercepts (*a*) and slopes (*b*) for species–area relationship (species richness against $\log_{10}$ area) in Holm oak fragments and pine plantations for all bird species (ST), ubiquitous species (SU), forest generalists (SF) and forest specialists (SFF)

		a^a	t	b^b	t	R	100 ha
Northern plateau							
ST	Oak fragments	6.19	21.45*	7.60	23.57*	0.907*	21.4
	Pine plantations	4.85	6.32*	11.14	10.62*	0.874*	27.1
SU	Oak fragments	2.84	14.92*	3.60	16.93*	0.840*	10.1
	Pine plantations	2.91	7.89*	4.72	9.36*	0.850*	12.3
SF	Oak fragments	1.83	16.08*	2.40	18.80*	0.864*	6.6
	Pine plantations	1.50	3.86*	3.91	7.36*	0.780*	9.3
SFF	Oak fragments	1.51	16.00*	1.60	15.14*	0.810*	4.7
	Pine plantations	0.44	$1.69^{0.1}$	2.51	7.14*	0.770*	5.5
Southern plateau							
ST	Oak fragments	4.59	14.57*	5.49	15.15*	0.867*	15.6
	Pine plantations	3.27	7.86*	6.48	13.23*	0.896*	16.2
SU	Oak fragments	2.60	13.02*	2.81	12.25*	0.815*	8.2
	Pine plantations	2.28	8.11*	2.98	9.01*	0.809*	8.2
SF	Oak fragments	1.08	7.47*	1.19	7.16*	0.635*	3.5
	Pine plantations	0.70	3.63*	2.22	9.71*	0.829*	5.1
SFF	Oak fragments	0.92	8.35*	1.49	11.83*	0.805*	3.9
	Pine plantations	0.29	$2.04^{0.05}$	1.29	7.67*	0.760*	2.9

Asterisks associated to *t* and *R* values indicate highly significant differences ($p<0.001$) and figures exact probabilities. Estimated richness values for patches of 1 ha (*a*: *Y*-intercept, since $\log_{10}(1) = 0$) and 100 ha are also given (see Fig. 1).

[a]ANCOVA results showed that intercepts differed significantly between oak fragments and pine plantations for ST ($F_{1,120} = 3.86$; $P = 0.05$) and SFF ($F_{1,120} = 17.79$; $P<0.001$) in the southern plateau, whereas in the northern plateau significant differences were found for SFF only ($t_{156} = 2.50$, $p<0.02$; Zar, 1984).

[b]Parallelism tests were significant ($P = 0.001$) for ST, SF and SFF and near significant for SU ($P = 0.056$) in the northern plateau ($F_{1,155}$), but only for SF in the southern plateau ($F_{1,119}$).

