

The impact of high-altitude ski-runs on alpine grassland bird communities

ANTONIO ROLANDO, ENRICO CAPRIO, ENRICO RINALDI and
IVAN ELLENA

Dipartimento di Biologia Animale e dell'Uomo, via Accademia Albertina 13, I-10123 Torino, Italy

Summary

1. Treeless mountainous areas at high altitudes have increased in value as wildlife habitat, but they are affected and increasingly threatened by ski-resort developments, in particular by the construction and enlargement of ski-pistes. To assess the extent of this threat, we compared bird diversity and community composition in circular plots centred on (i) ski-runs of recent construction, (ii) grassland habitats adjacent to ski-runs and (iii) natural grassland habitats far from the ski-runs.

2. Plots located in natural grasslands supported the greatest bird species richness and diversity and the greatest grassland species density, whereas those set in ski-pistes presented the lowest values. Plots located beside ski-runs did not support smaller numbers of bird species and diversity than plots of natural areas, but they supported a significantly lower bird density. This suggests that ski-pistes, besides exerting a negative direct effect on the structure of local bird communities, may also exert an indirect, detrimental effect on bird density in nearby patches.

3. Generalized linear models showed that species richness and diversity, and abundance of grassland species were best modelled by combinations of factors, including habitat type (the three categories defined above) and altitude. The category ski-run, in particular, was negatively correlated with species richness, diversity and abundance, and altitude was negatively associated with richness and diversity.

4. Richness and abundance of arthropods were significantly lower in ski-pistes than in the other plot types. Given that many invertebrates were preyed upon by birds, low food availability on ski-runs may be one of the factors reducing the attractiveness of these patches to birds.

5. *Synthesis and applications.* Retaining the avifauna around ski-resorts is likely to involve developing new, environmentally friendly ways of constructing pistes, such as only removing rocks and/or levelling the roughest ground surfaces, to preserve as much soil and natural vegetation as possible. Restoration of ski-pistes should promote the recovery and maintenance of local vegetation to enhance invertebrate and bird assemblages. In order to not compromise the safety of the ski-runs, it may be necessary to control encroaching shrubs through pruning and/or cattle grazing.

Key-words: alpine bird communities, diversity, grasslands, high altitudes, ski-pistes

Journal of Applied Ecology (2007) **44**, 210–219

doi: 10.1111/j.1365-2664.2006.01253.x

Introduction

High-altitude treeless zones are, by definition, alpine zones, irrespective of whether they are located in the

Alps or not. These zones have increased in value as wildlife habitat, given changes in habitats and processes at lower altitudes (Martin 2001). In Europe, high-altitude environments in the Alps remained relatively undisturbed until the beginning of the last century. Although alpine pastoral activities have been practised for at least 6000 years (Lichtenberger 1994), it is likely that their influence at high altitudes was, and still is, relatively weak. Recent data suggesting that grazing in

the alpine belt has little effect on typical open habitat bird species confirm this assumption (Laiolo *et al.* 2004). Pastoral activities have shaped the alpine environment mostly below the timberline, where shrublands and forests have been cleared or burned over large areas to increase the availability of grasslands for livestock. Hence the impact of human activities on the high-altitude environment is more recent, and is essentially the result of the development of winter sport resorts. From the beginning of the last century, the construction of buildings, access roads and ski-pistes, and the direct damage to fragile vegetation by machinery and skiers, has severely disturbed this environment (Simons 1988; Mackenzie 1989). Ski-runs, in particular, have brought about considerable damage: bulldozers and power shovels are used for soil removal to provide suitable slopes for skiers, and artificial seeding, along with the application of chemicals, is carried out to control for soil erosion (Urbanska, Erdt & Fattorini 1998; Barni, Siniscalco & Montacchini 2002; Tsuyuzaki 2002). After construction, skiing and ski-piste preparation by snow-grooming vehicles are likely to cause further mechanical and thermic damage to the vegetation and the soil (Cernusca *et al.* 1990; Rixen *et al.* 2004; Rixen, Haeberli & Stoeckli 2004). Moreover, in summer, cutting of shrubs and machine-grading are carried out at regular intervals to remove obstacles and level rough or bumpy soil surfaces, producing further damage to vegetation (Bayfield 1996; Titus & Tsuyuzaki 1999). As ski-tourism is one of the most important economic factors in European alpine regions, the area affected by ski-pistes is still increasing (Abegg *et al.* 1997; Elsasser & Messerli 2001; Wipf *et al.* 2005).

However, although ski-runs have become a common feature in alpine landscapes, there is a paucity of research on their effects on wildlife (Delmas 1986; Watson & Moss 2004). Ski-pistes may be found along a broad altitudinal range that touches, in particular, the montane and alpine belt (the woodland and the treeless zones, respectively). The construction of ski-runs below the timberline is particularly damaging to wildlife because forest tracts are abruptly clear-cut, thus producing habitat fragmentation and creation of edges (i.e. transition zones between adjacent habitats). Despite the general assumption that species richness and density increase at the border between different habitats (Odum 1971), previous results have demonstrated that ski-runs in the montane belt can produce a negative edge effect on birds. Plots at the edges of coniferous forests, in fact, can have lower bird species richness and diversity than those located in the forest interior or at the edges of pastures (Laiolo & Rolando 2005). In theory, the impact of ski-developments above the timberline may be less detrimental because the construction of ski-pistes does not require the permanent elimination of the habitat itself, as occurs in the montane belt (where the forest is totally and permanently removed from the track). In the short term, ski-pistes at high altitude may be restored through artificial seeding and, in the long

term, they may be slowly recolonized by the original local grassland vegetation. However, the success of revegetation declines with altitude (Urbanska 1997a), and restoration is also constrained by management to keep pistes smooth and without shrubs. Furthermore, environmental conditions at high altitudes are severe and avian communities, which are made up of the few species that can live there, mostly during the short summer period, may consequently be more fragile than communities at lower altitudes (Martin & Wiebe 2004).

