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Landscape changes caused by high altitude ski-pistes affect bird species richness and distribution in the Alps

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## Abstract

There is a paucity of research on the wider landscape-level effects of ski-piste construction on alpine fauna. In this study, the response of alpine bird communities to the landscape changes induced by the construction of ski-pistes was investigated in the western Italian Alps. The aims were: (i) to test the hypothesis that ski-pistes have a detrimental effect on alpine grassland bird communities at a landscape-scale; and, (ii) to model local probability distributions of bird species according to different scenarios of ski-piste restoration and ski-piste proliferation above the treeline. Species richness and presence/absence data from point counts were analyzed in relation to GIS-derived landscape variables based on a 16 ha hexagon grid. Predictive variables were selected through the LASSO model selection procedure, and logistic regression was used to estimate the probability of occurrence of each species in each hexagon.

Grassland species richness, and probability of occurrence of water pipit, wheatear and black redstart, significantly decreased with increasing extent of ski-piste edge. Length of ski-piste edge was in the set of best models considering a large range of habitat and landscape predictors, and are therefore clearly features that exert a strong negative effect on high alpine grassland bird communities. Predictions of species occurrence were made by applying the models to different scenarios of habitat change. These showed predicted detrimental impacts of a relatively small 10% increase in ski-piste extent, but also that grassland restoration on existing ski-pistes could result in significantly increased occurrence rates of alpine grassland species. This study suggests that ski-pistes are perceived by birds as detrimental features of the alpine landscape. To minimize their impact, new, environmentally-friendly ways of constructing pistes should be developed, which could include habitat restoration and management to obtain a level of grass cover such that edges of ski-pistes are no longer perceived by birds.

## 1. Introduction

Over recent decades, there have been steady declines in global biodiversity, both in terms of populations of wild species and in terms of the extent and condition of habitats (Butchart et al., 2010; Rands et al., 2010). Habitat destruction and degradation caused by human activities, and the subsequent fragmentation and loss of connectivity of ecosystems, are key factors in driving biodiversity loss (Balmford et al., 2005; Rands et al., 2010), and there are numerous examples of such impacts on wildlife (e.g. Villard et al., 1999; Lindenmayer et al., 2002; Becker et al., 2010; Benitez-Lopez et al., 2010; Irwin et al., 2010; Peres et al., 2010).

The European Alps, in common with many of the world's ecosystems, face multiple and increasing pressures on biodiversity through significant loss and degradation of natural habitats (EEA, 2010), including changes in land use (e.g. changes in grazing practices and land abandonment, Laiolo et al., 2004), climate change (e.g. Wilson et al., 2005; Pauli et al., 2007), and increasing human disturbance (Arlettaz et al., 2007; Patthey et al., 2008; Thiel et al., 2007). For high altitude alpine ecosystems, one of the main causes of habitat alteration is the construction of ski-pistes, and associated infrastructures, which significantly impact natural ecosystems by degrading soils (Freppaz et al., 2002; Delgado et al., 2007), destroying natural habitats for high altitude grassland and forest animals (Rolando et al., 2007; Negro et al., 2009) and plants (Barni et al., 2002; Tsuyuzaki, 2002), fragmenting natural habitats by creating barriers to dispersal (Negro et al., 2009) and reducing the quality of adjacent natural habitats due to edge effects (Laiolo and Rolando, 2005;

Rolando et al., 2007; but see Burt and Rice, 2009). The latter is important, as it implies that the construction of ski-pistes does not only have a local effect, but may have impacts over scales much larger than the extent of habitat directly modified, although the mechanism by which the edge effect occurs is yet to be identified (Rolando et al., 2007). The skiing industry is of major economic importance to the alpine region, and the industry has recently experienced a period of great expansion (Abegg et al., 1997; Elsasser and Messerli, 2001; Wipf et al., 2005). Furthermore, the pressures on alpine biodiversity are interacting – concerns are particularly relevant above the treeline, because climate changes will probably induce a shift in skiing activities and ski runs to higher altitudes (Elsasser and Messerli, 2001; Fukushima et al., 2002; Bicknell and McManus, 2006; Scott et al., 2008). In order to reconcile the economic benefits of the skiing industry to human populations with its current clear negative impacts on biodiversity, a number of initiatives have been suggested (Lorite et al., 2010; Braunisch et al., 2011), including more sympathetic management of grassland on the piste or identifying areas subject to various intensities of recreational disturbance. Current management of ski-pistes is highly destructive, yet new techniques for less intensive management and a restoration of grassland communities on ski-pistes are being developed (Urbanska, 1997; Florineth, 2002; Conlin and Ebersole, 2001; Ebersole et al., 2002; Isselin-Nondedeu et al., 2006; Lorite et al., 2010). However, a landscape-scale approach from a bird community perspective really needs to be adopted in order to integrate biodiversity issues into land-use planning (Butchart et al., 2010), particularly for animals that typically range over wider areas, including an appreciation of the landscape-scale impacts of skipistes.

In this paper, we analyse the effects of ski-pistes on alpine bird communities at a landscape scale. Rolando et al. (2007) and Laiolo and Rolando (2005) previously demonstrated significant differences in bird communities between ski-pistes, piste/natural habitat edges and natural habitat far from ski-pistes, which included analyses of fine-scale habitat characteristics at the plot level (50 m radius), but did not attempt to elucidate relationships with landscape structural variables. Here, we re-analyse the data of Rolando et al. (2007) in order to derive predictive models at a landscape scale. We consider relationships between bird species richness and individual species' occurrence, and GIS-derived topography and landscape descriptors, at a landscape mesoscale (hexagons of 16 ha around each plot). Through this approach, we aim: (i) to test the hypothesis, based on the negative edge effects observed at smaller scales, that ski-pistes have a detrimental effect on alpine grassland bird communities at a landscape-scale, and (ii) to model local probability distributions of grassland bird species according to four distinct scenarios, i.e. the actual situation (after ski-piste construction), a simulated one considering the original landscape patches (before ski-piste construction or assuming an ideal ski-piste removal with simulated restoration of original habitat elements), a further simulation that considers restoration of grassland-only (which is likely to be the most feasible practical option), and a final simulation that considers a conservative (10%) increase in ski-piste length based on the likely proliferation of the ski industry above the treeline (Elsasser and Messerli, 2001).

