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Management tradeoff between endangered species and biodiversity conservation: the case of *Carabus olympiae* (Coleoptera: Carabidae) and carabid diversity in north-western Italian Alps

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Abstract

The worldwide loss of biodiversity is of great concern, and this has lead to strong policy frameworks to promote the maintenance of biodiversity through protecting species and habitats in many countries. Often however, rare species with legal protection provide a conservation focus, and therefore may dictate management policy. Co-occurring species, especially those without formal protected status, may contribute significantly to overall biodiversity, but may nonetheless receive limited attention. In such cases, conflicts may arise between the needs of individual species and those of wider biodiversity. We examine such a conflict in alpine ground beetles of the Sessera Valley (NW Italian Alps). This area is the type locality of the endangered species *Carabus olympiae* Sella, 1855, an endemic species known only from two restricted sites in the Alps. In order to assess priorities for *C.olympiae* conservation we used habitat suitability modeling, and identified potential indicator species that could serve as an indicator of the occurrence of the target species. In order to assess priorities for ground beetle diversity conservation we described the local ground beetle community by calculating abundance, species richness, average taxonomic distinctness (Δ+), taxonomic distinctness (Δ*) and functional diversity (FD). The best management option for the conservation of ground beetle diversity was found to be maintenance of unforested habitats, in particular pastures, which showed the highest values for all the parameters considered. Forested habitats (artificial plantation and beech forest) had the lowest ground beetle diversity. However, the preferred habitat for *C. olympiae* was beech forest, especially at medium altitudes (800 to 1400 m) and with sloping terrain. Moreover, *C. olympiae* presence could be indirectly assessed by the co-occurrence of four indicator species: *Tanythrix senilis*, *Carabus depressuss*, *Cychrus italicus* and *Pterostichus appenninus*. Our main conclusion is that the most promising landscape-scale approach to preserve both the relict steno-endemic species *C. olympiae* and the high local syntopic ground

beetle diversity is to maintain traditional pastoral activities along with the preservation of the unmanaged beech forests.

Keywords: relict species, invertebrate conservation, GLMMs, suitability model, indicator power

Introduction

The worldwide loss of biodiversity is of great concern and has increasingly become a research focus since the United Nations Convention on Biological Diversity in 1992 (Spray et al., 2004). The European Union (EU) has a relatively strong and comprehensive biodiversity conservation policy framework, with key instruments including the Birds (79/409/EEC) and Habitats (92/43/EEC) Directives. The main aim of the Habitats Directive is to promote the maintenance of biodiversity by requiring Member States to take measures to maintain or restore natural habitats and wild species listed in the Annexes to the Directive at a favourable conservation status, introducing robust protection for those habitats and species of European importance.

Research underpins implementation and management, although at times it seems oblivious to what can actually be achieved in practice. Conversely, management practitioners need to be more aware of research findings, and how they might improve management strategies (Samways, 2005). Within this framework, managers, policy makers and regulatory authorities need tools for assessing the probability of presence of rare species in order to monitor and preserve local populations. Often, rare species with legal protection and listed in the Annexes of European Directives or international Conventions are the main conservation focus and thus provide the main bases for funding, while co-occurring species without formal status may be of perceived lesser concern and receive limited attention, if any. However, land management practices that focus on one or a limited number of target species could place local biodiversity at risk (Lambeck, 1997). There are numerous examples of such conflicts from a range of taxa (e.g. plants, Severn and Moldenke, 2010; invertebrates, Launer and Murphy, 1994; mammals, Berger, 1997) where the protection of a single rare target species through habitat management provides no benefit, and possibly may even harm, other key species or species groups in protected areas. Particular cases apart, conservation ethics and the need to secure ecosystem functioning suggest that management choices about a target species should not lead to a conflict with local, syntopic diversity.

The endangered ground beetle species *Carabus olympiae* Sella, 1855, is the epitome of an alpine endemic, with a very restricted range and low population density. Moreover, this carabid species, a descendant of a once widespread taxon that now has a narrow geographic distribution, is considered a biogeographic relict (Lomolino et al., 2006; Habel and Assmann, 2010). This steno-endemic species is known only from two sites (a few hectares each) in the western Italian Alps. Because of this very restricted distribution, and an alarming population decline suffered in the years 1930-42 (Malausa et al., 1983), *C. olympiae* is classed as priority species (i.e. the European Community has a particular responsibility for the conservation of the species) and is listed in the Annexes II and IV of the Habitat Directive, and it is also protected by international conventions such as Bern and Washington (CITES). Older (Sturani, 1947) and more recent studies regarding habitat use and movement patterns (Negro et al., 2007, 2008) have been carried out in the type locality, i.e. the location where the type specimen was originally found. This site has been recently classified, following the Habitats (92/43/EEC) Directive, as a Site of Community Importance (S.C.I.), that is a site which, in the biogeographical region or regions to which it belongs, contributes significantly to the maintenance or restoration at a favourable conservation status of a natural habitat type in Annex I, or of a species in Annex II. Due to the restricted range, low population densities and the increase in human activity (mainly due to the development of a winter sports resort), conservation and management of this rare insect species is particularly difficult. Moreover, because previous research

and collections only focused on the type locality, nothing was known about the occurrence of this species in the rest of the S.C.I., which extends for about 10,787 ha.
In addition to *C. olympiae*, this S.C.I. also houses a species-rich ground beetle assemblage, with several species that are of conservation interest as they are endemic to a more-or-less restricted geographic area (Negro et al., 2007). The occurrence of species with restricted ranges is particularly evident in the Alps, where ancient lineages of carabids survived in Pleistocene refugia. About 800 ground beetle species are known in the Alps, 220 of which are endemic to the southern (Italian) side of the chain (Casale and Vigna Taglianti, 2005). Many of these carabid species are strongly linked to open habitats (Brandmayr et al., 2005). In the Alps, such habitats are threatened through the abandonment of traditional pastoral practices (Chauchard et al., 2007; Mottet et al., 2006), where an absence of grazing by domestic animals leads to shrub encroachment and eventual forestation, which has negative consequences for animal diversity (birds: Laiolo et al., 2004; grasshoppers and butterflies: Marini et al., 2009; bats: Obrist et al., 2011).
This study focuses upon the possible conflict between the conservation of the target species *C. olympiae* and the conservation of local syntopic carabid diversity, using Genetic Algorithm for Rule-set Prediction (GARP) (Stockwell and Noble, 1992) and a Maximum Entropy approach (MAXENT) (Phillips et al., 2006) to model *C. olympiae* distributions in the S.C.I. The study aimed to identify:

i) the habitats which deserved the highest priority for the conservation of the rare and endangered species *C.olympiae* (by habitat suitability modeling),
ii) indicator species, the presence of which indicate the occurrence of *C. olympiae* (by means of Indicator Power analysis) and
iii) the habitats which deserved the highest priority for the conservation of local carabid diversity (by considering abundance, species richness, taxonomic and functional diversity indices).
Finally, the need for the conservation of ground beetle diversity was contrasted with that for the conservation of the target (and indicator) species, and the appropriate management for the conservation of both was identified.

Material and methods

Study area

The study was carried out in the Sessera Valley (45°40´16″N; 8°05´07″E, Biella province, Piedmont, north western Italy) which, together with the contiguous top end of the Cervo Valley, has been recently classified as a Site of Community Importance (S.C.I., IT113002, Val Sessera).
The five most extensive habitats in the S.I.C. were identified: pasture (dominated by graminaceous plants), shrub (alpen rose *Rhododendron ferrugineum* L. and bilberry *Vaccinium myrtillus* L.), pioneer forest (dominated mainly by European white birch *Betulla alba* L. and common hazel *Corylus avellana* L.), beech (*Fagus sylvatica* L.) forest (namely, the *Luzulo-Fagetum* phytosociological association) and artificial plantations (dominated mainly by Norway spruce *Picea abies* and, in a few small patches, by European silver fir *Abies alba*). Selected pastures were mainly dominated by *Nardus stricta* and extended over north, north-east facing slopes, between 1200-1400 m a.s.l. They were not intensively grazed: during the study period, in particular, only 120 cows were present in the study area.

Sampling design

A stratified sampling design was used to select plots in all five main habitats, with a random plot selection in each habitat, and the number of plots being set in relation to the area occupied by that

habitat in the S.C.I. A total of 101 plots were selected, each set at a minimum of 200 m linear distance from the nearest neighboring plot.
In every sampling plot, five pitfall traps were arranged in a Latin square of total size 20 x 20 m, one for each vertex and the last at the point of intersection of the two diagonals of the square. Pitfall traps were placed at the beginning of June 2010 and emptied fortnightly until the end of September (five sampling periods). They were 7.5 cm in mouth diameter and 9 cm deep. Four small holes (0.2 cm in diameter) were drilled 2.5 cm below the upper brim of the pitfall, so that excess rainwater could flow out. Each trap was filled with 150 ml of a mixed fluid (vinegar and salts) to preserve individuals (van den Berghe, 1992). A flat stone was placed 3 cm above each trap to prevent rainwater from entering the traps. However, this sampling method may exert negative effects on the population size of rare and/or endangered species. Therefore, considering the high conservation relevance of *C. olympiae*, in the type locality, a capture-and-release approach, using double-bottom pitfall traps to keep animals alive (Negro et al., 2008) was used in the type locality, where its presence was known for certain. The exact location of pitfall traps was established in the field by means of a Global Positioning System (GPS) Garmin eTrex navigator.
Ground beetles were sorted and identified to species level using updated standard keys or specialist publications. Nomenclature follows Audisio and Vigna Taglianti (2004).
Habitat characteristics for each sampling plot, were recorded in the 20 x 20 m square area: percentage of grass, litter, soil, shrub and rock cover (estimated by eye).