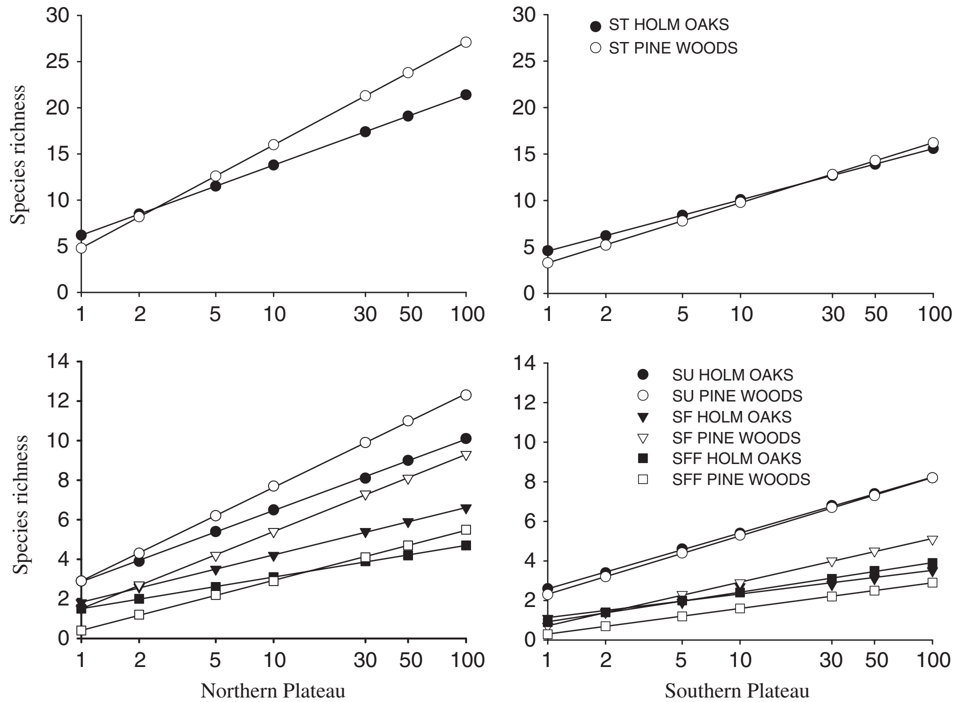

Figure 1. Estimated species richness for a range of woodland sizes (in hectares) according to woodland type (oak woodlands and pine plantations), plateau (northern and southern) and ecological group (ST, SU, SF and SFF as in Table 2). Estimates were derived from the regression equations of Table 2.

Methods), which were sampled with a roughly similar effort, were more species-rich in pine plantations than in oak woodlands (unpublished data): 27 vs. 24 bird species in the northern plateau (87.1 vs. 91.0 ha censused), and 21 vs. 19 species in the southern plateau (43.0 vs. 51.6 ha censused), respectively. Bearing in mind this sampling bias, we did not compare the distributional patterns of all bird species. Instead, we focused on bird species that were either frequent-exclusive or near-exclusive to each woodland type (see Methods and Table 5). Seven species were frequent-exclusive for the oak assemblages, but only four for pine plantations (Table 5). In addition, six near-exclusive species were widespread in the oak woodlands but only one was in pine plantations (Table 5). Out of these 18 species, the seven which are qualified as SPECs were frequent-exclusive (*Galerida cristata*, *Emberiza hortulana*, *Miliaria calandra* and *Phylloscopus bonelli*) or near-exclusive (*Galerida theklae*, *Sylvia undata*, *Sylvia hortensis*) from oak fragments. The first three species are ubiquitous rather than true forest birds and *G. theklae* may be considered even a farmland-steppe bird. These species were present at forest edges or in very degraded forest patches in the study area, whereas the two last species are Mediterranean warblers associated with shrublands and open oak woods (Purroy, 1997; Tellería, Asensio, & Díaz, 1999).

Discussion

Forest type and bird diversity

Bird diversities maintained by the mature pine plantations studied were rather similar to values recorded from oak fragments, whose higher diversity (12 species more than in pine plantations) was most likely due to the larger number of oak fragments sampled. Additionally, 12 out of 17 species restricted to oak assemblages (see Table 3) were recorded from just one plateau and 10 species occurred very infrequently in fragments (from 0.8% to 4.1%, Table 5). Moreover, the seven forest species classified as frequent-exclusive to oak woodlands (Table 5) could nest in non-sampled extensive pine forests (many thousand hectares) in the studied regions (Martí & del Moral, 2003), and two, the Robin *Erithacus rubecula* and the Firecrest *Regulus ignicapillus*, have been recorded as breeders in plantations over 100 ha (Díaz et al., 1998). In contrast, three out of five species