This study is part of a large project investigating the effects of ski-pistes on alpine avifauna and was undertaken to provide information on the effect of ski-runs on high-altitude grassland bird communities in the Alps. We compared bird diversity and community composition in circular plots centred on (i) ski-runs of recent construction, (ii) grassland habitats adjacent to ski-runs and (iii) natural grassland habitats far from the ski-runs.

In particular, we examined whether avian diversity is lower in the pistes than in the other habitats (as expected, assuming the construction of ski-runs causes a profound direct disturbance) and whether it was higher in grasslands further from rather than near the pistes (as expected assuming ski-runs also exert an indirect disturbance). We also investigated bird-habitat relationships, to identify the important floristic or structural components of the habitat that determine the structure and organization of grassland bird communities at high altitudes, and we surveyed habitat use to detect whether artificial human structures are avoided by birds. Finally, to provide some information on the effect of ski-run construction on arthropods, i.e. potential bird prey, we also measured insect and spider abundance in the three plot types considered.

Methods

STUDY AREA

The study was carried out in the grasslands of the western Italian Alps, located at the top end of Susa Valley (Piedmont) and around the Monte Bianco and Monte Rosa massifs (Aosta Valley). In particular, we surveyed seven localities, at altitudes between 2010 m and 2892 m a.s.l., usually above the treeline. In February 2006, the XX Olympic Winter Games took place in the ski-localities at the top end of Susa valley. Since the 1970s, ski-runs have been constructed on high-altitude grasslands by scraping off the vegetation and topsoil and, in some instances, artificially seeding with herbs. Around the timberline, coniferous forests are thin and dominated by larch *Larix decidua*. Above the timberline, pastures are dominated by grasses (Gramineae, *Festuca* and *Nardus* spp.). During summer, spring gentian *Gentiana verna*, yellow gagea *Gagea fistulosa*, mountain buttercup *Ranunculus montanus*, alpine pasque flower *Pulsatilla alpina* and bistorta *Polygonum bistorta* are abundant. Shrubs are represented by juniper *Juniperus communis*,

alpen rose *Rhododendron ferrugineum* and, to a lesser extent, bilberry *Vaccinium myrtillos* and bearberry *Vaccinium uliginosum*. Above 3000 m a.s.l., rocks, scree and snowbeds dominate the alpine landscape, up to the highest peaks and glaciers of Monte Bianco (4810 m a.s.l.), Monte Rosa (4634 m a.s.l.) and Cervino (4478 m a.s.l.).

BIRD AND HABITAT SAMPLING DESIGN

Field work was carried out in June and July 2004 and 2005. Grasslands in this study were relatively heterogeneous landscapes, with different topography, rock and shrub cover. In these heterogeneous conditions, traditional survey methods, such as fixed radius point counts and transects, may be inappropriate, as differences in bird detectability would bias intersite comparisons. In keeping with a previous study (Laiolo *et al.* 2004), we therefore used a standardized area count method (Bibby *et al.* 2000), surveying birds in circular plots of radius 50 m. Counts lasted 15 min, during the first 10 min of which the observer stood still and quiet at the centre of the plot, as in standard point counts. In the last 5 min of the count, the observer moved around, to flush secretive and non-singing individuals, and stopped at suitable vantage points to look and listen, recording all birds seen or heard within the plot. This method is particularly useful when comparing the community structure of habitats that differ in vegetation and rock density. Each census plot was visited twice (in June and July of the same year); the total number of species from the two censuses was used as a measure of species richness, and the highest numbers of individuals were used as a measure of bird species abundance per plot; overall 223 plots were visited (104 in 2004 and 119 in 2005).

The exact location of plots was established in the field by means of a global positioning system (GPS), Garmin eTrex® Navigator (Garmin Corporation, Olathe, KS). Plots along daily transects were located at a minimum distance of 400 m from each other; we were very careful not to record the same individuals on neighbouring plots by tracking birds with binoculars if they were flushed off the plot.

Three types of plots were defined: plots centred on the ski-runs ($n = 80$), grassland habitats adjacent to ski-runs ($n = 71$, at a distance of 100–200 m from pistes) and natural grassland habitats far from the ski-runs and located in districts without ski-developments ($n = 72$, at a distance of 1–10 km from pistes).

As for ski-pistes, we chose plots where the impact of ski-run construction was severe because the pre-existent habitat had been removed (in the case of shrub-beries and screes) or heavily damaged (in the case of meadows); hence we mostly selected ski-runs or tracts of ski-runs of recent construction (1 year in the case of a few Olympic pistes, 2–10 years in the other cases) that were characterized by poor grass cover and no bush cover. Ski-run strips were 20–200 m wide (mean 73.3 ± 6.22 SE); when the strip was narrower than 100 m, a variable portion of habitat at the side of the

ski-run was included in the plot. The plots located to the side of the pistes and in natural habitats were placed at random, each plot containing only one habitat type.

Avian communities were described in terms of bird species richness (S), species diversity (Shannon index $H' = -\sum p_i \ln p_i$, where p_i is the relative frequency of species i) and density of two ecological groups of birds: grassland species (birds that require open fields for feeding and, in most instances, for breeding) and non-grassland species (species that require shrubby or open forest areas and species that use both grassland and woodland).

