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## Modelling geographic distribution and detecting conservation gaps in Italy for the threatened beetle *Rosalia alpina*

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### ABSTRACT

Presence-only models can aid conservation and management of threatened, elusive species. We developed a Maxent model for the rare cerambycid beetle *Rosalia longicorn* *Rosalia alpina* L. in Italy and neighbouring regions and identified the variables best explaining the species' occurrence on a large scale. Once successfully validated, we used the model to (a) evaluate the current degree of fragmentation of *R. alpina* range in Italy; and (b) quantify the amount of the Italian territory with the highest probability of beetle presence within the existing national conservation areas (Natura 2000 network, parks and reserves). Low (<0.5) probability scores of *R. alpina* presence corresponded to 89% of the total area considered, whereas high scores (>0.9) covered only 2.5%. *R. alpina* was predicted to occur mostly in broadleaved deciduous forest at 1000–1700 m a.s.l. with warm maximum spring temperatures and May and November precipitation >80 mm. We found a high degree of fragmentation; gaps were mainly covered with farmland or other unsuitable habitat. Over 52% of potential habitat is unprotected. While the Natura 2000 network protects 42% of potential habitat, parks and reserve covers less than 29%. To preserve *R. alpina*, we urge to create, or restore, forest corridors to bridge the otherwise impermeable gaps our model detected and grant protection to the still largely unprotected area of the Italian territory e.g. by including it in further Natura 2000 sites. Models such as ours may also help focus field surveys in selected areas to save resources and increase survey success.

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### Introduction

Modelling species distribution offers a major approach to identify the key environmental factors determining both current and future spatial patterns of occurrence. Besides exploring basic ecological questions (Guisan & Zimmermann 2000) such models also have practical applications to nature management, such as detecting new populations of threatened species (e.g. Rebelo & Jones 2010), providing support to species conservation or reserve planning (Carvalho et al. 2010; Doko et al. 2011), identifying gaps in geographic distribution and assessing the degree of protection coverage granted by nature reserves (Doko et al. 2011; Domínguez-Vega et al. 2012). Therefore, models can be successfully employed to establish new protected areas, ecological corridors or

to assess the effects of changes in land management (Sowa et al. 2007).

Some of the more recent modelling approaches are based on presence data only, overcoming the often serious limitations posed by the requirement of absence data (Brotons et al. 2004; Phillips et al. 2006). Large-scale, presence-only models are vital to complement the information obtained by local studies: they reveal otherwise overlooked ecological requirements by establishing which environmental parameters influence species distribution on a broad geographical scale (Razgour et al., 2011). For regional-scale management of biodiversity, such models have been used to detect core regions of geographic distribution, recognise or plan continuous corridors or stepping-stone interconnections (Baum et al. 2004; Drag et al. 2011), as well as identify sites where restoration actions are more urgent.

Another promising process which such models may effectively support is gap analysis. The objective in this case is to recognise features in the biota (from species to communities) that are either under represented or not present in the current system of conservation areas (reviewed in Jennings 2000).

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A first step of gap analysis (Burley 1988) is to detect selected features of biodiversity and analyse the current system of protected areas. Then, all elements insufficiently covered by the latter are identified so that priorities for conservation actions (e.g. implementation of reserve networks) can be adequately pursued (Jennings 2000). In practice, gap analysis often needs state-of-art approaches to overcome the difficulties in obtaining the necessary large datasets (Jennings 2000). Specifically, for organisms whose distribution is poorly known, the identification of “conservation gaps” based on the mere observed distribution may be ineffective: in such cases geographic distribution models may be successfully used (e.g. Doko et al. 2011; Domínguez-Vega et al. 2012).

Models calculated for specialist taxa tend to provide greater accuracy than for generalists (Elith et al. 2006): the former usually occur in a more restricted range of ecological conditions than the latter. Hence, it is more likely that a smaller dataset may cover much of the species' niche (Brotons et al. 2004). This is the case with the species addressed in our study: the threatened, emblematic long-horned beetle, the Rosalia longicorn *Rosalia alpina* L., a most popular, conspicuously coloured cerambycid beetle often mentioned as an invertebrate flagship taxon (Duelli & Wermelinger 2005). *R. alpina* is an Annex II priority species under the EC/92/43 Habitats Directive. Presence-only modelling appeared best suited to deal with this beetle's low (at least in cluttered habitat) likelihood of detection and its uneven distribution across habitats (Russo et al. 2011; but see Drag et al. 2011). The large-scale *R. alpina* distribution is still insufficiently known, so that any analysis based on its occurrence range is likely to underestimate the species' actual presence. The sites where this beetle occurs are often overlooked due to insufficient search effort, difficult detection in forest (Russo et al. 2011) and short life span of adults (Drag et al. 2011) which restrict the chances of direct observation. Consequently, potentially large areas where the species persists may not be receiving appropriate management and protection and thus fall outside nature reserves.

