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The effects of grassland management using fire on habitat occupancy and conservation of birds in a mosaic landscape

PERE PONS[1,*], BERNARD LAMBERT[2], ERIC RIGOLOT[3] and ROGER PRODON[4]

*[1]Departament de Ciències Ambientals, Universitat de Girona, Campus de Montilivi, E-17071 Girona, Catalonia, Spain; [2]Service Interdépartamentale Montagne Elevage, 3 rue du Soleillé, F-66650 Prades, France; [3]Unité de Recherches Forestières Méditerranéennes, INRA, Avenue A. Vivaldi, F-84000 Avignon, France; [4]Biogéographie et Ecologie des Vertébrés, EPHE, Université Montpellier II, Cc. 94, F-34095 Montpellier cedex 5, France; *Author for correspondence (e-mail: pere.pons@udg.es; fax: +34-972418150)*

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Abstract. Prescribed burning is routinely used to improve grazing in Pyrenean rangelands affected by an overall trend of land abandonment. This study considers the environmental variables influencing habitat occupancy by birds and the consequences of the use of fire in range management for bird conservation. Bird use and habitat structure of 11 cover types, the result of specific management regimes, were monitored for two breeding seasons in a mosaic landscape. Three main gradients of avian composition, corresponding to tree cover, shrub volume and grazing intensity, were identified from canonical correspondence analysis. The structure of the bird community seemed more intensely affected by species-specific selection of cover types than by the birds' use of multiple patches. Out of a total of 10 bird species analysed by a simultaneous confidence intervals procedure, four species with an unfavourable conservation status in Europe (*Emberiza cia*, *Lullula arborea*, *Saxicola torquata* and *Lanius collurio*) preferred managed grassland. Three types of grassland with shrubs (derived from single or repeated burning) had the highest bird conservation index (taking into account specific status and abundance of the bird assemblage), whereas forests showed middle or low values. The relation (P = 0.054) of this index to the logarithm of the pastoral value (which includes density and grazing quality of grasses) in currently managed cover types suggests that the objectives of grassland recovery by appropriate management practices and those of bird conservation coincide in our study area.

Introduction

Important landscape changes have taken place in the northern Mediterranean during the second half of the 20th century as a result of rural depopulation (Barbero et al. 1990; Debussche et al. 1999). Mountain ranges have become less and less used for agriculture, grazing and forestry. This trend is particularly evident in the Pyrenees, where extensive land abandonment is occurring on both Spanish and French hillslopes. As a result, shrublands and pine forests are currently expanding to the detriment of formerly arable land and grasslands (García-Ruiz and Lasanta 1990;

Métailié et al. 2003). Besides the socio-economic implications of such a change in land use, the ecological impact on a landscape scale is still not fully understood.

Cattle raising is among the few existing possibilities for maintaining a sustainable and traditional economy in the Pyrenees. But the necessary recovery of livestock numbers requires part of the encroached lands to be transformed into grassland (García-Ruiz et al. 1996). Such a transformation, traditionally brought about by shepherd fires, is now achieved over large areas by means of prescribed burning. In the eastern French Pyrenees, range management since 1987 (Lambert and Parmain 1990) has led to an annual burned area of 528 ha (mean of 1990–2000). The increasing use of fire as a management tool, however, has led to growing concern for environmental consequences. Specifically, the potential effects of prescribed burning on biodiversity require careful study.

Birds, particularly breeding passerines, have been successfully used for evaluating ecological changes at the landscape level owing to the species-specific selection of landscape units according to the vegetation structure (e.g. Hansen and Urban 1992; Flather 1996; Preiss et al. 1997). Within the current northern Mediterranean context of shrubland progression and afforestation, species living in open habitats and low shrublands decrease, whereas the abundance of forest species tends to increase (Preiss et al. 1997). At lower elevations, many of the Mediterranean endemics tend to be mainly restricted to early stages of succession (Prodon and Lebreton 1981; Blondel and Farré 1988). Since rural depopulation is expected to continue, land abandonment has become one of the main threats to this avifauna (Tellería et al. 1992; Tucker and Heath 1994; Prodon 2000). On the other hand, the impact of prescribed burning on bird communities, which has begun to be assessed in Mediterranean shrublands (Pons 1998, 1999), has yet to be studied at higher elevations. The only existing data in the Pyrenees concern the grey partridge *Perdix perdix hispaniensis*, which is negatively affected by large fires in the short term but probably not in the long term (Novoa et al. 1998). In contrast, studies on the effect of shrubland and grassland burning on birds are numerous in North America, South Africa and Australia, even though their findings sometimes differ (e.g. Robel et al. 1998; Shriver and Vickery 2001).

This study has been carried out in a montane study area successively cultivated, abandoned, extensively grazed, burned by wildfires, and now regularly fire-managed and grazed. The aims of the present research are: (a) to investigate the habitat occupancy of breeding passerines in the resulting mosaic landscape, (b) to analyse the effect of habitat structural variables on bird communities, and (c) to evaluate the current and expected consequences of the different management and disturbance regimes for bird conservation.