For each plot we calculated seven habitat structure variables: percentage of shrub cover, percentage of stone–rock cover, percentage of soil–rubble cover (percentages estimated by eye), diversity of the vegetation and of the habitat [Shannon index, where p_i is the relative frequency of vegetation (grass, juniper, alpen rose and other bushes) and habitat (grass, bush, rock and soil–rubble) categories], mean height of the vertical component of the habitat (mean of 20 measurements of grass, shrubs, rocks and soil–rubble per plot, shared according to the relative cover percentages and recorded with a wooden dowel subdivided into 1-cm units) and heterogeneity of the vertical component ($CV = SD/\text{mean} \times 100$). We also took into account the distance between the centre of the plot and the nearest building (cable-car and ski-lift stations, Alpine huts, refuges, etc.) and altitude (both in metres).

A Bushnell laser telemeter (accuracy ± 1 m) (Bushnell Outdoor Product-Overland Park, KS) was used to estimate distances of birds and buildings from the centre of the plot.

We also assessed insect and spider availability per plot by counting all individuals detected in 1 m² of vegetation (four categories: grass, juniper, alpen rose and other shrubs). Six 1-m² counts (three per visit) were made for each category of vegetation. The total number of invertebrates found in each kind of vegetation was multiplied by the relative vegetation cover in the plot; a final invertebrate abundance for each plot was obtained by summing up abundances of each category of vegetation.

DATA ANALYSIS

Differences between plot types

We tested for differences in mean bird species richness and diversity and density of grassland and shrub/woodland species between the three plot types (ski-run, adjacent to ski-run, natural habitat) by means of nested ANOVA (plots nested into seven localities nested into 2 years). The values of Shannon diversity were transformed to logarithms [$y = \log(x + 1)$] and those of species richness and density of the three ecological groups to square roots [$y = \sqrt{(x + 0.5)}$] to attain normal distributions (Sokal & Rohlf 1995).

The same kinds of analysis and data transformation were also used to test for differences in mean arthropod richness, diversity and density between the three plot

types. Richness and diversity measures, in this case, did not refer to the different species (which would have required an impractical sampling and classification effort), but simply to the seven major arthropod groups (spiders, grasshoppers, butterflies, flies, ants, bees/wasps and others) we found in the study area.

Nested ANOVA was computed in an SPSS package for Windows (SPSS Inc. 2003).

Relationships between bird community and habitat

Principal component analysis (PCA; Gaunch 1984) was chosen to minimize the effects of multicollinearity and to reveal patterns in the data for habitat structure (i.e. the seven variables listed above) using standardized data (zero mean and unit standard deviation; Legendre & Legendre 1998).

We used generalized linear models (GLM) to test for the effects of habitat structure (three scores of PCA), distance from buildings, altitude and plot type (categorical variable defining ski-pistes, plots to the side of the ski-pistes and natural grassland) on bird species richness and diversity and abundance of the two ecological groups of birds. Akaike's information criterion (AIC; Akaike 1973) was used to select the most appropriate models, i.e. the best fit to the available data set. AIC is based on the principle of parsimony and helps to identify the model that accounts for the most variation with the fewest variables; the model that best explains the data is that with the lowest AIC. This information criterion is one of the most powerful approaches for model selection from a set of alternative plausible models and it solves the problems of stepwise model selection because no sequential statistical test is conducted (Burnham & Anderson 1998).

We also included arthropod indices of abundances in the AIC models. The rationale for this approach is that the arthropods we recorded in the different vegetation types were eaten by at least some of the bird species. Therefore we kept GLM models including arthropod abundance separate from the others.

Generalized linear models and AIC were calculated using the R package (Ihaka & Gentleman 1996; R Development Core Team 2005).

Relationships between bird species and habitat

The density of bird species at each plot was compared to local habitat characteristics using canonical correspondence analysis (CCA). This is a multivariate technique that ordines plots using both a primary matrix of species densities and a secondary matrix of environmental variation (ter Braak 1986; Legendre & Legendre 1998). The primary matrix included 21 species, i.e. those contributing at least four individuals to the data set; the secondary matrix included habitat descriptors (habitat structure variables, altitude and distance from buildings). To assess the significance in the CCA axes, we used Monte Carlo simulations to test the hypothesis

that there was no correlation between the primary (bird) and secondary (environment) matrices: *P*-values were based on the proportion of 1000 Monte Carlo simulations with an eigenvalue greater than the observed eigenvalue. CCA was performed using PC-ORD (McCune & Mefford 1999).

Results

Overall, 889 individuals from 33 species were sampled in 223 plots (see Appendix 1 in the supplementary material for a complete list of bird species and relative frequencies of occurrence, and Appendix 2 in the supplementary material for the distribution of points within plot types and localities). Water pipit *Anthus spinoletta* (L.), northern wheatear *Oenanthe oenanthe* (L.), black redstart *Phoenicurus ochruros* (S.G. Gmelin), snow finch *Montifringilla nivalis* (L.) and skylark *Alauda arvensis* L. were the most common species, with frequencies of occurrence higher than 10%.

DIFFERENCES BETWEEN PLOT TYPES

Ski-run, side of ski-run and natural habitat plots showed significant differences in terms of species richness and diversity, while no significant differences between localities were found (nested ANOVA). Plots located in natural open habitats supported the greatest bird species richness and diversity and the greatest grassland species density, whereas those set in the ski-pistes had the lowest values (Table 1 and Fig. 1). Least-squares deviation (LSD) posthoc tests showed that species richness and diversity were not significantly different between plots located in natural habitat and those adjacent to ski-runs, whereas the density of grassland species was significantly greater in natural grassland.

