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Land-use and bird occurrence at the urban margins in the Italian Alps: implications for planning and conservation.

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Abstract. Negative effects of urban sprawl on biodiversity are widely recognised, but information on how to counteract such effects are still scarce, especially for urban ecosystems in mountain regions.
We evaluated the effect of land-use and topography on the occurrence of eight passerine species in five urbanized study areas in the Italian Alps, which have undergone recent urban sprawl. Our aim was to identify the best planning and management practices to favour these indicator species in mountain peri-urban areas.
We surveyed the species' presence/absence at 142 point counts during the 2013 breeding season and evaluated the effects of environmental variables on occurrence using binomial multivariate adaptive regression splines (MARS). Species occurrence was mostly affected by meadows (positively) and woodland (negative effect, except in contexts with high meadow cover or high habitat diversity).
In Alpine valleys affected by recent sprawl, the conservation or restoration of grassland patches and the maintenance of heterogeneous landscapes can contribute to the conservation of some species of conservation concern found in suburban habitats. Afforestation, a common mitigation practice in and around urban centres, could be counter-indicated when urbanization occurs at the expense of open and semi-open habitats.

Key words: Alps, meadows, mountain landscape, urban sprawl, Trentino.

Introduction

Though urban surfaces cover only 0.51% of the world's land surface (1.78% in the highly urbanized Europe) (Schneider et al. 2009), currently more than 50% of the human population lives in urban settlements (United Nation 2012), a figure expected to rapidly increase leading to a dramatic increase of urban areas (Cohen 2006).

Urban districts grow mainly in suburban areas due to a process known as urban "sprawl", a pattern of human settlement characterized by a patchwork of low-density housing development (Gillham 2002, Robinson et al. 2005), which leads to a disproportionately large increase in the spatial extent of the urban areas (Marzluff 2001) with detrimental effects on natural resources and ecosystems (Gillham 2002). This process consumes natural or agricultural habitats in suburban and exurban lands (*sensu* Marzluff et al. 2001a), reducing their availability for wildlife. It also reduces overall habitat quality, increases fragmentation, isolation and degradation, often facilitating invasion of native habitats by exotic species (Marzluff 2001, McKinney 2002, Speir & Stephenson 2002, Ewing et al. 2005, Salvati et al. 2014).

The increment in the amount of urban areas increases in turn the surface of urban ecosystems (Magle et al. 2012). They are sometimes rich in biodiversity, especially in suburban areas with plenty of gardens and subjected to an intermediate level of urbanization (Crooks 2004, Daniels & Kirkpatrick 2006, McKinney 2006), though the landscape is unlike the original one and vegetation is often dominated by non-native species (Cadenasso & Pickett 2001, van Heezik et al. 2008). Only a subset of native species can cope with changes induced by urbanization (Kark et al. 2007), and generally urban development produces major habitat-loss, increases local extinction rates of native species and results in ecosystem homogenization (McKinney 2002, 2006). In addition, urban development also incorporates within urban boundaries natural or semi-natural habitats, creating small and scattered non-urban remnants (Miller & Hobbs 2002, Wittemyer et al. 2008, Ramalho & Hobbs 2012). The conservation of such remnants virtually provides the only remaining habitats for many wild species and it is important to preserve local biodiversity, ecosystem services, ecological connectivity and to improve the overall urban environment and life quality of people living in cities (Dearborn & Kark 2010, Ramalho & Hobbs 2012). Therefore, conserving biodiversity in and around urban settlements is a big challenge as well as a basic need in an urbanizing world (Miller & Hobbs 2002).

Birds are widely acknowledged as good bio-indicators (e.g. Gregory et al. 2005, Padoa-Schioppa et al. 2006) and are the most studied group in urban areas (Chace & Walsh 2006), consequently the majority of our understanding on the ecological implications of urbanization on biodiversity derives from ornithology. Impacts of urbanization on avian communities are deep and still not completely understood, despite a growing body of information (Chace & Walsh 2006). Urban sprawl effects are a relevant driver of bird decrease (Valiela & Martinetto 2007), although some adaptable species have been reported to have higher population density in cities than in their ancestral habitat (Møller et al., 2012). As a consequence, well-adapted urban birds, due to their ecological plasticity, were also the most successful species when established outside their native range (Møller et al., 2015).