Today the wide geographical range of *R. alpina* appears largely fragmented (Jurc et al. 2008; Sama 2002) due to the loss of preferred habitat, i.e. deciduous forest rich in dead wood, the obligate reproductive substrate used by this saproxylic beetle (Čížek et al. 2009; Drag et al. 2011; Michalcewicz & Ciach 2012; Russo et al. 2011). Intensive forestry, implying the removal of dead wood or defective and veteran trees (whose death supports the cycling of naturally degraded dead wood), also degrades the areas occupied by suitable habitats by causing a marked decrease in dead wood availability, in turn strongly affecting the survival of *R. alpina* and other saproxylic organisms (Čížek et al. 2009; Duelli & Wermelinger 2005). Another consequence of forestry is the detrimental effect of stacked wood, attracting egg-laying adult beetles, which eventually constitutes a trap if it is burnt or processed before the larvae develop into adults (Duelli & Wermelinger 2005).

Due to the strong decline shown by some populations, *R. alpina* is classified as vulnerable on a global scale (Baillie & Groombridge 1996), whereas in Europe is regarded as a least concern species (Nieto & Alexander 2010) given its overall wide range and the high number of available records. However, the species' status largely differs across the various regions of its European range, in several of which it is seriously threatened, whereas in others populations are increasing (e.g. Čížek et al. 2009). Land abandonment, leading to woody vegetation regrowth (Russo 2007) may only partly explain *R. alpina* population increase since in most cases the vegetation found in abandoned areas is not yet old enough to provide favourable habitat to the beetle.

On a local scale, studies on habitat selection by *R. alpina* are not numerous (Čížek et al. 2009; Drag et al. 2011; Russo et al. 2011). Russo et al. (2011) produced a habitat selection assessment for a population of the Italian central Apennines and found that grazed open beech forest managed traditionally, providing a large amount

of dead wood exposed to sun irradiation, was positively selected. Drag et al. (2011) performed a detailed assessment of demography and mobility of *R. alpina* in a patchy forest territory of the Czech Republic and showed that this beetle has limited dispersal capabilities so it crucially depends on sufficient stepping-stone interconnections. Both studies advise on local management practices which may improve this species' conservation status such as an increase in semi-open beech woodlands, yet they remark on the importance of appropriate habitat management at larger spatial scales too. No study has so far addressed the ecological requirements of this beetle on a macro-regional scale.

In this study, we develop a maximum entropy (Maxent; Phillips et al. 2006) presence-only distribution model for *R. alpina* to reach the following goals:

1. to carry out the first geographical distribution analysis for *R. alpina* in the southern sector of its distribution range, essentially in Italy and neighbouring regions;
2. to determine which ecological factors may be limiting the species distribution in the study area;
3. to evaluate the current degree of fragmentation of *R. alpina* habitat in Italy;
4. to quantify the extent of the Italian territory where this beetle is likely to occur falling within the existing national conservation areas, namely the Natura 2000 network and the system of protected areas (parks and reserves).

## Methods

### Study area

We selected an area centred on Italy and including neighbouring regions such as south-eastern France, Switzerland, Italy, Austria and Slovenia, approximately between 49°N and 36°N and 4°E and 19°E. Although our aim was to provide a model for Italy, we extended it to large neighbouring regions outside the country to reduce the risk of under-representing some areas of species' potential distribution as may happen when only strictly national data are used (Sánchez-Fernández et al. 2011). Therefore, besides Italian observations of *R. alpina*, our dataset also featured all records falling within the territory of the above states located within 300 km from the Italian borders (Fig. 1).

The study areas, totalling ca. 492,979 km<sup>2</sup> and corresponding to an elevation range of 0–4810 m a.s.l., include regions in the Alpine, Continental and Mediterranean biogeographical regions according to the EC/92/43 Habitats Directive classification.

### Selection of presence data

We used several sources of *R. alpina* presence records: (1) public access databases, including Natura 2000 (<http://www.minambiente.it>), Corine Biotopes (<http://biodiversity-chm.eea.europa.eu>), Global Biodiversity Information Facility (<http://www.gbif.org>), Swiss Biological Records Center (<http://www.cscf.ch>), Fauna Europea (<http://www.faunaeur.org>), CKmap Fauna Italiana (<http://www.faunaitalia.it/ckmap>) and European Environmental Agency (<http://www.eea.europa.eu>); (2) scientific articles and reports (Angelini & Montemurro 1986; Angelini 1986, 1991; Biscaccianti & Casalini 2007; Cecchi & Bartolozzi 1997; Dutto 2005; Favilli et al. 1999; Gobbi et al. 2007; Gobbi 1994; Sama & Schurmann 1980; Sama 1988, 2002; Sforzi & Bartolozzi 2001; Vitali 1999); and, (3) unpublished information, including our own records, those of colleagues and private collectors, museums, nature reserves and parks. We then critically reviewed this dataset by deleting unreliable, ambiguous or duplicate records and excluding those whose



**Fig. 1.** Presence records (black dots) considered for the development of a maximum entropy model predicting habitat suitability for *Rosalia alpina*.

geographical location was not precisely defined; for our analysis we only observations collected in the 31 year period (1980–2011) were used. The resulting database included 260 records scattered across the whole area investigated (Fig. 1). For all records we obtained latitude, longitude (expressed as U.T.M. projection system coordinates in the WGS84 datum source) and year. Care was taken to select presence records at a resolution matching as far as possible that available for the environmental digital representation. We used ArcGis version 9.2 to generate point distribution maps.