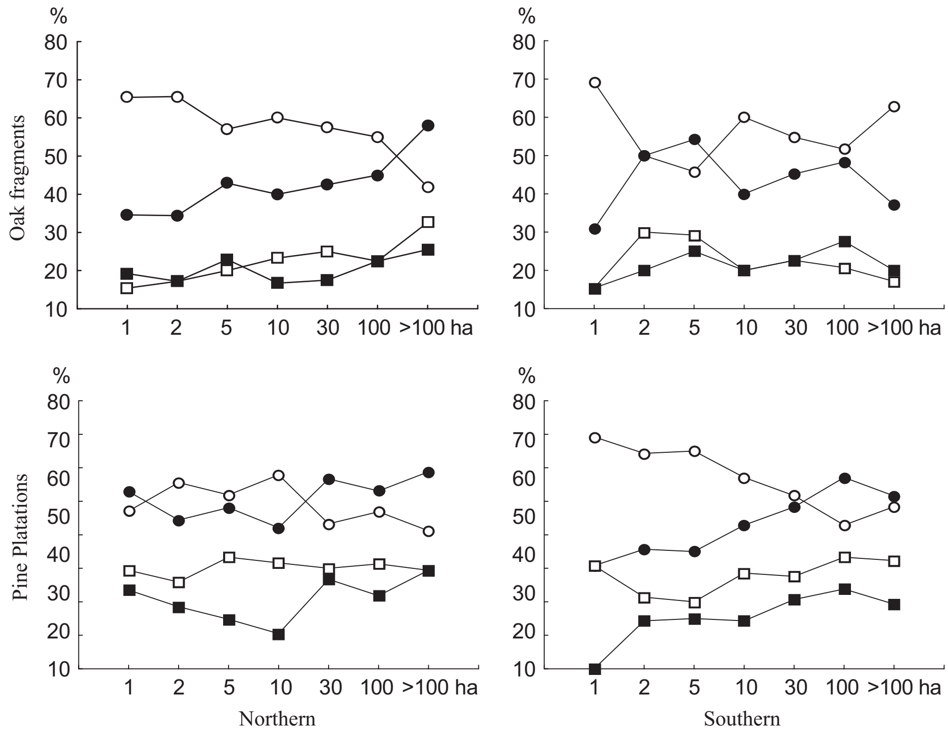

Figure 2. Variation in the cumulative proportions of ubiquitous (open circles) vs. forest species (closed circles) according to patch size for each woodland type and plateau; note that proportions of forest species are the sums of proportions of forest generalists (open squares) and forest specialists (closed squares). Proportions were directly computed from the data set obtained from the full sample of studied woodlands (see sampling section in Materials and Methods); for that, species lists were compiled for oak and pine patches $\leqslant 1$, $\leqslant 2$ ha, etc, and the proportions calculated from these accumulated species lists (note that forests >100 ha were included in a single size class). Differences in proportions were tested by means of three χ^2 tests (4 d.f.) per plateau and forest type (U–forest (F+FF); U–F; U–FF), excluding data for patches larger than 100 ha. Significant or near-significant differences in proportions of ubiquitous vs. forest birds as patch size changed were: southern Holm oak fragments (ubiquitous–forest = 14.02, $P = 0.015$; ubiquitous–forest generalists = 11.70, $P = 0.04$; ubiquitous–forest specialists = 8.09, $P = 0.15$); northern pine plantations (ubiquitous–forest specialists = 11.97, $P = 0.04$); southern pine plantations (ubiquitous–forest = 19.83, $P = 0.001$; ubiquitous–forest generalists = 10.0, $P = 0.08$; ubiquitous–forest specialists = 31.01, $P < 0.001$).

frequent-exclusive to pine woodlands were present in at least 30% of the northern plantations, and one of them, *Coccothraustes coccothraustes*, might be totally absent from flatland Holm oak remnants in spite of its preference for breeding in broadleaved as compared to pine forests (Martí & del Moral, 2003). Maybe this absence can be attributed to the fact that tree height in oak fragments is too low for this species, which tends to place its nests in tall trees (Purroy, 1997).

Many studies have shown that native woodlands tend to support higher bird richness than pine plantations (see Introduction), so that the similarity in the bird richness maintained by natural and planted woodlands is an unexpected result. However, this result may be explained by the greater forest development of the studied plantations, which are much taller and have a higher tree cover than oak fragments (Table 1). This difference is due to the degradation of oak forests by long-lasting human impacts such as grazing, clearing and firewood extraction (Santos & Tellería, 1998; Santos et al., 2002), which have strongly decreased the expected higher suitability of native forest vegetation for breeding birds (Berg, 1997; Rodewald & Abrams, 2002; Shochat et al., 2001). Nevertheless, it should be borne in mind that new pine afforestations will maintain much less forest species than the studied mature plantations during their growing period. For instance, available evidence from montane pine plantations in central Spain (Potti, 1985) indicates that young plantations are colonized mostly by bird species typical of open or shrubby habitats. These communities are replaced slowly by the true forest birds composing the final assemblages associated with old, full-

Table 3. Breeding bird assemblages of oak fragments and pine plantations (⩽100 ha) in central Spain according to ecological bird groups: ubiquitous (U); forest generalists (F); forest specialists (FF)

