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Habitat selection by the highly endangered long-horned beetle *Rosalia alpina* in Southern Europe: a multiple spatial scale assessment

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Abstract Despite the popularity of the saproxylic cerambycid *Rosalia alpina* as a flagship species, its ecology is still poorly know, especially in the southern part of its range. Detailed information on its habitat preferences is needed to plan appropriate management. We set our multiple spatial scale assessment of habitat preferences in a beech forest of central Italy whose landscape, featuring both unmanaged forest and two types of grazed open forest, allowed us to look at the influence of different land uses. Preferred trees occurred in open sites, and those close to tall undergrowth were avoided. A range of moribund or dead trees were used: those preferred had a lower percentage canopy closure, significantly thicker bark, and were more sun-exposed, than the average. Logistic regression showed that the most important variables for selection were distance from nearest occupied tree, bark thickness, undergrowth height and irradiation. Occurrence likelihood augmented as the distance from nearest other occupied tree increased. Despite being mostly unmanaged, forest was avoided, whereas open forest (with trees pruned by 'shredding') was used more than expected. Although intensive forestry limits the availability of dead wood, closed forest may be unsuitable when shadowing useful substrate. The disappearance of traditional forms of forest management as shredding and moderate cattle grazing may lead to woody vegetation expansion and habitat closure eventually threatening the persistence of *R. alpina*. The return to traditional habitat management would be beneficial to *R. alpina*, an issue that conservation plans should carefully take it into account.

Introduction

Beetles (Coleoptera) are the most diverse animal taxon worldwide and thus key contributors to insect global diversity (Groombridge 1992). Large numbers of species are currently exposed to the risk of extinction because of habitat loss or alteration (Vié et al. 2009). Habitat specialists such as saproxylic beetles are especially vulnerable to human pressure (e.g. Berg et al. 1994) and their sensitivity makes some of them effective environmental indicators (McGeoch 1998). We define saproxylic insects as "dependent, during some part of their life cycle, upon the dead or dying wood of moribund or dead trees or upon wood-inhabiting fungi, or upon the presence of other saproxylics" (Speight 1989; but see also Buse et al. 2007).

In managed forests in many areas of the world, concerns over forest safety, fire risk, productivity and pest species lead to the removal of dead or defective trees by foresters (Dudley et al. 2004). Although some forms of commercial

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forestry such as selective logging may appear sustainable because the overall forest cover—and thus the extent of available habitat—is not significantly changed, they often result in a strong decline in dead wood availability even in protected areas (Dudley et al. 2004). Besides being deliberately removed, dead trees are less available in managed forests, since trees are logged well before completion of their natural cycle. The dependence on dead wood of saproxylic beetles exposes them to the detrimental effects of intensive forest management (Jonsson et al. 2005, 2006). Removal of veteran, moribund or dead trees thus strongly affects the amount of breeding substrate available to saproxylic beetles (Jonsson et al. 2005). Besides, when suitable breeding substrate, or micro-habitat conditions, are lost, even apparently continuous forest corridors can be interrupted, obstructing gene flow. To assess the actual impact of forestry on saproxylic beetles, analyses should therefore be made at multiple spatial scales, from micro-habitat to landscape (Buse et al. 2007).

In this study we analysed habitat selection in one of the most endangered long-horned beetles (Cerambycidae) of Europe, the rosalia longicorn *Rosalia alpina* L. This colourful beetle is often cited as a flagship species, i.e. a taxon "that can gain public sympathy for broader conservation needs" (New 1997). Flagship species are typically charismatic large vertebrates rather than small-sized animals such as insects (Leather et al. 2008), but *R. alpina* is well-known and popular, has beautiful and distinctive colouring, and is featured on stamps and coins in several European countries (Duelli and Wermelinger 2005). Although new records suggest local expansion of central European populations (Čížek et al. 2009), *R. alpina* has undergone a marked decline across much of its range (Luce 1996; Jurc et al. 2008). This cerambycid is listed as a Priority Species under Annex II of the EC/92/43 Directive, and classified as Vulnerable on a global scale (IUCN 2009). Overall the species' range is still wide, but it has shrunk significantly because of human impact (Luce 1996). Moreover, the area of occupancy is largely fragmented, and the species' presence is now restricted to patches of mature forest featuring significant amounts of the dead wood necessary for larval development (Duelli and Wermelinger 2005). Although *R. alpina* is also observed in forests where beech trees (*Fagus sylvatica*) are absent (Čížek et al. 2009), in several regions of its range such as the Italian Peninsula it is currently mostly restricted to beech forest—a fact believed to result from past overexploitation of other lowland forest types where the species once occurred (Luce 1996). Threats to this beetle operate at different spatial scales. On a landscape level, habitat fragmentation is likely to affect especially relatively sedentary organisms such as *R. alpina*, whose limited dispersal range (up to 1 km; Gatter 1997) means they may not manage to cross habitat gaps. Reforestation, in which beech forest is replaced with red spruce (*Picea rubens*), has also resulted in large-scale habitat loss for *R. alpina* (Duelli and Wermelinger 2005). On a smaller scale, the removal of dead wood or defective and veteran trees (whose death sustains the cycling of dead wood degraded by natural agents) is a major threat. Besides, stacked wood attracts adult beetles selecting suitable substrate to lay eggs, but turns into a trap if the wood is burnt or processed before the new generation emerges (Duelli and Wermelinger 2005).