#### Environmental variables

We took the following Eco-Geographical Variables (EGV) from the WorldClim – Global Climate Data (version 1.4 available at <http://www.worldclim.org>; for more details see Hijmans et al. 2005): mean, maximum and minimum monthly temperature (°C), monthly precipitation (mm), altitude a.s.l. (m), and those in the “bioclim” dataset (available from <http://www.worldclim.org/bioclim>; see also e.g. Echarri et al. 2009). The latter variables are derived from the monthly temperature and rainfall values in order to generate more biologically meaningful variables (Busby 1991). Land cover variables were taken from Global Land Cover 2000 (available from <http://bioval.jrc.ec.europa.eu/products/glc2000/products.php>). A total of 69 variables in an ASCII file format were used for modelling. All digital information had a resolution of 30 arc second ( $0.93 \times 0.93 \text{ km} = 0.86 \text{ km}^2$  at the equator). The land cover information we used for modelling was based on recent years whereas our dataset featured presence records from as early as 1980. However, this did not affect our model’s performance because (a) the vast majority (87%) of the presence data were collected during the period 2000–2011; and (b) the remaining

species observations referred to sites which did not undergo significant changes in land use, i.e. they are still characterised by mature forest as they were when *R. alpina* was recorded.

#### Modelling procedure

To model the geographic distribution of *R. alpina*, we adopted Maximum Entropy Species Distribution Modelling (Maxent; Phillips et al. 2006) ver.3.3.3k (<http://www.cs.princeton.edu/~schapire/maxent>). It estimates the range of a species by finding the species distribution of maximum entropy (i.e. closest to the uniform) given the constraint that the expected value for each EGV closely matches the empirical average of the occurrence data (Phillips et al. 2006). This approach is especially valuable for species whose available presence data are limited and false absence in surveys is a significant risk (Elith et al. 2006; Phillips & Dudik 2008). To build the model, we used the 260 *R. alpina* presence records and the above mentioned 69 environmental variables. We selected the logistic output format to generate response curves and Jackknife results. In the setting panel, the following options were selected: random seed; write plot data; regularisation multiplier (fixed at 1); random test percentage 20% (percentage of records to be randomly set aside as test points); 10,000 maximum number of background points; 1000 maximum iterations; and, finally 20 replicate effects with bootstrap replicated run type. This replicated run type makes it possible to replicate sample sets selected by sampling with replacement. After the 20 replicates are run, Maxent outputs a single average model. The final map obtained had a logistic format providing the probability of occurrence according to a 0–1 scale. The 10th percentile (the value above which the model classifies correctly 90% of the training locations) was selected as the threshold value for defining the species’ presence. This is a conservative value that is commonly used in species distribution modelling studies especially when considering datasets gathered over a long time by different observers and methods of collection (e.g.: Rebelo & Jones 2010). This threshold was used to reclassify our model into binary presence/absence maps and subsequently to perform the gap analysis.

#### Validation of predictive models

The obtained model was tested with receiver operated characteristics (ROC) plots to evaluate its predictive ability. The area under curve (AUC) of the ROC analysis provides a single measure of model performance (Fielding & Bell 1997) and ranges from 0.5 (randomness) to 1 (perfect discrimination). Additionally, the model was also tested to check whether it differed significantly from what would be expected by chance using the approach proposed by Raes and ter Steege (2007). First, null models were generated by randomly drawing 260 localities in the study area (the same number of presence data as used in the above mentioned distribution models). This procedure was repeated 100 times to obtain a frequency histogram of AUC from which it was possible to determine a probability of AUC value other than chance. Subsequently, the model was also tested for environmental bias in presence data. For that, a distribution model using all presence data was tested 1000 times against a null model with an equal number of random points for the entire study area. Additionally a Jackknife analysis was adopted to estimate which variables were most important for model building. During this process, we generated a number of models. First, each EGV was excluded in turn and a model created with the remaining variables in order to check which variable was most informative. Then, a model was created for each individual EGV to find which variable has the most information that is not present in the others, i.e., the most uncorrelated variable.

Fragmentation of geographic distribution and conservation gap analyses

In the analysis of *R. alpina* range fragmentation we did for Italy, we regarded as a “gap” any distance >1.7 km between consecutive patches in the presence/absence map. This value corresponds to the longest dispersal move recorded for *R. alpina* (1.628 km; Drag et al. 2011). For each gap, we assessed size (minimum distance between consecutive patches) and land use composition occurring in a circular area bridging consecutive patches of high probability of *R. alpina* presence whose diameter corresponded to the distance between them. We also established which land use types occur within such circles, i.e. within discontinuities in species presence. For this aim we used the Global Land Cover 2000 IV level (available from <http://www.sinanet.isprambiente.it/Members/mais/Corine/>) as well as satellite images and aerial photographs (<http://www.pcn.minambiente.it/GN/>). We also measured the total number of patches with a high probability of presence as well as their average area and perimeter from the 10th percentile threshold binary map.

To assess the degree of protection granted to *R. alpina* by the protected area network of Italy, we overlaid the Maxent presence/absence map with the shape files containing the boundaries of the Italian (1) Natura 2000 network such as special areas of conservation (SAC) and special protection areas (SPA) and (2) protected areas (PA), i.e. national parks; regional parks; and, state reserves. The Natura 2000 network included 2287 SACs and 601 SPAs; 974 PAs were considered for analysis (data from <http://www.minambiente.it/>).

