High mountain rangelands host important populations of threatened bird species, but can be affected by extensive changes in land use. I studied the breeding bird community of two shrubland plots at 1,850–2,100 m a.s.l. in the Pyrenees. Breeding territories were mapped for four years, before and after the prescribed burning, the aim of which was to increase the grazing value of the study area. The most abundant species (reaching $\geq 3$ breeding pairs/10 ha in at least one plot and one year) were Dunnock *Prunella modularis*, Dartford Warbler *Sylvia undata*, Stonechat *Saxicola torquatus*, Rock Bunting *Emberiza cia* and Ortolan Bunting *E. hortulana*. The open-shrubland plot contained a similar number of breeding species (10 vs. 9), but a lower overall density (23 vs. 28 breeding pairs/10 ha) than the dense-shrubland plot. Most breeding species also occurred in winter. After fire, the number of bird species, overall density and conservation value (an index that takes into account all species’ densities and their categories of conservation concern in Europe) decreased, but tended to recover afterwards. These results may help understand the composition and dynamics of bird assemblages in managed mountain areas.

Key words: mountain shrublands, grassland management, territory mapping, breeding bird densities, conservation value

Pere Pons, Departament de Ciències Ambientals, Universitat de Girona, Campus de Montilivi, 17071 Girona, Catalonia, Spain. email: pere.pons@udg.edu

Received: 04.04.11; Accepted: 15.06.11 / Edited by S. Herrando.
tion status in Europe (Tucker & Evans 1997). Available publications, concerning the Pyrenees and the Sierra Nevada, highlight the value of burned rangelands for bird conservation (Pons et al. 2003), reveal that the short-term impact of fire on birds depends on its severity (Pons & Clavero 2010), demonstrate that logging burned trees can be detrimental to avifauna (Castro et al. 2010), and suggest that avian communities in upland areas recover more slowly after fire than those in lowland areas (Pons & Clavero 2010).

The general aim of this work was to study the breeding bird community of a subalpine rangeland in the Pyrenees. Thus, I mapped bird territories and measured vegetation structure in two plots, one with high and the other with low shrub density, for four years. Both plots were managed by prescribed burning and grazed during the study period. The specific objectives were to contrast bird density and vegetation volume, and to describe the composition and conservation value of the bird community before and after the fires. In addition, due to the lack of available studies at this altitude, I also verified which species occurred in winter.

Study Area

The study area at Err (42°27’N, 2°4’E) is located on the Puigmal massif (2,913 m a.s.l.) in the French Pyrenees. These south-west-facing slopes are used for summer grazing and managed by local shepherds and the SUAMME (Service d’Utilité Agricole Montagne Méditerranéenne et Elevage in Prada de Conflent). Two plots (A and B), each of 10 ha (300 x 333 m), were established in spring 1997 in subalpine shrublands dominated by Pyrenean Broom *Cytisus balansae* and a grass, *Festuca paniculata*. The plots were 30 m apart at their corners (Figure 1) and divided into a grid of 50 x 50 m using 1.5-m high stakes planted in the ground. Plots and their surroundings were affected by prescribed burning during

![Figure 1. Location and altitude above sea level of the Err study plots in the Pyrenees. The photograph was taken a few days after the prescribed burning of Plot A in November 1997. The corner of plot B is shown in the bottom left-hand corner of the picture.](image-url)

Ubicació i altitud sobre el nivell del mar de les parcel·les d’estudi d’Err, als Pirineus. La fotografia va ser presa pocs dies després de la crema prescrita de la parcel·la A, el novembre de 1997. La cantonada de la parcel·la B s’observa a la part inferior esquerra de la imatge.
the first (plot A) and the second (plot B) winters after the beginning of the study. These fires were part of a long-term programme run by SUAMME aimed at controlling shrub encroachment in stock-breeding areas in the Eastern Pyrenees.