Data analyses

1) Assessing priorities for biodiversity conservation

Species richness estimators
Species richness was estimated for each habitat by using the bias-corrected Chao1 (Chao, 1984, 2005) and the abundance-based richness estimator ACE (Chao and Lee, 1992).
Inventory completeness for each habitat was measured as the percentage of species observed from the total number of species predicted by the estimators.
This analyses was performed by using R software with the *vegan* package (Oksanen et al., 2005; R Development Core Team, 2010).

Diversity differences between habitats
To test for differences in diversity between habitats we calculated abundance (N), species richness (S), average taxonomic distinctness (Δ+), taxonomic distinctness (Δ*) and functional diversity (FD) for each trap (i.e. the single trap was the sampling unit).

The two taxonomic indices quantify diversity as relatedness of the species within a sample, attempting to capture phylogenetic diversity rather than simple richness of species. According to Clarke and Warwick (1998), taxonomic relatedness is more closely linked to functional diversity and, given their statistical properties (mainly robustness to sampling variation and independence of sampling effort) and their sensitivity to environmental degradation, they may represent useful parameters for bio-diagnostic purposes (Clarke and Warwick, 1999, 2001). Taxonomic distinctness (Δ*) is defined as a measure of pure taxonomic relatedness, being the average phylogenetic path length between any two randomly chosen individuals within a sample.
The algebraic definition of taxonomic distinctness is

$$\Delta^* = \left[\sum\sum_{i<j} \omega_{ij} x_i x_j\right] / \left[\sum\sum_{i<j} x_i x_j\right]$$

where ω are taxonomic distances among taxa *i* and *j*, and x are species abundances.
Average taxonomic distinctness (Δ+) represents the case of taxonomic distinctness (Δ*) when calculated on presence/absence data only (Clarke and Warwick, 1998).

$$\Delta^{+} = \left[\sum\sum_{i<j} \omega_{ij}\right] / \left[s(s-1)/2\right]$$

where ω are taxonomic distances among taxa *i* and *j* and *s* is the number of species present.

Functional diversity refers to those components of diversity that influence how an ecosystem operates or functions. Here we used the measure of functional diversity (FD) proposed by Patchey and Gaston (2002, 2006), that is the total branch length of the functional dendrogram that is often used in multivariate approaches to dividing species among functional groups.
Four traits were considered to describe differences in ecology, behaviour and morphology of ground beetles (Cole et al., 2002): size [very small (<5 mm); small (5-9 mm); medium (9-15 mm); large (>15 mm)], food of the adult (collembola specialists; generalist predators; mixed diet beetles; mostly vegetation feeders), breeding season (spring/summer; autumn/winter) and wing morphology (apterous or brachypterous; dimorphic; macropterous).This analysis requires a preliminary dendrogram of species traits as input. We used the UPGMA (Unweighted Pair Group with Arithmetic Mean) clustering method since species traits were factor variables (Podani and Schmera 2006).
Phylogenetic and traits classification of the full set of species involved follow Audisio and Vigna Taglianti (2004).
The two taxonomic indices (Δ+ and Δ*) and functional diversity (FD) were calculated by using R software with the *vegan* package (Oksanen et al., 2005; R Development Core Team, 2010).
Generalized linear mixed models (GLMMs) were used to test for differences in the dependent variable (N, S, Δ+, Δ* and FD) between habitats. Sampling period was set as a random effect (Bolker et al., 2009).
The best distribution to describe abundance and species richness was assumed to be the quasiPoisson with logarithmic link function (Agresti, 1996), while for taxonomic indices (Δ* and Δ+) and functional diversity (FD) the best distribution was assumed to be Gamma with an inverse link function (McIntyre and Lavorel, 1994; Zuur et al., 2009).
Significance tests were performed using the Wald statistic (Dobson, 1990). Moreover, the habitat factor estimates, their standard errors and individual statistical significance were explored. Pasture was used as the reference category. GLMMs were run using R software with the *MASS* package (Venables and Ripley, 2002; R Development Core Team, 2010).

Species turnover
Species turnover between habitats was analysed with a ternary plot (Koleff et al., 2003) by using R software with the *vegan* package (Oksanen et al., 2005; R Development Core Team, 2010). We considered the presence/absence of species in a focal habitat type compared with each neighbouring habitat in turn. The number of species that are present in both habitats is the pairwise matching component *a*. The number of species that are present only in the neighbouring habitat is *b*, while the number present only in the focal habitat is *c*. The total number of species for the pair of habitats is therefore $a + b + c$; in percentage terms the notation becomes $a' + b' + c' = 100\%$. The original equations for the measures of beta Whittaker diversity (Magurran, 1988; Whittaker, 1960) have also been re-expressed in terms of *a*, *b* and *c* (Krebs, 1999; Koleff et al., 2003). Beta Whittaker diversity was also calculated for each pair of habitats.

2) Assessing priorities for *C. olympiae* conservation

Univariate Regression Tree

The Univariate Regression Tree method (URT) was used to identify the interactions between environmental variables (in our case habitat type, sampling period, percentage of grass, litter, soil, shrub and rock cover) and the relative *C.olympiae* abundance (Ripley, 2009). This method performs a binary recursive partitioning of the dataset and offers the opportunity to identify the influential explanatory variables (Crawley, 2007; Gallé et al., 2011). URT was run using R software with *mvpart* package (R Development Core Team, 2010)

Habitat Suitability models

Habitat suitability modeling (HSM) is likely to provide the best tool for predicting the suitability of habitat for a given rare and endangered species based on known affinities with environmental parameters. GIS-based habitat suitability models are regularly used in wildlife management. However, these models are poorly applied to invertebrates, with the exception of macrobenthos (Degraer et al., 2008; Meissner et al., 2008; Li et al., 2009; Magris and Destro, 2010) and vectors of parasites (Kamdem et al., 2007; Laporta et al., 2011; Fisher et al., 2011). As for non-vector insects, a few studies have used models for the conservation of grasshoppers and crickets (Hein et al., 2007a,b; Altmoos et al., 2010) to increase forest habitats for several red-listed species without substantial losses in timber production (Tikkanen et al., 2007) and to predict the occurrence of ground-beetles in arable landscapes (Petit et al., 2003). This method may be usefully employed to study and protect target species and, in fact has been applied to the conservation of the rare ground beetle *Carabus variolosus* (Matern et al., 2007).

Several approaches have been used to approximate species' ecological niches (Elith et al., 2006). In this study, we used Genetic Algorithm for Rule-set Prediction (GARP) (Stockwell and Noble, 1992) and a Maximum Entropy approach (MAXENT) (Phillips et al. 2006) to model species distributions. Both methods use known occurrences and pseudo-absence data resampled from the set of pixels where the species in question is not known to occur.

We employed these algorithms in concert with openModeller Desktop software (available online, http://openmodeller.sourceforge.net/) and Geographic Information Systems (GIS) software package, ArcGIS 9.3 (ESRI, 2008) to provide an accurate assessment of the potential spatial distribution of the target species within the S.C.I.

GARP relates ecological characteristics of known species occurrence points to those of points that are randomly sampled from the remaining areas within the study region, developing a series of decision rules that best summarize those factors that are associated with the species' presence (Stockwell and Peters, 1999; Stockwell and Townsend Peterson, 2002). In openModeller Desktop software, we selected the algorithm GARP with Best Subsets (Anderson et al., 2003). This algorithm applies the Best Subsets procedure using the new openModeller implementation in each GARP run.

MAXENT focuses on fitting a probability distribution for occurrence of the species in question to the set of pixels across the study region, based on the idea that the best explanation to unknown phenomena will maximize the entropy of the probability distribution, subject to the appropriate constraints. In the case of modeling ecological niches of species, these constraints consist of the values of those pixels at which the species has been detected (Phillips et al., 2004, 2006). Both GARP and MAXENT, based on the ecological niche theory, require the presence data of a species and the environmental parameter layers with the same spatial resolution.

In the present study, models were based on 101 sampling points and four map layers representing parameters that, based on previous results (Negro et al., 2007, 2008), are known to influence the distribution of this steno-endemic species. The environmental variables were all obtained from

various online sources and included habitat type (pasture, shrub, pioneer forest, beech forest and artificial plantation), elevation, aspect and slope. All data layers were clipped in ArcGIS 9.3 using a mask of our study area. Occurrence points were evenly divided into training and test data sets, and they underwent an iterative process of rule selection, evaluation, testing, and incorporation or rejection. To evaluate GARP and MAXENT models, we plotted the receiver operating characteristic (ROC) curves, which differentiate model performance at all possible thresholds by a single number, the area under the curve (AUC). ROC curves plot sensitivity against (1 - specificity), where sensitivity is the proportion of observed presences correctly predicted and (1 - specificity) is the proportion of absences incorrectly predicted (Anderson et al., 2003). Further statistical assessments were derived from the Kappa statistic, which is often used as a measure of overall accuracy because it incorporates all of the information contained within the confusion matrix (Fielding and Bell, 1997). Landis & Koch (1977) have suggested the following ranges of agreement for the Kappa statistic: poor $K < 0.4$; good $0.4 < K < 0.75$ and excellent $K > 0.75$.