Several studies that have assessed the level of protection offered by nature reserves consider species' occurrence location (Maiorano et al., 2007; Parra-Quijano et al. 2012). Besides using the Maxent model of occurrence likelihood, we also conducted this analysis to compare the outcome provided by the two approaches (Campedelli et al. 2010; Doko et al. 2011).

All analyses were carried out with ArcGis 9.2 and DIVA-GIS version 7.5 (<http://www.diva-gis.org/>; Hijmans et al. 2001).

Results

Assessment of probability of *R. alpina* presence

The model identified substantially uninterrupted areas of geographic distribution in Switzerland, central Austria, southwest Germany, southeast France, Corsica, Balkans, Italian Alps and Apennines. In the remaining area, only very limited and scattered sites were detected (Fig. 2).

We also detected some areas characterised by a high (>0.70) presence likelihood where no records for the species were available (Figs. 1 and 2), such as a small area across south Germany and north Switzerland (Baden-Württemberg, Aargau and Schaffhausen regions), southern Austria near the borders with Slovenia and Italy (among Salzburg, Kärnten and Tirol), and some areas along the southern Italian Alps (from Piedmont and Val d'Aosta to Veneto). The model also predicted that *R. alpina* does not occur on Sardinia.

Our model predicted that 89% of the total study area has a low (<0.5) probability of presence. Over 57% of the total area was classified as of very low (0–0.1) presence probability whereas only 11.4% obtained a species presence score >0.6, with probability of presence values of 0.9–1 accounting for 2.5% of the total area (Fig. 3). Presence records mainly (80% of total sample) corresponded to sites whose probability of presence was >0.6 (Fig. 3).

The model achieved a 1.8 regularised gain value indicating good fit to presence data. Eight variables contributed to a total 70% of model prediction (Fig. 4). The analysis of single variable contribution (Fig. 4) showed that altitude (18%) and land cover (12.6%) were the main factors influencing model performance. May

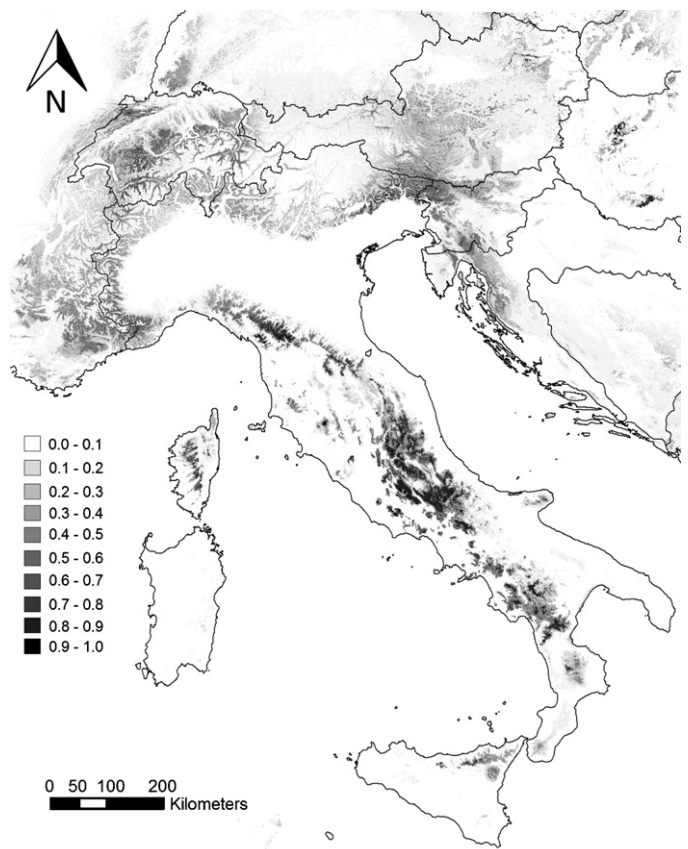


Fig. 2. Maximum entropy model developed for *R. alpina* in the southern sector of its geographical range. Percent probability of occurrence is pooled in ten categories and expressed as different shades of grey as in legend.