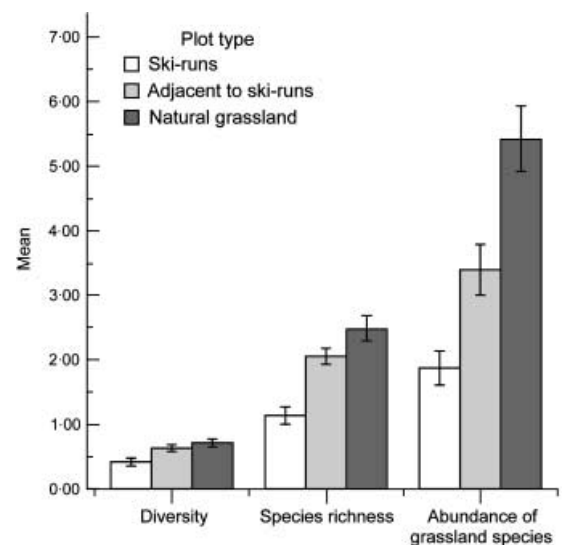


Fig. 1. Differences in diversity, richness and abundance of grassland bird species between plot types (natural habitats far from the ski-runs, habitats adjacent to ski-runs and ski-runs). Values are means. Error bars are \pm standard errors.

Table 1. Mean \pm SE bird species richness and diversity and abundance of grassland and non-grassland species on natural grassland habitats far from ski-runs, grassland habitats adjacent to ski-runs and ski-runs. Interplot differences were tested with nested ANOVAs (plot type nested into locality). LSD posthoc tests were used for pairwise comparisons of means. * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$. In nested ANOVAs, comparisons between years (species richness $F_{1,12} = 0.16$, species diversity $F_{1,12} = 0.18$, abundance of grassland species $F_{1,12} = 0.74$ and abundance of non-grassland species $F_{1,12} = 0.93$) and between localities ($F_{12,25} = 1.45$, $F_{12,22} = 1.73$, $F_{12,25} = 0.90$, $F_{12,25} = 1.40$, respectively) were not significant

	(1) Natural grassland	(2) Adjacent to ski-runs	(3) Ski-runs	Interplot differences (nested ANOVA)	Significant pairwise comparison at $P < 0.05$
Species richness	2.49 \pm 0.18	2.03 \pm 0.13	1.12 \pm 0.13	$F_{25,184} = 3.45^{***}$	(1) vs. (3) (2) vs. (3)
Species diversity	0.71 \pm 0.06	0.63 \pm 0.05	0.42 \pm 0.060	$F_{22,148} = 1.75^*$	(1) vs. (3) (2) vs. (3)
Abundance of grassland species	5.43 \pm 0.85	3.39 \pm 0.38	1.85 \pm 0.30	$F_{25,184} = 2.04^{**}$	(1) vs. (3) (2) vs. (3) (1) vs. (2)
Abundance of non-grassland species	0.53 \pm 0.17	0.67 \pm 0.16	0.29 \pm 0.12	$F_{25,184} = 1.52$	

Table 2. Mean \pm SE of arthropod richness, diversity and abundance on natural grassland habitats far from ski-runs, grassland habitats adjacent to ski-runs and ski-runs. Interplot differences were tested with nested ANOVAs (plot type nested into locality). LSD posthoc tests were used for pairwise comparisons of means. In nested ANOVAs, statistical comparison between years (arthropod richness $F_{1,12} = 0.75$, diversity $F_{1,12} = 0.69$, abundance $F_{1,12} = 0.23$) were not significant. Arthropod richness ($F_{12,25} = 1.92$) and diversity ($F_{12,25} = 2.01$) did not differ between locations but abundances did differ significantly ($F_{12,25} = 2.72$, $P = 0.015$). * $P < 0.05$, *** $P < 0.001$

	(1) Natural grassland	(2) Adjacent to ski-runs	(3) Ski-runs	Interplot differences (nested ANOVA)	Significant pairwise comparison at $P < 0.05$
Richness	3.47 \pm 0.19	3.39 \pm 0.18	2.44 \pm 0.19	$F_{25,184} = 2.26^{***}$	(1) vs. (3) (2) vs. (3)
Diversity	0.74 \pm 0.06	0.70 \pm 0.05	0.60 \pm 0.05	$F_{25,168} = 1.56^*$	NS
Abundance	31.00 \pm 5.05	33.90 \pm 4.47	16.84 \pm 3.63	$F_{25,184} = 1.77^*$	(1) vs. (3) (2) vs. (3)

Table 3. Results of PCA carried out on seven habitat structure variables. The highest factor loadings are in bold type

Variables	Factor loadings		
	PC1	PC2	PC3
Percentage of soil-rubble cover (SoC)	-0.570	0.507	0.353
Percentage of stone-rock cover (StC)	0.509	-0.637	0.443
Percentage of shrub cover (ShC)	0.742	0.562	-0.099
Diversity of the habitat (H'hab)	0.609	0.290	0.310
Diversity of the vegetation (H'veg)	0.772	0.544	-0.056
Heterogeneity of the vertical component (CVHe)	-0.328	0.297	0.729
Mean height of the vertical component (He)	0.607	-0.501	0.234
Percentage variation explained	36.8	24.3	14.5

Differences between localities and years were not significant (Table 1). The great scarcity of birds present in ski-run areas was emphasized even more when we consider that in 35% of ski-run plots not one bird was detected and that 69.2% of birds recorded on these plots were located outside the actual strip of the ski-run.