Ubiquitous	Forest generalists	Forest specialists
Alectoris rufa	*Lullula arborea*	*Dendrocopos major*
Columba oenas	*Prunella modularis* (HO)	*Sylvia undata*
C. palumbus	*Erithacus rubecula* (HO)	*S. cantillans*
Streptopelia turtur	*Luscinia megarhynchos*	*S. melanocephala* (HO)
Clamator glandarius	*Turdus merula*	*S. hortensis*
Cuculus canorus	*T. viscivorus*	*S. atricapilla* (HO)
Caprimulgus europaeus	*Oriolus oriolus*	*Phylloscopus bonelli* (HO)
C. ruficollis	*Garrulus glandarius*	*P. ibericus* (HO)
Upupa epops	*Cyanopica cyana (PP)*	*Regulus ignicapillus* (HO)
Picus viridis	*Fringilla coelebs*	*Aegithalos caudatus*
Galerida cristata (HO)	*Serinus serinus*	*Parus ater* (PP)
G. theklae	*Carduelis chloris*	*P. caeruleus*
Phoenicurus ochruros (HO)	*Coccothraustes coccothraustes* (PP)	*P. major*
Saxicola torquata (HO)		*Certhia brachydactyla* (PP)
Oenanthe oenanthe (HO)		
O. hispanica (HO)		
Cettia cetti (HO)		
Hippolais polyglotta (HO)		
Lanius meridionalis		
L. senator		
Pica pica		
Corvus monedula (HO)		
C. corone		
C. corax (PP)		
Sturnus unicolor		
Passer domesticus		
P. montanus		
Petronia petronia		
Carduelis carduelis		
C. cannabina		
Emberiza cirlus		
E. hortulana (HO)		
Miliaria calandra (HO)		

Restricted species: HO (Holm oak fragments), PP (pine plantations). Three additional species were found in large forest tracts (>100 ha): the Spotted Flycatcher *Muscicapa striata* (F) and the Rock Bunting *Emberiza cia* (F) in oak forests, and the Crested Tit *Parus cristatus* (FF) in pine plantations.

Table 4. Distribution of bird richness according to woodland type, plateau and ecological group

Bird groups	Holm oak fragments		Pine plantations	
	Northern	Southern	Northern	Southern
Ubiquitous (U)	30	22	22	19
Forest generalists (F)	11	8	10	9
Forest specialists (FF)	11	8	8	6
Total	52	38	40	34

All χ^2 tests comparing number of species between pine plantations and oak woodlands (three tests per plateau: U–F–FF; U–F+FF; and F–FF) were non-significant ($\chi^2_{1,\text{Yates}} = 0.001-0.34$; $P = 0.80-0.97$).

grown plantations. Given the prevailing climatic and soil conditions of the Spanish plateaux, that limit seriously tree growth (Rivas-Martínez, 1981), the establishment of bird assemblages similar to those found in the studied plantations would take decades.

Table 5. Number of fragments or plantations with presence of breeding bird species exclusive or near-exclusive of each woodland type

Holm oak fragments				Pine plantations			
Bird species	SPEC	North (122)	South (78)	Bird species	SPEC	North (37)	South (45)
Rare-exclusive							
Corvus monedula (U)	—	1	1	*Corvus corax* (U)	—	2	0
P. ochruros (U)	—	3	0				
Saxicola torquata (U)	—	3	0				
Oenanthe oenanthe (U)	3 (Dc: MRD)	1	0				
Oenanthe hispanica (U)	2 (Dp: LHD)	2	1				
Cettia cetti (U)	—	5	0				
Prunella modularis (F)	—	3	0				
Sylvia atricapilla (FF)	—	1	0				
P. ibericus (FF)	—	3	0				
Regulus ignicapillus (FF)	—	4	0				
Frequent-exclusive							
Galerida cristata (U)	3 (Dp: MHD)	2	6	*Cyanopica cyana* (F)	—	12	0
Hippolais polyglotta (U)	—	17	0	*C. coccothraustes* (F)	—	11	1
Emberiza hortulana (U)	2 (Dp: LHD)	16	0	*Parus ater* (FF)	—	3	0
Miliaria calandra (U)	2 (Dc: MRD)	9	1	*C. brachydactyla* (FF)	—	15	0
Erithacus rubecula (F)	—	13	0				
S. melanocephala (FF)	—	0	14				
P. bonelli (FF)	2 (Dc: MRD)	29	3				
Near-exclusive							
Galerida theklae (U)	3 (Dp: LHD)	0(1)	13(0)	*Dendrocopus major* (FF)	-	8(2)	2(0)
Emberiza cirlus (U)	—	49(2)	0(1)				
L. megarhynchos (F)	—	60(0)	1(3)				
Garrulus glandarius (F)	—	9(1)	0(0)				
Sylvia undata (FF)	2 (Dp: LHD)	13(0)	4(2)				
Sylvia hortensis (FF)	3 (Dp: LHD)	13(1)	5(3)				