Indicator power

Another useful method to evaluate the presence of rare species is that of indirectly assessing the occurrence of the species of interest by defining a common species that could serve as an indicator of the likely occurrence of the target species. A reliable indicator species can therefore function as a tool that saves time and money. Indicator power (IP) analysis (Halme et al., 2009) was used to provide a list of indicator species, the presence of which indicate the occurrence of other species. The index of each indicator species is calculated with information about the frequency with which the indicator and the target species occur in the matrix of sampling points and the frequency of their co-occurrence.

The equation for the IP of an indicator species for the target species *C. olympiae* is:

$$IP_I = \sqrt{\left(\frac{S}{O_I}\right)\left(1 - \frac{O_{Co} - S}{N - O_I}\right)}$$

where O_I is the frequency of occurrence of the indicator species *I*, O_{Co} is the frequency of occurrence of the target species, S is the frequency of shared occurrences of the indicator species *I* and the target species, and N is the total number of sampling points surveyed.

The first part of the equation (S/O_I) is the proportion of shared occurrences (S) of the two species out of all the occurrences of the indicator species (O_I). This represents the strength of the positive prediction (presence-presense prediction). The second part of the equation, $[1 - (O_{Co} - S) / (N - O_I)]$, assesses how often *C.olympiae* occurs without the indicator species (absence-absence prediction) (Halme et al., 2009).

The statistical significance of the Indicator power (IP) for each indicator species in respect of the steno-endemic target species *C. olympiae* was evaluated by a randomization test (1,000 runs).

We carried out the indicator power analysis using R software with *vegan* package (Oksanen et al., 2005; R Development Core Team 2010).

Results

1) Assessing priorities for biodiversity conservation

Species richness estimator

A total of 46 species (13.494 individuals) of seven sub-families (Carabinae, Harpalinae, Lebiinae, Nebriinae, Platyninae, Pterostichinae and Trechinae) were collected (Table 1, Appendix 1) during the five sampling periods. We identified nine species that were dominant, i.e. with the number of sampled individuals > 1% of the total carabid beetles sampled. They were: *Pterostichus flavofemoratus* (36.4%), *Abax exaratus* (14.0%), *Pterostichus spinolae* (12.4%), *Carabus monticola* (9.7%), *Pterostichus apenninus* (9.5%), *Carabus depressus* (5.4%), *Cychrus italicus* (3.3%), *Tanythrix senilis* (2.1%) and *Carabus concolor* (1.9%). Most of these species were of conservation interest as they are endemic to fairly restricted geographic areas. *Tanythrix senilis*, in particular, is a steno-endemic species of the southern slopes of the Monta Rosa massif (as is *C. olympiae*), found only in seven sites (Negro et al., 2007).

Species richness varied across habitats, but all species richness estimators were consistently higher in pastures. As most of the expected species were caught at each habitat type (completeness ranging from 79 to 94% for Chao1 and from 71 to 87% for ACE), we assumed that the sampling effort was sufficient for correctly describing local assemblages of carabid beetles (Table 1).

Diversity differences between habitats

Abundance (N), species richness (S), average taxonomic distinctness (Δ+), taxonomic distinctness (Δ*) and functional diversity (FD) differed significantly between habitats. Habitat estimates showed that all parameters in pasture (set as a reference category) were significantly higher than those associated with shrub, pioneer forest, beech forest and artificial plantation (Fig.1, Fig.2, Table 2).

The correlation between the functional and taxonomical diversity, evaluated by means of Speraman correlation coefficient, was very high (FD vs. Δ+ = 0.88, $P<0.001$; FD vs. Δ* = 0.87, $P<0.001$).

Species turnover

The ternary plot showed that open and semi-open habitat, i.e. shrub-pioneer forest, pasture-shrub and pasture-pioneer forest pairs, shared a greater number of species (ranging from 60 to 67%) than the others pairs of habitats (Fig.3). This was confirmed by the lower species turnover (measured by beta Whittaker index, β_w) that was equal to 0.19, 0.24 and 0.25, respectively.

The minimum percentage of shared species (29%) occurred between pasture and artificial plantation ($\beta w = 0.54$).

In each comparison, c' percentage was always greater than b' due to the increased number of exclusive species in the first habitat type of the pair, which was always represented by an open or semi-open habitat, with the exception of the beech–coniferous forest pair.

2) Assessing priorities for *C.olympiae* conservation

Univariate Regression Tree

In the URT, built considering mean abundance of *C.olympiae*, the only dichotomy separated beech forest sampling points (N = 42) from sampling points set in the remaining other four habitat types (N = 59; pasture, shrub, pioneer forest and artificial plantation), hence suggesting a strong habitat selection for the areas covered by European Beech *Fagus sylvatica*.

Habitat suitability models

According to the ROC plot (Fig.4), the areas under the curves (AUC) for the GARP and MAXENT models were 0.90 and 0.78 respectively, suggesting the first algorithm was more predictive than the second. The Kappa values also suggested that the GARP model (K = 0.61) performed better than the MAXENT model (K = 0.50).

The potential distribution of *C.olympiae* in the S.C.I. (Fig.5), predicted by the GARP model, showed that the area with the greater average probability of presence (> 80%) covered 1854.13 ha (17.19% of the study area). This predicted distribution was strongly dominated by beech forests (1773.26 ha) followed by pioneer forest (46.32 ha), pasture (14.61 ha), shrub (10.46 ha) and artificial plantation (9.48 ha). Radar charts showed that areas with the highest value of habitat suitability had a mean slope of 28° (Fig.6b) and ranged in elevation from 800 to 1400 m a.s.l. (Fig.6a); they also showed that south, southeastern and southwestern facing slopes were almost completely avoided (Fig.6c).

Indicator Power

The measure of Indicator Power (IP) for the target species *C. olympiae* was calculated for each of the nine dominant species. Observed IP values ranged from 0.24 in *C. concolor* to 0.55 in *T. senilis*. For each indicator species (Table 3), the values of the first part of the equation (S/OI), that represents the strength of the positive prediction (co-presence), were lower than the second part [$1 - (O_T - S) / (N - O_I)$], that corresponds to negative prediction (co-absence).

The randomization procedure showed that only four species (*T. senilis*, *C. depressus*, *C. italicus* and *P. appenninus*) had a significant difference between observed IP and the mean IP value for the target species *C. olympiae* ($P<0.001$, $P<0.001$, $P<0.01$ and $P<0.05$, respectively, Table 3).

All these species, in common with *C. olympiae*, were abundant in beech forests (Appendix 1).

Discussion

Relict species with legal protection, such as *C. olympiae* (listed in the Annexes of the Habitat Directive), usually receive a greater conservation focus than co-occurring species without formal status. Restricted distributions place relict species under particular threat of extinction, especially in the face of ongoing climatic changes and other ecological perturbations (Habel and Assmann, 2010).

However, the conservation and proper management of the assemblage as a whole is also desirable because of the risk of losing local biodiversity when ecosystem management is focused on a single species. In fact, biological communities are threatened through a variety of activities that increase rates of species invasions and species extinctions at all scales, from local to global (Hooper et al. 2005). These changes have the potential to alter ecosystem properties and the goods and services they provide (Loreau et al., 2001; Hooper et al., 2005), known as the biodiversity ecosystem functioning (BEF) hypothesis. Within this framework, the conservation of ground beetles deserves high priority because they are of prime importance for ecosystem functioning. Most species are predators that play a key role in regulating populations of soil invertebrates and serving as prey for amphibians, birds and small mammals. Much research has previously focused on the impact of forest management practices (Vance and Nol, 2003; Brouat et al., 2004; Barton et al., 2009), whereas little has been devoted to the study of the effect of grassland management (Grandchamp et al., 2005; Cole et al., 2007).

Our results showed that the conservation of the relict steno-endemic species *C. olympiae* and that of syntopic ground beetle biodiversity could conflict. In the S.C.I., the best management option for the conservation of the greatest part of ground beetle diversity is that of maintaining unforested habitats such as pastures, which were characterized by significantly higher abundance, species richness, taxonomic diversity and functional diversity than all other habitat types. The conservation value of pastures may also be significant (albeit inferior than that of beech forests), given that certain species of restricted geographic areas, such as *Carabus concolor* and *Pterostichus flavofemoratus*, are typical of this habitat type.

Results from the species turnover analysis showed that open habitat types (pasture and shrub) housed a higher number of exclusive species. Some of these species were rather rare (just 1-2 individuals sampled) and this suggests that, in addition to the preservation of pastures, their management should also be considered to avoid the possible local extinction of exclusive species. Overgrazing, for instance, should be attentively avoided because the abundance of certain ground beetle species in overgrazed areas significantly decreases (Negro et al., 2007).