Plot A was located at 1,900–2,100 m a.s.l. and consisted of a one-meter high dense shrubland, containing 138 young Mountain Pines Pinus uncinata, mostly in the highest part. Its vegetation was 35 years old in 1997, the last wildfire having occurred in 1962-63 (B. Lambert pers. comm. based on aerial photographs). An intense fire set on 28 November 1997 burnt 88% of the plot area (Figure 1) and only seven pines survived this prescribed burning. The plot had been grazed infrequently by cattle and wild ungulates (Cervus elaphus and Rupicapra pyrenaica) before the fire. Afterwards, grazing intensity increased. Plot B was a 7-year-old open shrubland with grassland at 1,850–2,000 m a.s.l. (prescribed burning took place in winter 1989-90, the last wildfire also having occurred in 1962-63). There were <10 pines in this plot, but there were also some old junipers Juniperus communis, that mostly survived the prescribed burning. In all, 78% of the plot area was burnt by a relatively mild fire on 11 March 1999. The plot was quite intensively grazed by horses and cattle both before and after the fire. After fire, a detailed map of burned and unburned vegetation was drawn for both plots and used in both the bird and vegetation studies.

Methods

Vegetation and birds were studied in June 1997–2000, the main period of bird song in the Pyrenees. Vegetation structure was measured once a year before the fire in each plot at 13 fixed sampling stations, regularly distributed at ~100-m intervals. After the fire in each plot 10 stations were selected in well-burned areas and eight in areas containing unburned patches. Each sampling station covered an area of ~1,200 m², in which the foliage cover (in %) of six vegetation layers (0–0.25 m, 0.25–0.5 m, 0.5–1 m, 1–2 m, 2–4 m and 4–8 m) was estimated by comparison with a template (Prodon & Lebreton 1981). A vegetation volume index was then constructed by multiplying cover values by the height of the layer and then summing up the products. After the fire, volume was calculated separately for burned and unburned vegetation, weighted by their corresponding areas, and then pooled to calculate a global index for each plot.

Birds were studied by a mapping method which allows detailed maps of the breeding territories found in a given plot to be generated (Bibby et al. 1997). Using this method, I concentrated on simultaneous songs in order to distinguish neighbouring territorial males. Observations from daily visits were cumulated until there was very little doubt about the precise location of territorial boundaries. The number of visits ranged from eight to 10 per year. These visits lasted on average five hours (including mornings and sometimes afternoons); both plots were mapped every day. Specific densities (breeding pairs/10 ha) were calculated by taking into account not only the territories totally included in the plots, but also the within-plot proportion (0.25-0.5-0.75) of edge territories (i.e. those also extending outside the plot). In addition, global density (total number of breeding pairs/10 ha), bird species richness (number of breeding species) and conservation value were calculated for every plot and year. The conservation value was calculated following the index in Pons et al. (2003), which takes into account the European conservation status or SPEC (BirdLife International 2004) and the log-transformed density (abundance in the original index) of the bird species recorded in the plot. As well, the structure of the bird community was described using a Correspondence Analysis (CA) performed on bird species’ densities per plot-year. The CA was focused on inter-species distances and was performed with Canoco (ter Braak & Smilauer 1998). Finally, five visits to the plots in November 1997 to February 1998 were used to confirm the occurrence of breeding species in the cold season.

Results

Plot A had the densest cover before the fire and as a result the fire here was more severe than in Plot B. Prescribed burning noticeably affected the four vegetation layers from the ground to two meters in height in plot A. Moreover, cover values of burned areas did not reach those of unburned areas during the study (Figure 2). By contrast, cover values of burned areas in plot B...
were similar to those of unburned areas by the second year after the fire.

The most abundant species of both plots was the Dunnock *Prunella modularis*, which reached its highest density (14.5 b.p./10 ha) in the dense shrubland of plot A before the fire (Table 1). Other species present in high densities (≥ 3.0 b.p./10 ha) in particular plots and years were Dartford Warbler *Sylvia undata*, Stonechat *Saxicola torquatus*, Rock Bunting *Emberiza cia* and Ortolan Bunting *E. hortulana*. Nests were found or behaviour indicating nest construction, nest defence or feeding young were observed for all the commonest breeding species. Overall bird density ranged between 13.2 and 28.0 b.p./10 ha and the number of breeding species varied between seven and 11, while the bird conservation value varied between 10.0 and 21.4. All these community variables decreased in the first year after fire, especially in plot A, although tended to return to former values in the following years. Furthermore, there was a remarkable similarity between changes in overall bird density and changes in vegetation volume in both plots (Figure 3).