Rare-exclusive species were present in less that 7% of fragments or plantations in one or both plateaux, whereas Frequent-exclusive species were present in more than 7%. Near-exclusive species are birds widespread in one woodland type (>7% fragments or plantations occupied) which also occupied a few patches (<7%) of the other (in brackets, number of patches occupied of the secondary woodland type). North and south indicate the northern and southern plateaux, respectively (number of patches studied in brackets). SPEC category (species names in bold) indicates European conservation status according to Burfield and van Bommel (2004). 2: species whose global populations are concentrated in Europe and which have an unfavourable conservation status in Europe; 3: species whose global populations are not concentrated in Europe, but which have an unfavourable conservation status in Europe. European threat status are Dp (depleted) and Dc (declining) in function of criteria large historical decline (LHD), moderate historical decline (MHD) and moderate recent decline (MRD).

Changes in bird diversity with isolation, patch size and habitat structure

Former extensive studies carried out in the oak woodlands (Santos et al., 2002) and old plantations (Díaz et al., 1998) of the Spanish plateaux have shown that patch size is the main factor determining both species richness and species composition of bird assemblages, whereas distances of woodland patches to large forest tracts showed just minor effects on some species. Other studies carried out in Mediterranean landscapes have found rather similar results (Brotons & Herrando, 2001), contrasting with the significant negative effects of isolation on forest birds in the Mesic woodland remnants of central-western Europe. This finding, which can be attributed to the relative scarcity of forest specialists in Mediterranean forests (Blondel & Aronson, 1999; see Santos et al., 2002 for a full discussion), would imply that isolation is a negligible factor in the development of afforestation strategies designed to maintain high bird diversities in Mediterranean forests.

Bird diversity was higher in oak fragments than in plantations for small patches (1–2 ha), whereas the reverse was true for medium-sized patches (5–10 ha) in the northern plateau and for large patches (50–100 ha) in the southern plateau (Fig. 1). Significant differences in the *Y*-intercepts of species–area relationships were only found for forest specialist species (FF; Table 2), and were directly attributable to the higher diversities of Mediterranean warblers (*Sylvia* spp.) in small (up to 2 ha) oak fragments than in pine plantations (0.46 species per fragment on average for 115 oak tracts as compared to 0.07 species per plantation for 44 plantations; Mann-Whitney *U*-test, $z = 3.509$, $P < 0.001$). Bearing in mind that most Mediterranean forest specialists are just as reliant upon shrubs than on the development of the tree layer (Blondel & Aronson, 1999; Santos et al., 2002), larger diversities in small oak fragments as compared to small pine afforestations could be due to a greater shrub development in the former, which is suppressed by management in the latter. In fact, shrub cover (woody vegetation up to 2 m tall) was significantly larger in small (2 ha or less) oak fragments than in pine plantations (15.7% and 5.8% respectively, $t_{157} = 5.35$, $p < 0.001$; unpublished data).

Conservation value of the bird assemblages of pine and oak archipelagoes

There were no SPEC-1 species (species of global conservation concern; see Burfield and van Bommel (2004)) in the studied fragmented woodlands. The seven SPEC-2 and -3 species frequent-exclusive or "near-exclusive" to oak woodlands are, with the exception of *Emberiza hortulana*, Mediterranean or south-European birds whose main European populations are concentrated in Spain (Burfield & van Bommel, 2004; Martí & del Moral, 2003). Although the expansion of pine plantations might have local negative effects on *Galerida cristata*, *M. calandra* and *Emberiza hortulana* (Table 6), two facts reduce the significance of oak fragments as critical habitats for the conservation of these species. On the one hand, four of them (*G. cristata*, *G. theklae*, *M. calandra* and *Emberiza hortulana*) are rather farmland and/or shrubland birds (Martí & del Moral, 2003; Tellería et al., 1999), so that the relative importance of the populations supported by oak woodlands can be considered as very low. On the other hand, the three other species are concentrated in habitats (*Phylloscopus bonelli*) and regions (*S. undata* and *S. hortensis*) different to those considered here (Martí & del Moral, 2003). To sum up, fragmented oak woodlands do not seem to be critical habitats for the conservation of birds of European concern.