Taxonomic diversity and functional diversity were strongly and significantly correlated, in keeping with predictions by Clarke and Warwick (1988). Functional diversity, like all the other ecological parameters, was significantly higher in pastures than in beech forests, hence furtherly confirming the opportunity to preserve this open habitat type.

Both univariate regression tree and habitat suitability models showed that beech forest is the preferred habitat for the relict species *C. olympiae*. Habitat suitability analysis, in particular, showed the area with the greater average probability of presence was dominated by beech (96% of predicted suitable area) and that the species was predicted to prefer slopes of about 30°, at an elevation between 800 and 1400 m which were not south-facing. Moreover, the presence of *C. olympiae* could be indirectly assessed by the occurrence of four indicator species. These indicator species were also of conservation relevance because they are endemic to more-or-less restricted zones in the Alps (*T. senilis*, *C. depressuss*) or endemic to Italy (*C. italicus*, *P. appenninus*). For each indicator species, the strength of the positive prediction (co-presence) was lower than that corresponding to the negative prediction (co-absence), suggesting that areas where all the indicator species are absent may be reliably considered as areas avoided by the target species *C. olympiae* as well.

Artificial plantations may be surrogates of native forest and may also contribute to target species conservation as suggested by two studies (Brockerhoff, 2005; Berndt et al., 2008) where exotic pine

forests play a positive role in the conservation of critically endangered ground beetle species. This is not the case in the Sessera Valley, whose artificial plantations, although partly colonized by beech and other deciduous trees, were little used by the target species *C. olympiae* (only two individuals sampled) and by most of the indicator species as well.

In the Sessera Valley, no significant environmental management interventions are foreseen in the next few years. However, this valley, like many others in the Alps, is experiencing profound habitat changes mainly due to natural reforestation following pastoral abandonment. After the abandonment of pasture, vegetation succession develops to shrub, pioneer forest and local climax forest, therefore causing the disappearance of open habitats (Tasser and Tappeiner, 2002; Chemini and Rizzoli, 2003; Tasser et al., 2007). Several studies have confirmed the importance of traditional pastoral practices in preserving open areas and maintaining animal diversity (birds: Laiolo et al., 2004; grasshoppers and butterflies: Marini et al., 2009; bats: Obrist et al., 2011). A previous study carried out in the same area demonstrated that most local dung beetle (Coleoptera: Scarabaeoidea) species positively selected pastures, whereas a few other species preferred beech forests, and concluded that to preserve local dung beetle assemblages both habitats should be maintained (Tocco et al., 2012). Conclusions of the present study are quite similar. A management tradeoff between the need for conserving the relict steno-endemic species *C. olympiae* and that of preserving wider biodiversity and ecosystem functionality, is the only possible option. Reforestation processes should therefore be controlled by favouring interventions which support local traditional pastoral activities. The retention of traditional pastoral grazing may not be the only option to retaining/restoring open habitat in the Alps. The choice of mowing technique, although more expensive than pastoral activities, may be important for conserving open habitats and carabid beetle assemblages. However, the use of cutter bar mowers (finger or double blade) are recommended over rotary and flail mowers, because they cause around half as much mortality of invertebrates (Humbert et al., 2008). Moreover, because no practicable harvesting processes are damage free, leaving uncut grass strips is a simple practice that will benefit many ground-welling arthropods (Humbert et al., 2012).

Several studies have shown that clear-cutting or single-tree selection cutting exert significant short and medium term negative impacts on forest carabid species (Vance and Nol, 2003; Brouat et al., 2004). In particular, forest management causes a loss of large-bodied species, and substantial reductions in activity densities, the latter effect being correlated with significant reductions in leaf litter (Vance and Nol, 2003). The Sessera Valley beech forests, at the moment, are not heavily managed.

This study has taken a habitat scale approach, even though the potential distribution of *C.olympiae* in the S.C.I. may also represent an example of a landscape approach. There is an increasing awareness that biodiversity needs to be managed at the landscape scale, which necessarily includes conserving representative examples of all habitat types. Thus the conflict might actually be narrow *vs* more holistic approaches to conservation and landscape management.

All the above considered, our conclusion is that the most promising landscape scale approach to preserve both the relict steno-endemic species *C. olympiae* and the high local, syntopic ground beetle diversity is to retain traditional low-intensity pastoral activities along with the preservation of the unmanaged beech forests.

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	Pasture	Shrub	Pioneer forest	Beech forest	Artificial plantation
Sampling points	15	10	26	42	8
Total Number	3151	1850	2977	4887	629
Specie richness	34	32	30	20	14
Chao1	43.00	35.5	31.67	23.33	15.5
ACE	47.71	37.79	34.42	25.42	17.33
Completeness - Chao1 (%)	79.07	90.14	94.74	85.71	90.32
Completeness - ACE (%)	71.26	84.67	87.15	78.67	80.77

Table 1: Number of sampling points, total number of individuals collected, observed species richness, Chao1 and ACE estimate of expected richness for each habitat. Inventory completeness was measured as the percentage of species observed from the total number of species predicted by the estimators.

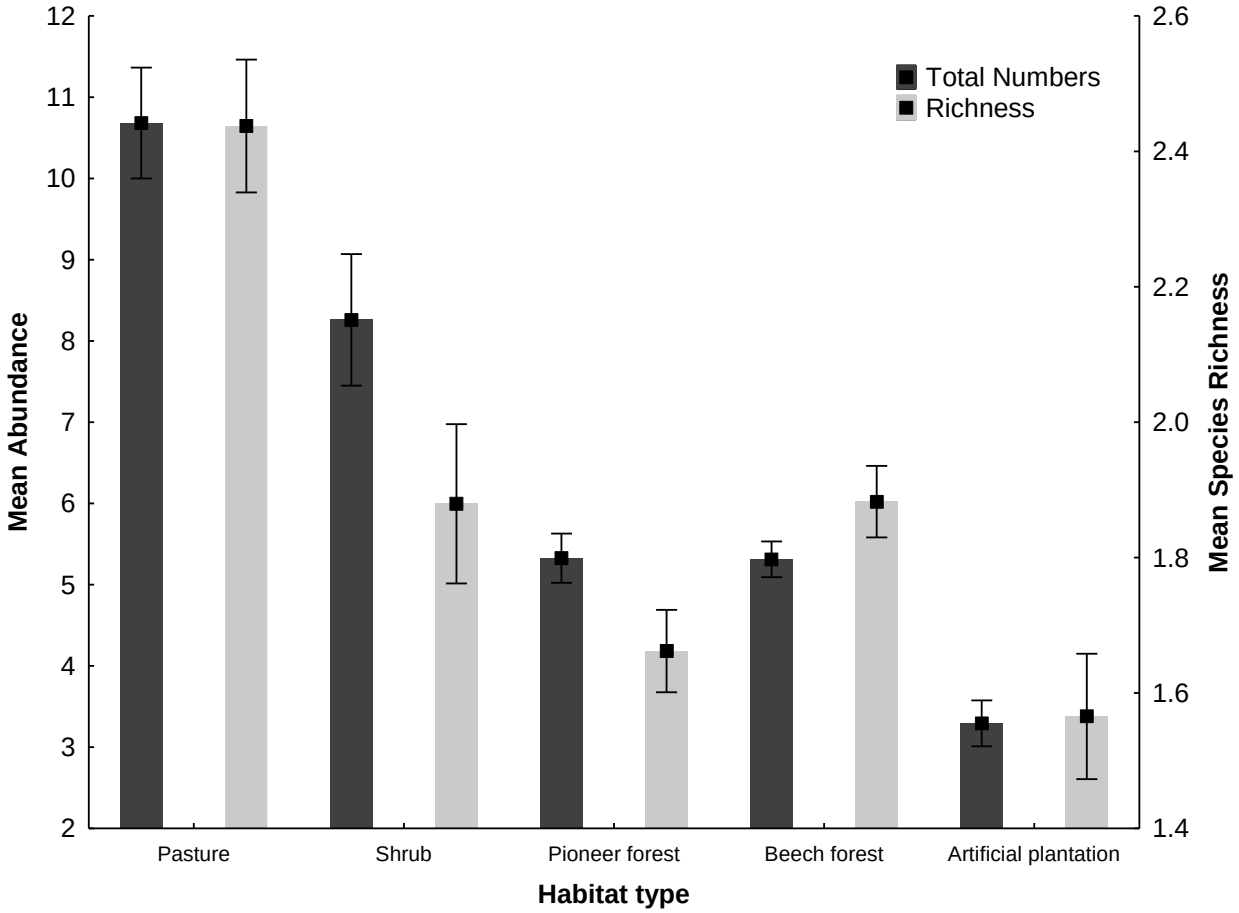

Fig.1: Mean species richness and abundance of carabid beetle per pitfall trap in the different habitats. Bars are ± standard errors.