The three plot types also showed significant differences in terms of richness, diversity and abundance of arthropods (Table 2), with a single significant difference between localities in the case of abundance ($F_{12,25} = 2.72$, $P = 0.015$). LSD posthoc tests showed that arthropod richness and abundance were significantly lower in ski-pistes than in the other plot types, whereas they were not significantly different between plots located in natural habitat and those adjacent to the ski-runs (Table 2).

RELATIONSHIPS BETWEEN BIRD COMMUNITY AND HABITAT

The first three principal components (PC1, PC2, PC3) accounted for 75.6% of the total variation in the habitat structure matrix, with eigenvalues > 1 (Table 3). The percentage of shrubs and vegetation heterogeneity showed the highest correlation with PC1 scores (suggesting a gradient of shrub encroachment, from meadows to shrubby areas), percentage of stone-rock cover provided the major negative loading on PC2, and heterogeneity of the vertical component showed a high positive correlation with PC3 (Table 3). The relative position of centroids (i.e. the average location of survey plots in ordination space) in the biplot determined by

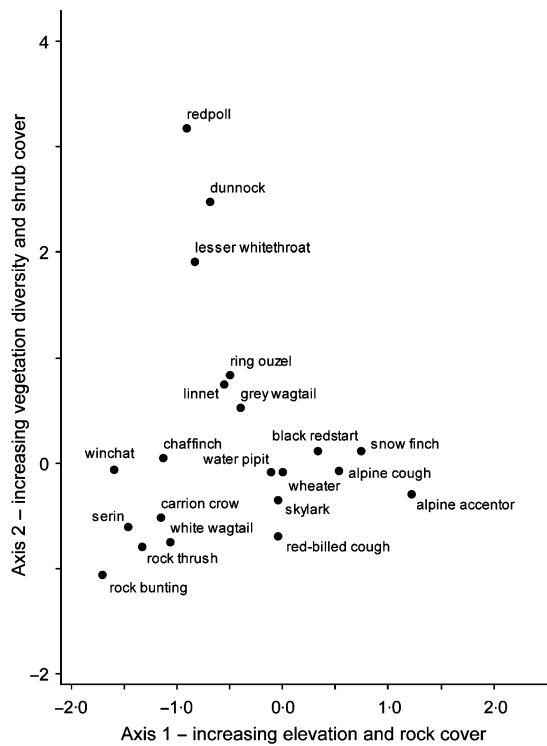


Fig. 3. Ordination plot of bird community distribution along the first two axes of a canonical correspondence analysis. The first axis represents an altitude and rock cover gradient (increasing from left to right) whereas the second axis represents a vegetation diversity and shrub cover gradient (increasing from bottom to top).

the black redstart, alpine chough and snow-finch also frequently nested in buildings and pillars (11, five and four nests, respectively).

Discussion

Winter sports, because of changes they cause both at habitat and at landscape levels, represent a potentially serious threat to the conservation of wildlife habitat in the Alps. Vegetation is heavily damaged by skiing and ski-piste preparation by snow-grooming vehicles (Cernusca *et al.* 1990; Rixen *et al.* 2004; Rixen, Haeberli & Stoeckli 2004). Also, plant diversity in ski-runs is lower than that of nearby grasslands (Urbanska, Erdt & Fattorini 1998) and successional stages are disrupted (Tsuyuzaki 2002).

The effects of ski-runs on the alpine fauna of these areas are poorly studied and more varied. It has been demonstrated previously that, as a consequence of ski-development, the breeding success of the ptarmigan *Lagopus mutus* (Montin) can decline (Watson & Moss 2004) and the local range of the black grouse *Tetrao tetrix* L. may shrink (Menoni & Magnani 1998; Zeitler & Glanzer 1998). A previous study of ours also showed that ski-runs lower forest bird species richness and diversity through a negative edge effect (Laiolo & Rolando 2005).

Alpine areas have increased in relative value as wildlife habitat, given changes in habitats and processes at low altitudes (Martin 2001). To date, the effects of ski-developments on high-altitude animal communities are poorly known. This study focuses for the first time on birds that, in summer, occupy open habitats set above the timberline. The results show that bird species richness and diversity and abundance of grassland species are significantly lower on ski-runs than in areas adjacent to ski-runs and in natural areas without ski-developments. Species richness and diversity and abundance of grassland birds were best modelled by combinations of factors including plot type, and confirmed, in particular, that ski-pistes were negatively associated with all three community parameters considered. In addition, no birds were detected in about one-third of ski-run plots and most birds recorded on these plots were located outside the actual ski-run. Ski-pistes also exert a direct, detrimental effect on local arthropods, whose richness, abundance and diversity were significantly lower than those found in grasslands off-piste and in natural areas. Many insects sampled in the study area (especially grasshoppers and beetles) are known to be part of the diet of several grassland alpine bird species (Cramp 1988; Cramp & Perrins 1994; Rolando & Laiolo 1997). GLM analyses also suggested arthropod abundance can affect the organization of bird communities. The low food availability of ski-runs may thus be a key reason why few birds occur on these sites. We acknowledge that differences in the detectability of insects and spiders between different vegetation types may have affected these counts. However, differential prey detectability can reflect on birds' feeding performances as well. Moreover, arthropod availability in pistes, given the paucity of vegetation cover, may have been overestimated in comparison with the other plot types, and this strengthens our conclusions about the negative effect of pistes on arthropods and birds.