The ecology of *R. alpina* has received little attention. Specifically, most available information comes from the European Alps (Duelli and Wermelinger 2005), but despite its specific epithet "alpina", the species is widely distributed also in southern Europe, where knowledge of its habitat requirements is anecdotal. In such areas, the species has been targeted by conservation actions supported by little or no knowledge of local ecology. To illustrate this, we conducted a web search of LIFE projects funded by the EC in 1992–2008 directly or indirectly targeting *R. alpina* and its occurrence sites (http://www.ec.europa.eu/environment/life/project/Projects/index.cfm, search made on 7 June 2010), and retrieved 15 projects, many of which target areas in the southern part of the species' range where habitat preferences have not been systematically assessed. Factors such as habitat availability, landscape connectivity, type of forest management, climate and patterns of land use change (including land use de-intensification) may all vary profoundly in the different parts of the species' range, and may in turn lead to locally unique ecological responses by *R. alpina*. Besides, no researchers have determined habitat selection by comparing the use of spatial resources with their availability, adopting a multiple scale approach.

Here, we address these crucial issues, thus making a major contribution to knowledge on the ecology of *R. alpina*. First, our study was set in the Italian central Apennine Mountains, the southern part of the species' range. The population we examined is one the most important in the Italian Apennines, whose presence there has been long reported (Luigioni 1931). Second, to take spatial scale into account in the analyses, we determined the selection of spatial features at site, tree and habitat levels by adopting both univariate and multivariate statistical approaches. We compared forested environments that have been long subject to little or no management with others, the structure of which was determined by past traditional management. In our habitat selection analysis, we quantitatively compared the relative value of beech forest subject to no or very little management (the climax successional stage under land abandonment) with that of two neighbouring habitats. The latter were both characterized by open forest cover, the structure of which has been shaped by a long history of past traditional management and moderate cattle grazing.

In summary, we quantify habitat selection by *R. alpina* at multiple spatial scales, ask how different forest use may impact *R. alpina* in beech forests of southern Europe, and give clear recommendations for the conservation management of this rare saproxylic beetle species.

Materials and methods

Study area

The study was carried out in a beech forest area of the Abruzzo, Lazio and Molise National Park, in central Italy (Lat 41°48′N, Long 13°46′E), where the occurrence of *R. alpina* has been long recorded by entomologists and park staff (Luigioni 1931; R. Latini, pers. com.). We defined the study area by choosing limits corresponding to main mountain ridges and other topographic features. The area is mountainous, with a mean elevation of ca. 1,500 m a.s.l. (range 1,278–1,924 m a.s.l.). The substrate is mainly limestone, and is extensively covered with beech forest; other habitats occurring in the area are pastures, often associated with open forests or with scattered trees. Other tree species, such as sycamore (*Acer pseudoplatanus*), are rare. For the aims of our study, we distinguished three habitat types with structures resulting from differences in forest management and land utilization:

a. Forest (total surface 229 ha), mainly unlogged since 1956 or subject to occasional very limited and highly selective logging in some stands;
b. Forested pasture (156 ha), occurring at lower altitudes and today subject to limited grazing by livestock and wild ungulates. In the past, woodland was thinned to favour the presence of pasture. Pastures are interspersed with (or surrounded by) old trees occurring at low densities and in some parts are being covered with expanding shrubs following a reduction in grazing pressure;
c. Open shredded forest (107 ha). In this land use type, trees—occurring at low density—used to be pruned by 'shredding' (removing all side branches), a traditional form of management adopted to maintain open space for pasture, in which branches and foliage were used for animal fodder. This habitat still maintains its typical structure although the area has not been shredded for over 40 years and is now subject only to moderate cattle grazing. Some areas are being encroached by expanding woody vegetation following a reduction in grazing pressure.