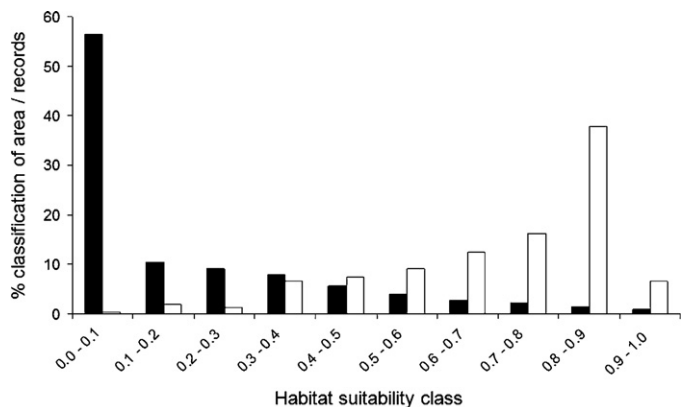
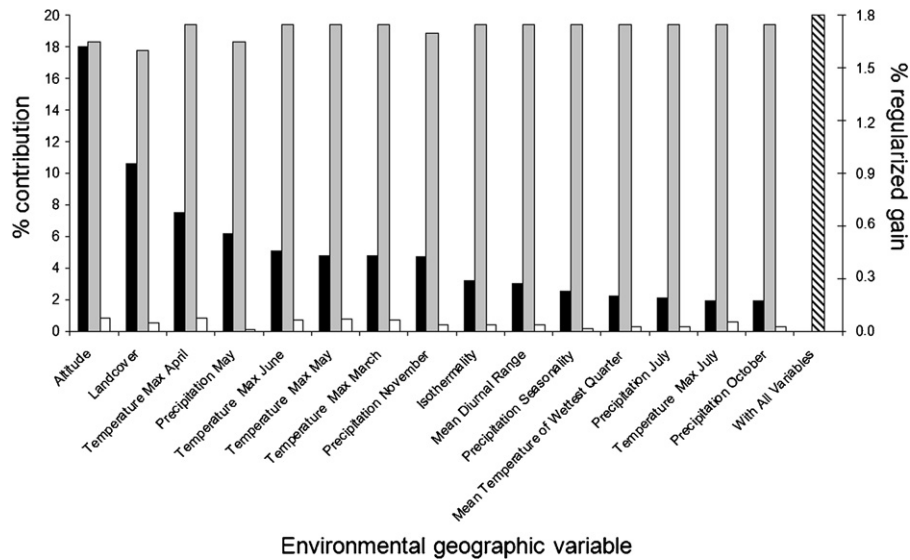


Fig. 3. Percent classification of study area surface (black) and presence records (white) according to 10% habitat suitability intervals derived from the maximum entropy model developed for *Rosalia alpina*.

and November precipitations accounted for 8.2 and 6.7% of total contribution, whereas maximum temperatures for spring months (March–June), ranging from 7.5% (April) to 4.8% (May) provided a total contribution of 23.6%. We verified that the variables which proved most important for the model (especially altitude, land cover, and May and November precipitations) had a lower degree of correlation than those contributing less by observing that only when the former were withdrawn from calculations the overall gain decreased the most (Fig. 4).

Based on the model's predictions *R. alpina* has a higher (>60%) probability of occurring in broadleaved deciduous forest at relatively high altitudes (1000–1700 m a.s.l., peaking between 1400



**Fig. 4.** Representation of the contribution provided by the first 15 environmental variables considered to develop the Maxent model for *R. alpina*. Black bars show the percent contribution of each variable to the model and corresponding values are given on the left axis. Jackknife results for the model (values on the right axis) are also shown for single variables (white), for all variables except the one selected (grey) and for all variables (diagonal shade).

and 1500 m a.s.l.) characterised by warm maximum spring temperatures ( $>0^{\circ}\text{C}$  in March,  $>5^{\circ}\text{C}$  in April,  $>10^{\circ}\text{C}$  in May, and  $>13^{\circ}\text{C}$  in June). Moreover, *R. alpina* selects areas with a narrow range of precipitation values (80–120 mm) in November characterised by heavier precipitation ( $>100$  mm) in May (spring).

#### Validation of Maxent model

The AUC was high, with mean values of 0.991 for the training data and 0.977 for test data i.e. values indicate an excellent predictive ability. The very low AUC standard deviation (0.001) shows there was no overfitting around the presence data. This AUC value was significantly different from what could be expected by chance alone ( $p < 0.05$ ), i.e. presence data were environmentally biased. The second set of null models developed gave a 95% C.I. AUC value of 0.961, also corresponding to a very good modelling performance.

#### Fragmentation of geographic distribution and conservation gap analyses

From the 10th percentile threshold map (Fig. 5) we identified 2029 patches representing the geographic distribution for *R. alpina* across the entire Italian territory. The analysis revealed a high degree of fragmentation: 70.4% of patches cover less than  $1\text{ km}^2$ ; 22.1% lie between 1 and  $9\text{ km}^2$ ; and, only 7.5% cover  $10\text{ km}^2$  or more. The mean area and perimeter of patches are  $10.84 \pm 124.57\text{ km}^2$  (range: 0.44–4595.65) and  $10.37 \pm 58.69\text{ km}$  (range: 3.12–1885.25), respectively. Land use types occurring in such patches are mainly beech forest (30%) followed by deciduous mixed oaks woods (9%), grasslands continuous and discontinuous (7.90 and 7.70%) and mixed forests dominated by mesophilic and mesotermophilic species (7.50%). Several other land use types occurred with land cover values (Table 1).

We identified 421 gaps (patches spaced out  $>1.7\text{ km}$ ) measuring  $14.77 \pm 10.68\text{ km}$  (range: 1.79–115.1). Although the largest gaps occur in southern Italy (the maximum is a 115-km gap in the Calabria region) the habitat also appears heavily fragmented in the northern sector of the country where most small patches occur. In the Alps, several gaps only slightly exceeding 1.7 km were found, i.e. connectivity in the Alps appears to be more favourable to the beetle's presence. Within gaps (Table 1) 52.90% of the land is