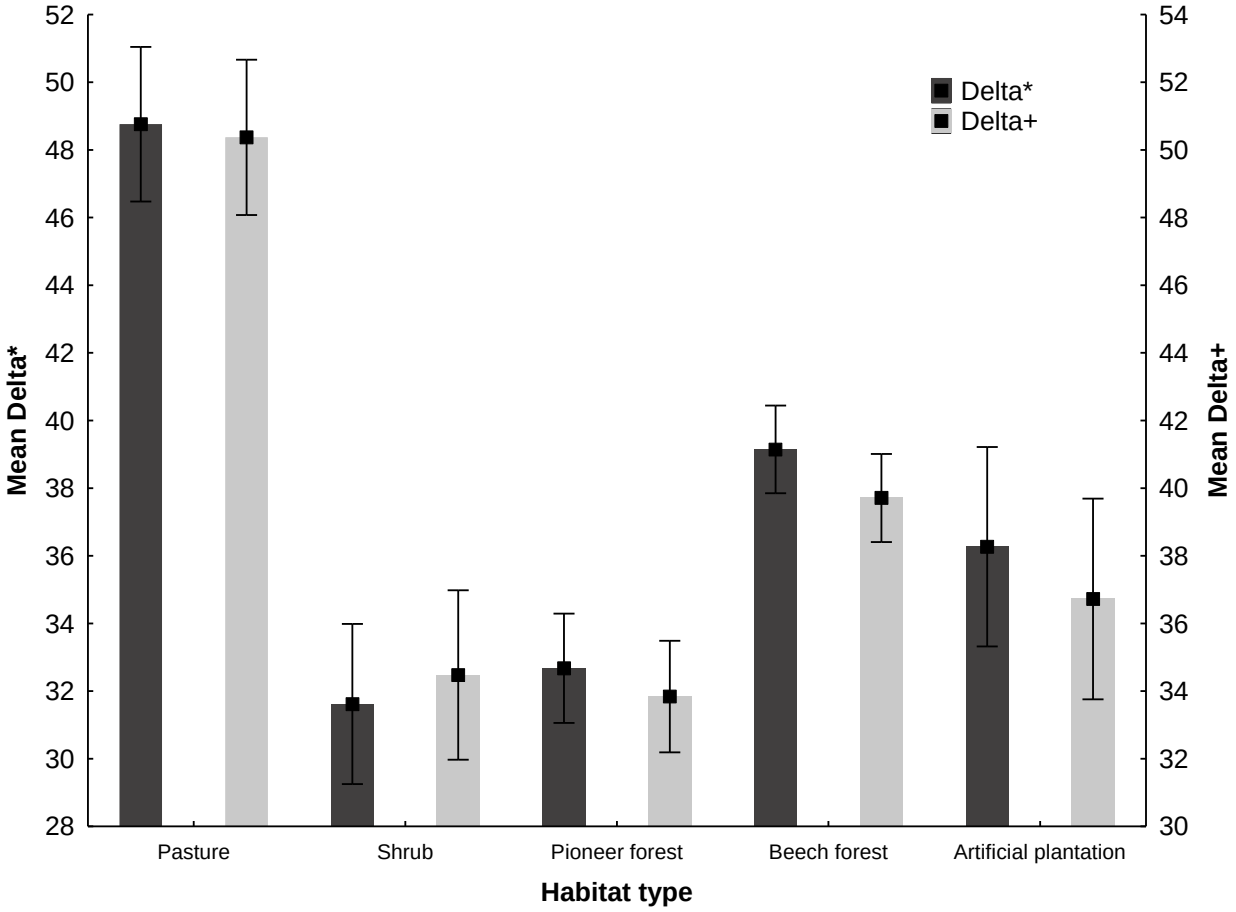

Fig.2: Mean taxonomic distinctness (Δ*) and average taxonomic distinctness (Δ+) of carabid beetles per pitfall trap in pasture, shrub, pioneer forest, beech forest and coniferous forest. Bars are ± standard errors.

	N			S			Δ*			Δ+			FD		
	Distribution: quasiPOISSON			Distribution: quasiPOISSON			Distribution: GAMMA			Distribution: GAMMA			Distribution: GAMMA		
	Link function: Log			Link function: Log			Link function: Inverse			Link function: Inverse			Link function: Inverse		
Habitat type	Est.	S.E.	*p*	Est.	S.E.	*p*	Est.	S.E.	*p*	Est.	S.E.	*p*	Est.	S.E.	*p*
Artificial plantation	-1.159	0.130	**<0.001**	-0.430	0.075	**<0.001**	0.006	0.002	**<0.01**	0.007	0.002	**<0.01**	0.008	0.002	**<0.001**
Beech forest	-0.686	0.068	**<0.001**	-0.247	0.048	**<0.001**	0.004	0.001	**<0.01**	0.005	0.002	**<0.01**	0.005	0.001	**<0.001**
Pioneer forest	-0.694	0.076	**<0.001**	-0.382	0.054	**<0.001**	0.009	0.002	**<0.001**	0.009	0.002	**<0.001**	0.007	0.001	**<0.001**
Shrub	-0.265	0.087	**<0.01**	-0.265	0.067	**<0.001**	0.011	0.002	**<0.001**	0.009	0.002	**<0.001**	0.004	0.001	**<0.001**

Table 2: Habitat estimates and statistical significance (GLMMs) for abundance (N), species richness (S), taxonomic distinctness (Δ*), average taxonomic distinctness (Δ+) and functional diversity (FD) parameters. In this analysis pasture is set as reference category. Therefore, in the GLMMs run using R software, the parameter value for the pasture is set to zero. Significant comparisons are in bold type.

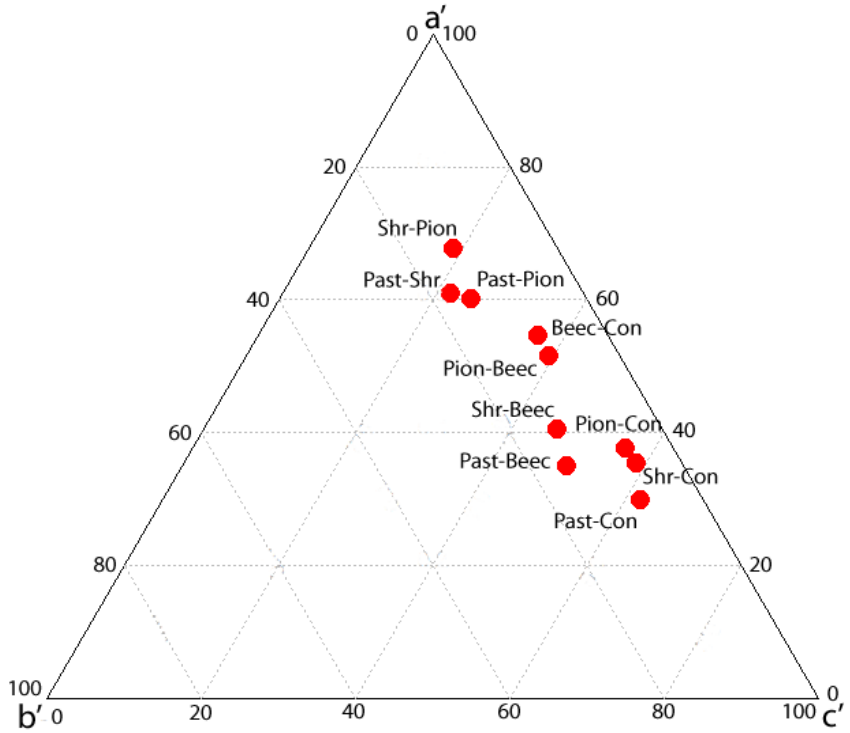

Fig.3: Ternary plot representing species turnover between pairs of habitats. Numbers on sides are percentages. The percentage of species that are present in both habitats is the pairwise matching component *a'*. The percentage of species that are present only in the neighbouring habitat is *b'*, while the number present only in the focal habitat is *c'*. Past: Pasture; Shr: Shrub; Pion: Pioner forest; Beec: Beech forest; Con: Coniferous forest.

	observed values						randomization procedure				
Indicator species	**O_i**	**Oco**	**S**	**Pres**	**Abs**	**IP**	**mean IP**	**min IP**	**max IP**	**S.D.**	***P***
Tanythrix senilis	57	20	18	0.32	0.95	0.55	0.40	0.21	0.53	0.05	**<0.001**
Carabus depressus	63	20	19	0.30	0.97	0.54	0.40	0.27	0.50	0.04	**<0.001**
Cychrus italicus	73	20	19	0.26	0.96	0.50	0.40	0.27	0.52	0.04	**<0.01**
Pterostichus apenninus	80	20	19	0.24	0.95	0.48	0.40	0.26	0.50	0.04	**<0.05**
Pterostichus spinolae	58	20	15	0.26	0.88	0.48	0.40	0.24	0.57	0.05	0.13
Abax exaratus	91	20	19	0.21	0.90	0.43	0.40	0.25	0.47	0.04	0.74
Pterostichus flavofemoratus	76	20	13	0.17	0.72	0.35	0.40	0.26	0.51	0.04	0.34
Carabus monticola	68	20	11	0.16	0.73	0.34	0.40	0.25	0.52	0.04	0.28
Carabus concolor	14	20	1	0.07	0.78	0.24	0.39	0.00	0.70	0.12	0.36

Table 3: The nine potential indicator species with the highest indicator power for the target species *C.olympiae*. O_i: occurrences of indicator species; O_{co}: occurrences of target species; S: shared occurrences of the indicator species and the target species; Pres: value of presence-presence prediction for the target species; Abs: value of absence-absence prediction for the target species; IP: indicator power of the indicator species for the target species; mean IP, min IP, max IP: mean, minimum and maximum indicator power of the indicator species for the target species in the randomization test (1,000 runs); S.D.: standard deviation; *P*: significance of the IP value.