Study area and methods

Study area

The study was conducted at Railleu (2°10′ E, 42°36′ N), in the eastern Pyrenees

(Catalonia, France). Railleu consists of a southwest facing hillside of 145 ha between 1450 and 1850 m a.s.l., near the upper limit of Scots pine *Pinus sylvestris* forest, with a mean annual temperature and rainfall of 6.8 °C and 744 mm, respectively. The study site is a mosaic of grassland, broom, *Cytisus* (*purgans*) *oromediterraneus*, shrubland, and forest fragments; the result of a complex history of human activities. At the beginning of the 20th century the hillside was cultivated, especially on the lower slopes, while scattered sheep flocks pastured on the marginal areas. Aerial photographs show general land abandonment from the 1940s onwards, resulting in the progression of native vegetation despite extensive cattle breeding. As a result of the extensive encroachment, an uncontrolled fire burned about half of the hillside in November 1980. From 1990 onwards a management schedule, consisting both of prescribed winter burning and the regulation of cattle grazing, was set up to improve grassland value and to decrease wildfire risk for the surrounding forests (Rigolot et al. 1998). During the study (1998 and 1999) the site was intensively grazed, mostly in summer, for around 170 livestock unit grazing days (LUGD)/ha. Cattle, as well as wild deer *Cervus elaphus*, could move freely through the different vegetation patches. Clearing fires, set on shrubland patches previously unburned or burned a long time ago, were intense and spread quickly. Maintenance fires, set on patches burned recently and with discontinuous and low fuel load, were of low intensity. This pattern of land abandonment, grazing, wildfires and prescribed burning of different intensities has resulted in the current mosaic of vegetation patches (Figure 1).

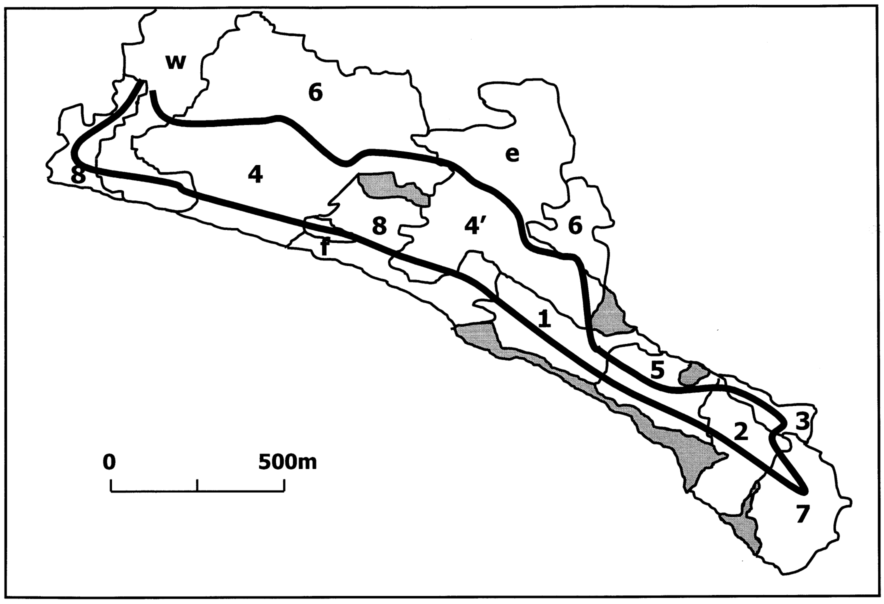


Figure 1. Map of the study area showing the extent of the habitat patches studied. Numbers/letters refer to the cover types in Table 1. The bold line represents the transect used for bird counts.

Table 1. Recent history of disturbances (date of the last wildfire and years with prescribed burning, p.b.) and area surveyed for the cover types studied; see text for details.

Cover code	Cover type	Wildfire	Clearing p.b.	Maintenance p.b.	Area (m^2) surveyed
8	Wet grassland	NO	NO	NO	34 780
5	Grassland with shrubs	1994	1990	NO	32 930
1	Grassland with shrubs	NO	1990	1997	25 900
4	Grassland with shrubs	1980	1990	1995, 1996	40 700
4′	Grassland with shrubs	1980	1990	1995, 1997	28 120
2	Grassland with shrubs	NO	1993	NO	25 160
3	4-year-old shrubland	1994	NO	NO	8880
6	18-year-old shrubland	1980	NO	NO	26 455
7	45-year-old shrubland	NO	NO	NO	13 320
w	>50-year-old pine forest (west)	1960	NO	NO	21 090
e	>50-year-old pine forest (east)	1960	NO	NO	8695
f	40-year-old pine forest fragment	NO	NO	NO	8695

Habitat description

The 11 cover types studied were defined as areas by an exclusive combination of disturbance events and management regimes, outlined in Table 1. Previous work in the study area showed that such a classification corresponded well to plant composition or to vegetation structure (Rigolot et al. 1998). The 11 cover types were distributed in 14 patches, ranging in size from 0.9 to 16.5 ha (Figure 1); smaller or marginal patches were not sampled. The edges between vegetation covers were sharp, without gradual transition or hedgerows. Sparse stone walls, limiting ancient terraces, existed at the bottom of the hillside, but these seldom acted as boundaries of different patches. The area surveyed for birds (see below) in a given patch was measured on aerial photographs. This was the product of the length of the transect in that patch by a width of 25 m if the patch extended on only one side of the pathway (a different patch extending on the other side), or by 50 m if the patch extended on both sides of the pathway. A sample of these measurements was then checked in the field.

Vegetation was sampled once each summer, in 1998 and 1999. Three or four 20-m line transects were randomly located in each cover type, except in more homogeneous forest where one transect was used. Contacts of habitat items with a stick at 100 points, evenly distributed at 20-cm intervals along the transect, were used to calculate relative covers of bare ground, litter and herbaceous and shrubby species. An index of the pastoral value of the herbaceous layer (varying from 0 to 100) was deduced from the grass cover and the relative frequency of individual species (Daget and Poissonet 1971): PV = 0.2 grass cover $\Sigma_i\ f_i q_i$, where f_i is the relative frequency of each grass species and q_i is a specific grazing quality index (see Appendix 1) that varies from 0 to 5 depending on the species growth speed, nutritive value, palatability, digestibility, etc. Shrub cover was estimated, in 20×0.5 m plots adjacent to each transect, by counting the number of squares of 100 cm^2 included in the ground projection of shrubs. Height of the foliage volume (H in cm)

Table 2. Mean values of several habitat descriptors for the cover types studied; pooled data for 1998 and 1999.