The European Alps, with a population of 14.2 million inhabitants, are among the world's most densely populated mountain regions (Perlik et al. 2001) and nearly two-thirds of the population in the European Alps lives in towns or peri-urban municipalities which are largely undergoing widespread processes of sprawl (Perlik et al. 2001, Perlik & Messerli 2004, Astrade et al. 2007). The Alps are also a centre of biodiversity, exhibiting a high heterogeneity of habitat and climatic conditions along reduced spatial scales and altitudinal gradients (Chemini & Rizzoli 2003). All the negative consequences on environment caused by urban sprawl outlined above, are evident in mountain areas, and in the Alps in particular (Romero & Ordenes 2004, Carruthers & Vias 2005, Perlik 2006, Zimmermann et al. 2010).

Human activities have been shaping Alpine landscapes for centuries, and thus biodiversity conservation cannot disregard managing and regulating human land use, by applying appropriate planning and management practices to enhance biodiversity in human-altered landscapes (Chemini & Rizzoli 2003). Breeding birds in and around Alpine urban settlements become an ideal model to evaluate the importance of maintaining patches of remaining natural or semi-natural habitats for conservation. If different habitats (e.g. forests, grasslands, different crop types) have different values for wildlife, it would be important to understand what habitat(s) should be primarily conserved or promoted in areas subjected to urban expansion.

In this work we focused on the effects of the main natural, semi-natural and agricultural land-cover types which are eroded (or, on the contrary, restored) in the case of new urbanization. In particular, we investigated what environmental variables most affected the occurrence of eight passerine species (six of which are species of conservation concern at the European level) in the Italian Alps. We considered areas that have undergone recent urban sprawl in order to identify the best planning and management practices to favour these indicator species in peri-urban settlements of mountain regions. This is a neglected but potentially relevant topic, as urban sprawl is impacting most Alpine valleys and it is therefore urgent to evaluate what measures for non-urban habitat conservation or restoration should be primarily implemented in urbanized or urbanizing landscapes.

Material and methods

Study system

We considered eight passerine species that in Italy and Trento Province, as in other parts of Europe, are found exclusively or at least regularly in urban and, especially, peri-urban settlements: barn swallow *Hirundo rustica*, black redstart *Phoenicurus ochruros*, common redstart *Phoenicurus phoenicurus*, spotted flycatcher *Muscicapa striata*, common starling *Sturnus vulgaris*, serin *Serinus serinus*, Italian sparrow *Passer italiae* and tree sparrow *Passer montanus*. These species show a various degree of synanthropy, from the obligate commensalism of the Italian sparrow (Summers-Smith 1988), to the often reported dependence of barn swallow on cattle farming (Møller 2001, Ambrosini et al. 2012), to an only moderate adaptation to urban ecosystems of others, that however commonly occur in urban areas, using buildings or gardens to nest.

Six of them are considered of European conservation concern by BirdLife (2004): barn swallow (SPEC 3), common redstart (SPEC 2), spotted flycatcher (SPEC 3), common starling (SPEC 3), Italian sparrow (not evaluated according to SPEC category as recently split from house sparrow (Trier et al. 2014); being endemic to Italy and declining, it should likely be considered as a SPEC 2) and tree sparrow (SPEC 3).

Study area

Trento province (NE Italy) is located in the Italian Alps (6,206 km^2, elevations 67–3,769 m a.s.l. and with 50% of the area lying between 1,000 and 2,000 m). Valley bottoms are intensively cultivated and partly urbanized; mountainsides are covered by woodlands, interspersed with pastures and vineyards (up to 800 m) and with anthropogenic grasslands (800 – 2,000 m); highest areas (2,000 m) are covered by alpine grasslands, rocks and snow.

Mean human density (85 inhabitants/km²; Servizio Statistica PAT 2013) is relatively low, especially in rural and mountain areas.

Urban sprawl in Trento Province is considered a serious threat for Alpine environment (Diamantini & Zanon 2000): although urbanized surface is only about 2.5% of the province, between 1990 and 2006 it increased by 18%, with a major increase of "discontinuous urban fabric", most likely caused by this process (data based on comparison of CORINE Land Cover data for 1990 and 2006).