The ski-pistes that we sampled were devastated environmental patches, from which shrubby and herbaceous native vegetation had been removed and/or severely damaged and artificial seeding, if any, had produced a very poor grass cover. Hence, in theory, we may have underestimated the value of such areas to bird populations. However, in the Alps, ski-pistes completely recolonized by original local grassland vegetation are infrequent because restoration is regularly counteracted by management to keep pistes smooth and shrub-free. Recovery may also be severely limited by maintenance interventions on ski-facilities. During our surveys, for instance, old ski-facilities had been replaced by new ones (e.g. ski-lifts substituted by chair-lifts) while pistes, after being used as access roads, were often levelled in several places. Most of the pistes we selected had artificial snow applied to them. It has been demonstrated that ski-piste preparation by machine-grading and the use of artificial snow can result in deviations from the natural structure and composition

of alpine vegetation (Wipf *et al.* 2005). We are therefore confident that the pistes that we selected accurately represent the damage that may be suffered by high-altitude alpine ecosystems as a result of highly developed ski-resorts.

Plots adjacent to the ski-runs did not support lower numbers of bird species and diversity than plots in natural areas, but they supported a significantly lower bird density. This suggests that ski-pistes at high altitudes, besides exerting a direct negative effect upon the structure of their avifauna, may also exert an indirect, detrimental effect on the number of individuals utilizing nearby areas. Similar results were obtained for pistes in the montane belt, which produce a negative edge effect on birds of coniferous forests (Laiolo & Rolando 2005). Hence the impact of ski-runs upon birds is always negative, irrespective of altitude and the communities considered. This is of particular concern because in Europe the area affected by ski-pistes is large and increasing (Abegg *et al.* 1997; Elsasser & Messerli 2001; Wipf *et al.* 2005). It should be borne in mind that the area of the Alps is about 190 000 km² and that the total length of pistes for Alpine skiing might be several thousands of kilometres. For instance, the total length of pistes is about 4000 km in Italy, 2000 in Switzerland and 6000 in Austria (data inferred from the official regional or national tourist web sites).

Altitude is a good predictor of species richness and diversity, which are lower at high altitudes. Typical alpine species, which are probably better adapted to harsh environmental conditions of high altitudes, are very few (see below, individual species distribution). Alpine zones are extreme environments for many bird species, so human interference should be reduced to a minimum to benefit wildlife conservation.

Altitude was also a good predictor of individual species' distribution. Only typical alpine species, such as the alpine chough, alpine accentor and snow finch, were found at very high altitudes, whereas other species, such as the rock bunting, whinchat and citril finch, only occurred at lower altitudes, around the timberline. The distribution of these species may also be explained by the proportion of stone-rock cover, although this correlation may also result simply from the increasing abundance of scree with increasing altitude. The presence of shrub species, for example linnet, dunnock and lesser whitethroat, was strongly dependent on high vegetation diversity and shrub cover.

In this study, in keeping with previous works (Laiolo *et al.* 2004; Laiolo & Rolando 2005), we measured bird densities and species richness and diversity, rather than breeding success (Watson & Moss 2004). The aim was to investigate the value of different grassland plots for the bird community as a whole. The main result is that ski-runs in the alpine belt do negatively affect bird diversity *per se* and this has to be viewed as detrimental, irrespective of the effects on breeding success.

CONSERVATION IMPLICATIONS

Alpine grasslands of high altitude represent a semi-natural habitat, where pastoral activities have been practised for thousands of years (Lichtenberger 1994) and, although declining (Cernusca, Tappeiner & Bayfield 1999; Dirnböck, Dullinger & Grabherr 2003), are still practised. Abandonment of grazing has few effects on bird species diversity and abundance in the alpine belt (Laiolo *et al.* 2004) and therefore the most detrimental present-day anthropologic threat to bird communities of these habitats is most probably represented by the development of ski-resorts. The construction of ski-pistes, in particular, can cause a substantial loss of feeding habitats for grassland birds. Throughout Europe, habitat loss of semi-natural grasslands is recognized as a serious threat to many rare and declining animal and plant species (Fuller 1987; Labaune & Magnin 2002). More than one-quarter (7/26) of the grassland and shrub species recorded in this study have been classed as species of European conservation concern, i.e. SPEC2 and SPEC3 categories (*sensu* BirdLife International 2004). In particular, rock partridge *Alectoris graeca* (Meisner), red-billed chough *Pyrrhocorax pyrrhocorax* (L.) and wheatear are declining species, whereas black grouse, rock-thrush *Monticola saxatilis* (L.), skylark and rock bunting are considered as depleted. Moreover, other species, although not classified as under threat (i.e. not SPEC species), are known to be declining in several European countries [notably whinchat, ring ouzel *Turdus torquatus* L., water pipit, tree pipit *Anthus trivialis* (L.), linnet and yellowhammer *Emberiza citrinella* L.].