Field surveys, data recording and detection probability assessment

We carried out our observations in July and August 2007. At this time of year when adults of *R. alpina* perform courtship and mate, and females lay eggs, sightings by park rangers peak in our study area (R. Latini, pers. com); this phenology is also confirmed for other European regions (Duelli and Wermelinger 2005).

We carefully searched the study area for trees used by the study species by walking strip transects ca. 50 m wide (i.e., a 50-m buffer around each transect was inspected). Transects were mostly parallel in each land use patch and their buffer areas never overlapped. In all, we walked 49 transects with a total length of 17,900 m (366 ± 134 m, mean ± SD); search effort, measured as transect length walked in each habitat type, was stratified according to habitat type occurrence (46% forest, 32% forested pasture, 22% open shredded forest). We examined all suitable trees (termed "available trees"), including large live trees with decaying parts, snags, stumps (i.e. standing dead wood) and logs (fallen trunks). The geographic coordinates and altitude of all such trees were recorded with a Global Positioning System receiver. We did not examine live trees without any dead parts, because during preliminary observations carried out in the same areas no signs of occurrence were observed in these trees.

For the habitat selection, we compared two analyses. The former was based on direct observation of adult insects, i.e. trees were only counted as "occupied" when adult insects were sighted on them. The tops of tall trees were inspected by two observers using 10×42 binoculars. In four cases, we found *R. alpina* remains (possibly resulting from predation by insectivorous birds; Duelli and Wermelinger 2005) on tree bark. We regarded them as evidence for recent use and recorded the tree as occupied. The species creates emergence holes of potentially diagnostic shape (Duelli and Wermelinger 2005), but direct observation of adults may avoid all risk of confusion with holes made by other beetles or woodpeckers, minimize the influence of subjective identification on data collection, and (most importantly) especially exclude trees which may have been occupied in the past, but which may not, at the time of data collection, have provided suitable conditions and surrounding microhabitat for the species.

To check for the effects of differences in surveying using either emergence holes or direct observation we repeated the habitat selection analysis classifying as "occupied" a subset of available trees with emergence holes we attributed to *R. alpina*, and as "unoccupied" all other trees without holes.

We randomly selected 75 previously inspected trees (25 from each of the three habitat categories) from the trees which were available, but not occupied by *R. alpina*, to compare their features with those of occupied trees. Such trees are hereafter termed "unoccupied trees".

For both occupied and randomly chosen unoccupied trees, we recorded the following variables [adapting the sampling protocol of Buse et al. (2007)]: (a) tree diameter

at breast height (DBH, in m); (b) tree height (in m, obtained with a clinometer); (c) percentage canopy closure (assessed visually as the percentage of the tree's branches and foliage which was in contact with those of surrounding trees); (d) mean bark thickness (in mm, calculated as the average value of three bark thickness measurements taken with a calliper); (e) sun index (estimated on a 0–12 scale: 0, fully shaded tree; 12, sun from all directions); (f) tree condition (live; recent snag, loose bark snag, and clean snag, including snags whose trunk was not broken; snag broken above breast height, snag broken below breast height, dead and fallen tree); (g) crown condition (full foliage, foliage or branches partly missing, crown absent); (h) tree position (in forest, in clearing, on forest edge); (i) undergrowth height (0, between 0 and 1 m, >1 m); (j) percentage of the trunk's surface covered with bark; (k) number of emergence holes we attributed to *R. alpina*; (l) distance to nearest occupied tree; (m) elevation (in m above sea level, a.s.l.); (n) % slope; (o) aspect (null, for flat terrain; north; and south); (p) minimum straight distance to closest mountain peak; and (q) minimum distance to habitat edge (forest—open habitat interface). Parameters (n) and (o) were extracted from a digital terrain model developed with software ArcView rel. 3.2 (ESRI Inc., CA). To confirm the representativeness of the tree set selected at random, for parameters (n–q) we also carried out comparisons between used trees ad all available trees and checked whether any divergence in the outcome occurred between the two types of comparison.