Within the Province, we identified five valley-bottom study areas (Fig. 1) characterized by small urban settlements which have known recent urban expansion, mainly at the expense of farmland (and in particular meadows). These are found between 360 m a.s.l. and 1375 m a.s.l., but only Talpina area is wholly under the 800 m a.s.l.

Bird data and habitat variables

We defined a total of 142 random points scattered in each of the five study areas according to their extent: 34 in Val di Non, 45 in Valle di Fiemme, 13 in Primiero, 37 in Tesino and 13 in Talpina.

Point counts were at a minimum distance of 300 m one from another and at a maximum distance of 1050 m (mean 262 m, 1SE: 19 m) from the urban patches defined in the most recent land cover map of the area (Servizio Urbanistica e Tutela del Paesaggio 2003). Therefore, all points can be considered to be located at the interface between urban areas and semi-natural landscape.

At each of the point counts we surveyed the presence/absence of our target species in the breeding season 2013 (22 May - 19 July), early in the morning (6.00-11.00 a.m.), in clear weather and without wind. We considered a species occurring at a given point when territorial or foraging birds were observed, whereas we excluded species only contacted as overflying individuals.

Each point count was surveyed twice (Primiero), three (Val di Non, Tesino, Talpina) or five times (Val di Fiemme). Birds were recorded within a fixed radius from the point (100 m; Bibby 2000).

We then measured the land cover as percentage cover of each land-use within a 100-m buffer from each point using the most recent data available (Servizio Urbanistica e Tutela del Paesaggio 2003) and merged some of the original layers to obtain seven land-cover types: woodland, quarries, crops, wooded meadows and pasture, meadows, vineyards and other natural habitats. We also calculated the habitat heterogeneity at each site using the Shannon diversity index of land uses ($H' = -\sum pi \times \log pi$), where pi is the relative proportion of land use i (Krebs 1998). Finally, we computed the mean elevation and the mean slope at each site, using a digital terrain model (resolution: 10 m).

Statistical analyses

To model the effect of habitat variables on species occurrence we used the multivariate adaptive regression splines (MARS), a rather recent machine-learning technique (Friedman 1991, Hastie et al. 2009), which is now increasingly used in ecology (e.g. Heinänen & von Numers 2009, Brambilla et al. 2013) thanks to its flexibility and ability to model complex relationships in species-habitat studies (Elith & Leathwick 2007).

MARS fits non-linear functions, by fitting linear segments to the data, breaking predictors at knots, while connecting adjacent segments at knots, so that the full fitted function does not have gaps, steps or breaks. This approach is ideal to analyse our dataset because it is a non-parametric regression method, which efficiently model non-linear relationships, which were expected in our study system. Moreover, it allows multi-response models, which are particularly useful in our work because some of our target species were present only at few points, this resulting in a low sample sizes and, most importantly, because we are particularly interested in determining what factors are more likely to be important across all species. Multiresponse models are built and pruned in exactly the same way as a single-response MARS model, except that the residual squared errors are averaged across all response variables, with individual basis functions selected to give the best average improvement in performance. The final multiresponse model uses a common set of basic functions for all species, while the variables included in the species-habitat models are the same for all species (Elith and Leathwick 2007). Multi-species models have been shown to better perform than single-species models for poorly represented taxa (Elith & Leathwick 2007, Brambilla & Gobbi 2014).

We used the earth package ver. 3.2-7 (Milborrow 2014a) in R 3.0.3 (R Development Core Team) to fit MARS models. There was no pair of variables highly collinear ($|r| < 0.7$ for all possible pairs of variables). We modelled the potential effect of the ten habitat variables described above and of their interaction on the occurrence of bird species by running: 1) four species-specific models for barn swallow, common redstart, serin and tree sparrow, which had an adequate sample size, and 2) a multi-species model for black redstart, spotted flycatcher, common starling and Italian sparrow, that had a smaller sample size.