## 2. Materials and methods

### 2.1. Study area

The study was carried out in prairies (high altitude natural grasslands and seasonally grazed pastures) of the western Italian Alps, in Aosta Valley (Monte Bianco and Monte Rosa massifs) and in Piedmont (top end of Susa Valley). Ski-pistes in the study area are constructed in high altitude prairies by removing the vegetation and topsoil, and are continuously managed to modify slopes and keep vegetation (mainly grasses such as *Festuca* and *Nardus* spp.) under control. The resulting habitat is a sparse and patchy vegetation layer interspersed with small rocks and bare soil, especially on the more impacted pistes. The wider landscape is characterized by thin coniferous forest patches dominated by larch *Larix decidua* around the timberline; above the timberline prairies are dominated by grasses (Gramineae, *Festuca* and *Nardus* spp.). Shrub patches are represented by juniper *Juniperus communis*, alpen rose *Rhododendron ferrugineum* and, to a lesser extent, bilberry

*Vaccinium myrtilloides* and bearberry *Vaccinium uliginosum*. At higher altitudes, the landscape is dominated by rocks, scree and snowbeds.

## 2.2. Bird surveys

Field work was carried out in June and July 2004 and 2005 as described in detail in Rolando et al. (2007). We sampled birds within, and in the vicinity of, seven large ski resorts (Fig. 1) using a standardized area count method, surveying birds in circular plots of 50 m radius. Selection was based on locating accessible plots with suitable vantage points (i.e. visibility of the surrounding 50 m radius was required), while maintaining a minimum 300 m between each plot. While a random point selection would be theoretically ideal, it is not logistically feasible in high alpine landscapes because many selected plots will be located in inaccessible areas, or areas where views and hence detectability are severely restricted (and in such a potentially hostile environment, the health and safety of the fieldworker must be paramount, so sites need to be as safe as possible). Nevertheless, the plot selection was not based on factors likely to strongly influence the bird community (i.e. plots close to roads or paths or buildings) and we therefore assume that the selection of plots was no different to a random selection of plots with respect to the birds recorded. We visited 203 plots (99 in 2004 and 104 in 2005) at altitudes between 2010 m and 2892 m a.s.l., mostly above the treeline, and 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 per plot. Three types of plots were defined: plots centered on the ski-pistes ( $n = 71$ ), grassland habitats (i.e. pastures and/or prairies) adjacent to ski-pistes ( $n = 61$ , a distance of 100–200 m) and natural grassland habitats far from the ski-pistes located in areas without ski-development ( $n = 71$ , at a distance of 1–10 km from pistes). Ski-pistes were 20–200 m wide (mean  $73.3 \pm 6.22$  SE); when the piste was narrower than 100 m, a portion of habitat at its side was included in the plot. The plots located to the side of the pistes and in natural habitats were dominated by grassland, but bushes, scree and sparsely vegetated areas were also present. Birds requiring open prairies for feeding and, in most instances, for breeding, were classed as alpine grassland species (Table 1).

## 2.3. Landscape data analyses

We used 1:10,000 regional vectorial maps to describe land use in the study area. For each sampling plot, land use was described according to six habitat classes based on Corine landcover terminology (CEC, 1995): prairies (P), coniferous forests (CF), shrubs (B), bare rock (BR), sparsely vegetated areas (SVA, high altitude areas of bare soil or scree with some patchy ground vegetation) and ski-pistes (SKI). When necessary, ski-pistes were digitized from aerial photographs and overlaid on the original (vectorial) maps. Ten topographic variables (mean altitude, mean slope, percentage of North, North-East, East, South-East, South, South-West, West and North-West expositions) were computed within 16 ha hexagons, centered on each point count, using a regional DEM (Digital Elevation Model) with a 50 m accuracy. One climatic variable (mean annual temperature) was also considered and computed using the WorldClim dataset (Hijmans et al., 2005).

FRAGSTATS 3.3 (McGarigal et al., 2002) was used to calculate landscape variables (or metrics) describing landscape structure and fragmentation induced by ski-pistes (the complete list of metrics is reported in Appendix A). Metrics were derived for all patches irrespective of habitat (termed Landscape-level metrics, following the terminology of McGarigal et al., 2002), and also separately for each of six habitat classes (i.e. they referred to patches of prairie, forest, etc.), termed Class-level metrics. For example, a landscape with four grassland patches and two forest patches at the Class-level would have six patches at the Landscape-level. We consider two classifications because we expect birds may respond both to the extent and spatial arrangement of particular habitat types and to habitat structure independent of habitat type (e.g. Guadagnin and Maltchik, 2007; Caprio et al., 2009). To avoid any nomenclatorial confusion with the term “landscape”, hereafter we refer to the

above metrics using italic types. For landscape-level metrics, a greater range of variables were calculated (several were not applicable to class-level data, e.g. diversity indices; see Appendix A). A total of 165 metrics was computed, 108 class-level metrics (18 metrics per six habitat types) and 57 landscape-level metrics for each spatial hexagon. Predictive models of the distribution of alpine grassland species were constructed for different land use scenarios (see below) using a data set of landscape-level and class-level metrics from 17 ski resorts in the whole Aosta Valley region, where we had access to digitized regional maps that included tracks of ski-pistes. To achieve homogenous spatial units for proper statistical analysis, we produced a 16 ha hexagon grid and clipped the land use and topography data set into 640 hexagons comprising ski-pistes at an altitude ranging from 2000 m up to 2900 m asl. This dataset was independent of the bird survey sites, with the exception of a small percentage of hexagons (8% of 640 hexagons contained bird survey points).