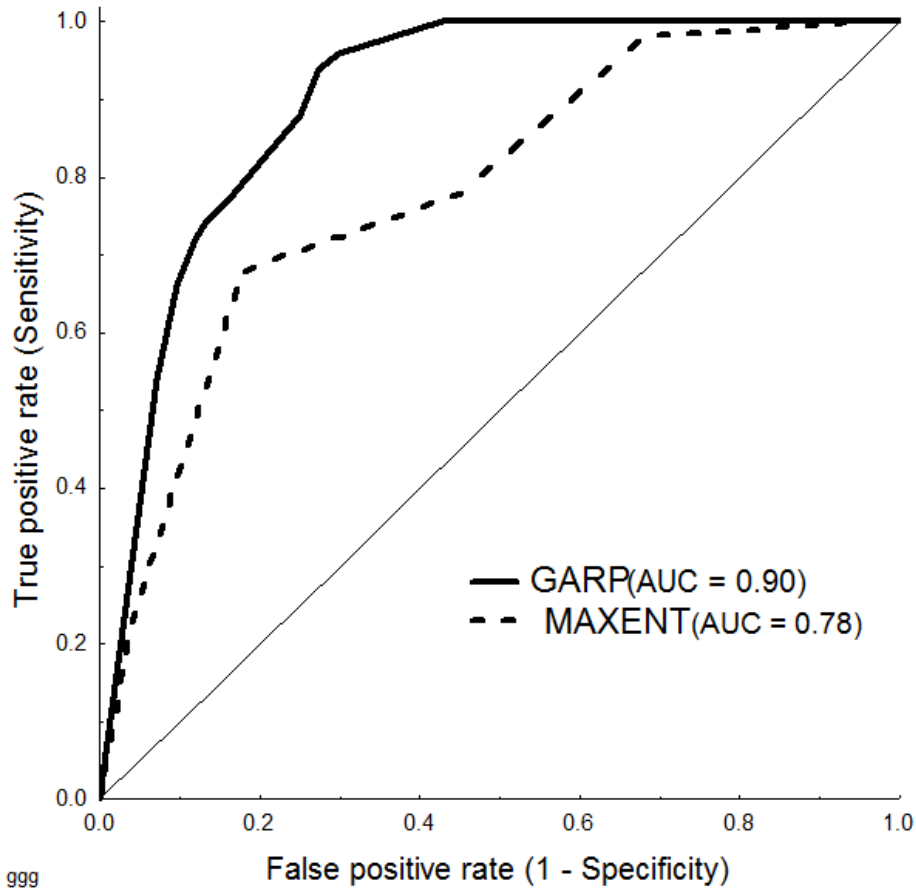

Fig.4: ROC curves for the habitat suitability models (GARP and MAXENT). For each model, the ROC curve was generated by comparing the model predictions against observed presence/pseudo-absence data. True and false positive rates were used as measures of model performance.
The straight line under the curve represents the frequencies of positive and negative results generated by a random model. The more the model curve departs from the straight line (i.e. the larger the AUC), the higher is the average model performance.

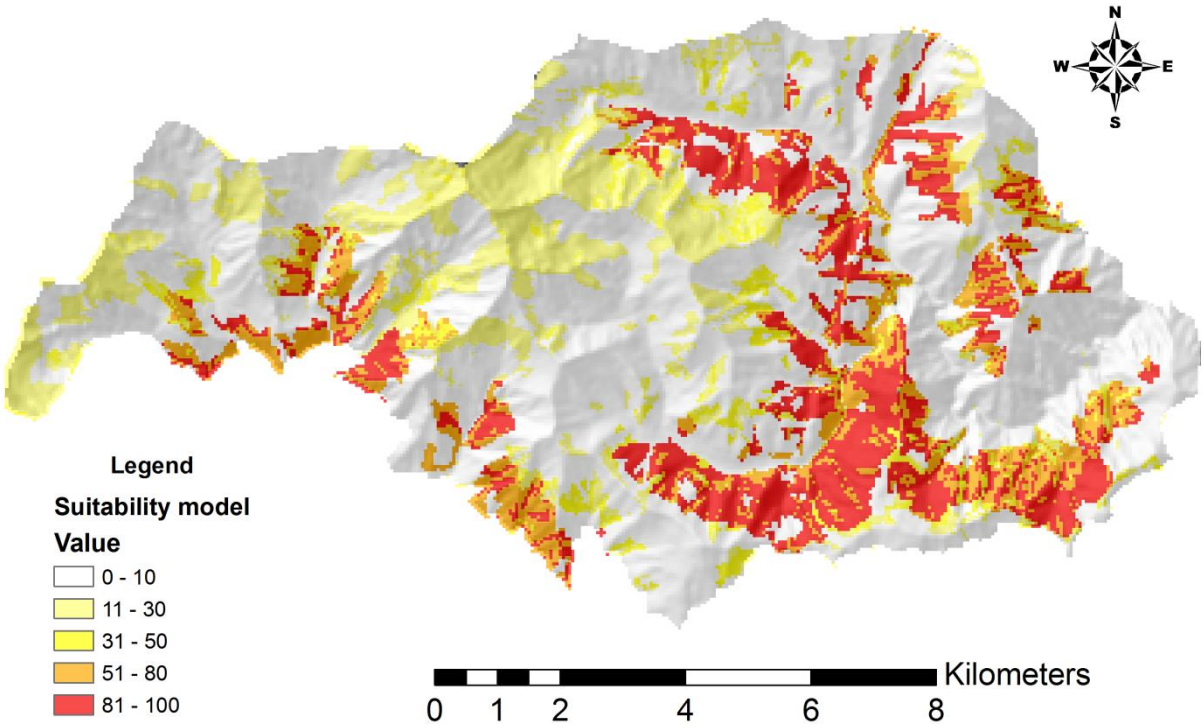

Fig.5: A suitability map, based on GARP model, for the steno-endemic species *Carabus olympiae*. Warmer colors represent increasing probability levels of presence.

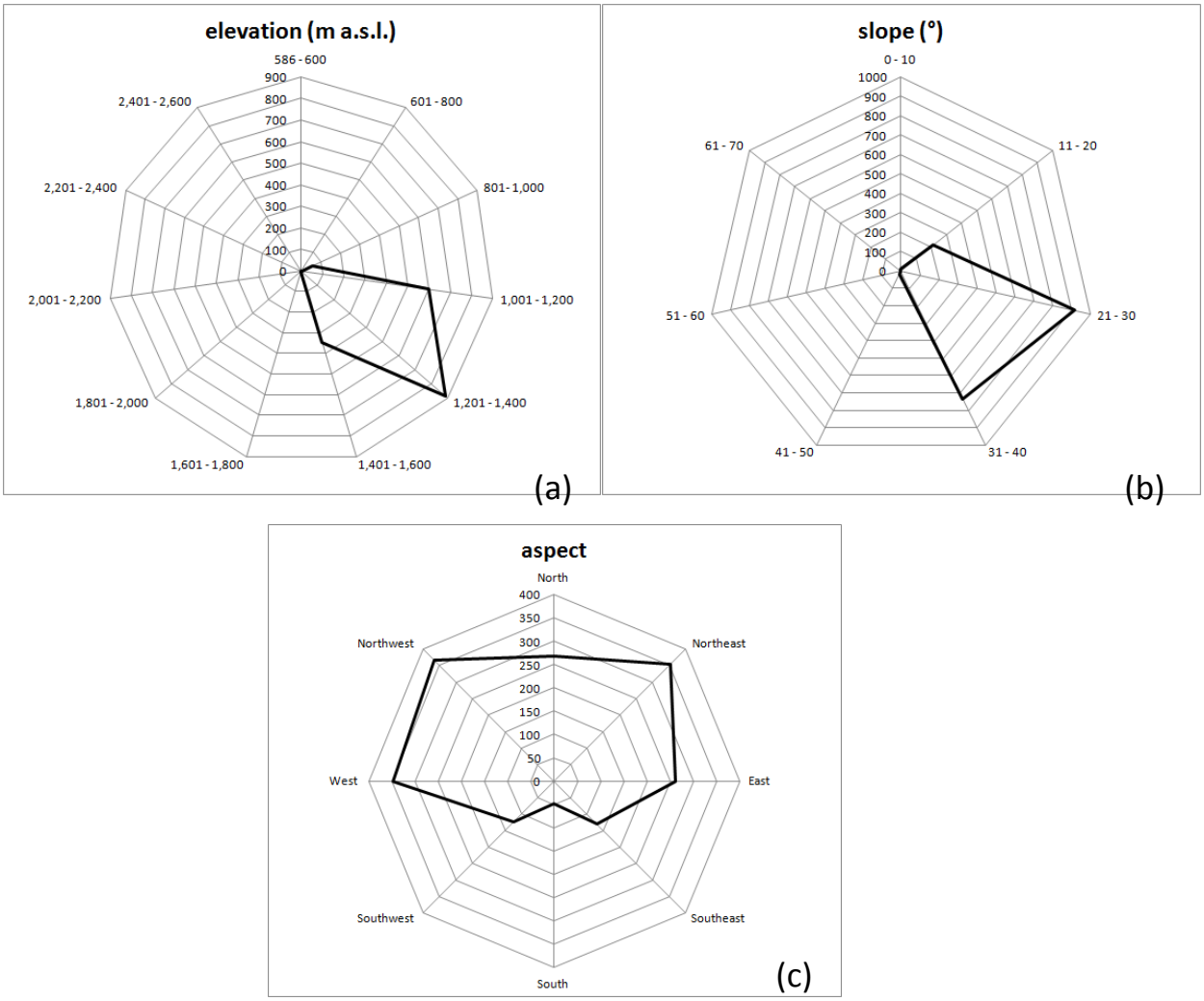

Fig.6: Radar chart of the elevation (a), slope (b) and aspect (c) for the area with a greater average probability of presence (> 80%). The charts show the relative surface (expressed in hectares) for each class.