To weigh the effect of the development of ski-resorts above the timberline, the construction of ski-facilities, which may have no immediate negative impact on certain passerines species (which use anthropogenic structures for perching and nesting), should be discerned from the construction of ski-pistes, which have a detrimental effect on all bird species of the alpine belt. From this point of view, retaining the avifauna of these zones is likely to involve developing new, environmentally friendly ways of constructing pistes, such as only removing rocks and/or levelling the roughest (and most dangerous for skiers) ground surfaces, in order to preserve as much soil and natural vegetation as possible. Where necessary, ski-pistes should be restored through management to prevent soil erosion and promote the recovery of local vegetation. Transplantation of single plants (Urbanska 1997b; Florineth 2002) or turf transplants of wild alpine species to create a wholly autochthonous community (Urbanska 1997a, 1997b; Conlin & Ebersole 2001; Ebersole, Bay & Conlin 2002) are particularly promising methods. Mount Rainier National Park (Washington, USA), for instance, has adopted an ambitious programme of alpine herb and shrub restoration (Rochefort & Gibbons 1992). Once the natural revegetation of pistes is achieved, vegetation should be preserved without compromising the safety of the

ski-runs. Shrubby vegetation, for instance, can be kept low through direct pruning and/or cattle-grazing. The value of these measures in promoting natural diversity of the avifauna should be tested experimentally, through co-ordinated management and improved co-operation between sport- and land-management agencies, which should have the mandate to maintain native biodiversity while simultaneously managing land for commodity production and recreation. Their effectiveness, in particular, should be verified on large resident species (namely the black grouse and ptarmigan), whose preservation is dramatically threatened by ski-resort developments, which may increase the abundance of generalist predators (Marti 1985; Delmas 1986; Menoni & Magnani 1998; Zeitler & Glanzer 1998; Watson & Moss 2004). Eventually, studies on the ecological impacts of alpine sport resorts on wildlife should also be encouraged, to understand whether disturbed ecosystems have already moved outside of their natural range of variation and to identify threshold levels within which human activities can be sustainable.

Acknowledgements

This research was partly funded by TOROC, the Organizing Committee of the XX Olympic Winter Games in Turin. We are grateful to Cervinia SpA and Monterosa Ski-SpA for their kind assistance. We also thank Valeria Ardito and Patrizio Frederic for providing useful support in statistical analyses and Consolata Siniscalco for useful suggestions regarding alpine vegetation. Kathy Martin and John Mallord provided useful suggestions and comments on the manuscript.

References

- Abegg, B., Koenig, U., Buerki, R. & Elsasser, H. (1997) Climate impact assessment in tourism. *Die Erde*, **128**, 105–116.
- Akaike, H. (1973) Information theory and an extension of the maximum likelihood principle. *International Symposium on Information Theory* (eds B.N. Petran & F. Csari), 2nd edn, pp. 267–281. Akademiai Kiado, Budapest, Hungary.
- Barni, E., Siniscalco, C. & Montacchini, F. (2002) *Vegetation Dynamics on Restored Ski Runs (North-Western Italian Alps) and the Relation to Soil Stability*. Interdisciplinary Mountain Research, Europäische Akademie Bozen. Blackwell, Verlag, London, UK.
- Bayfield, N.G. (1996) Long-term changes in colonization of bulldozed ski pistes at Cairn Gorm. *Scotland Journal of Applied Ecology*, **33**, 1359–1365.
- Bibby, C.J., Burgess, N.D., Hill, D.A. & Mustoe, S.H. (2000) *Bird Census Techniques*, 2nd edn. Academic Press, London, UK.
- BirdLife International (2004) *Birds in Europe: Population Estimates, Trends and Conservation Status*. BirdLife Conservation Series No. 12. BirdLife International, Cambridge, UK.
- ter Braak, C.J.F. (1986) Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. *Ecology*, **67**, 1167–1179.
- Burnham, K.P. & Anderson, D.R. (1998) *Model Selection and Inference: A Practical Information-Theoretic Approach*. Springer, New York, NY.
- Cernusca, A., Angerer, H., Newesely, C. & Tappeiner, U. (1990) Auswirkungen von Kunstschnee eine Kausalanalyse der Belastungsfaktoren. *Verhandlungen der Gesellschaft für Ökologie*, **19**, 746–757.
- Cernusca, A., Tappeiner, U. & Bayfield, N. (1999) *Land-Use Changes in European Mountain Ecosystems*. Blackwell, Wissenschafts-Verlag, Berlin, Germany.
- Conlin, D.B. & Ebersole, J.J. (2001) Restoration of an alpine disturbance: differential success of species in turf transplants, Colorado, USA. *Arctic, Antarctic and Alpine Research*, **33**, 340–347.
- Cramp, S. (1988) *The Birds of the Western Palearctic*, Vol. V. Oxford University Press, Oxford, New York.
- Cramp, S. & Perrins, C.M. (1994) *The Birds of the Western Palearctic*, Vol. VIII. Oxford University Press, Oxford, New York.
- Delmas, M. (1986) Tétralyre et stations de ski. I. Résultats de six années de dénombrement de Tétralyre au chant en Haute-Tarentaise (Savoie). *Boll Mensuel de L'office National de la Chasse*, **99**, 17–21.
- Dirnböck, T., Dullinger, S. & Grabherr, G. (2003) A regional impact assessment of climate and land-use change on alpine vegetation. *Journal of Biogeography*, **30**, 401–407.
- Ebersole, J.J., Bay, R.F. & Conlin, D.K. (2002) Restoring high-alpine social trails on the Colorado Fourteeners. *Handbook of Ecological Restoration* (eds M.R. Perrow & A.J. Davy), Vol. 2, pp. 389–391. Cambridge University Press, Cambridge, UK.
- Elsasser, H. & Messerli, P. (2001) The vulnerability of the snow industry in the Swiss Alps. *Mountain Research and Development*, **21**, 335–339.
- Florineth, F. (2002) Pioneers in the height. *Acer*, **5**, 67–69.
- Fuller, R.M. (1987) The changing extent and conservation interest of lowland grasslands in England and Wales: a review of grassland surveys 1930–84. *Biological Conservation*, **40**, 281–300.
- Gaunch, H.G. Jr (1984) *Multivariate Analysis in Community Ecology*. Cambridge University Press, Cambridge, UK.
- Ihaka, R. & Gentleman, R. (1996) R: a language for data analysis and graphics. *Journal of Computational Graph Statistics*, **5**, 299–314.
- Labaune, C. & Magnin, F. (2002) Pastoral management vs. land abandonment in Mediterranean uplands: impact on land snail communities. *Global Ecology and Biogeography*, **11**, 237–245.
- Laiolo, P. & Rolando, A. (2005) Forest bird diversity and ski-runs: a case of negative edge effect. *Animal Conservation*, **7**, 9–16.
- Laiolo, P., Dondero, F., Ciliento, E. & Rolando, A. (2004) Consequences of pastoral abandonment for the structure and diversity of the alpine avifauna. *Journal of Applied Ecology*, **41**, 294–304.
- Legendre, P. & Legendre, L. (1998) *Numerical Ecology*. 2nd English edn. Elsevier Science B.V., Amsterdam, the Netherlands.
- Lichtenberger, E. (1994) Die Alpen in Europa. Österreichische Akademie der Wissenschaften. *Veröffentlichungen der Kommission für Humanökologie*, **5**, 53–86.
- McCune, B. & Mefford, M.J. (1999) *PC-ORD Multivariate Analysis of Ecological Data*, Version 4. MJM Software Design, Gleneden Beach, OR.
- Mackenzie, D. (1989) Alpine countries seek controls on skiers, builder and roads. *New Scientist*, **124** (1686), 22.
- Marti, C. (1985) Unterschiede in der Winterökologie von Hahn und Henne des Birkhuhns *Tetrao tetrix* im Aletschgebiet (Zentralalpen). *Der Ornithologische Beobachter*, **82**, 1–30.
- Martin, K. (2001) Wildlife in alpine and sub-alpine habitats. *Wildlife-Habitat Relationships in Oregon and Washington* (eds D.H. Johnson & T.A. O'Neil), pp. 285–310. Oregon State University Press, Corvallis, OR.