Cover code	Cover type	Rock cover (%)	Grass cover (%)	Shrub cover (%)	Tree cover (%)	Pastoral value	Grazing intensity
8	Wet grassland	0.7	97.8	1.2	4.4	52.0	3.8
5	Grassland with shrubs	7.0	85.2	5.0	5.9	31.5	3.7
1	Grassland with shrubs	1.9	85.7	14.0	1.9	31.3	3.5
4	Grassland with shrubs	4.6	93.2	21.0	2.0	45.5	3.2
2	Grassland with shrubs	5.7	79.8	49.3	0.4	33.5	2.0
3	4-year-old shrubland	3.6	45.3	37.7	2.3	9.0	3.0
6	18-year-old shrubland	4.0	31.9	80.0	4.3	17.8	1.0
7	45-year-old shrubland	3.5	43.5	66.8	7.5	22.9	2.4
w	>50-year-old pine forest (west)	6.7	61.5	0.0	76.7	10.5	1.0
e	>50-year-old pine forest (east)	5.2	52.0	0.0	72.0	9.0	0.0
f	40-year-old pine forest fragment	5.0	33.5	2.0	70.0	21.5	1.0

The index of pastoral value follows Daget and Poissonet (1971) and depends on grass cover and grazing quality of each plant species. Grazing intensity is a categorical variable that can range from 0 to 5 and measures the degree of plant consumption by cattle.

and basal area (S in cm^2) were also measured for each shrubby clump. Cover was estimated: (a) for individual shrubby species (mainly *Cytisus oromediterraneus* and *Rosa canina*), (b) for five vertical layers (0–25, 25–50, 50–100, 100–200, and 200–400 cm) by adding the individual foliage cover of shrubby clumps at each layer, and (c) for the overall shrub layer. Shrub volume (SV in cm^3) was then estimated as the product of cover and height: $SV = \Sigma_i S_i H_i$.

Apparent landscape features such as trees and rock outcrops, that were unevenly distributed in the study area, may affect habitat occupancy by birds. Since vegetation transects and plots were not adequate to measure tree and rock cover, these were evaluated from aerial photographs within a 100-m wide band centred on the bird transect. At the end of the 'grazing season' in autumn, grazing intensity was estimated (always by the same observer) in each cover type as a whole, according to an index of plant consumption by cattle. This index varies from 0 to 5 depending on the degree to which grasses have been predated (Etienne et al. 1996). For all variables used in the analyses, values were the mean of the different measures per cover type and per year. Principal habitat descriptors of the study area are shown in Table 2. For the sake of simplicity, values in the table are averaged for the 2 years of the study, with the exception of rock and tree cover, which was measured only once.

Bird sampling

Birds were censused within a 50-m-wide and 5.6-km-long transect which passed through the cover types described in Table 1; the sampled area covered 28 ha. Markers were scattered throughout the vegetation in order to make record location easier. Transect censuses were always performed by the same observer, on mornings with good weather conditions, and were repeated six times each breeding season, i.e.

in June 1998 and 1999. Records included: number of birds, species identity, cover type code, use of rocks or trees, and movements of individuals between patches. Unequal probabilities of bird detection in the different cover types may bias the comparisons between vegetation types (Bibby and Buckland 1987). In order to minimise this bias we defined a rather narrow count band (25 m on each side of the path) and excluded records not preceded by any call, song or flight. For example, a bird simply standing on a shrub in a grassland was not recorded because it could have been overlooked, should this shrub be situated in a shrubland instead of a grassland. Special attention was also paid to maintaining a constant speed throughout the transect. This speed was measured within five 1.1-km-long sections. The mean speed was 1.40 km/h ($\pm$ 0.14 standard deviation (SD), $n = 60$), and there were no significant differences between the sections (single factor ANOVA, $F = 0.063$; df = 4, 55; $P = 0.99$). Counts totalled a duration of 49 h 35 min, and 1156 independent records of individual birds were obtained.

Data analyses

The avifauna matrix of 34 bird species $\times$ 22 samples (the result of the 2-year census of the 11 cover types) was submitted first to correspondence analysis (CA) to detect the main avifaunal gradients. The relationships between these gradients and the environmental variables, measured in each cover type and year, were then investigated with canonical correspondence analysis (CCA) using the software Praxis (Praxème, France). Due to the relatively small number of samples in our data set, the number of environmental variables (initially 12) had to be reduced before carrying out the analysis (Lebreton et al. 1988). We selected five structural characteristics of the vegetation which depend on the history of the habitat and thus may be modified by appropriate management practices. These were: bare ground cover, grazing intensity (which depends, in practice, on grass cover and herbaceous species composition), *Cytisus oromediterraneus* volume, *Rosa canina* volume, and tree (mainly *Pinus sylvestris*) cover. The percentage of variance of the avifauna matrix explained by the environmental variables was then estimated as the ratio of the inertia (sum of the canonical eigenvalues) of the CCA to the inertia of the unconstrained CA.

Supposing that (i) the surveyed area of each cover type was correctly estimated, (ii) detectability bias was reduced, and (iii) the speed of the observer was constant, habitat availability can be defined as the area of a given cover type within the transect belt in proportion to the total transect area, and habitat use as the number of observations of a given species in a given cover type in proportion to the total number of observations of that species. Habitat selection was then analysed by using a simultaneous confidence intervals procedure (Alldredge et al. 1998). When the available proportion lies within the confidence interval of the proportion used by a given species, the use of a specific habitat is non-selective; when it is lower the habitat is preferred and when it is higher the habitat is avoided. Bailey intervals (Bailey 1980) provide a good combination of low error rates and interval length. These intervals are based on large sample properties, but are fairly robust and not particularly sensitive to small sample sizes (Cherry 1996). A necessary assumption of the method, however, is that of independent observations. This requirement was met by our data, because each individual provided no more than one observation per visit. The continuity corrected formulas for Bailey's intervals were applied to all species pooled (with the exception of aerial feeders) and to individual species with a sufficient number of records. In order to meet the standard rule-of-thumb of an expected number of observations in each cover type to be ≥ 5, cover types of reduced extent were grouped together or deleted in the analyses of most species. Cover types were grouped according to their closeness in the F1–F2 plot of the CA. Cover type 3, burned recently by a wildfire, was deleted from most analyses because of its dissimilarity in the CA to the remaining cover types. Depending on the number of records of each species, the number of cover types finally considered in the analyses was 12, 9 or 5.