Habitat	Pasture	Shrub	Pioneer forest	Beech forest	Artificial plantation
Carabinae					
Carabus olympiae Sella, 1855	0	1	2	67	2
Calosoma sycophanta (Linné, 1758)	0	0	0	1	0
Carabus cancellatus Illiger, 1798	2	0	1	0	0
Carabus concolor Fabricius, 1792	219	34	3	0	0
Carabus depressus Bonelli, 1810	35	9	57	495	135
Carabus germarii Sturm, 1815	37	50	38	1	0
Carabus glabratus Paykull, 1790	26	0	10	0	0
Carabus monticola Dejean, 1826	125	66	477	626	14
Cychrus angustatus Hoppe & Hornschuch, 1825	0	0	0	0	0
Cychrus caraboides (Linné, 1758)	4	6	10	0	0
Cychrus cordicollis Chaudoir, 1835	1	1	1	0	0
Cychrus italicus Bonelli, 1810	5	3	92	300	45
Harpalinae					
Harpalus affinis (Schrank, 1781)	1	0	0	0	0
Harpalus marginellus Dejean, 1829	20	7	0	0	0
Ophonus azureus (Fabricius, 1775)	0	1	0	0	0
Pseudophonus rufipes (De Geer, 1774)	32	9	0	1	0
Trichotichnus rimanus Schauberger, 1936	0	0	23	2	0
Lebiinae					
Cymindis cingulata Dejean, 1825	1	2	4	1	1
Cymindis scapularis Schaum, 1857	1	0	0	0	0
Syntomus truncatellus (Linné, 1761)	0	1	0	0	0
Nebriinae					
Leistus nitidus (Duftschmid, 1812)	1	5	2	0	0
Notiophilus aquaticus (Linné, 1758)	0	3	8	4	0
Notiophilus biguttatus (Fabricius, 1779)	0	0	1	5	3
Platyninae					
Calathus melanocephalus (Linné, 1758)	42	9	2	0	0
Laemostenus janthinus (Duftschmid, 1812)	9	1	1	0	0
Platynus complanatus Dejean, 1828	2	0	0	16	3
Sphodropsis ghilianii (Schaum, 1858)	0	0	2	2	0
Synuchus vivalis (Illiger, 1798)	12	4	3	1	0
Pterostichinae					
Abax exaratus (Dejean, 1828)	223	129	473	947	120
Amara convexior Stephens, 1828	1	2	1	0	0
Amara curta Dejean, 1828	4	3	0	0	0
Amara aenea (De Geer, 1774)	6	0	0	0	0
Amara equestris (Duftschmid, 1812)	13	2	4	0	1
Amara lucida (Duftschmid, 1812)	1	0	0	0	0
Amara lunicollis Schiödte, 1837	23	7	3	0	0
Amara nitida Sturm, 1825	1	0	0	0	0
Poecilus versicolor (Sturm, 1824)	51	22	39	0	0
Pterostichus apenninus (Dejean, 1831)	72	36	212	815	147
Pterostichus cribratus (Dejean, 1828)	0	2	2	0	0
Pterostichus flavofemoratus (Dejean, 1828)	1801	1030	1062	881	135
Pterostichus rutilans (Dejean, 1828)	0	1	0	0	0
Pterostichus spinolae (Dejean, 1828)	335	340	411	575	7
Stomis roccae Schatzmayr, 1925	1	2	5	5	0
Tanythrix senilis (Schaum, 1859)	42	61	28	142	15
Trechinae					
Metallina lampros (Herbst, 1784)	2	0	0	0	0
Trechus lepontinus Ganglbauer, 1891	0	1	0	0	1
Sampling points	15	10	26	42	8

Appendix 1: Number of individuals of carabid beetle species collected in the five habitats. In every sampling plot, five pitfall traps were placed at the beginning of June 2010 and emptied fortnightly until the end of September (five sampling periods).

References

Agresti, A., 1996. An introduction to categorical data analysis. Wiley.

Anderson, R. P., Lew, D., Peterson, A. T., 2003. Evaluating predictive models of species' distributions: criteria for selecting optimal models. Ecol. Model. 162 (3), 211-232.

Audisio, P., Vigna Taglianti A., 2004. Fauna Europaea: Coleoptera, Carabidae. Fauna Europaea version 1.1, http://www.faunaeur.org

Altmoos, M., Henle K., 2010. Relevance of multiple spatial scales in habitat models: a case study with amphibians and grasshoppers. Acta Oecol. 36(6), 548-560.

Berger, J., 1997. Population constraints associated with the use of black rhinos as an umbrella species for desert herbivores. Conserv. Biol. 11, 69-78.

Barton, P.S., Manning, A.D., Gibb, H., Lindenmayer, D.B., Cunningham, S.A., 2009. Conserving ground-dwelling beetles in an endangered woodland community: multi-scale habitat effects on assemblage diversity. Biol. Conserv. 142(8), 1701-1709.

Berndt, L.A., Brockerhoff, E.G., Jactel, H., 2008. Relevance of exotic pine plantations as a surrogate habitat for ground beetles (Carabidae) where native forest is rare. Biodiver. Conserv. 17(5), 1171-1185.

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H., White, J.S., 2009. Generalized linear mixed models: a practical guide for ecology and evolution. Trends Ecol. Evol. 24(3), 127–135.

Brandmayr, P., Zetto, T., Pizzolotto, R., Casale, A., Vigna Taglianti, A. 2005. I Coleotteri Carabidi per la valutazione ambientale e la conservazione della biodiversita`. Manuale operativo, APAT, Roma, p 34

Brockerhoff, E.G., Berndt, L.A., Jactel, H., 2005. Role of exotic pine forests in the conservation of the critically endangered New Zealand ground beetle *Holcaspis brevicula* (Coleoptera : Carabidae). New Zeal. J. Ecol. 29(1), 37-43.

Brouat, C., Meusnier, S., Rasplus, J.Y., 2004. Impact of forest management practices on carabids in European fir forests. Forestry 77(2), 85-97.

Casale, A., Vigna Taglianti, A., 2005. Coleotteri Caraboidei delle Alpi e Prealpi centrali e orientali, e loro significato biogeografico (Coleoptera, Caraboidea). Biogeographia 24, 129-201.

Chao, A.1984, Nonparametric estimation of the number of classes in a population. Scand. J. Stat. 11, 265-270.

Chao, A., 2005. Species estimation and applications, In: Balakrishnan, N., Read, C.B.,Vidakovic, B. (Eds.), Encyclopedia of Statistical Sciences12, second Edition., Wiley, New York, pp.7907-7916.

Chao, A., Lee, S.M., 1992. Estimating the Number of Classes via Sample Coverage. J. Am. Stat. Assoc. **87**(417), 210-217.

Chauchard, S., Carcaillet, C., Guibal, F., 2007. Patterns of land-use abandonment control tree-recruitment and forest dynamics in Mediterranean mountains. Ecosystems 10**,** 936–948.

Chemini, C., Rizzoli A., 2003. Land use change and biodiversity conservation in the Alps. J. Mt. Ecol. 7, 1-7.

Clarke, K.R., Warwick, R.M., 1998. A taxonomic distinctness index and its statistical properties. J. Appl. Ecol. 35(4), 523-531.

Clarke, K.R., Warwick, R.M., 1999. The taxonomic distinctness measure of biodiversity: weighting of step lengths between hierarchical levels. Mar. Ecol. – Prog. Ser. 184, 21-29.

Clarke, K.R., Warwick, R.M. 2001**.** A further biodiversity index applicable to species lists: variation in taxonomic distinctness. Mar. Ecol. – Prog. Ser. 216**:** 265–278.

Cole, L.J., McCracken, D.I., Dennis, P., Downie, I.S., Griffin, A.L., Foster, G.N., Murphy, K.J., Waterhouse, T., 2002. Relationships between agricultural management and ecological groups of ground beetles (Coleoptera: Carabidae) on Scottish farmland. Agric. Ecosyst. Environ. 93, 323–336.

Cole, L.J., McCracken D.I., Baker, L., Parish, D. 2007. Grassland conservation headlands: their impact on invertebrate assemblages in intensively managed grasslands. Agr. Ecosyst. Environ. 122: 252-258.

Crawley, M. J., 2007. The R Book. John Wiley, New York pp: 627–685.