- Martin, K. & Wiebe, K.L. (2004) Coping mechanisms of alpine and arctic breeding birds: extreme weather and limitation to reproductive resilience. *Integrative and Comparative Biology*, **44**, 177–185.
- Menoni, E. & Magnani, Y. (1998) Human disturbance of grouse in France. *Grouse News*, **15**, 4–8.
- Odum, E.P. (1971) *Fundamentals of Ecology*. Saunders, Philadelphia, PA.
- R Development Core Team (2005) *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria. <http://www.R-project.org>, accessed 3 November 2005.
- Rixen, C., Casteller, A., Schweingruber, F.H. & Stoeckli, V. (2004) Age analysis helps to estimate plant performance on ski pistes. *Botanica Helvetica*, **114**, 127–138.
- Rixen, C., Haeberli, W. & Stoeckli, V. (2004) Ground temperatures under ski pistes with artificial and natural snow. *Arctic, Antarctic and Alpine Research*, **36**, 403–411.
- Rocheftort, R.M. & Gibbons, S.T. (1992) Mending the meadow. High altitude meadow restoration in Mount Rainier National Park. *Restoration and Management Notes*, **10**, 120–126.
- Rolando, A. & Laiolo, P. (1997) A comparative analysis of the diet of the chough *Pyrrhocorax pyrrhocorax* and the Alpine chough *P. graculus* co-existing in the Alps. *Ibis*, **139**, 388–395.
- Simons, P. (1988) Après ski le deluge. *New Scientist*, **117** (1595), 49–52.
- Sokal, R.R. & Rohlf, F.J. (1995) *Biometry*, 3rd edn. W.H. Freeman, New York, NY.
- SPSS Inc. (2003) *12.0-1 for Windows*. SPSS Inc., Chicago, IL.
- Titus, J.H. & Tsuyuzaki, S. (1999) Ski slope vegetation of Mount Hood, Oregon, USA. *Arctic, Antarctic and Alpine Research*, **31**, 283–292.
- Tsuyuzaki, S. (2002) Vegetation development patterns on skiruns in lowland Hokkaido, northern Japan. *Biological Conservation*, **108**, 239–246.
- Urbanska, K.M. (1997a) Restoration ecology research above the timberline: colonization of safety islands on a machine-graded alpine ski run. *Biodiversity and Conservation*, **6**, 1655–1670.
- Urbanska, K.M. (1997b) Restoration ecology of alpine and arctic areas: are the classical concepts of niche and succession directly applicable? *Opera Botanica*, **132**, 189–200.
- Urbanska, K.M., Erdt, S. & Fattorini, M. (1998) Seed rain in natural grassland and adjacent ski run in the Swiss Alps: a preliminary report. *Restoration Ecology*, **6**, 159–165.
- Watson, A. & Moss, R. (2004) Impacts of ski-developments on ptarmigan (*Lagopus mutus*) at Cairn Gorm, Scotland. *Biological Conservation*, **116**, 267–275.
- Wipf, S., Rixen, C., Fischer, M., Schmid, B. & Stoeckli, V. (2005) Effects of ski piste preparation on alpine vegetation. *Journal of Applied Ecology*, **42**, 306–316.
- Zeitler, A. & Glanzer, U. (1998) Skiing and grouse in the Bavarian Alps. *Grouse News*, **15**, 8–12.

Received 17 January 2006; final copy received 28 September 2006
Editor: Jenny Gill

Supplementary material

The following supplementary material is available as part of the online article (full text) from <http://www.blackwell-synergy.com>.

Appendix 1. List of the bird species recorded in the 223 plots.

Appendix 2. Number of points sampled in each plot type and locality.