Because some values in the frequency distribution of 'tree condition' were too small for statistical analysis, we adopted larger categories, i.e. "live", "snag" (including recent snag, loose bark snag and clean snag), and "broken snag/dead and fallen tree" (including the remaining condition categories).

The problem of detecting individual animals for population size assessment or habitat selection studies in habitats with potentially different likelihoods of direct observation is well known to field ecologists. Occupancy modelling may assess habitat effects on direct observation by analysing detection histories to support, or confute, conclusions gathered from collected data (Milner-Gulland and Rowcliffe 2007). In our case, we assessed the site occupancy probability according to the model by MacKenzie et al. (2002), by employing the software Presence ver. 2.3. Within the study period, 3 weeks after the first visit, we checked occupied trees and randomly chosen unoccupied trees again for the presence of adult beetles. The method is flexible enough to allow for missing observations and thus to use a sample subset for which repeated observations are available; it also allows the user to adjust the probability estimate for habitat-related biases in detection by entering habitat type as a covariate. Model fitting is estimated through the Akaike Information Criterion (AIC). We fitted two models to our

dataset, one assuming detection probability to be constant in all habitat types, the other including the effect of habitat type on detection. To compare the two models quantitatively and evaluate the influence of habitat type on detection, we performed a likelihood ratio test of the null hypothesis of detection probability being constant, the alternative hypothesis being that detection probability varies between habitats.

Habitat selection

We mapped the occurrence of the three habitat types and measured the corresponding areas with Autocad release 14.0. We applied a χ^2 goodness-of-fit test to see whether the observed proportion of use (number of trees used by *R. alpina* occurring in a habitat type divided by the total sample size in that habitat type) departed from the expected proportion (calculated as the number of available trees in each habitat type, divided by the total sample size). We then established selection by calculating Bonferroni's confidence intervals for the occurrence of each land use category (Neu et al. 1974), by using the standard normal deviate or Z statistic. In one analysis, occupied trees represented those for which direct sightings of adults were available, in another the presence of emergence holes was used as a sign of occupancy for comparison.

Differences between occupied and unoccupied sites and trees

We compared the measured variables between occupied and randomly chosen unoccupied sites and trees. Percent slope, aspect, distance to closest peak and distance from nearest habitat edge were also compared between occupied and all available trees. Variables in which differences in mean values occurred were subjected to ANOVA. Conformity to the assumptions of ANOVA was checked by inspecting homogeneity in variance values between compared samples, and by assessing the approximation to normal distribution of the residuals by means of a Ryan-Joiner test. When assumptions were not met, we used a Kruskal–Wallis test. Frequency differences were explored with a χ^2 goodness-of-fit test. When the test showed significant results, we assessed selection by calculating Bonferroni's confidence intervals for each category as explained above. When the data did not conform to the assumptions of the χ^2 goodness-of-fit test, we used a Fisher's exact test.

Multivariate analysis of selection

To explain the multivariate importance of factors that differed significantly between occupied and unoccupied

trees, we used binary logistic regression to model the binary condition (occupied or unoccupied trees) based on a set of predictors including both continuous and discrete variables. We assessed the relative weight of each factor for the overall model significance by generating several models, removing one factor at a time, and observing the percent change in residual deviance (expressed as a χ^2 value). Analyses were performed with MINITAB 13.

Results

Species occurrence and detection of adults

We inspected a total of 737 available beech trees (4.0 ± 2.88 trees/100 m of transect length). We recorded the presence of live adult *R. alpina* ($n = 38$) or remains ($n = 4$) in 5.7% of the available trees (0.22 ± 0.30 occupied trees/100 m of transect length). The detection probability model without habitat covariates produced an AIC score of 324.89. The estimated proportion of occupied sites was 0.055 (0.0088 S.E.) and the probability of detection was 0.95 (0.03 S.E.). The model featuring habitat types produced an AIC value of 320.00, and there was insufficient evidence ($df = 4$, $P > 0.05$) to reject the hypothesis that detection probability varied between habitats. Therefore, the probability of detecting adult insects was high, and constant between the habitat types, i.e. habitat effects on observation proved negligible.