Table 6. Main habitats and recommended management measures for the conservation of the seven SPEC species classified as frequent-exclusive or near-exclusive to Holm oak woodlands (see Table 5) in the landscapes studied

	Main habitats in Spain*	Main habitats in the studied landscape	Conservation measures recommended
G. cristata (U) SPEC 3	Non-intensive farmland	Non-intensive cropland	Concentrate new afforestations in landscapes with sparse but large pine patches (>10 ha)
G. theklae (U) SPEC 3	Semi-natural steppes, Holm-oak dehesas	Hilly set-asides, oak woodlands edges	Keep out the hilly set-asides from afforestation and maintain a low grazing pressure
E. hortulana (U) SPEC 2	Non-intensive farmland, upland shrublands	Oak woodlands edges and sparse oak trees	Preserve the isolated trees, and locate new afforestations far away from oak fragments
M. calandra (U) SPEC 2	Non-intensive farmland	Non-intensive cropland, farmland-oak forest edges	Preserve edges among fields, isolated trees and shrubs, and the non-cultivated elements of the farming landscape
P. bonelli (FF) SPEC 2	Open Mediterranean forests, mainly upland deciduous oak woodlands	Restricted to oak fragments	Preserve the oak woodlands
S. undata (FF) SPEC 2	Mediterranean maquis	Oak woodlands, shrubby set-asides	Preserve the oak woodlands
S. hortensis (FF) SPEC 3	Open oak woodlands, olive and fruit-tree groves	Oak woodlands	Preserve the oak woodlands

*According to Tellería et al. (1999) and Martí and del Moral (2003).

Implications for management

Although fragmentation usually produces negative effects on biodiversity at local and regional scales (Harrison & Bruna, 1999; Turner, 1996), the importance of small habitat patches such as forest fragments has been repeatedly emphasized as a way to maintain some biodiversity in human-dominated environments (see Bennett, Hinsley, Bellamy, Swetnam, & Mac Nally, 2004; Duelli & Obrist, 2003 and references therein for birds). Focusing on animal diversity in forest plantations, Lindenmayer and Hobbs (2004) have recently identified several key reasons for including wildlife conservation within the primary goals of afforestations, highlighting that many species can be favoured by implementing proper management modifications. In fact, studies carried out in fragmented European forest landscapes have shown that bird diversity can be increased substantially by just managing some landscape attributes such as the size and distance between forest patches, or the structure and plant composition of forest vegetation (Bellamy, Hinsley, & Newton, 1996; Díaz et al., 1998; McCollin, 1993; Opdam, Rijsdijk, & Hudtings, 1985; Santos et al., 2002; see Strijker, 2005; Van Diggelen, Sijtsma, Strijker, & Van den Burg, 2005, for complementary views). Here, we propose some specific management measures to the seven SPECs birds frequent-exclusive and near-exclusive to oak woodlands (Table 6).

Our results show that small patches of native Holm oak woodland maintain higher bird diversities than equivalent pine plantations, while the opposite was true for larger patches. This result appears to be associated to size-related differences in management, that enhance the shrub layer at the expense of trees in oak fragments whereas shrubs are suppressed in small pine plantations but not in larger ones. In fact, large pine tracts have been commonly planted on degraded Holm oak woodlands, and are currently used at low intensity as hunting or leisure areas rather than for timber production (unpublished data). López and Moro (1997) and Brotons and Herrando (2001) have also found strong positive effects of the oak-shrubby layer on bird diversity in *Pinus halepensis* plantations of eastern Spain.

Summing up, management of both pine plantations and oak remnants could play a role in the restoration and maintenance of forest bird assemblages in farming landscapes of the Spanish plateaux. Regeneration, expansion and re-creation of Holm oak woodlands seem to be difficult under current management practices due to the dependence of Holm oaks seedlings on nurse plants for surviving the Mediterranean summer drought and on acorn dispersers to these safe sites (Pulido & Díaz, 2005; Santos & Tellería, 1997; Truscott, Mitchell, Palmer, & Welch, 2004). Pines are much easier to establish, but pine plantations have the drawback of the long time needed for their development into mature woods (at least from the point of view of forest birds) and of their low ability to maintain Mediterranean bird species (e.g. *Sylvia* warblers). Pine plantations could be subsidized for conservation goals only if large (>2 ha, or preferably much more, especially in the Southern plateau; Díaz et al., 1998) and shrubby. Management of oak fragments will be focussed on increasing its size, preferably by subsidizing shrub development around them (or connecting neighbouring small fragments) using set-aside programs, before planting oaks under shrubs using reafforestation schemes, if needed (Gómez-Aparicio et al., 2004).