#### 2.4. Statistical analyses

Avian diversity was described for each census plot in terms of bird species richness. Spatial autocorrelation among plots was tested by comparing UTM coordinates of each plot and species richness and presence/absence of individual species with the Mantel test, which evaluates the similarity between an ecological distance matrix and a geometric distance matrix (Koenig, 1999). If autocorrelation exists, then the closer the points are in the geometric space, the more similar should be the values of the community parameters. We used 9999 Monte Carlo simulations to test the hypothesis that there was no correlation between the bird community parameter matrix and the UTM coordinate matrix. The Mantel test was performed using PC-ORD (McCune and Mefford, 1999). No strong spatial autocorrelation ( $r = 0.05$ ,  $P = 0.09$ ) for bird species richness was found, suggesting that plots could be considered as effectively independent. Subsequently, we tested for site effects by running a one-way ANOVA, that showed no significant variation between sites (Appendix B). This is in agreement with Rolando et al. (2007) who found no site effect nor interactions between site and ski-piste metrics. The models were therefore initially fitted without site effects, but we subsequently tested for such effects by adding site as a categorical variable, and interactions with skipiste variables when relevant, to the final selected models (see below). Both the significance of the added effects (likelihood ratio tests) and model performance (AIC) were considered in assessing the extent to which site effects and interactions were important predictors. There was some tendency for sites to be orientated towards the west (Fig. 2), although this tendency was applicable to all plots, so effects of ski-pistes on birds will not have arisen due to differential exposition between survey plot types.

A generalized linear modeling (GLM) approach was used to analyse species richness and the probability of occurrence of individual species in relation to landscape-level and class-level predictors.

LASSO model selection based on the cyclical coordinate descent method (Tibshirani, 1996; Friedman et al., 2009) was used, running glmnet 1.6 (R package), which allows fitting of generalized linear models with elastic-net penalties (Zou and Hastie, 2005). The number of predictors selected by LASSO depends on the value of the tuning parameter  $k$ , which was selected using AIC. LASSO is somewhat indifferent to highly correlated predictors, and will tend to pick a single variable and ignore the rest, therefore we decided to reduce the number of correlated landscape and topographic metrics. We first examined all pair-wise correlations to identify strongly correlated pairs ( $r > |0.7|$ ). Then, following Riitters et al. (1995), one variable was selected to represent each group of highly correlated variables; selection criteria included biological interpretability and the distribution of the data (in particular, favouring approximately normal or uniform distributions assessed through visual inspection of PP plots, and excluding variables with highly skewed distributions, clear outliers or bimodality). This procedure reduced the number of class-level variables from 108 to 34 and the number of landscape-level variables from 57 to 18.

To model the distribution of species richness, count (i.e. number of individual species) was modeled specifying a Poisson error distribution and a log link function. We ran LASSO model selection to

test for the effects of uncorrelated variables (independent variables or covariates) on overall community species richness and on grassland species richness (dependent variable); variables selected with the LASSO procedure were used in GLMs to obtain non-penalized coefficients and the lowest ranked models were compared to those having a DAIC < 2 (full results of those models are shown in Appendix C).

For individual alpine grassland species, the probability of occurrence was modeled by analyzing presence/absence with binomial logistic regression. We considered the most frequent species (i.e. overall frequencies of occurrence higher than 9%, Table 1) and focused therefore upon the water pipit *Anthus spinoletta* (L.), northern wheatear *Oenanthe oenanthe* (L.), black redstart *Phoenicurus ochrurus* (S.G. Gmelin), snow finch *Montifringilla nivalis* (L.), linnet *Carduelis cannabina* L. and skylark *Alauda arvensis* L.

No spatial autocorrelation was found ( $|r| < 0.03$ ,  $P > 0.20$  in each case), with the exception of skylark ( $r = 0.096$ ,  $P < 0.01$ ) which was more common in Piedmont than in the Aosta Valley. In common with the analysis of species richness, we initially tested for site effects through a logistic regression and found no significant effects (Appendix D). Additionally, site effects and interactions were tested by adding them to the final models (as above). Variables selected with the LASSO procedure were used in GLMs to obtain non-penalized coefficients that were used to estimate the probability of occurrence of individual species. The dataset was divided randomly into model building (70%) and model validation (30%) subsets. For each model, we obtained a Receiver Operating Characteristic (ROC) curve, based on the true and false positive rates as measures of model performance. We used a linear stretch to scale the predicted values ( $w$ ) of each model between 0 and 1. The linear stretch is a common transformation for image enhancement and interpretation (Lillesand and Kiefer, 1994). We assessed the discrimination ability of the predictive models by using: (i) the area under the curve (AUC) of a ROC plot (Fielding and Bell, 1997) and (ii) the corresponding True Skill Statistic (TSS, Allouche et al., 2006) and the cut-off value that maximized the classification accuracy of the models as a break value to discriminate presence and pseudo-absence, (sensu Engler et al., 2004).

According to the response to predictors selected with the glmnet procedure, a first map of relative probability of occurrence was generated for each species for the 17 ski areas in the Aosta Valley.

Then, within the framework of a simulated scenario in which ski-pistes were removed and land use assigned to the class the ski-piste crossed in that hexagon, a second map of relative probability of occurrence was generated. In terms of habitat creation, grassland is the most feasible option (e.g. Isselin-Nondedeu et al., 2006).

To evaluate the possible effect of grassland restoration in active ski-pistes, we created a third scenario computing the probability of presence of each of the species when grassland-only could be reinstated in ski-pistes. Finally, a fourth scenario was constructed which assumed a 10% increase in ski-piste extent for hexagons already containing ski-pistes.