Habitat selection

Available trees were most numerous in forest and least frequent in forested pasture, whereas occupied trees were most numerous in open shredded forest, followed by forest and forested pasture (Fig. 1). Available trees had a similar density in forest (1.83/ha) and open shredded forest (1.78/ha), but were more sparse in forested pasture (0.81/ha). Occupied trees were not randomly distributed in the different habitat types: open shredded forest was the only positively selected habitat; forest was avoided and forested pasture used in proportion to its availability ($\chi^2 = 9.58$, $df = 2$, $P < 0.05$, $Z = 2.4$). An analysis carried out after removing the four trees classified as occupied due to the presence of adult remains led to the same selection pattern ($\chi^2 = 14.13$, $df = 2$, $P < 0.05$, $Z = 2.4$). Likewise, when trees were classified as occupied based on the presence of emergence holes ($n = 117$), we obtained identical results, i.e. open shredded forest was used more than expected (75% of all trees), forest less (32.5%) and forested pasture (48.5%) in proportion to its availability ($\chi^2 = 7.32$, $df = 2$, $P < 0.05$, $Z = 2.4$).
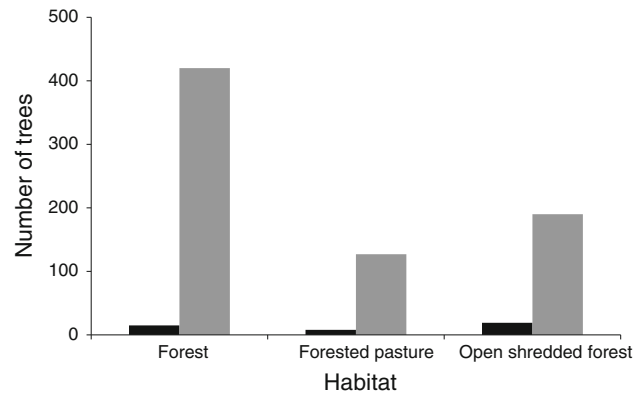
Fig. 1 Total number of trees occupied and unoccupied by *Rosalia alpina* categorized by habitat type ($n = 737$). *Black* columns = occupied trees; *grey* columns = unoccupied trees

Differences between occupied and unoccupied sites and trees

Within habitats, the selection of tree location was non-random ($\chi^2 = 5.58$, $df = 2$, $P < 0.05$, $Z = 2.41$): clearings and open spaces were positively selected, whereas forest interior and edges were used in proportion to their availability. Trees surrounded by tall (>1 m) undergrowth were less frequently used than expected; those surrounded by short or no undergrowth were selected for ($\chi^2 = 4.33$, $df = 2$, $P < 0.05$). Distance to the nearest occupied tree differed significantly between occupied and unoccupied trees, and was significantly larger for the former (114 ± 82 m vs. 32 ± 24 m, $F_{1, 82} = 38.79$, $P < 0.001$).

A two-way ANOVA showed that the altitude of tree sites differed between habitat types ($F_{2, 113} = 97.7$, $P < 0.005$) but not ($F_{1, 113} = 2.7$, n.s.) between unoccupied and occupied trees. A significant interaction ($F_{2, 113} = 5.1$, $P < 0.01$) occurred between habitat type and occupation status (occupied vs. randomly chosen unoccupied trees). A visual examination of the interaction plot showed that this was the effect of unoccupied trees being at higher elevations than occupied trees in forest ($1,583 \pm 85$ m a.s.l. vs. $1,489 \pm 169$ m a.s.l.), but not in forested pasture ($1,295 \pm 22$ m a.s.l. vs. $1,294 \pm 23$ m a.s.l.) or in open shredded forest ($1,384 \pm 37$ m a.s.l. vs. $1,401 \pm 79$ m a.s.l.).

Similarly, the minimum distance to closest peak differed between habitat types ($F_{2, 113} = 8.02$, $P < 0.005$) but not ($F_{1, 113} = 0.1$, n.s.) between unoccupied and occupied trees. The same result was obtained when this variable was compared between occupied and all available trees (habitat; $F_{2, 733} = 20.42$, $P < 0.001$; distance to closest peak $F_{1, 733} = 2.39$, n.s.).