The first and second axes of the CA accounted for 55.7% and 20.9%, respectively, of the variance of the dataset. The first axis

![Figure 2](image_url)
separates the two plots and appears to be related from right to left to increasing vegetation cover. This is supported by species position (e.g. Dartford Warbler to the left and Skylark *Alauda arvensis* to the right), but also by changes in plot coordinates after the fires (Figure 4). The second axis seems mainly to be related from bottom to top with increasing herbaceous cover in areas with sparse trees, burned or otherwise, in plot A. This interpretation is supported by the position on the graphic of *Tree Pipit* *Anthus trivialis* and Whinchat *Saxicola rubetra* (Figure 4).

If we exclude long distance migrants, most species also occurred in winter, the most important exception being the Dunnock (Table 1). In complete contrast, the Dartford Warbler, a Mediterranean species, was present even after snowfall. Other species that did not breed in the plots but occurred in winter (some of them also in spring) were Meadow Pipit *Anthus pratensis*, Goldfinch *Carduelis carduelis*, Chaffinch *Fringilla coelebs*, Coal Tit *Periparus ater*, Crested Tit *Lophophanes cristatus*, Alpine accentor *Prunella collaris*, Citril Finch *Carduelis citrinella* and Mistle Thrush *Turdus viscivorus*.

### Discussion

Density estimates derived from mapping methods have been used to assess the response of bird communities to disturbances such as timber

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Alauda arvensis</em></td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>NO</td>
</tr>
<tr>
<td><em>Alectoris rufa</em></td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.25</td>
<td>NO</td>
</tr>
<tr>
<td><em>Anthus trivialis</em></td>
<td>-</td>
<td>0</td>
<td>0</td>
<td>0.5</td>
<td>1</td>
<td>0.25</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>LDM</td>
</tr>
<tr>
<td><em>Carduelis cannabina</em></td>
<td>2</td>
<td>1.5</td>
<td>1</td>
<td>1</td>
<td>0.75</td>
<td>0.5</td>
<td>1</td>
<td>0.5</td>
<td>0.5</td>
<td>LDM</td>
</tr>
<tr>
<td><em>Coturnix coturnix</em></td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>LDM</td>
</tr>
<tr>
<td><em>Emberiza cia</em></td>
<td>3</td>
<td>0.75</td>
<td>2</td>
<td>2</td>
<td>3.75</td>
<td>2.5</td>
<td>2.25</td>
<td>3.25</td>
<td>2.25</td>
<td>YES</td>
</tr>
<tr>
<td><em>Emberiza hortulana</em></td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.75</td>
<td>2.5</td>
<td>2.5</td>
<td>2</td>
<td>3</td>
<td>2.5</td>
</tr>
<tr>
<td><em>Lanius collurio</em></td>
<td>3</td>
<td>0</td>
<td>0.75</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>0.75</td>
<td>0.25</td>
<td>2.5</td>
<td>LDM</td>
</tr>
<tr>
<td><em>Lullula arborea</em></td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.25</td>
<td>0.75</td>
<td>1</td>
<td>1.25</td>
<td>1.25</td>
<td>YES</td>
</tr>
<tr>
<td><em>Prunella modularis</em></td>
<td>E</td>
<td>14.5</td>
<td>7.25</td>
<td>6</td>
<td>6.75</td>
<td>8.25</td>
<td>7.25</td>
<td>4.25</td>
<td>2.25</td>
<td>NO</td>
</tr>
<tr>
<td><em>Saxicola rubetra</em></td>
<td>E</td>
<td>0.75</td>
<td>0.25</td>
<td>2</td>
<td>1.75</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>LDM</td>
</tr>
<tr>
<td><em>Saxicola torquatus</em></td>
<td>0</td>
<td>2.25</td>
<td>1</td>
<td>2</td>
<td>1.5</td>
<td>4.75</td>
<td>4.5</td>
<td>2</td>
<td>3</td>
<td>YES</td>
</tr>
<tr>
<td><em>Sylvia communis</em></td>
<td>2</td>
<td>5</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>0</td>
<td>1.5</td>
<td>0</td>
<td>0.5</td>
<td>YES</td>
</tr>
<tr>
<td><em>Sylvia undata</em></td>
<td>2</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.25</td>
<td>0.5</td>
<td>0.75</td>
<td>0.75</td>
<td>YES</td>
</tr>
<tr>
<td><em>Turdus merula</em></td>
<td>E</td>
<td>0.25</td>
<td>0</td>
<td>0</td>
<td>0.5</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>YES</td>
</tr>
<tr>
<td><em>Turdus torquatus</em></td>
<td>E</td>
<td>9</td>
<td>7</td>
<td>8</td>
<td>11</td>
<td>23.25</td>
<td>24.25</td>
<td>17</td>
<td>17.25</td>
<td></td>
</tr>
<tr>
<td>Number of species</td>
<td></td>
<td>57</td>
<td>53</td>
<td>52</td>
<td>52</td>
<td>57</td>
<td>53</td>
<td>52</td>
<td>52</td>
<td></td>
</tr>
</tbody>
</table>