We used in all models the following settings for model selection: pmethod=backward (default), penalty=3, degree of interactions=2, maximum number of MARS terms (nk) = 10. Variable importance was evaluated on the basis of the evimp command in earth package (Jedlikowski et al. 2014, Milborrow 2014a). The evimp command estimates the importance of a variable in a MARS model according to three criteria: (i) the number of model subsets generated by the pruning pass, which include a given variable: variables included in more subsets are considered more important; (ii) the decrease in the residual sum-of-squares (RSS) for each subset relative to the previous subset: for each variable evimp sums these decrease over all subsets that include the variable and rescales the summed decreases to a percentage scale (largest one equal to 100); (iii) the generalized cross validation (GCV) of the model, calculated using the penalty argument, which considers the increase or decrease in the

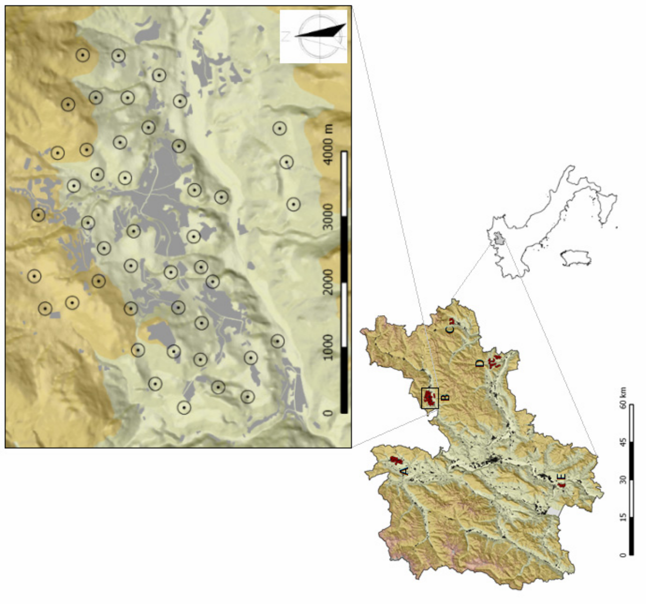

Figure 1. Location of the study sub-areas in Trento Province. A: Val di Non, B: Val di Fiemme, C: Primiero, D: Tesino, E: Talpina. Topography and urban centers in the Province are schematically outlined. The box shows as an example the distribution of 45 point counts and their relative 100 m buffers in Val di Fiemme study sub-area. The position of Trento in Italy is also shown.

GCV associated with a variable being added to the model; the evimp command uses GCV criterion exactly like the RSS criterion (Milborrow 2014a). The plotmo package ver. 1.3-3 (Milborrow 2014b) was used to plot the fitted functions.

Because sampling efforts were not uniform in the five study areas, we ran also a model with sampling effort and study areas as factorial predictors of presence/absence of birds. None of them produced significant output, and then we assumed that the differences in sampling efforts were irrelevant (details not shown for brevity).

Results

The effect and importance of predictors on the occurrence of the eight model species is summarised in Table 1 and Figures 2-3.

The multiresponse model suggested that the occurrence of the four species analysed was mainly driven by woodlands, meadows and elevation. However, such a model had a fairly low explanatory power, except for the spotted flycatcher (Table 1).

The species-specific models had a fairly good performance ($R^2 > 0.2$), except for serin (Table 1).

Only five out of the 10 predictors considered were selected by the two model groups: percentage cover of woodlands, meadows and crops; Shannon diversity index of land cover; elevation (Table 1).

Meadow cover was selected for all species except for the tree sparrow. High meadow coverage

Table 1. Evimp summary of the selected MARS models for the eight species considered. * indicates the variables that appear only in interactions with another. Abbreviation: no. of subset: number of model subsets generated by the pruning pass, which include a given variable; RSS: decrease in the residual sum-of-squares; R^2: model's R square. See text for details.