We also studied the selection by birds of landscape features (trees and rocks) in the eight unwooded cover types. Here we considered not only the first observation of a given individual, but the first observation of this individual in each vegetation patch if it moved between patches (Carrascal 1983). Such repeated observations of the same individual accounted for only 7.7% of the data set (n = 571, seven species). Because it would have been difficult to estimate the availability of rocks and trees in the landscape in comparison to that of other substrates (bare ground, grass and shrubs), selection was assessed by testing the distribution of the observation frequencies within contingency tables (see for instance Martin and Thibault 1996). To identify the categories responsible for a significant χ^2 value, we analysed their adjusted residuals (Everitt 1977). Assuming that the variables forming the table are independent, the residuals are approximately normally distributed with mean 0 and SD 1. Statistical significance at the 5% level of residuals can then be estimated by comparing their absolute values with the 5% standard normal deviate, namely 1.96.

In order to assess the conservation value of the different cover types, we constructed a simple index (Equation 1) which takes into account the status and abundance of species observed in each cover type. The status was based on the classification of Tucker and Heath (1994) in categories of 'Species of European Conservation Concern', hereafter SPECs. A SPEC value, in geometric progression of increasing conservation concern, was assigned to each species (SPEC value$_i$: Non-SPEC = 1, SPEC-4 = 2, SPEC-3 = 4, SPEC-2 = 8, SPEC-1 = 16; species belonging to all categories except SPEC-1 were present at Railleu, see Appendix 2). Abundance was logarithm-transformed to balance its contribution to the global index.

$$\text{Conservation index of the cover type} = \sum_{i=1}^{k} [\log(A_i + 1) \times \text{SPEC value}_i] \quad (1)$$

where k is the species richness and A_i the abundance of species i relative to an area of 1 ha.

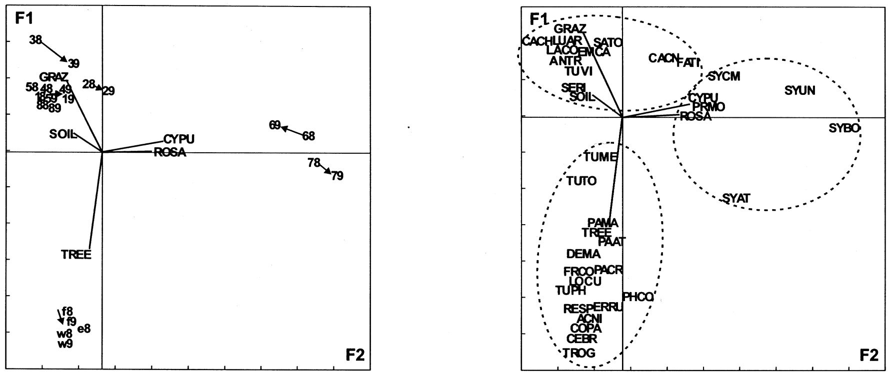
The influence of environmental variables on the conservation index was then approached by multiple regression. Eight quantitative independent variables were considered in the analysis: covers of bare ground, rock, grass and trees, volume of *Cytisus oromediterraneus* and *Rosa canina*, indices of pastoral value and grazing intensity. Year was used as an indicator variable to account for the use of two dates for each cover type without committing temporal pseudoreplication (Hurlbert 1984).

Results

The first two CA axes of the avifauna matrix (not shown) correctly discriminated the 11 cover types. Conversely, bird samples of the same cover type in different years were close together. These results suggest (i) that the spatial scale of the management on the one hand, and of the sampling on the other hand, both proved adequate to induce and assess, respectively, clear avian responses, (ii) that between-year within-type variation in the composition of bird communities was low when compared to between-type variation. The inertia of the CCA accounted for 52.4% of the inertia of the unconstrained CA, and the first two axes alone for 42.2%. The corresponding CCA biplots showed a triangular pattern (Figure 2). The first vertex consisted of forest samples (codes w, e, f) and strict forest birds (such as *Troglodytes troglodytes*, *Regulus* sp. and *Certhia brachydactyla*), tree cover being the main explicative variable. The second vertex consisted of old shrublands in which the transition from a 15-year-old (code 6) to a 45-year-old community (code 7) was determined by a decrease in *Cytisus oromediterraneus* volume, due to senescence, but also by an increase in *Rosa canina* volume. *Sylvia borin* and *S. undata* are representative species of these mature shrublands. The third vertex consisted of recently burned shrubland (code 3) and, to a lesser extent, grassland with sparse shrub cover (codes 5, 4, 1 and 8), the related variables being grazing intensity and, secondarily, bare ground cover. Bird species characteristic of farmland habitats and ecotones (*Lullula arborea*, *Sturnus vulgaris*, *Emberiza cia*, *Saxicola torquata* . . .) were close to this vertex. Finally, the between-year change of cover types in the CCA biplot (arrows in Figure 2a) was slight and mostly linked to an increase in shrub volume.