Table 1. The bird community before and after fire (arrow) at the two study plots at Err. Columns show breeding species, SPEC category (BirdLife International 2004), study years for both plots and occurrence in winter (LDM = long-distance migrant overwintering abroad). Rows show data for every species (including breeding densities in b.p./10 ha), number of breeding species, overall density (b.p./10 ha) and the conservation value index (Pons et al. 2003). Specific densities equal to or above 3 b.p./10 ha are shown in bold.
harvesting (Sallabanks et al. 2000, Moorman & Guynn 2001, Fink et al. 2006), coppicing (Fuller & Moreton 1987), farming practices (Genghini et al. 2006), grassland management (Gottschalk et al. 2007) and building development (Jokimaki & Suhonen 1993). However, standard mapping method produces a single density value per plot and year, and plot replication is very time consuming. In the present study, density figures for those species that sing mostly in flight or have large territories in relation to plot size (basically Skylark, Woodlark Lullula arborea and Red-legged Partridge Alectoris rufa), are probably only coarse estimates. Density estimation is also a coarse approximation for semi-colonial and for polygamous species (Bibby et al. 1997, Sutherland et al. 2004). This is the case of the semi-colonial Linnet Carduelis cannabina (Drachmann et al. 2002), which often moved around in small groups in the Err plots, and the Dunnock, which has a highly variable mating system that includes monogamy and different forms of polygamy. In a 10-year study in Britain, there were 21% more males than female Dunnocks in a population (Davies 1992). If this proportion were applied to the Err populations before the fire, the number of females and of potential nests would be smaller. For example, the 14.5 male territories/10 ha in plot A before the fire would potentially correspond to 12 nests. It is possible that this figure could be a more reliable estimator of the density of reproductive units for this species, although it would not change the marked dominance of the Dunnock in this rangeland bird community.

The avifauna of the subalpine rangelands studied was composed of a large number of temperate and boreal species, with the addition of a few Mediterranean elements such as Red-legged Partridge and Dartford Warbler. Most breeding species also occurred in winter. The Err plots lay a few kilometres from the edge of the study area of the Catalan Winter Bird Atlas (Herrando et al. 2011) and three species were found at even higher altitudes than the maximum recorded in this atlas: Dartford Warbler was observed during the five ‘winter’ visits to the plots (reaching 2,020 m a.s.l. on 23-11-1997), Stonechat was recorded at 1,900 and 1,780 m a.s.l. on 22-11-1997 and 16-1-1998, respectively, and Meadow Pipit was found at 1,900 m a.s.l. on 28-11-1997. The mild microclimate of this southwest-facing hillside, where the snow does not last long (B. Lambert pers. comm.), may help explain this pattern. In severe weather (cold snaps, high wind, heavy snowfall) birds can temporarily descend in altitude to feed and avoid cold stress (Senar & Borras 2004). In the eastern Pyrenees these movements can be sudden, due to the high elevation gradient and to the proximity of Mediterranean-climate lowlands.