Sites where occupied trees were recorded had a % slope ($45.0 \pm 17.9\%$) similar to that of randomly selected ($46.1 \pm 16.0\%$, $H = 0.12$, $df = 1$; n.s.) or all available

trees (48.5 ± 19.1% $H = 2.68$, $df = 1$; n.s.). Terrain aspect of occupied trees also showed no difference relative to both random ($\chi^2 = 1.16$, $df = 2$, n.s.) and all available trees ($\chi^2 = 0.7$, $df = 2$, n.s.).

No obvious selection pattern for tree condition emerged. Occupied trees comprised four live trees showing decaying parts, 9 snags and 29 broken snags/dead and fallen trees, reflecting the corresponding condition of unoccupied trees (respectively 14 live trees, 16 snags and 45 broken snags/ dead and fallen trees; $\chi^2 = 1.81$, n.s.). Only two dead and fallen trees were occupied, while none of the unoccupied trees selected for analysis showed this condition. Occupied and unoccupied trees were also similar in terms of height, DBH, crown condition and extent of trunk surface covered with bark, but differed significantly in percentage canopy closure, sun index, and bark depth (Table 1). Occupied trees were more exposed to the sun than unoccupied ones, as shown by the lower canopy closure surrounding them and the higher sun index. They also had significantly thicker bark (on average 3 mm). The distance to habitat edge was significantly influenced only by habitat type ($F_{2, 113} = 40.7$, $P < 0.001$), but did not differ between occupied and random trees (Table 1). The comparison between occupied and all available trees (distance to edge 197 ± 203 m) confirmed this result (habitat $F_{2, 733} = 310.76$, $P < 0.001$, occupied vs. available, $F_{2, 733} = 0.36$, n.s.).

On trees where we observed adult *R. alpina*, we found significantly (over eight times) more emergence holes than on those where insects were not seen (70.88 ± 88.3 vs. 8.2 ± 14.8, $F_{1, 115} = 36.18$, $P < 0.001$). We did not use this variable for multivariate selection analysis since holes are a consequence, rather than a predictor, of tree occupancy.

Multivariate analysis of selection

We generated a full logistic regression model (Table 2) by entering all variables that differed significantly between occupied and unoccupied trees. We also entered elevation in the model to account for possible influences on tree selection, since at least in forest this variable differed between occupied and randomly chosen unoccupied trees. The model was non-random, with a full model deviance $\chi^2 = 80.261$.

Four factors were significant determinants of habitat selection: models generated by removing one variable at a time in an iterative process showed that the most important was "distance from occupied tree", followed by "bark thickness", then "sun index" and "undergrowth". The removal of each of these factors from the model caused a % χ^2 deviance change >5% (Table 3). Elevation gave a practically null contribution to model.

Discussion

Habitat preferences at multiple spatial scales

Though our study area supported abundant dead wood and moribund trees, *R. alpina* was present at low density (less than 6% of all inspected trees were occupied). The low density of the study species seems difficult to explain, since we found *R. alpina* occupying a range of different substrate types (from live trees with rotten parts, to fallen trunks) which were widely available in the study area. However, environmental conditions which were revealed at different scales of analysis, and were mainly, directly or indirectly, associated with exposure to sun and micro-climate, explained the observed pattern.

We achieved a high probability of adult detection which was not significantly influenced by habitat type, so direct observation proved robust for habitat selection analysis. This was also confirmed by the full agreement between the analysis carried out on direct sightings and that based on the observation of emergence holes. In fact, analysis of both adult beetles and emergence holes showed that open shredded forest was used more than expected, whereas forest was avoided and forested pasture used in proportion to availability. The selection pattern at the habitat level is

Table 1 Mean values ± standard deviation (SD) for continuous variables taken from trees occupied by *Rosalia alpina* ($n = 42$) and randomly chosen unoccupied trees ($n = 75$), and results of ANOVA of the difference between categories

Variable	Occupied trees (mean ± SD)	Unoccupied trees (mean ± SD)	F	P
Mean DBH (cm)	103.7 ± 30.6	97.1 ± 27.8	1.42	n.s.
Height (m)	8.1 ± 4.8	8.2 ± 6.3	0.01	n.s.
% canopy closure	13.8 ± 29.4	34.7 ± 41.2	8.41	<0.005
Bark thickness	13.7 ± 4.2	10.7 ± 5.2	9.92	<0.002
Sun index	9.9 ± 2.9	6.8 ± 4.3	16.82	<0.001
Distance to habitat edge (m)	272 ± 244	193 ± 181	0.01	n.s.