Variable	no. of subsets	GCV	RSS
multiresponse model (Black restart: R^2 =0.09; Spotted flycatcher: R^2 = 0.26; Common starling: R^2 =0.11; Italian sparrow: R^2 = 0.06).			
woodland	3	100	100
meadows*	2	48.3	73.4
elevation	1	32.7	51.3
Barn swallow (R^2 =0.21)			
meadows*	1	100	100
elevations*	1	100	100
Common redstart (R^2 =0.33)			
Shannon d. i.	5	100	100
meadows	4	33.7	62.9
elevation*	2	30.6	46.1
Serin (R^2 =0.11)			
elevation	3	100	100
crops	2	83.3	81.3
meadows*	2	83.3	81.3
Tree sparrow (R^2 =0.39)			
elevation	5	100	100
woodland	4	74.9	78.8
crops*	1	17.3	28.1

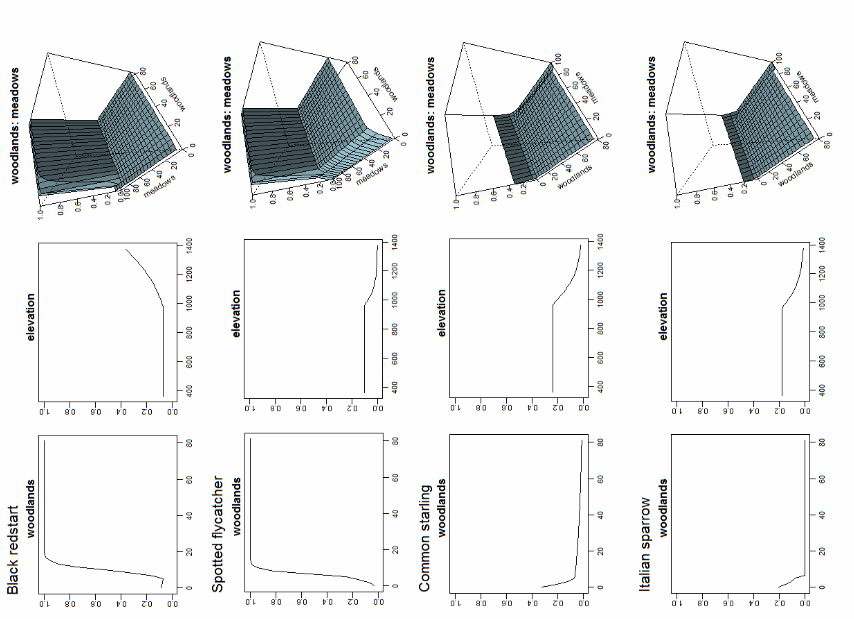

Figure 2. Graphical representation of the effects of predictors selected by the multiresponse model on the species occurrence.

was particularly relevant for barn swallow, black redstart and spotted flycatcher and, to a lesser extent, for serin and common redstart. The latter species was particularly favoured by meadows in heterogeneous landscape, as shown by the interaction of meadow cover and Shannon diversity index.

On the other hand, high percentage cover of woodlands had a negative effect on the occurrence of common starling, Italian sparrow and tree sparrow, whereas woodland cover had a positive effect on the occurrence of spotted flycatcher and black redstart, but only if also meadows occurred in the plot.

Crop cover was selected only for serin, with a negative effect.

Elevation was selected for all the species, generally with a negative effect (Figs. 2-3), with the remarkable exceptions of black redstart (positive effect) and tree sparrow (positive effect at intermediate elevation, but with a curve shape suggesting overfitting; see Fig. 3).

Discussion

Urban sprawl in mountain regions is an increasing process with possible serious consequence on the environment (Romero & Ordenes 2004, Zimmermann et al. 2010). Other types of human impact on mountain ecosystems and species have received a considerably higher attention. Threats as disturbance to wildlife (Arlettaz et al. 2007), soil and vegetation trampling and waste dumping (Geneletti & Dawa 2009), deforestation (Laiolo 2004), soil degradation, fragmentation of natural habitats, barriers to dispersal and edge effects (Rolando et al. 2006, Patthey et al. 2008, Caprio et al. 2011, Walzer et al. 2013), agricultural transformation (Mottet et al. 2006, Agnoletti 2007) have been the subject of specific research, whereas the effect of urbanization has been hardly considered in the Alps. Our work then provides evidences on the importance of conserving or restoring some habitat types rather than others in an area which experienced recent urban expansion.

In particular, we showed that in Alpine urban settlements, which recently underwent expansion at the expense of semi-natural habitats, some synanthropic bird species of conservation concern are mostly favoured by the presence of meadows, whereas woodland has a mainly negative effect and is tolerated only when there is a high meadow cover, or in a context of high habitat diversity.