Table 3 shows habitat selection by common species in the study area. For all species pooled a significant preference for the three forests (codes w, e, f) existed, whereas a significant avoidance of cover types with the lowest shrub volume (codes 5, 8) or recently burned by wildfire (code 3) appeared. At a specific level, four species preferred managed grassland with shrubs (*Emberiza cia*, *Lullula arborea*, *Saxicola torquata* and *Lanius collurio*), three species selected forest positively (*Parus ater*, *P. cristatus* and *Fringilla coelebs*), two selected shrubland (*Prunella modularis* and *Sylvia communis*) and one preferred both wet grassland and grassland with shrubs (*Turdus viscivorus*). It is worth noting that the four species showing exclusive preference for grassland with shrubs are also those with an unfavourable conservation status in Europe (SPECs 2 and 3). The other six have a favourable conservation status (Non-SPEC and SPEC-4).

Table 4 shows the selection of landscape features by six common bird species in



(a) (b)

Figure 2. Biplots of the first two CCA axes. (a) Environmental variables (SOIL = % cover of bare ground, GRAZ = grazing intensity [plant consumption index], CYPU= volume of *C. oromediterraneus*, ROSA = volume of *Rosa canina*, TREE = % tree cover) vs. cover types (the first number refers to cover code in Table 2, the second number to year: 8 = 1998 and 9 = 1999; e9 is hidden under e8). Arrows show CCA trajectories of cover types from 1998 to 1999. (b) Environmental variables vs. bird species. The top oval groups together birds found mainly in managed grasslands; the right oval groups birds occurring mainly in mature shrubland; the bottom oval groups birds of pine forest (see Appendix 2 for the meaning of acronyms; GAGL is hidden under SOIL; EMCT and STVU are hidden under LACO).

unwooded cover types. Significant preference for trees can be related to territorial signalling in *Lullula arborea*. In *Turdus viscivorus* isolated trees are frequently used as intermediate perching sites between grassland (main feeding areas) and forest (breeding habitat). Rocks were used as territorial posts and possibly for foraging in *L. arborea* and *Saxicola torquata*, both with significant preference. *Sylvia communis* and *Emberiza cia* showed significant avoidance of rocks; the first one forages and sings in shrubs, the second one was usually seen on the ground or alarm calling on the top of shrubs. Scattered trees seemed to allow the use of unwooded habitats by tits (*Parus ater* and *P. cristatus*). Two nests of *P. ater* were even found in holes of stone walls at grassland in the study area. The relatively small number of observations in unwooded habitats, however, precluded testing the importance of trees as perching substrate in tits.

The last step of the analyses concerned the comparison of conservation value, taking into account all recorded species, among cover types. Figure 3 shows that the conservation value index (Equation 1) of a given cover type was in general not concordant with a positive selection of that cover by the avifauna (i.e. cover types that concentrated species of conservation concern were not densely populated by birds). The conservation index mainly depends, therefore, on species composition and status. The highest indices were found in currently managed cover types, particularly in codes 4′, 2 and 1. The relatively high value of the shrubland code 6 was mainly due to the presence of *Sylvia undata*, a SPEC-2. By contrast, wet grassland (code 8) and cover type 3 (severely affected by wildfire) had the lowest values. Among forests, the fragment (f) had the highest index, probably due to a

Table 3. Habitat use, habitat availability, Bailey confidence intervals of the proportion used and habitat selection for all birds pooled and for the commonest species in the study area.

Species	Selection	Cover types					
		8	5	1	4	4′	2
A							
Overall (1127)	Used/available	0.062/0.127	0.074/0.120	0.086/0.094	0.138/0.148	0.129/0.102	0.101/0.092
	Bailey intervals	(0.043, 0.085)	(0.053, 0.098)	(0.063, 0.112)	(0.110, 0.170)	(0.101, 0.159)	(0.077, 0.129)
	Preference	−	−	0	0	0	0
Emberiza cia (183)	Used/available	0.011/0.127	0.098/0.120	0.169/0.094	0.235/0.148	0.240/0.102	0.115/0.092
	Bailey intervals	(0, 0.052)	(0.044, 0.174)	(0.096, 0.258)	(0.149, 0.332)	(0.154, 0.338)	(0.055, 0.194)
	Preference	−	0	+	+	+	0
		3	6	7	w	e	f
Overall (1127)	Used/available	0.017/0.032	0.107/0.096	0.050/0.048	0.125/0.077	0.057/0.032	0.054/0.032
	Bailey intervals	(0.008, 0.031)	(0.082, 0.136)	(0.033, 0.071)	(0.098, 0.175)	(0.038, 0.079)	(0.036, 0.076)
	Preference	−	0	0	+	+	+
Emberiza cia (183)	Used/available	0.033/0.032	0.082/0.096	0.005/0.048	0/0.077	0.011/0.032	0/0.032
	Bailey intervals	(0.005, 0.087)	(0.033, 0.154)	(0, 0.042)	(0, 0.030)	(0, 0.052)	(0, 0.030)
	Preference	0	0	−	−	0	−
		8	5	1	4	4′	
B							
Prunella modularis (104)	Used/available	0.039/0.131	0.010/0.124	0.049/0.097	0.087/0.153	0.068/0.106	
	Bailey intervals	(0.003, 0.119)	(0, 0.071)	(0.007, 0.134)	(0.025, 0.187)	(0.015, 0.161)	
	Preference	−	−	0	0	0	
Turdus viscivorus (100)	Used/available	0.260/0.131	0.080/0.124	0.070/0.097	0.130/0.153	0.240/0.106	
	Bailey intervals	(0.144, 0.392)	(0.021, 0.179)	(0.015, 0.166)	(0.050, 0.243)	(0.129, 0.370)	
	Preference	+	0	0	0	+	