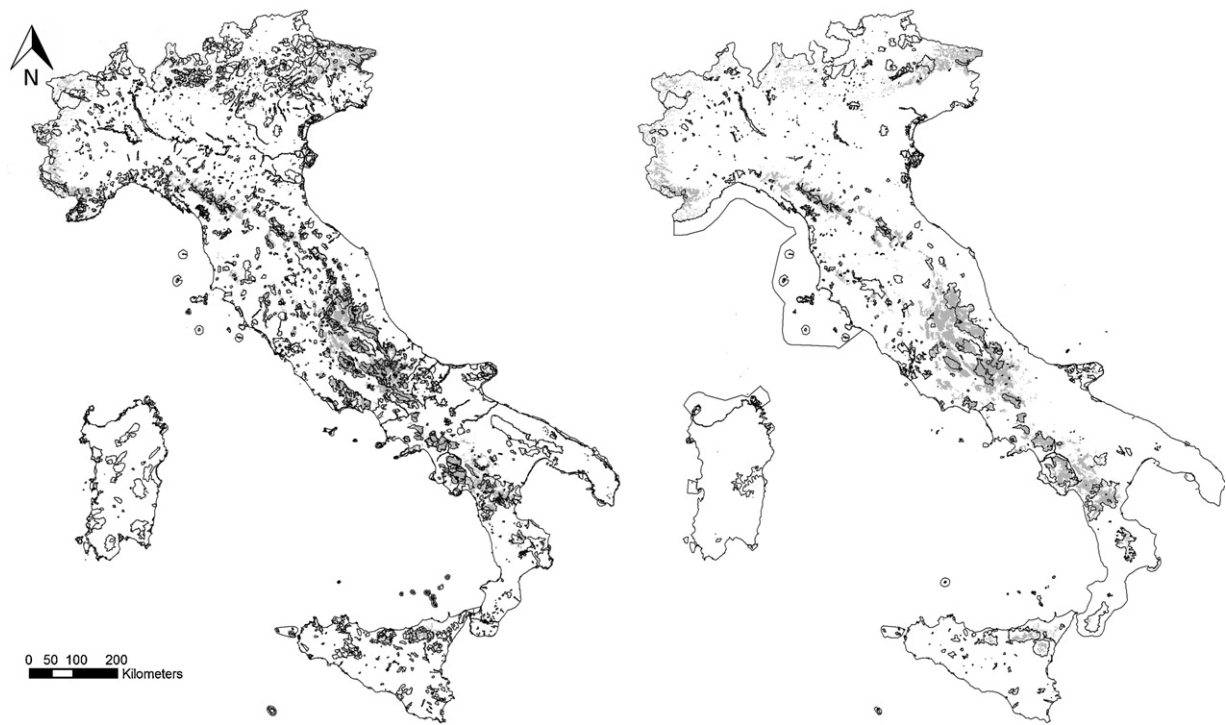
represented by agricultural areas, namely by intensive agriculture (15.49%), olive groves (10.50%), complex cultivation (10.20%) and extensive agriculture (8.54%). Another 31% of gap areas are covered by forest and semi-natural areas with a higher presence of conifers (8.60%), oaks (7.8%), chestnuts (6.20%) and hornbeams-maples woodland complex (4.30%). The remaining 15% of gap area features Mediterranean woodland (12%) and artificial surfaces (3%).

The overlay between the existing system of conservation areas (SACs, SPAs and PAs) and the binary presence/absence map showed that over 52% of potential habitat is unprotected (Fig. 5). Taken separately, SACs and SPAs respectively protected ca. 32% and 33% of potential habitat. However, because the two site types often spatially overlap, when taken together the situation only slightly improved, i.e. the entire Natura 2000 network protects 42% of potential habitat. PAs offer a lower degree of protection, covering less than 29%. The analysis based on recorded (observed) presence offers a more optimistic scenario, with an overall 82% of presence points represented in the existing conservation areas (Fig. 5). SACs, SPAs and PAs cover 60%, 67% and 60% of the record dataset. Overall, based on this analysis the Natura 2000 network includes ca. 76% of record points.

## Discussion

### Assessment of geographic distribution

We succeeded in developing a model which detects a set of environmental variables that, on a broad spatial scale, explain a non-random pattern of geographic distribution. Model's predicted power was first of all supported by the high gain value achieved. In general, models whose AUC  $> 0.75$  are regarded as reliable (Elith 2002). In our case, AUC values  $> 0.998$  demonstrate an especially high predictive capacity – in fact among the highest obtained in published models (e.g.: Domínguez-Vega et al. 2012; Rebelo & Jones 2010). We also gathered practical evidence of our model's performance by identifying previously unknown areas of presence. For example, in one Italian region where no published record was available, Val d'Aosta, we were recently informed of the extraction of a *R. alpina* elytra from a barn owl (*Tyto alba*) pellet (M. Dutto, pers. comm.): given the frequent pellet ejection of this bird (Bunn et al. 1982), we assume the beetle remain to be local, thus confirming our



**Fig. 5.** Overlay between suitable areas for *R. alpina* (grey) and boundaries of Natura 2000 network (left) and protected areas (national parks, regional parks and nature reserves; right). Suitable areas were extracted from the 10th percentile threshold binary map of the Maxent model.

prediction. Although our model provides a robust picture of the species' distribution and is satisfactorily validated, we are aware that some limitations to its full representation may arise from the absence of “realised niche” predicting factors, such as e.g. biotic interactions and dispersal. The latter could certainly play an additional role in determining population viability (Elith & Leathwick 2009).