Recent analyses of the total economic value of Mediterranean forests emphasize that non-commercial economic values such as recreation, conservation and carbon fixation can be more important than commercial values such as timber production, grazing or hunting (Caparrós, Campos, & Montero, 2003). In fact, multiple uses (including conservation) rather than intensification of timber production or hunting appear to be the only way of maintaining the economic sustainability of Mediterranean forests (Díaz, Pulido, & Marañón, 2003; Díaz, Campos, & Pulido, 2006). This economic reality would justify CAP reforms aimed at subsidizing the proposed management measures.

Acknowledgements

Financial support was provided by the Spanish Ministry of Science and Technology (Project BOS2000-0556). This paper is a contribution to the project 'Evaluating current European agri-environment schemes to quantify and improve nature conservation efforts in agricultural landscapes' (EASY) financed by the V Framework Programme of the EU. Two anonymous referees made valuable comments on early drafts of the manuscript.

References

Barbero, A., González, F., & Catalán, G. (Eds.). (1994). *Manual de forestación en tierras agrícolas*. Madrid: Ministerio de Agricultura, Pesca y Alimentación.

Bellamy, P. E., Hinsley, S. A., & Newton, I. (1996). Factors affecting birds species numbers in small woods in south-east England. *Journal of Applied Ecology, 33*, 243–262.

Bennett, A. F., Hinsley, S. A., Bellamy, P. E., Swetnam, R. D., & Mac Nally, R. (2004). Do regional gradient in land-use influence richness, composition and turnover of bird assemblages in small woods? *Biological Conservation*, *119*, 191–206.

Berg, Å. (1997). Diversity and abundance of birds in relation to forest fragmentation, habitat quality and heterogeneity. *Bird Study, 44*, 355–366.

Blondel, J., & Aronson, J. (1999). *Biology and wildlife of the Mediterranean region*. Oxford: Oxford University Press.

Brotons, L., & Herrando, S. (2001). Factors affecting bird communities in fragments of secondary pine forests in the north-western Mediterranean basin. *Acta Oecologica*, *22*, 21–31.

Burfield, I., & van Bommel, F. (2004). *Birds in Europe: Populations estimates, trends and conservation status*. Cambridge: BirdLife International.

Caparrós, A., Campos, P., & Montero, G. (2003). An operative framework for total Hicksian income measurement. Application to a multiple-use forest. *Environmental and Resource Economics*, *26*, 173–198.

Díaz, M., Campos, P., & Pulido, F.J. (2006). Importancia de la caza en el desarrollo sustentable y en la conservación de la biodiversidad. In: J. Carranza, & M. Sáez de Buruaga, (Eds.), *Manual para la gestión integral de los ecosistemas mediterráneos en los terrenos cinegéticos de Andalucía* (pp. 000–000). Junta de Andalucía: Sevilla.

Díaz, M., Carbonell, R., Santos, T., & Tellería, J. L. (1998). Breeding birds communities in pine plantations of the Spanish plateaux: Biogeography, landscape and vegetation effects. *Journal of Applied Ecology, 35*, 562–574.

Díaz, M., Pulido, F.J., Marañón, T. (2003). Diversidad biológica y sostenibilidad ecológica y económica de los sistemas adehesados. *Ecosistemas*, *2003*(3), (URL: http://www.aeet.org/ecosistemas/033/investigacion4.htm).

Duelli, P., & Obrist, M. K. (2003). Regional biodiversity in an agricultural landscape: The contribution of seminatural habitat islands. *Basic and Applied Ecology, 4*, 129–138.

Gómez-Aparicio, L., Zamora, R., Gómez, J. M., Hódar, J. A., Castro, J., & Baraza, E. (2004). Applying plant facilitation to forest restoration in Mediterranean ecosystems: A meta-analysis of the use of shrubs as nurse plants. *Ecological Applications, 14*, 1128–1138.

Harrison, S., & Bruna, E. (1999). Habitat fragmentation and large-scale conservation: What do we know for sure? *Ecography, 22*, 225–232.

ICONA. (1979). *Las coníferas en el primer inventario forestal nacional*. Madrid: Ministerio de Agricultura.

Lindenmayer, D. B., & Hobbs, R. J. (2004). Fauna conservation in Australian plantation forests – A review. *Biological Conservation, 119*, 151–168.