DBH Diameter at breast height

Table 2 Statistics and levels of significance of the estimated Z coefficients for the binary logistic model (cases: 42 trees occupied by *Rosalia alpina*; 75 unoccupied trees) including habitat and tree variables that proved significant in univariate analyses

Predictor	Coefficient	Standard error	Z	P	Odds ratio
Constant	−7.340	4.505	−31.63	0.103	
Percent canopy closure	0.028	0.016	1.71	0.087	1.03
Nearest occupied tree (m)	0.0303	0.007	4.16	0.000	1.03
Sun index	0.419	0.191	2.19	0.028	1.52
Bark thickness (mm)	0.189	0.068	2.79	0.005	1.21
Altitude (m a.s.l.)	−0.0004	0.003	−0.15	0.885	1.00
Undergrowth height (mm)					
<1	−1.138	0.980	−1.16	0.246	0.32
>1	−1.915	0.884	−2.17	0.030	0.15
Position					
Interior	−0.029	1.385	−0.02	0.983	0.94
Edge	−0.556	0.827	−0.67	0.501	0.57

Table 3 Importance of the independent variables in occurrence on trees of *Rosalia alpina* observed in binary logistic models as percentage change in residual deviation obtained after removing one term at a time from the complete model

Variable	% deviance change
Distance from nearest occupied tree	36.9
Bark thickness	11.3
Sun index	7.2
Undergrowth	6.8
Canopy	4.1
Position	0.8
Elevation	0.0

in agreement with those detected at other scales, because in our study area forest was too dense to host many sun-exposed sites, and possibly too dense to allow free movement of adults. The difference observed between the two open forest habitat types is probably linked with the effects of shredding, which by the removal of foliage and branches allows sunlight to penetrate to the tree. Shredded trees were often very old, had a large diameter and many decaying parts. In open shredded forest, trees potentially suitable to *R. alpina* were more frequently found in the patch interior (probably because edge areas used to be less subject to management and now host fewer shredded trees), whereas in the remaining habitat types they were also common close to margins, a fact that explains the observed effect of habitat on the "distance to edge" parameter. However, no difference in this distance occurred between used and unused trees.

The tree-level variables most important for selection at least in part also reflected the role played by sun exposure. Trees with a high sun index were selected, and most other factors that differed significantly between occupied and unoccupied trees were indirectly related to the degree of sun exposure. Occupied trees had characteristics allowing a high level of sun exposure: they were found at open sites, had a low canopy closure and occurred more frequently in the open shredded forest, characterized by a scattered forest cover allowing more sun to penetrate. The importance of sun-exposed substrates for the conservation of several species of saproxylic beetles has been highlighted in other studies at both species and community levels (e.g. Kaila et al. 1997; Buse et al. 2007; Lindhe et al. 2005; Vodka et al. 2009). The level of exposure to sun may also explain the fact that trees surrounded by tall undergrowth were avoided, and those where undergrowth was either short or absent were preferred, since in the former case tree trunks may be partly or fully shaded and the resulting microclimate too cool for the larvae to develop. However, it is also possible that the movement of a species with such a low flight manoeuvrability as *R. alpina* is impaired by tall undergrowth, and that trees surrounded by shrubs may be avoided simply because they cannot be reached. In fact, the outcome of logistic regression suggests that the effect of undergrowth may be independent of its interaction with irradiation, since both sun index (a direct estimate of irradiation) and undergrowth height concurred in determining the selection at a multivariate level. Noticeably, in the model only the state "taller than 1 m" of the undergrowth height factor proved significant, a result coherent with the hypothesis that the strongest effect of undergrowth is to hinder adult movement.

We found no significant difference in the distance to closest peak (a variable related to elevation) between used and unused trees, although not surprisingly this variable differed across habitats because pastured habitat is mainly found at lower elevation. Similarly, logistic regression showed that altitude (a determinant of ambient temperature) is unimportant for selection, at least within the elevation range found in our study area. However, when we compared the altitude of occupied and unoccupied trees,

we found a significant interaction between habitat type and occupation status. Hence, although across habitats the effect of altitude on tree choice may be outweighed by the effect of habitat type, altitude might play a role at least in sub-optimal habitat such as dense forest. In central Europe, *R. alpina* is reported from *ca* 150 m a.s.l. in lowland floodplain areas (Jendek and Jendek 2006; Čížek et al. 2009), and up to 1,900 m a.s.l. in Austria (Mairhuber 2005). In our study area we observed it up to ca. 1,750 m a.s.l., but records up to 1,850 m a.s.l. are available for the central Apennines (A. Biscaccianti, pers. com.).