A model developed for a broad geographical scale such as ours is not aimed at resolving locally crucial features of preferred habitats, such as those visible on a plot scale (Russo et al. 2011), but one of its scopes is to detect the influence of factors such as climatic variables otherwise missed in analyses based on a finer-grained (local) scale (e.g. Lomba et al. 2010). We provided important knowledge on a broader scale complementing what is so far known about local

**Table 1**  
 Proportion of land use classified after the Corine Global Land Cover 2000 IV occurring in suitable habitat patches and distribution gaps in Italy according to the MaxEnt model developed for *R. alpina*. Land use types <1% in both suitable habitat patches and distribution gaps are pooled in the “Other” category. Land use types are ordered according to decreasing proportion values in suitable habitat patches.

Corine land cover IV level	Suitable habitat patches (%)	Distribution gaps (%)
Beech forest	31.05	0.00
Deciduous mixed oaks woods	9.9	7.38
Mesophilic and mesotermophilous mixed forest	8.54	3.25
Grasslands discontinuous	7.94	2.73
Grasslands continuous	7.73	2.73
Other	6.29	5.08
Heath and scrub	5.82	1.89
Land principally occupied by agriculture with significant areas of natural vegetation	4.97	4.01
Chestnut forest	4.06	6.2
Silver fir and/or spruce forest	3.41	0.00
Holm oak and evergreen woods	2.74	0.00
Mountain pine forest	2.11	0.00
Coniferous forest	2.08	8.6
Larch and/or pine forest	1.94	0.00
Bare rocks, cliffs, rocks, outcrops	1.42	1.18
Discontinuous urban fabric	0.00	2.44
Intensive agriculture	0.00	15.05
Extensive agriculture	0.00	5.54
Permanently irrigated land	0.00	3.55
Fruit trees and berry plantations	0.00	2.72
Olive groves	0.00	9.5
Pastures	0.00	1.51
Annual crops associated with permanent crops	0.00	3.54
Complex cultivation patterns	0.00	9.2
Mediterranean shrubs (high)	0.00	1.78
Mediterranean shrubs (low)	0.00	2.12
Total	100.00	100.00

patterns of habitat selection by *R. alpina* (e.g. Duelli & Wermelinger 2005; Russo et al. 2011) and overall offering an ecologically comprehensive picture of the species' habitat requirements.

The first two variables characterising the predicted broad scale distribution of *R. alpina* were land use and altitude, contributing together ca. 31%. Areas of highest probability of presence corresponded to deciduous broadleaved forests situated at elevations between ca. 1000 and 1700 m a.s.l. In most of the study area (such as in the Italian Apennines) these are represented by beech (*Fagus sylvatica*) forest. This elevation belt is most likely to shift locally within the range we detected according to climate and topographic factors, so our outcome refers to the overall altitudinal distribution of beech forests across the study area. Beech forest is well known to be the main forest type used by *R. alpina* both in western (Duelli & Wermelinger 2005; Russo et al. 2011) and eastern (Drag et al. 2011) Europe. The importance of significant levels of precipitation in May and November, contributing both ca. 11%, may likewise be explained in terms of climatic requirements of beech forest (Fang & Lechovicz 2006) since they may prevent late spring water stress, especially in lower elevation forest stands, and favour radial growth (e.g. Piovesan et al. 2005).

Noticeably, when taken together monthly maximum temperature variables for spring months provided a significant contribution to model performance, addressing the importance of at least some daily hours of warmth in that season. In insects, developmental rates are affected by temperature levels within a given thermal window, i.e. the range in temperature between the minimum and maximum rate of development for individual species (Dixon et al. 2009). In our case, the most likely biological explanation is that warm spring temperatures may help the final stage of larval development, because before the last winter *R. alpina* larvae move to the surface and in spring build pupal cells beneath bark where they will eventually undergo metamorphosis in summer (Duelli & Wermelinger 2005). Warm temperatures also make thermophilous species more active, and this may partly influence the likelihood of recording the beetle's presence.

The large-scale thermal requirement highlighted in our model is paralleled by the microclimatic preferences to emerge from local habitat selection analysis. For instance in beech forests of central Italy (Russo et al. 2011) trees occupied by *R. alpina* were found at open sites, had a low canopy closure and occurred more frequently in an open shredded forest allowing more sun to penetrate. Selected trees were also surrounded by little or no undergrowth, a feature exposing their trunk to sun and determining a microenvironment more suitable for larval development.

#### Fragmentation of geographic distribution and conservation gap analyses

Our model succeeded in detecting the numerous *R. alpina* populations reported for mountainous regions and especially in beech forests, where the species is recorded over the vast majority of its range, often following the distribution of such forests.