Both plots had suffered previous fires and were grazed by livestock. However, the more recent fire and the higher grazing pressure in plot B may explain why this plot was covered by a very open shrubland in contrast to the dense shrubland of plot A. Bird species composition
was therefore different and only five (Linnet, Rock Bunting, Dunnock, Stonechat and Blackbird *Turdus merula*) out of 14 species were shared between plots. The nine exclusive species were linked to grassland habitats (five species) in plot B and mostly to shrubland (four species) in plot A. The trend towards an increase in bird species as shrub encroaches in mountain rangelands (Laiolo *et al.* 2004) does not seem to be fulfilled when comparing before the fire plot A with plot B. This is probably because plot B consisted of a mosaic of shrub and grass patches that favours a higher beta diversity of birds.

Grass, shrub and tree layers in plot A were severely affected by fire and did not reach their former values for three years after the fire, although the cover below 50 cm was already important at the end of the study. As a consequence, shrubland bird species disappeared or severely reduced their densities (the case of the Whitethroat *Sylvia communis*), whereas open-habitat species density increased. In addition, four new species arrived in plot A in the period from the first to the third year after the fire. Prescribed burning was less severe in plot B and cover reduction was fairly moderate in its burned areas. Accordingly, the bird species composition in this plot was modified less, although changes were apparent along the first axis of the CA (Figure 4) probably due to the post-fire disappearance of the Dartford Warbler. Furthermore, overall bird density changed in parallel with vegetation volume (Figure 3). This pattern, consistent between plots and before and after fire, suggests that resources required by birds are strongly associated to variations in habitat structure (James & Wamer 1982). Functional changes in the bird communities
have not been evaluated in this study, but are interesting aspects that could be explored further after shrubland burning.

Before the fire, the bird conservation value was slightly higher in plot B, despite the lower bird density, since this plot had more SPEC 2 and 3 species (those with unfavourable conservation status in Europe). The conservation value decreased after fire but tended to recover afterwards in both plots, with greater species richness and abundance, a finding that coincides with larger scale studies (Pons & Clavero 2010). The present study, in particular, contributes to the knowledge of bird assemblages in managed high mountains. In these areas, the preservation of traditional activities (Lasanta-Martínez et al. 2005), including burning and livestock grazing, helps to maintain the open habitats necessary for several declining bird species, and helps counteract the current general trend of afforestation (Roura et al. 2005).

Acknowledgements

Thanks are due to Bernard Lambert (SUAMME) for providing information and making the prescribed burning possible, as well as to Bernard Batailler, Sussi Abel and Roger Prodon for support during the fieldwork. The manuscript benefited from the comments made by Sergi Herrando and two anonymous referees. This study was supported by DIREN (Evaluation des MAE-Estive) and by Spanish Ministry of Science and Innovation (CGL2008-05506/BOS).

Resum

La comunitat d’ocells d’una pastura subalpina abans i després del foc: cremes prescrites als Pirineus Orientals

Les pastures d’alta muntanya acullen poblacions importants d’espècies amenaçades d’ocells, alhora que es veuen afectades per importants canvis en els usos del sòl. Aquest treball estudia la comunitat d’ocells nidificants de dues parcel·les de matollar a 1.850-2.100 metres sobre el nivell del mar als Pirineus. Es van cartografiar els territoris de reproducció durant quatre anys, abans i després d’una crema prescrita encaminada a augmentar el valor de pastura de la zona. Les espècies més abundants (amb ≥ 3 parelles /10 hectàrees en almenys una parcel·la i un any) van ser el Pardal de Bardissa Saxicola torquatus, el Sit Negre Emberiza cia i l’Hortolà Emberiza hortulana. La parcel·la de matollor obert contenia un nombre similar d’espècies d’ocells nidificants (10 vs. 9), però una menor densitat (23 vs. 28 parelles/10 ha) que la parcel·la de matoller dens. La majoria de les espècies reproductorres també hi van ser presents a l’hivern. Després del foc, el nombre d’espècies d’ocells, la densitat total i el valor de conservació (un índex que inclou la densitat de totes les espècies i les seves categories d’interés de conservació a Europa) es va reduir, però va tendir a recuperar-se posteriorment. Aquests resultats poden ajudar a entendre la composició i dinàmica de les comunitats d’ocells en àrees de muntanya gestionades.

References