		2	6	7	Forest	
Prunella modularis (104)	Used/available	0.126/0.095	0.359/0.100	0.146/0.050	0.117/0.145	
	Bailey intervals	(0.048, 0.236)	(0.228, 0.494)	(0.061, 0.260)	(0.042, 0.224)	
	Preference	0	+	+	0	
Turdus viscivorus (100)	Used/available	0.110/0.095	0.030/0.100	0/0.050	0.080/0.145	
	Bailey intervals	(0.037, 0.218)	(0.001, 0.107)	(0, 0.052)	(0.021, 0.179)	
	Preference	0	0	0	0	
		8	5+1	4+4′+2	6+7	Forest
C						
Parus ater (75)	Used/available	0.053/0.131	0.013/0.221	0.147/0.354	0.133/0.150	0.653/0.145
	Bailey intervals	(0.006, 0.153)	(0, 0.090)	(0.056, 0.275)	(0.047, 0.259)	(0.491, 0.782)
	Preference	0	−	−	0	+
Sylvia communis (58)	Used/available	0/0.131	0.052/0.221	0.414/0.354	0.534/0.150	0/0.145
	Bailey intervals	(0, 0.081)	(0.003, 0.170)	(0.244, 0.583)	(0.351, 0.695)	(0, 0.081)
	Preference	−	−	0	+	−
Lullula arborea (49)	Used/available	0/0.131	0.490/0.221	0.510/0.354	0/0.150	0/0.145
	Bailey intervals	(0, 0.095)	(0.294, 0.668)	(0.313, 0.687)	(0, 0.095)	(0, 0.095)
	Preference	−	+	0	−	−
Saxicola torquata (48)	Used/available	0.021/0.131	0.250/0.221	0.625/0.354	0.104/0.150	0/0.145
	Bailey intervals	(0, 0.137)	(0.103, 0.432)	(0.417, 0.787)	(0.018, 0.259)	(0, 0.097)
	Preference	0	0	+	0	−
Parus cristatus (45)	Used/available	0.044/0.131	0/0.221	0.067/0.354	0.111/0.150	0.778/0.145
	Bailey intervals	(0, 0.182)	(0, 0.103)	(0.004, 0.215)	(0.019, 0.275)	(0.567, 0.904)
	Preference	0	−	−	0	+
Lanius collurio (40)	Used/available	0.175/0.131	0.550/0.221	0.275/0.354	0/0.150	0/0.145
	Bailey intervals	(0.046, 0.366)	(0.326, 0.738)	(0.108, 0.478)	(0, 0.115)	(0, 0.115)
	Preference	0	+	0	−	−
Fringilla coelebs (39)	Used/available	0.077/0.131	0/0.221	0.128/0.354	0/0.150	0.795/0.145
	Bailey intervals	(0.004, 0.244)	(0, 0.118)	(0.022, 0.312)	(0, 0.118)	(0.566, 0.922)
	Preference	0	−	−	−	+

The number of independent registrations (in parentheses) is beside specific names. Plus sign = preference, minus = avoidance, zero = non-significant selection. Cover types are described inTables 1 and 2 (forest = w+e+f).

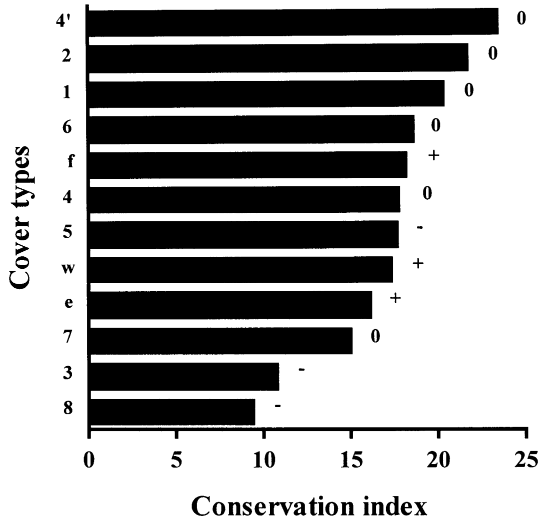


Figure 3. Bird conservation index (Equation 1) for the cover types studied, taking into account the abundance and conservation status (SPEC) of each species occurring in a given cover type; pooled data from 1998 and 1999. 0, +, −: indifferent, positive and negative selection by birds for the different cover types (see Table 3).

border effect. Multiple linear regression of conservation index with the eight environmental variables and year was non-significant (F = 1.20; df = 9, 12; P = 0.38), even though some individual coefficients were almost significant, either positive (rock cover t = 1.87, P = 0.086) or negative (bare ground cover t = −1.85, P = 0.089; *Rosa canina* volume t = −1.73, P = 0.11). It is interesting to note that post-wildfire cover type 3 combined the highest value of bare ground cover with the second lowest of the conservation index. In order to further analyse the possible relation between the pastoral and conservation objectives, we considered only those cover types that are currently being submitted to management (i.e. excluded forests and natural wet grassland). The regression between conservation index and the logarithm of pastoral value and year was almost significant (F = 2.75; df = 2, 11; P = 0.11), as was the individual coefficient for the logarithm of pastoral value (t = 2.16, P = 0.054).

Discussion

We initially used CCA to extract the differences in bird communities between cover types and, to a certain extent, the species turnover associated with management regimes. The main gradients identified, tree cover, shrub volume (which mainly depends on time since fire) and grazing intensity, account for a significant part of the variability of the avifauna composition and abundance among cover types. The unexplained variability of the bird matrix could be partly attributed to the historical component of habitat change (Knick and Rotenberry 2000), including important

Table 4. Number of observations and χ^2 residuals for the use of landscape features by six bird species in unwooded cover types.