### 3. Results

#### 3.1. Species richness

Overall, 856 individuals from 29 species were sampled in 203 plots (Table 1). Species richness had its lowest values in ski-piste centered plots (mean value:  $1.09 \pm 0.13$ ), while plots in prairies close to ski-pistes and far from ski-pistes had higher values ( $2.10 \pm 0.11$  and  $2.48 \pm 0.13$  respectively). Results of generalized linear models for bird species richness are shown in Table 2. Overall, species richness was positively associated with core area of prairie and forest edge length; it was negatively associated with the number of sparsely vegetated patches and the area covered by rocks. Grassland species richness was positively associated with plots with a southerly aspect and negatively associated with the length of the edge of ski-pistes, demonstrating that the number of grassland species decreased with increasing extent of ski-pistes.

The number of rock patches entered both models (whole community and grassland species), but it was never significant. When added to the above models, site effects were not significant and the

models showed lower predictive power (in terms of AIC) than those without site effects (Appendix E). The unique significant predictor in the whole community model was the number of sparsely vegetated patches. For grassland species richness, ski-piste edge was the only significant variable in the site effects model.

### 3.2. Individual species occurrence

Logistic regression models for the snow finch and skylark performed poorly (i.e.  $AUC < 0.5$ ), possibly due to inadequate sample size. Skylark was more abundant in grasslands adjacent to skipistes and less abundant in ski-pistes, while the snowfinch was the only species that was more abundant in plots located in skipistes. Conversely, models for the black redstart, water pipit and wheatear performed rather well (Appendix F). For black redstart, the cut-off that maximized the classification accuracy of the model was 0.43, with  $TSS = 0.22$  ( $AUC = 0.65$ ). Respective figures for water pipit were 0.48, with  $TSS = 0.40$  ( $AUC = 0.77$ ), and for wheatear 0.45, with  $TSS = 0.33$  ( $AUC = 0.70$ ).

The length of the ski-piste edge gave a significant negative contribution ( $P < 0.01$ ) for all three species, i.e. probability of presence significantly decreased with increasing extent of ski-pistes per sampling plot (Table 3). The black redstart model also revealed a significant positive contribution of sparsely vegetated core area, bare rock total area and shrub edge length. The water pipit model encompassed five other significant variables: prairie core area (positive contribution), forest total area, bare rock total area, total number of patches and mean annual temperature (negative contributions).

The wheatear model encompassed a further four significant variables: shrub core area (positive contribution), bare rock total area, total number of patches, and mean slope (negative contributions). It should be noted that most significant landscape predictors of both presence/absence and species richness were classlevel predictors, the only landscape-level predictor being the total number of patches (see Table 3). Logistic regression models to test for site effects showed much less predictive power than those without site effects in all the three species (Appendix G), and in no case was site, nor the interaction between site and ski-piste length, significant (Appendix G).

Estimates of metrics from our models were used to compute the relative probabilities of presence for each of the three species in before- and after-ski-piste scenarios for the Aosta Valley area. Differences were apparent, virtually all hexagons with ski-pistes having much lower probabilities of species occurrence than the same hexagons with ski-pistes removed ( $P < 0.001$ , Wilcoxon signed ranks tests; Fig. 3). Furthermore, the scenario of grassland-only restoration gave similar predictions to those for restoration of all original habitat, and for water pipit and wheatear, estimated probabilities were highest in the grassland only scenario (Fig. 3). The fourth scenario of a 10% increase in extent of existing ski-pistes resulted in a c. 10% decrease in the probability of occurrence of each of the three species relative to the current scenario, the effect being most marked for wheatear (Fig. 3). Examples of maps of relative probability of presence in one selected ski-resort areas are shown in Fig. 4 (maps for the six largest areas considered under the predictive models are given in Appendix H).

## 4. Discussion

Species richness of grassland birds, and the occurrence of three individual species, water pipit, wheatear and black redstart, showed clear negative effects of ski-piste extent, specifically the length of ski-piste edge present, at the landscape scale considered. Ski-pistes were in the set of best models considering a large range of habitat (i.e. class-level) and landscape-level predictors. They are therefore clearly features that exert a strong negative effect on high alpine grassland bird communities. The removal of much of the vegetation and rocks, and disturbance to the soil, creates a habitat that is almost totally unsuitable for nesting and also is likely to provide very few food resources for high alpine species that require both grass cover (water pipit, Brodmann et al., 1997) and rocks (black redstart, Kollinsky and Landmann, 1996; wheatear Zamora, 1990). There was, however, an implication that snowfinch, may derive some benefit from ski-pistes, although the



abundances were still higher in natural prairies (Rolando et al., 2007). The fact that ski-piste length, rather than the area of ski-pistes per se, was a consistent negative predictor suggests that edge effects may be important in driving the observed relationships. Our prediction that edge effects shown at small scales (Rolando et al., 2007) will manifest themselves at larger scales is therefore supported. The creation of edges, i.e. transition zones between adjacent habitats, is one of the main effects of habitat fragmentation (Murcia, 1995). Historically, land managers considered habitat fragmentation to be beneficial for biodiversity, on the basis of what was later called the “edge effect”, i.e. the hypothesis that species richness increases at the border between different habitats (Odum, 1971). Indeed, the species richness of all species recorded increased with increasing forest edge length in this study, and positive effects of predominantly forest ski-pistes in North America have been found (Burt and Rice, 2009), although a negative edge effect has been found affecting black grouse up to 1500 m from ski lifts (Patthey et al., 2008).

Notably, no ski-related landscape predictor entered the model regarding the whole bird community, because non-grassland species are not likely to be influenced by the landscape changes of alpine prairies. However, it is clear that anthropogenic edges and habitat fragmentation may also induce radical changes in community structure (Matlack and Litvaitis, 1999), improving the habitat for some species (Burt and Rice, 2009), but causing local extinctions for others (Andrén and Angelstam, 1988; Andrén, 1994; Simberloff, 1995; Kurosawa and Askins, 2003) and inhibit dispersal (Forman, 1997). For most of the above examples, edge effects have been demonstrated in fragmented forest ecosystems, where there is an abrupt change between forested and open areas. Our results suggest that such effects may also occur in open habitats and may affect community structure and alter grassland species distribution, with general negative effects, although further research is needed to determine if there is in fact a benefit in certain cases, most notably for the snowfinch.