Logistic regression also showed that some factors (percentage canopy closure, position), although differing significantly between occupied and unoccupied trees, had little influence at the multivariate level. Such variables may co-vary with others which are more directly linked with sun exposure and outweigh the former in the analysis.

Bark thickness, which ranked second in importance in our multivariate selection model, is important for tree selection by another threatened longicorn beetle, *Cerambyx cerdo* (Buse et al. 2007). In that case, the variable was strongly correlated with tree diameter, and the latter was a powerful predictor of tree occupancy. In our case, although the two factors showed a significant, albeit weak, correlation ($r = 0.275$, $P < 0.005$), DBH did not differ between occupied and unoccupied trees, so the importance of bark thickness must have another explanation, or perhaps relate to other factors (e.g. tree age or shredding). In *C. cerdo*, eggs are often laid in protected deep slits in bark (Döhring 1955; Buse et al. 2007) and this might also be the case for *R. alpina*.

Spatial patterns observed in occupied trees

In species with limited dispersal capacity, the likelihood of tree use is expected to decrease at increasing distances from other occupied trees which act as sources of emerging adults (see e.g. the case of *C. cerdo*; Buse et al. 2007). We found an opposite pattern, however, the greater the distance between a tree and the nearest other occupied tree, the higher its probability of occupancy. Although this result is apparently surprising, it is worth noticing that all distances were <1 km, i.e. within the species' maximum dispersal range (Gatter 1997). Therefore, in our study area *R. alpina* should not find it difficult to cover the distances between occupied trees. This factor appeared to be of great importance, since the distance to the next occupied tree was the most powerful predictor in our model. Although the presence of aggregation pheromones has not yet been documented for *R. alpina*, they are known for the congeneric *R. funebris* (Ray et al. 2009). Should similar compounds be also present in *R. alpina*, their action might lead to the observed distributional pattern. The

consequence of the distributional pattern we observed may be a natural rarity of this species, which, along with scarcity of reproductive sites, may account for the low population density observed and represent a further factor of vulnerability.

Implications for habitat management

Our results show clear selection patterns as determined in beech forest. However, we highlight that *R. alpina* may also use other forest types (Čížek et al. 2009), so our conclusions should not be generalized to other environments to develop appropriate management.

Our results imply that although standing dead wood is certainly important to preserve this species, and that its presence is undoubtedly more significant in unmanaged forest (Ranius and Jansson 2000), the complete closure of forest structure as resulting from the loss of traditional management and land abandonment may affect the species' persistence by shadowing otherwise suitable substrate. The persistence of moderate forms of management such as shredding and sustainable livestock grazing might help this beetle to survive, provided sufficient dead wood is retained. In our study area too, we noticed clear signs of vegetation re-growth in pastures which, in complete absence of grazing by herbivores, might lead to a loss of preferred sites and then a drop in *R. alpina* population in the long term, and the reintroduction of mild forms of traditional management at least at selected sites would help preserve sufficient habitat for the species.

Besides saproxylic insects, our study area supports populations of other animals of conservation concern, including a major population of the barbastelle bat (*Barbastella barbastellus*), another Annex II species under the EC/92/43 Habitats and Species Directive. Reproductive groups of barbastelles roost beneath the loose bark of snags (Russo et al. 2004) also used by *R. alpina*. Larvae of *R. alpina* are frequently eaten by the white-backed woodpecker (*Dendrocopos leucotos*; Aulén 1988). The latter too is a strict habitat specialist depending on deciduous trees and dead wood; for this reason, it has been suggested as a potentially useful flagship, and perhaps umbrella, species (Roberge et al. 2008).

Although belonging to unrelated taxonomical groups and different ecological guilds, these species share with *R. alpina* an observed population decline and a reliance upon dead trees. Our findings, and the above considerations, highlight that to sustain forest animal biodiversity and grant survival to vulnerable taxa, a multiple-scale perspective in land management is needed: dead wood should be retained in structurally diverse forests, within landscape mosaics also including adequately preserved open habitats.

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