Landscape features	*Emberiza cia*	*Turdus viscivorus*	*Prunella modularis*	*Sylvia communis*	*Lullula arborea*	*Saxicola torquata*
No use	166 (+1.13) NS	90 (−0.77) NS	80 (+0.87) NS	56 (+1.30) NS	45 (−2.33) *	37 (−3.05) *
Tree	16 (−6.91) *	18 (+8.94) *	11 (+0.19) NS	7 (−1.65) NS	9 (+6.76) *	6 (+0.06) NS
Rock	9 (−2.51) *	4 (−7.82) *	2 (−14.43) *	1 (−17.21) *	6 (+22.14) *	8 (+48.74) *

Numbers in parentheses are adjusted residuals of the χ^2 test. Residuals are significant at the 5% level (*) if their absolute value is higher than 1.96 (NS = non-significant residuals).

time lags in species responses to disturbance. It could also arise from the spatial characteristics of patches, such as their extent or the sharpness of the edges, which could not be considered in this study due to the limited amount of patches (usually one or two) available for each cover type. From a spatial point of view, however, the main message is that even such a fine mosaic (patches of 0.9–16.5 ha) results in a clear effect on bird diversity and community pattern; the size of the patches in our study site being mainly a result of the scale of current prescribed burning operations (burned plots averaged 12–19 ha in the last 7 years; unpublished results). The structure of the bird community was therefore more intensely affected by species-specific selection of cover types than by the use of multiple adjacent patches by some birds.

The differences in bird density among cover types at Railleu appear to be mainly a result of its well-known positive relation with vegetation volume, already measured in this region at the meso-Mediterranean (Prodon and Lebreton 1981) but not the montane bioclimatic level. At a specific level, a fairly good link seems to exist between habitat selection and requirements during the breeding season for the 10 species analysed. Such requirements, including foraging and nest sites, are usually found inside the breeding territory. As for *Turdus viscivorus*, however, its preference for patches of grassland (codes 8 and 4′) is exclusively for foraging, whereas the defended territory is located elsewhere in forests. The presence of rocks and trees in unwooded cover types also seems important, because: (a) they are used by species breeding in the patch for specific behaviours like territorial signalling, anti-predatory behaviour or specialised foraging, and (b) they allow the use of a patch by species usually absent from it. The latter is especially noticeable in generalist forest birds that are found in grassland and shrubland. An example is *Dendrocopos major*'s use of unwooded cover types, provided that some scattered dead trees exist. Even more extreme is *Parus ater*, which can breed in stone walls at a considerable distance from forest. Therefore, both species seem not especially concerned by grazing management as long as isolated trees are protected from burning.

A further analysis dealt with the conservation status in Europe of the bird community occupying each cover type. We have shown that the highest conservation values are found on managed grassland, whereas densely populated habitats, i.e.

forests, host few declining passerines. This is because many declining species in the Mediterranean are open-habitat birds (Prodon 2000). The conservation value of a cover type is, however, not linearly associated with a single environmental variable. The combination of grassland with scattered shrubs and trees, and a high rock cover is likely to increase both overall diversity and the abundance of SPECs 2 and 3 (cf. Appendix 2). At the scale of our study area such intra-patch heterogeneity is a result of both uneven burning of the shrub cover and important spatial differences in plant consumption by cattle. With the continuity of this dynamic management (four of the studied cover types were burned in the winters of 1999–2000 and 2000–2001, after this study was concluded), the maintenance or increase of patchy open habitats is most likely.

Our study demonstrates that land exploitation goals (i.e. grazing improvement here) can sometimes coincide with conservation objectives, although it does not seem common (e.g. Díaz et al. 1998). The preferred management regime of Railleu, consisting of a clearing burning of the shrubland, followed by cattle grazing and a maintenance fire 1–7 years afterwards, has proved to be sustainable for the plant community (Rigolot et al. 2002) and beneficial to bird conservation. Cover types unburned for a long time have, in general, a higher avian density but lower bird species richness and conservation value. When burning it is, however, important to avoid greatly affecting the grass layer, as this would increase the bare ground cover and therefore cause soil erosion; the replacement *per se* of shrubland by grassland probably does not produce erosion problems (García-Ruiz et al. 1996). To maintain bird diversity, trees and some shrubs, in particular the susceptible *Juniperus communis* (García et al. 1999), should be protected from fire. To achieve this, Pärt and Söderström (1999) recommended a retention of ≥10% shrub cover in farmland habitats of Sweden. In addition, it would be prudent to monitor the potential expansion of the invader *Senecio inaequidens* (*Compositae*, present in cover type 3 after wildfire) in burned and grazed areas. Similarly, the consequences of grazing intensity for bird richness and demography might be evaluated, as a negative relation may exist (e.g. Baines 1996; Jansen et al. 1999).

Population decline among farmland birds has been related to changes in agricultural practices in the 1970s (Fuller et al. 1995). Such changes, including an increased use of chemical pesticides, inorganic fertilisers and more intensive grassland management, are unlikely to occur in the process of grassland recovery in the eastern Pyrenees. These mountain habitats can therefore continue to host an array of open-habitat species under severe decline in intensified rural plains throughout the whole of Europe (Donald et al. 2001).

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Appendix 1

Occurrence of 102 plant species in 30 line transects in 11 cover types in 1998.