The mechanisms by which edge effects may operate in such open landscapes are unknown, but there are two likely general explanations, disturbance effects and predation effects. Disturbance to the habitat may arise due to alteration of the natural grassland at the habitat edge leading to an effectively greater habitat loss than the area of the ski-piste itself. Such effects could arise through physical disturbance associated with piste management in the summer or skiing in the winter, affecting grassland structure and hence the quality of habitat for ground-nesters, although there seems to be no apparent reason why such activity would not be almost wholly confined to the piste itself. However, the use of artificial snow generators has been shown to have detrimental impacts on piste vegetation composition and structure (Wipf et al., 2005), and it seems likely that artificial snow will commonly come to rest outside the piste. There could also be a direct effect of disturbance in the spring and summer if ski-pistes are used as paths by walkers, which have been shown to have negative effects on ground-nesting upland birds (Yalden and Yalden, 1990; Finney et al., 2005). The creation of a habitat edge may facilitate greater nest predation on the species considered here that nest on or near to the ground in open habitats. It is possible that the ski-piste could act as a predator corridor, enabling greater ease of movement of terrestrial predators from lower habitats, although such open habitats are presumably highly accessible throughout the grassland above the treeline. An alternative explanation is that structures associated with the ski-piste (e.g. lights, stanchions for ski-lifts, etc.) create lookouts for avian predators and brood parasites like the cuckoo (*Cuculus canorus*, Linnaeus 1758), or that predators are attracted by deposition of food remains, which may enhance predator density (Storch and Leidenberger, 2003). Certainly, vertical structures in the landscape are avoided by several ground-nesting species (e.g. Chamberlain et al., 1999, 2009; Reino et al., 2009; Massa and La Mantia, 2010), and it seems plausible that such effects could also operate for alpine ground-nesters such as water pipit and skylark.

The importance of any of the above factors will only become apparent when intensive studies are carried out on behavior, territory settlement and reproductive performance of target species, which must be considered the priority for future research in this field.

#### 4.1. Conservation implications

The models derived for the three target species suggested that grassland restoration on the ski-piste could result in a substantial expansion of suitable habitat as ski-piste characteristics in the summer are made more favorable. For water pipit and wheatear in particular, the predicted restoration of grasslands resulted in higher occurrence rates than restoration of the original habitat, probably because these species are most strongly linked to open grassland habitats. Management to restore and maintain grassland on existing ski-pistes may therefore have significant benefits to the high alpine bird community as a mitigation measure. The minimum aim should be to obtain a level of grass cover such that edges of ski-pistes are no longer perceived by birds. Transplantation of single plants (Florineth, 2002) or turf transplants of wild alpine species to create a wholly autonomous community (Urbanska, 1997; Conlin and Ebersole, 2001; Ebersole et al., 2002; Isselin-Nondedeu et al., 2006) seem to be particularly promising methods, particularly compared to sowing grass seed.

However, it should be noted that grassland restoration is typically only partially successful in that, although vegetation structure in terms of cover and plant size is restored (Lorite et al., 2010), the species composition is typically different from surrounding natural grasslands (Bayfield, 1996; Barni et al., 2007; Isselin-Nondedeu and Bedecarrats, 2009). For nesting habitat, this may not matter as the vegetation structure is likely to be of over-riding importance (e.g. Zamora, 1990). Similarly, restoration may result in invertebrate communities that differ from the natural habitat, which could have implications for chick food (Walker et al., 2004; Negro et al., 2010). Nevertheless, it seems likely that such restorations provide a greatly improved habitat for alpine ground-nesters.

The models also predicted a concomitant decrease in bird occurrence with an increase in ski-extent. Continued expansion of ski-pistes above the treeline will clearly have detrimental impacts on the grassland bird community. Furthermore, it should be noted that future expansion is likely to include more high altitude ski-pistes. This is undesirable not only from the point of view of impacts on biodiversity, but also because the probability of successful re-vegetation declines with altitude (Urbanska, 1997; Barni et al., 2007), thus management to encourage grass development and hence minimize ski-piste impacts is likely to be less successful.

The ski-pistes in our sample were centered on larger ski resorts, which are likely to experience relatively intensive management and be subject to high levels of disturbance. Furthermore, the piste management in our study area may be more intensive than that in adjacent countries (pers. obs.), and it has been shown that the impact of management can vary according to its intensity, in some cases even having positive effects (Burt and Rice, 2009). The simulation may therefore be regarded as a high-impact scenario, but we nonetheless feel our sites were representative of management on the vast majority of ski-pistes in the Italian Alps.

Throughout Europe, habitat loss of semi-natural grasslands has been recognized as a serious threat to many rare and declining animal and plant species (Labaune and Magnin, 2002; Pöyry et al., 2009; Krauss et al., 2010). Several grassland and shrub species recorded in the study area 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 rock thrush *Monticola saxatilis* (L), skylark and rock bunting *Emberiza cia* (L.) are considered as depleted (BirdLife International, 2004). Our results are of particular concern because in the European Alps, the area affected by ski-pistes is large, with a combined length totalling several thousands of kilometres (Rolando et al., 2007), and it is still increasing (Abegg et al., 1997; Elsasser and Messerli, 2001; Wipf et al., 2005). The predictive scenarios presented here suggest that, for the most abundant high alpine species, restoration of grassland alone would be sufficient to have a positive impact on their occurrence.

We suggest a landscape-level approach should also be taken to identify areas that are already affected, and hence maybe considered for mitigation measures, and also to target those areas to be considered for a moratorium on future ski developments.