Conversely, other authors suggested that in areas in which urban expansions eroded natural woodlands, conservation of remnant woodland patches and creation of new ones could be an adequate measure for forest specialists (Myczko et al. 2014), thus conservation strategies should be spe-

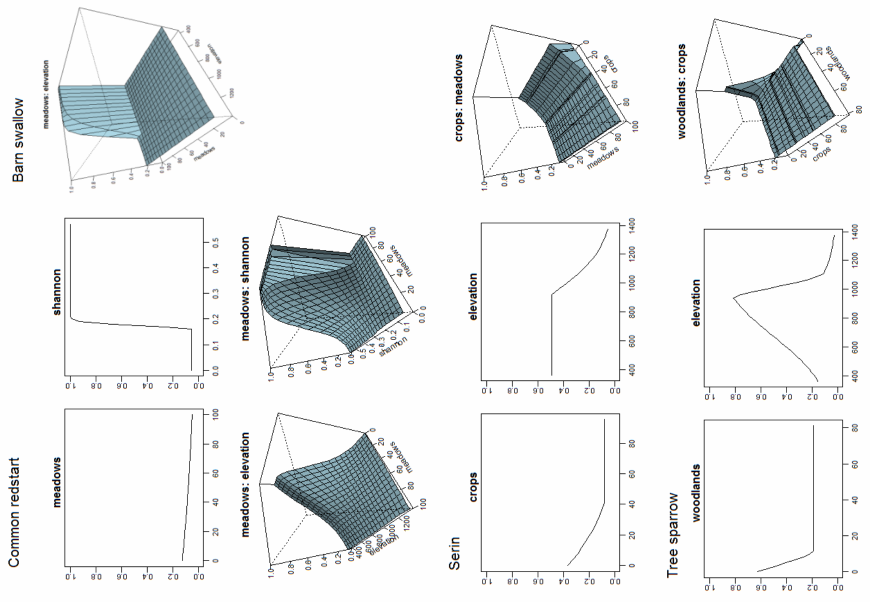

Figure 3. Graphical representation of the predictors selected by the species-specific model on the species occurrence.

cifically targeted.

Our results can have relevant conservation implications and could suggest best planning and management practices, with particular emphasis on open habitats. Urbanization has been reported to impact more on open ground birds than on woody or riparian species (Chace & Walsh 2006), and it is crucial to conserve non-urban remnants (and restore them when possible) in lands subjected to urban sprawl (McKinney 2002, Ramalho & Hobbs 2012). We outlined that the conservation or restoration of grassland patches and the maintenance of heterogeneous landscapes can contribute to favour some species of conservation concern found in suburban habitats. Currently, the commonest measure to buffer or mitigate the effect of urbanization is afforestation, which means planting trees to create stands of trees in areas where there was no forest (Alvey 2006). However, tree planting could be counter-indicated when urbanization occurs at the expense of open and semi-open habitats, as frequently found in the Alps, where villages expand mostly at the expense of secondary grasslands and pastures. In those cases, remnant meadow patches must be conserved and when possible restored in order to create larger patches (cf. Husté et al. 2006) and favour grassland and open-habitat species (Davis 2004; Meffert et al. 2012). The preference shown by open-habitat species for larger patches is explained by the negative impact exerted by the edge effect between cities and grassland, probably because of the loss of preferred grassland cover-types at the urban-grassland interface (Bock et al. 1999, Haire et al. 2000).

The importance of meadows in urban areas has been already emphasized in other geographical areas, where bird species richness is highest in areas with meadow cover, and consequently urban planners are expected to allocate more meadows in the most densely inhabited areas of the city to strengthen the connectivity between existing meadowlands (Lin et al. 2008).

In conclusion, our results and the consequent suggestions for planning are consistent with the conservation strategies suggested for biodiversity conservation in open habitats eroded by other factors, such as the progressive colonization by woodland due to forest encroachment as a consequence of land abandonment, one of the main threats for farmland birds and biodiversity in Europe (Sirami et al. 2006, Brambilla et al. 2010) and also in the Alps (Chemini & Rizzoli 2003, Laiolo et al. 2004).

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