López, G., & Moro, M. J. (1997). Birds of Aleppo pine plantations in south-east Spain in relation to vegetation composition and structure. *Journal of Applied Ecology, 34*, 1257–1272.

Martí, R., & del Moral, J. C. (Eds.). (2003). *Atlas de las Aves Reproductoras de España*. Madrid: Dirección General de Conservación de la Naturaleza-Sociedad Española de Ornitología.

McCollin, D. (1993). Avian distribution patterns in a fragmented wooded landscape (North Humberside, UK): The role of between-patch and within-patch structure. *Global Ecology and Biogeography Letters, 3*, 48–62.

Opdam, P., Rijsdijk, G., & Hudtings, F. (1985). Bird communities in small woods in an agricultural landscape: Effects of area and isolation. *Biological Conservation*, *34*, 333–352.

Potti, J. (1985). La sucesión de las comunidades de aves en los pinares repoblados de Pinus sylvestris del Macizo de Ayllón (Sistema Central). *Ardeola*, *32*, 253–277.

Pulido, F. J., Campos, P., & Montero, G. (Eds.). (2002). *La gestión forestal de las dehesas*. Mérida: IPROCOR.

Pulido, F. J., & Díaz, M. (2005). Regeneration of a Mediterranean oak: A whole-cycle approach. *EcoScience*, *12*, 92–102.

Purroy, F. J. (1997). *Atlas de las aves de España (1975–1995)*. Barcelona: Lynx Edicions.

Rivas-Martínez, S. (1981). Les étages bioclimatiques de la végétation de la Péninsule Iberique. *Anales del Jardín Botánico de Madrid*, *37*, 251–268.

Rodewald, A. D., & Abrams, M. D. (2002). Floristics and avian community structure: Implications for regional changes in eastern forest composition. *Forest Science*, *48*, 267–272.

Santos, T., & Tellería, J. L. (1997). Vertebrate predation on Holm Oak, *Quercus ilex*, acorns in a fragmented habitat: Effects on seedling recruitment. *Forest Ecology and Management*, *98*, 181–187.

Santos, T., & Tellería, J. L. (Eds.). (1998). *Efectos de la fragmentación de los bosques sobre los vertebrados en las mesetas ibéricas*. Madrid: Organismo Autónomo de Parques Nacionales, Ministerio de Medio Ambiente, Colección Técnica.

Santos, T., Tellería, J. L., & Carbonell, R. (2002). Bird conservation in fragmented Mediterranean forests of Spain: Effects of geographical location, habitat and landscape degradation. *Biological Conservation, 105*, 113–125.

Shochat, E., Abramsky, Z., & Pinshow, B. (2001). Breeding bird species diversity in the Negev: Effects of scrub fragmentation by planted forests. *Journal of Applied Ecology, 38*, 1135–1147.

Sokal, R. R., & Rohlf, F. J. (1981). *Biometry* (2nd ed.). New York: Freeman.

StatSoft, Inc. (1999). *STATISTICA for Windows*. Tulsa, OK: StatSoft Inc.

Strijker, D. (2005). Marginal lands in Europe – Causes of decline. *Basic and Applied Ecology, 6*, 99–106.

Tellería, J. L. (1992). Gestión forestal y conservación de las aves en España peninsular. *Ardeola*, *39*, 99–114.

Tellería, J. L., Asensio, B., & Díaz, M. (1999). *Aves Ibéricas II. Paseriformes*. Madrid: J.M. Reyero.

Tellería, J. L., & Santos, T. (1999). Distribution of birds in fragments of Mediterranean forests: The role of ecological densities. *Ecography*, *22*, 13–19.

Truscott, A. M., Mitchell, R. J., Palmer, S. C. F., & Welch, D. (2004). The expansion of native oakwoods into conifer cleared areas through planting. *Forest Ecology and Management*, *193*, 335–343.

Turner, I. M. (1996). Species loss in fragments of tropical rain forest: a review of the evidence. *Journal of Applied Ecology*, *33*, 200–209.

Van Diggelen, R., Sijtsma, F. J., Strijker, D., & Van den Burg, J. (2005). Relating land-use intensity and biodiversity at the regional scale. *Basic and Applied Ecology*, *6*, 145–159.

Wiens, J. A. (1989). *The ecology of bird communities. Volume I. Foundations and patterns*. Cambridge: Cambridge University Press.

Zar, J. H. (1984). *Biostatistical analysis* (2nd ed.). Englewood Cliffs, NJ: Prentice-Hall Inc.