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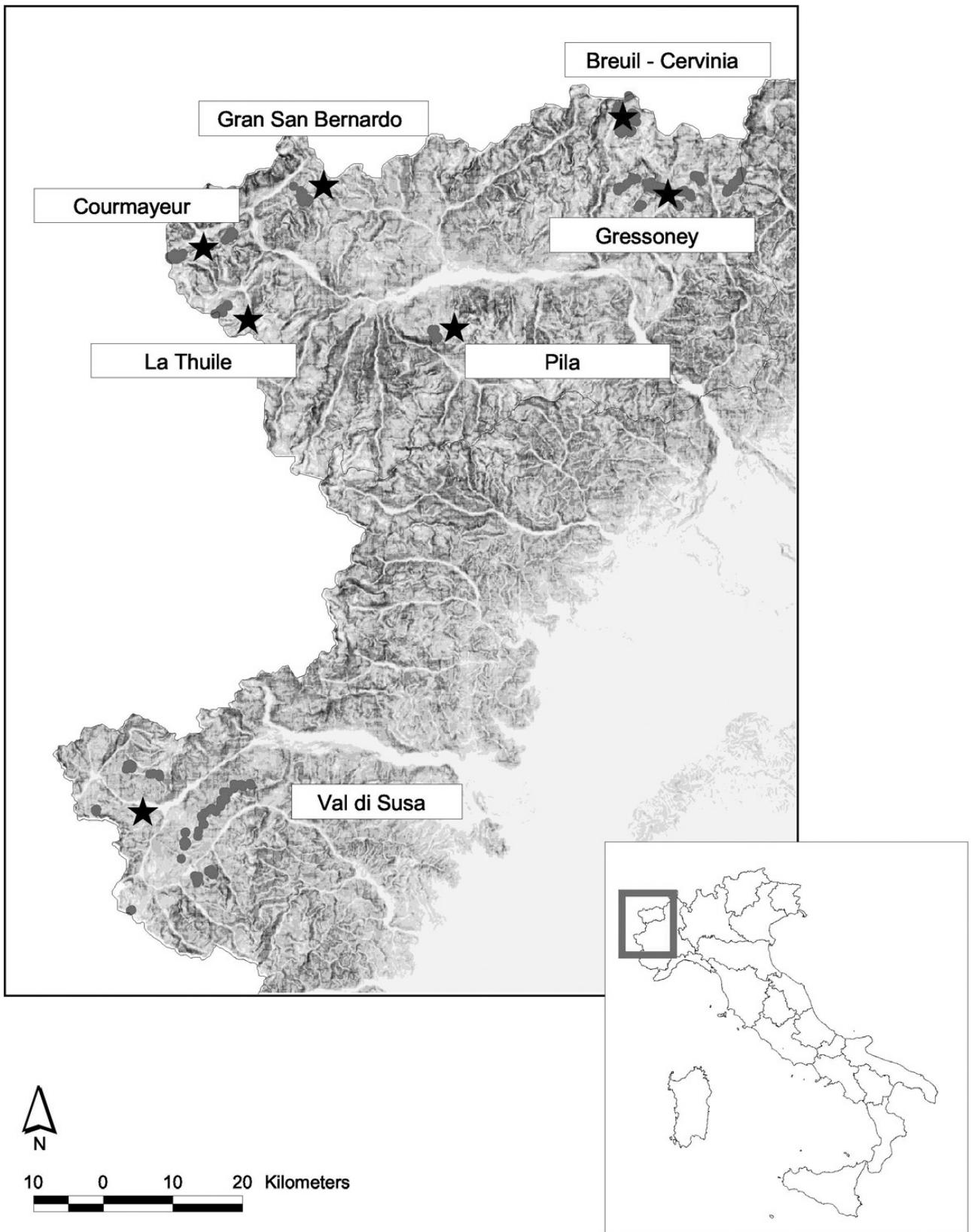


Fig. 1. Map of the study area with geographical location of plots (grey dots).

Table 1.

Bird species recorded during the surveys, their percentage frequency of occurrence ( $n = 203$  plots), and their habitat classification (grassland specialist or non-grassland) according to their local ecological preferences.

Common name	Scientific name	Ecological group	Frequency of occurrence (%)			
			Ski-pistes	Praires close to ski-pistes	Natural praires	Total
Water pipit	<i>Anthus spinoletta</i>	Grassland	26.76	55.74	56.34	46.28
Skylark	<i>Alauda arvensis</i>	Grassland	7.04	14.75	12.68	11.49
Black redstart	<i>Phoenicurus ochruros</i>	Grassland	18.31	24.59	36.62	26.51
Rock thrush	<i>Monticola saxatilis</i>	Grassland	1.41	0.00	1.41	0.94
Alpine chough	<i>Pyrhcorax graculus</i>	Grassland	4.23	8.20	9.86	7.43
Red-billed chough	<i>Pyrhcorax pyrrhcorax</i>	Grassland	1.41	1.64	1.41	1.49
Chaffinch	<i>Fringilla coelebs</i>	Non-grassland	1.41	1.64	1.41	1.49
Snow finch	<i>Montifringilla nivalis</i>	Grassland	15.49	8.20	5.63	9.77
Wheatear	<i>Oenanthe oenanthe</i>	Grassland	28.17	40.98	63.38	44.18
Grey wagtail	<i>Motacilla cinerea</i>	Grassland	0.00	1.64	4.23	1.95
White wagtail	<i>Motacilla alba</i>	Grassland	1.41	0.00	5.63	2.35
Linnet	<i>Carduelis cannabina</i>	Grassland	4.23	16.39	11.27	10.63
Alpine accentor	<i>Prunella collaris</i>	Grassland	0.00	6.56	8.45	5.00
Ptarmigan	<i>Lagopus mutus</i>	Grassland	0.00	0.00	4.23	1.41
Rock partridge	<i>Alectoris graeca</i>	Grassland	0.00	0.00	1.41	0.47
Raven	<i>Corvus corax</i>	Grassland	0.00	0.00	1.41	0.47
Carrion crow	<i>Corvus corone</i>	Non-grassland	0.00	3.28	0.00	1.09
Yellowhammer	<i>Emberiza citrinella</i>	Grassland	0.00	1.64	0.00	0.55
Lesser whitethroat	<i>Sylvia curruca</i>	Non-grassland	1.41	3.28	1.41	2.03
Goldfinch	<i>Carduelis carduelis</i>	Non-grassland	0.00	1.64	0.00	0.55
Citril finch	<i>Serinus citrinella</i>	Non-grassland	1.41	4.92	5.63	3.99
Winchat	<i>Saxicola rubetra</i>	Grassland	5.63	8.20	5.63	6.49