Species	GQ	Layer	Freq.	Species	GQ	Layer	Freq.
Achillea millefolium	2	G	462	*Malva moschata*	0	G	25
Agrostis capillaris	3	G	308	*Medicago suffruticosa*	2	G	16
Anthemis arvensis	0	G	20	*Ononis spinosa*	1	G	2
Anthyllis vulneraria	2	G	7	*Phleum pratense*	4	G	298
Armeria arenaria	1	G	13	*Picris hieracioides*	0	G	2
Avenula lodunensis	1	G	29	*Pimpinella major*	0	G	2
Campanula ficarioides	0	G	6	*Pinus sylvestris*	0	T	219
Campanula scheuchzeri	0	G	13	*Pinus uncinata*	0	T	1
Carex praecox	1	G	22	*Plantago holosteum*	1	G	11
Carlina acanthifolia	0	G	2	*Plantago lanceolata*	2	G	37
Centaurea alpina	0	G	35	*Plantago media*	2	G	3
Cirsium arvense	0	G	3	*Poa chaixii*	2	G	2
Clinopodium vulgare	0	G	1	*Poa compressa*	2	G	137
Convolvulus arvensis	1	G	5	*Potentilla aurea*	2	G	7
Coritospermum lucidum	0	G	4	*Prunella grandiflora*	0	G	25
Cytisus oromediterraneus	0	S	652	*Prunus spinosa*	0	S	5
Dactylis glomerata	5	G	220	*Pteridium aquilinum*	0	G	2
Deschampsia caespitosa	2	G	2	*Quercus robur*	0	T	1
Deschampsia flexuosa	1	G	90	*Ranunculus bulbosus*	0	G	1
Dianthus seguieri	0	G	18	*Rhinanthus angustifolius*	0	G	2
Echium vulgare	0	G	15	*Rosa canina*	0	S	47
Epilobium angustifolium	0	G	24	*Rostraria cristata*	1	G	10
Erodium cicutarium	1	G	5	*Rubus idaeus*	0	S	392
Euphorbia cyparissias	0	G	19	*Rumex acetosa*	0	G	4
Festuca ovina	1	G	54	*Rumex acetosella*	0	G	14
Festuca rubra	2	G	361	*Sanguisorba minor*	2	G	85
Filago vulgaris	0	G	3	*Scabiosa columbaria*	0	G	33
Fourraea alpina	0	G	20	*Sedum sediforme*	0	G	1
Fragaria vesca	0	G	1	*Senecio adonidifolius*	0	G	3
Galeopsis pyrenaica	0	G	2	*Senecio inaequidens*	0	G	2
Galium mollugo	0	G	69	*Senecio viscosus*	2	G	1
Galium rotundifolium	0	G	5	*Silene nutans*	0	G	8
Galium verum	0	G	98	*Silene vulgaris*	0	G	1
Genista sagittalis	1	S	111	*Solidago virgaurea*	0	G	1
Helianthemum nummularium	0	S	186	*Sonchus oleraceus*	1	G	2
Hieracium pilosella	0	G	10	*Taraxacum officinale*	1	G	15
Hippocrepis comosa	1	G	10	*Thymus serpyllum*	0	S	11
Hippocrepis unisiliquosa	1	G	1	*Tragopogon pratensis*	0	G	1
Holcus mollis	1	G	77	*Trifolium arvense*	1	G	19
Hypericum perforatum	0	G	13	*Trifolium campestre*	2	G	1
Jasione montana	0	G	3	*Trifolium ochroleucon*	3	G	33
Juniperus communis	0	S	7	*Trifolium pratense*	4	G	26
Knautia arvensis	0	G	5	*Trifolium repens*	4	G	38
Knautia integrifolia	0	G	1	*Trisetum flavescens*	3	G	2
Lathyrus cirrhosus	2	G	2	*Urtica dioica*	0	G	28
Lathyrus latifolius	2	G	13	*Valeriana officinalis*	0	G	8
Leontodon hispidus	2	G	14	*Verbascum pulverulentum*	0	G	3
Leucanthemum vulgare	0	G	1	*Veronica austriaca*	0	G	4
Linaria repens	0	G	11	*Veronica officinalis*	0	G	3
Lotus corniculatus	3	G	45	*Vicia cracca*	3	G	19
Luzula campestris	1	G	14	*Viola tricolor*	1	G	10

Frequence (Freq.) is expressed as the number of contacts of plant items with a stick over a total of 3000 points. GQ = specific grazing quality index (from 0 to 5). Layer: G = grass, S = shrub, T = tree. The plant nomenclature is according to Kerguelen (1999).

Appendix 2

Bird names, SPEC category (from Tucker and Heath 1994) and number of independent records at the Railleu study site for the 34 species included in the analyses.

Acronym	Scientific name	Common name	SPEC	*n*
ACNI	*Accipiter nisus*	Sparrowhawk	No	2
FATI	*Falco tinnunculus*	Kestrel	3	2
COPA	*Columba palumbus*	Woodpigeon	4	5
DEMA	*Dendrocopos major*	Great Spotted Woodpecker	No	4
LUAR	*Lullula arborea*	Woodlark	2	50
ANTR	*Anthus trivialis*	Tree Pipit	No	32
TROG	*Troglodytes troglodytes*	Wren	No	4
PRMO	*Prunella modularis*	Dunnock	4	104
ERRU	*Erithacus rubecula*	Robin	4	11
SATO	*Saxicola torquata*	Stonechat	3	48
TUTO	*Turdus torquatus*	Ring Ouzel	4	2
TUME	*Turdus merula*	Blackbird	4	33
TUPH	*Turdus philomelos*	Song Thrush	4	5
TUVI	*Turdus viscivorus*	Mistle Thrush	4	106
SYUN	*Sylvia undata*	Dartford Warbler	2	12
SYCM	*Sylvia communis*	Whitethroat	4	58
SYBO	*Sylvia borin*	Garden Warbler	4	2
SYAT	*Sylvia atricapilla*	Blackcap	4	14
PHCO	*Phylloscopus collybita*	Chiffchaff	No	10
RESP	*Regulus* spp.	Firecrest/Goldcrest	4	31
PACR	*Parus cristatus*	Crested Tit	4	45
PAAT	*Parus ater*	Coal Tit	No	75
PAMA	*Parus major*	Great Tit	No	8
CEBR	*Certhia brachydactyla*	Short-toed Treecreeper	4	8
LACO	*Lanius collurio*	Red-backed Shrike	3	40
GAGL	*Garrulus glandarius*	Jay	No	18
STVU	*Sturnus vulgaris*	Starling	No	20
FRCO	*Fringilla coelebs*	Chaffinch	4	39
SERI	*Serinus serinus*	Serin	4	39
CACH	*Carduelis chloris*	Greenfinch	4	2
CACN	*Carduelis cannabina*	Linnet	4	31
LOCU	*Loxia curvirostra*	Crossbill	No	13
EMCA	*Emberiza cia*	Rock Bunting	3	183
EMCT	*Emberiza citrinella*	Yellowhammer	4	13

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