Though *R. alpina* has been documented to be locally expanding its geographical range in some areas of central Europe (Čížek et al. 2009), our model made it possible to assess quantitatively the highly fragmented distribution pattern mentioned in the scientific literature for this beetle (e.g. Bense 2002; Binner & Bussler 2006). The main land use type nowadays occurring in gaps between fragments is farmland. Although agricultural land mostly occurs at medium to low elevation, large-scale deforestation and farming have certainly reduced *R. alpina* habitat continuity and limited its dispersal by affecting lowland populations which may occur in forest types other than beech (Čížek et al. 2009; Gobbi 2006) in several European areas, from central Europe to coastal Mediterranean. In forest landscapes of central Italy a negative relationship between

the occurrence of deciduous broadleaved forest and farmland is often observed (D. Russo, pers. obs.), i.e. cultivated areas are grown at the expense of *R. alpina* suitable habitat. It might be questioned whether the current preference for *F. sylvatica* forest results at least in part from the large scale deforestation, the recurrent wildfires (especially in the Mediterranean region) and the application of intensive forestry regimes that have profoundly affected lowland forests. As Horák et al. (2010) point out, trees suitable for *R. alpina* are often those that remain standing on difficult-to-access slopes, where logging is too laborious. Apart from the macro-scale dealt with by our model, it is also important to remark that on a habitat scale, *R. alpina* often selects large trees that are either dead or show significant portions of decaying wood (Duelli & Wermelinger 2005; Russo et al. 2011), features which are under represented or absent in intensively managed forests. Therefore, the mere presence of forest does not grant a sufficient degree of habitat suitability per se.

In this study, all discontinuities in the species distribution >1.7 km were regarded as a "gap". Data on *R. alpina* mobility are still limited. According to Gatter (1997) movements are generally <1 km; Drag et al. (2011) recorded the value we used for modelling by mark-recapture – an approach that, as the authors put it, "gives lowest estimates due to underestimation of long-distance movements". In our study, we felt that adopting a cautious approach for such a threatened, management – dependent species was important so we refrained from using hypothetical longer threshold distances. Although we may have of course overlooked some longer movements and their modelling implications, our results are nonetheless robust: the mean gaps in the Italian *R. alpina* distribution were ca. 15 km, most probably representing a significant obstacle to gene flow and colonisation of new areas even if the species were more mobile than it would be according to our estimation. The smallest gaps recorded, concerning the Alpine areas, suggest that fragmentation is a somewhat less significant threat to the beetle in that region.

Noticeably, the overall network of Italian areas is largely insufficient in granting protection to over half of *R. alpina* geographic distribution. We found that the Natura 2000 network does somewhat better than parks and reserves, confirming the importance of European environmental policies to protect threatened species. Rosati et al. (2008) found that over 59% of potential natural vegetation types occurring in Italy (which may act as a proxy of the country's environmental and biogeographical diversity) is not represented in the parks and reserves network yet 68% falls within the Natura 2000 network. Both this result and ours can be partly explained by the fact that Natura 2000 sites have been specifically designated to protect species and habitat of community importance. However, the effectiveness of the Natura 2000 network for the protection of *R. alpina* may only partly be seen as the result of a deliberate strategy. In fact, the network covers an area that is ca. twice as much that of national parks and reserves. Besides, both the number of records and the extent of the area with a high likelihood of species presence falling within SPAs are somewhat greater than those found in SACs. According to the EC/92/43 Habitats Directive, SACs are specifically designated to protect non-avian species, including *R. alpina*, whereas SPAs target birds only. From this perspective, the degree of protection offered by SPAs to *R. alpina* would be in fact a "by-product" of the bird conservation strategy. This has important implications because site management is usually planned according to the target species: it is therefore of utmost importance to consider the occurrence of other management-dependent species besides birds in SPAs so that such species will receive an appropriate degree of protection.

The conservation gap assessment based on presence points led to higher protection performances. This is not surprising since most distributional studies of threatened taxa such as *R. alpina* have

been carried out in (and often sponsored by) protected areas, so the known distribution of this beetle is largely biased towards such areas whereas large portions of geographic distribution (=high likelihood of species occurrence) inevitably fall outside them. This once again highlights the limits of basing conservation gap assessments on presence records only and the contribution of modelling probability of presence for threatened species to plan adequate protection and improve species management (e.g. Domínguez-Vega et al. 2012; Matern et al. 2007).

### Conservation implications

From our model it is apparent that the current degree of fragmentation may be an obstacle to gene flow between suitable patches affecting metapopulation dynamics but genetic analyses of population structure are needed to fully support this point. Besides locally improving forests by applying more sustainable management and increasing dead wood availability (Drag et al. 2011; Russo et al. 2011) our study urges that conservation measures on a large spatial scale should be adopted too. Specifically, creating, or restoring continuous or discontinuous (stepping stones) forest corridors to bridge the otherwise impermeable gaps detected by our model represent a key issue to connect habitat fragments and support gene flow. Besides planting new forest areas to restore connectivity on a regional scale, to favour *R. alpina* dispersal further improvements may include the replacement of conifer plantations with deciduous broadleaved trees (Duelli & Wermelinger 2005) as well as the adoption of a more sustainable management for logged forests. As proposed for other snag-dependent species (Russo et al. 2010), in many cases significant results might be reached by simply designating small groups of trees, or even single ones some hundred metres apart as snag replacement in managed woodland (Moorman et al. 1999) – i.e. well within *R. alpina* dispersal range (Drag et al. 2011) – to connect more effectively the areas occupied by the beetle. Based on our model, we also advise to carefully consider the still largely unprotected portion of the Italian territory of *R. alpina* for inclusion in further Natura 2000 sites to be designated in the future.

Finally, to best achieve the monitoring goals explicitly addressed by the EC/92/43 Habitats Directive, we recommend the application of models predicting the occurrence of *R. alpina* as an especially attractive method to optimise surveys where resources to support them are insufficient. This will help select the areas where to carry out field surveys and limit search effort.

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