Common name	Scientific name	Ecological group	Frequency of occurrence (%)			
			Ski-pistes	Praires close to ski-pistes	Natural praires	Total
Wren	<i>Troglodytes troglodytes</i>	Non-grassland	0.00	1.64	0.00	0.55
Redpoll	<i>Carduelis flammea</i>	Non-grassland	0.00	1.64	1.41	1.02
Ring ouzel	<i>Turdus torquatus</i>	Non-grassland	2.82	1.64	1.41	1.95
Mistle thrush	<i>Turdus viscivorus</i>	Non-grassland	0.00	1.64	0.00	0.55
Willow tit	<i>Parus montanus</i>	Non-grassland	0.00	1.64	0.00	0.55
Dunnock	<i>Prunella modularis</i>	Non-grassland	1.41	3.28	1.41	2.03
Rock bunting	<i>Emberiza cia</i>	Grassland	0.00	0.00	2.82	0.94

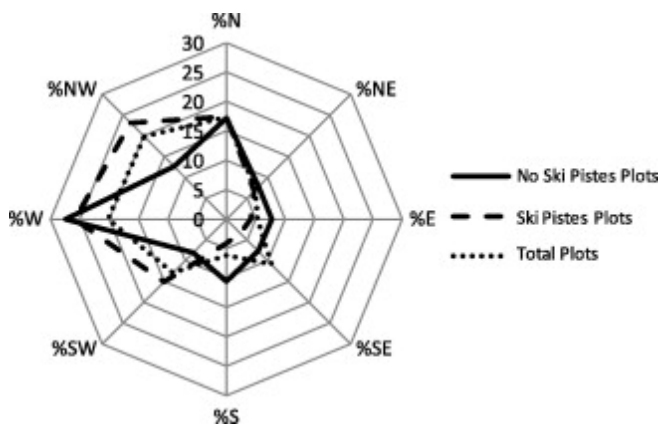


Fig. 2.

Radar plot of study site orientation, according to plot type.

Table 2.

GLM of bird species richness in relation to landscape and topographic predictors.  
 $P = * < 0.05$ ,  $** < 0.01$ ,  $*** < 0.001$ , NS = not significant.

<b>Predictor</b>	<b>Beta</b>	<b>SE</b>	<b>P</b>
<i>Whole community</i>			
Rock area	-0.024	0.011	*
Number of rock patches	-0.042	0.03	NS
Number of sparsely vegetated patches	-0.068	0.029	*
Forest edge length	0.002	5E-04	***
Prairie core area	0.021	0.013	*
AIC = 743.12			
<i>Grassland species</i>			
South exposition	0.006	3E-04	*
Ski-pistes edge length	-0.0083	4E-04	*
Number of rock patches	-0.043	0.03	NS
AIC = 674.83			

Table 3.

GLMs of presence/absence for the black redstart, water pipit and wheatear in relation to landscape and topographic predictors. (*Class-level* predictors indicated by: S = shrubs, CF = coniferous forests, BR = bare rocks, SKI = ski pistes, P = prairies, SVA = sparsely vegetated areas).  $P = ^* < 0.05$ ,  $^{**} < 0.01$ ,  $^{***} < 0.001$ , NS = not significant.

	Black redstart			Water pipit			Wheatear		
	Estimate	SE	P	Estimate	SE	P	Estimate	SE	P
Intercept	0.097	0.023	*	0.374	0.09	*	2.33031	1.138	*
S total area				0.119	0.35	NS			
S edge length	0.074	0.023	**	0.0013	0.0015	NS			
S core area							0.537	0.023	**
CF total area				-0.194	0.02	**			
BR total area	0.0383	0.017	*	-0.187	0.06	**	-0.033	0.012	*
SKI edge length	-0.0036	0.0015	**	-0.0036	0.0015	**	-0.034	0.002	**
P total area							0.088	0.042	*
P number of patches				-0.16	0.14	NS			

	<b>Black redstart</b>			<b>Water pipit</b>			<b>Wheatear</b>		
	<b>Estimate</b>	<b>SE</b>	<b>P</b>	<b>Estimate</b>	<b>SE</b>	<b>P</b>	<b>Estimate</b>	<b>SE</b>	<b>P</b>
P core area				0.158	0.055	**			
SVA core area	0.045	0.076	NS						
Total number of patches				-0.133	0.02	*	-0.157	0.058	**
Total number of classes				-0.0138	0.02	NS			
Mean slope				0.075	0.045	NS	-0.0647	0.041	NS
Mean annual temperature				-0.687	0.207	***			
AIC	214.72			202.1			189.24		

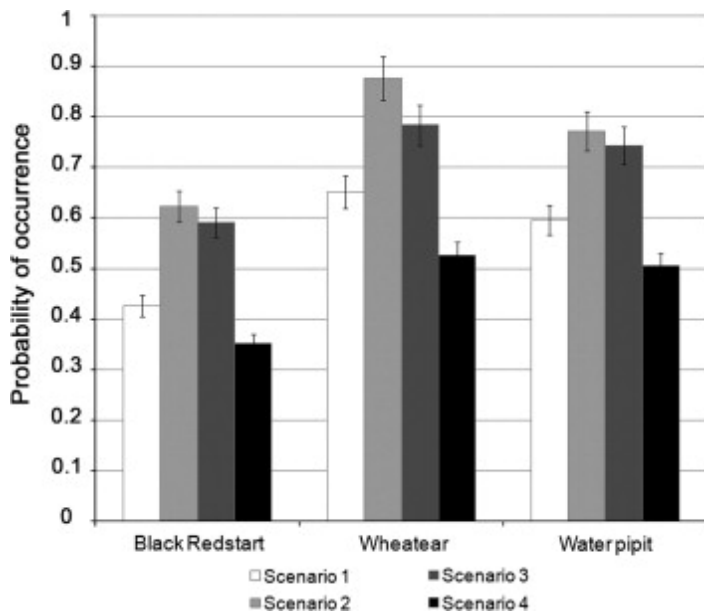


Fig. 3.

Differences in mean probability of occurrence of water pipit, wheatear and black redstart in the current landscape scenario with ski-pistes (Scenario 1, white bars), in a simulated one where ski-pistes are restored with grasslands (Scenario 2, light grey bars), in a simulated scenario with ski-pistes removed and replaced with dominant land use class in the landscape (Scenario 3, dark grey bars) and in a simulated scenario in which ski-pistes have been increased by 10% (Scenario 4, black bars). Error bars are 95% confidence intervals.

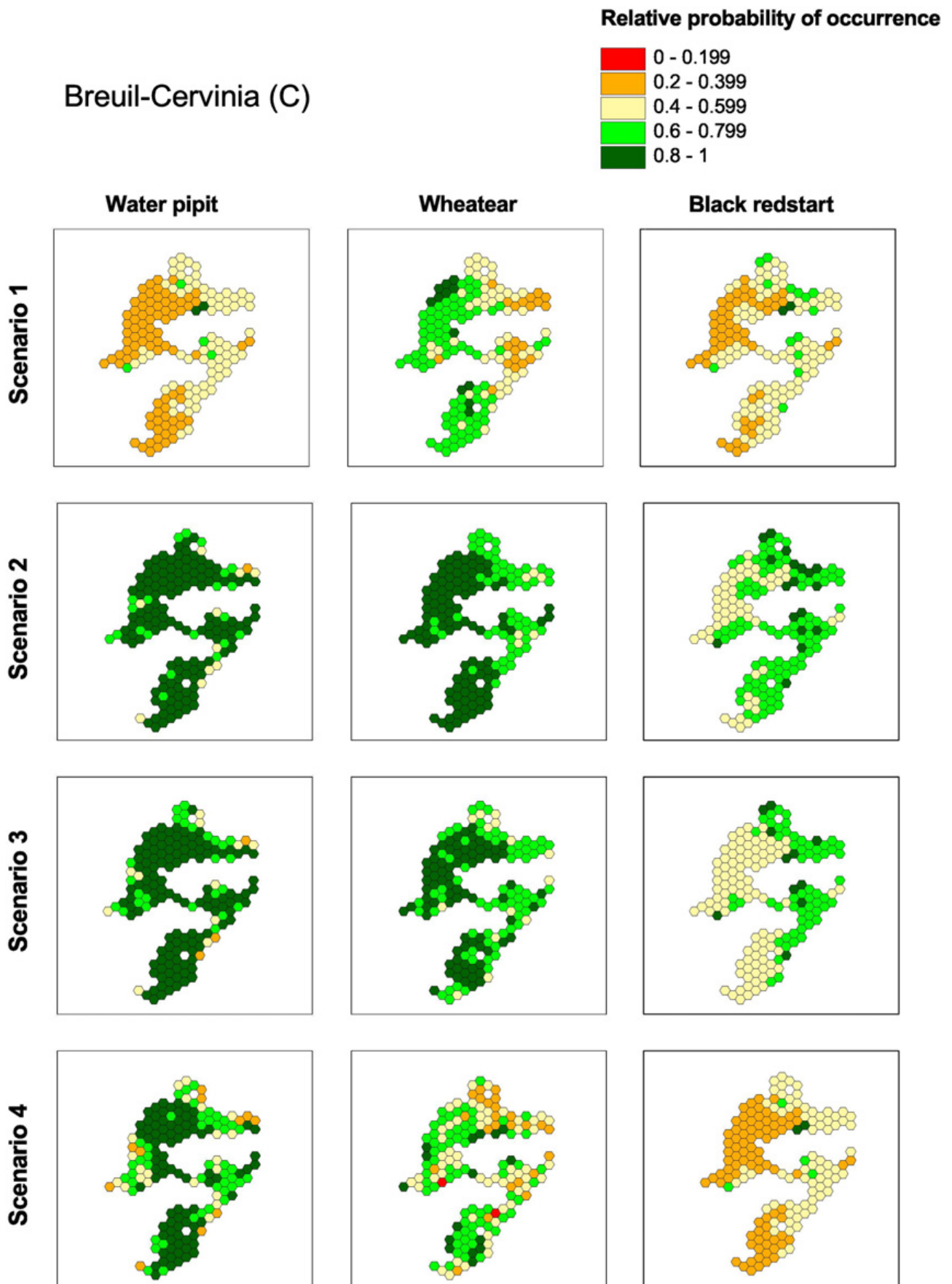


Fig. 4. Example maps of relative probability of occurrence for the water pipit, wheatear and black redstart at Breuil-Cervinia ski-resort. The present situation with ski-pistes

displayed in Scenario 1, Scenario 2 shows a simulated situation with praires restored, Scenario 3 shows a simulated situation with ski-pistes removed and Scenario 4 shows a 10% increase in extent of existing ski-pistes.