Degraer, S., Verfaillie, E.W., Adriaens, W.E., Vincx, M., Van Lancker, V., 2008. Habitat suitability modelling as a mapping tool for macrobenthic communities: an example from the Belgian part of the North Sea. Cont. Shelf Res. 28(3), 369-379.

Dobson A.J., 1990. An introduction to generalized linear models. Chapman & Hall, New York.

Elith, Jane; Graham, Catherine H.; Anderson, Robert P.; Dudik, Miroslav; Ferrier, Simon; Guisan, Antoine; Hijmans, Robert J.; Huettmann, Falk; Leathwick, John R.; Lehmann, Anthony; Li, Jin; Lohmann, Lucia G.; Loiselle, Bette A.; Lohmann, G.; Manion, Glenn; Moritz, Craig; Overton, Jacob McC.; Peterson, A. Townsend; Phillips, Steven J.; Nakamura, Miguel; Nakazawa, Yoshinori; Schapire, Robert E.; Wisz, Mary S.; Zimmermann, Niklaus E., 2006. Novel methods improve prediction of species' distributions from occurrence data. Ecography 29(2), 129-151.

ESRI, 2008. ArcGIS Desktop: Release 9.3. (Environmental Systems Research Institute, Redlands, CA.).

Fielding, A.H., Bell, J.F., 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environ. Conserv. 24, 38-49.

Fisher, D., Thomas, S.M., Niemitz, F., Reineking, B., Beierkuhnlein, C., 2011. Projection of climatic suitability for *Aedes albopictus* Skuse (Culicidae) in Europe under climate change conditions. Global Planet. Change 78(1-2), 54-64.

Gallé, R., Vesztergom, N., Somogyi, T., 2011. Environmental conditions affecting spiders in grasslands at the lower reach of the River Tisza in Hungary. Entomol. Fennica 22, 29-38.

Grandchamp, A.C., Bergamini, A., Stofer, S., Niemelä, J., Duelli, P., Scheidegger, C., 2005. The influence of grassland management on ground beetles (Carabidae, Coleoptera) in Swiss montane meadows. Agr. Ecosyst. Environ. 110(3-4), 307-317.

Habel, J.C., Assmann T., 2010. Relict Species. Phylogeography and Conservation Biology. Springer, 449 pp.

Halme, P., Mönkkönen, M., Kotiaho, J.S., Ylisirniö A.L., Markkanen, A., 2009. Quantifying the Indicator Power of an Indicator Species. Conserv. Biol. 23(4), 1008-1016.

Hein, S., Binzenhoefer, B., Poethke, H.-J., Schröder, B., Settele, J., Biedermann, R., 2007a. The generality of habitat suitability models: a practical test with two insect groups. Basic Appl. Ecol. 8(4), 310-320.

Hein, S., Voss, J., Poethke, H.J., Boris, S., 2007b. Habitat suitability models for the conservation of thermophilic grasshoppers and bush crickets - simple or complex? J. Insect Conserv. 11(3), 221-240.

Hooper, D.U., Chapin, F.S. III, Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecol. Monogr. 75, 3–35.

Humbert, J.Y, Ghasoul, J., Walter, T. 2008. Meadow harvesting techniques and their impacts on field fauna. Agric. Ecosyst. Environ. 130: 1-8.

Humbert, J.-Y., Ghazoul, J., Richner, N., Walter, T. 2012. Uncut grass refuges mitigate the impact of mechanical meadow harvesting on orthopterans. Biol. Conserv. 152, 96–101.

Kamdem, C.D., Etouna J., Ose, K., Ayala, D., Besansky, N., Fontenille, D., Costantini, C., Simard, F., 2007. Spatial modelling of habitat suitability, distribution, and relative abundance of species and molecular forms of the *Anopheles gambiae* complex in cameroon. Am. J. Trop. Med. Hyg. 77(5), 296-297.

Koleff, P., Gaston, K.J., Lennon, J.J., 2003. Measuring Beta Diversity for Presence-Absence Data. J. Anim. Ecol. 72(3), 367-382.

Krebs, C.J., 1999. Ecological Methodology, 2nd ed., Benjamin Cummings, Menlo Park, California, pp. 620.

Laiolo, P., Dondero, F., Ciliento, E., Rolando, A., 2004. Consequences of pastoral abandonment for the structure and diversity of the alpine avifauna. J. Appl. Ecol. 41(2), 294–304.

Lambeck, R.J., 1997. Focal species: a multi-species umbrella for nature conservation. Conserv. Biol. 11(4), 849-856.

Landis, J.R., Koch, G.G., 1977. The measurement of observer agreement for categorical data. Biometrics 33(1), 159–174.

Laporta, G.Z., Ramos, D.G., Ribeiro, M.C., Sallum, M.A.M., 2011. Habitat suitability of Anopheles vector species and association with human malaria in the Atlantic Forest in south-eastern brazil. Mem. I. Oswaldo Cruz 106, 239-249.

Launer, A.E, Murphy, D.D., 1994. Umbrella species and the conservation of habitat fragments: a case of a threatened butterfly and a vanishing grassland ecosystem. Biol. Conserv. 69(2), 145–153.

Li, F., Cai, Q., Fu, X.C., Liu, J.K., 2009. Construction of habitat suitability models (HSMs) for benthic macroinvertebrate and their applications to instream environmental flows: a case study in Xiangxi River of Three Gorges Reservior region, China. Progr. Nat. Sci. 19(3), 359-367.

Lichtenberg, F.R., 1994. Testing the Convergence Hypothesis. This Rev. 76, 576-579.

Lomolino, M.V., Sax, D.F., Riddle, B.R., Brown, J.H., 2006. The island rule and a research agenda for studying ecogeographical patterns. J. Biogeogr. 33, 1503–1510.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J.P., Hector, A., Hooper, D.U., Huston, M.A., Raffaelli, D., Schmid, B., Tilman, D., Wardle, D.A. 2001. Biodiversity and ecosystem functioning: Current knowledge and future challenges. Science. 294, 804–808.

Magris, R.A., Destro G.F.G., 2010. Predictive modeling of suitable habitats for threatened marine invertebrates and implications for conservation assessment in Brazil. Braz. J. Oceanogr. 58, 57-68.

Magurran, A.E., 1988. Ecological diversity and its measurement. Princeton University Press,. New Jersey, pp. 177.

Malausa, J.-C., Raviglione, M., Boggio, F., 1983. Il *Carabus olympiae* Sella dell'Alta Valle Sessera. Pro Natura Biellese, Biella (in coll. con O.P.I.E. Francia) 113 pp.

Marini, L., Fontana, P., Battisti, A., Gaston, K.J., 2009. Agricultural management, vegetation traits and landscape drive orthopteran and butterfly diversity in a grassland–forest mosaic: a multi-scale approach. Insect Conserv. Diver. 2(3), 213–220.

Matern, A., Drees, C., Kleinwächter, M., 2007. Habitat modelling for the conservation of the rare ground beetle species *Carabus variolosus* (Cololeoptera, Carabidae) in the riparian zones of headwaters. Biol. Conserv. 136(4), 618-627.

McIntyre, S., S. Lavorel., 1994. Predicting richness of native, rare and exotic plants in response to habitat and disturbance variables across a variegated landscape. Conserv. Biol. 8, 521-531.

Meissner, K., Darr, A., Rachor, E., 2008. Development of habitat models for Nephtys species (Polychaeta: Nephtyidae) in the German Bight (North Sea). J. Sea Res. 60(4), 271-286.

Mottet, A., Ladet, S., Coque, N., Gibon, A., 2006. Agricultural land-use change and its drivers in mountain landscapes: a case study in the Pyrenees. Agric. Ecosyst. Environ. 114, 296–310.

Negro, M., Casale, A., Migliore, L., Palestrini, C., Rolando, A., 2007. The effect of local anthropogenic habitat heterogeneity on assemblages of carabids (Coleoptera, Caraboidea) endemic to the Alps. Biodivers. Conserv. 16**,** 3919-2932.

Negro, M., Casale, A., Migliore, L., Palestrini, C., Rolando, A., 2008. Habitat use and movement patterns in the endangered ground beetle species, *Carabus olympiae* (Coleoptera: Carabidae). Eur. J. Entomol. 105, 105-112.

Obrist, M.K., Rathey, E., Christe, P., Bontadina, F.,Martinoli, A., Conedera, M., Moretti, M., 2011. Response of bat species to sylvo-pastoral abandonment. Forest Ecol. Manag. 261, 789-798.

Oksanen, J., Kindt, R., O'Hara, R.B., 2005. Vegan: community ecology, package version 1.6-10. http://cc.oulu.fi/~jarioksa/.

Petchey, O.L., Gaston, K.J., 2002. Functional diversity (FD), species richness and community composition. Ecol. Lett. 5, 402–411.

Petchey, O.L., Gaston, K.J., 2006. Functional diversity: back to basics and looking forward. Ecol. Lett. 9, 741–758.

Petit, S., Haysom, K., Pywell, R., Warman, L., Allen, D., Booth, R., Firbank, L., 2003. Habitat-based models for predicting the occurrence of ground-beetles in arable landscapes: two alternative approaches. Agr. Ecosyst. Environ. 95(1), 19-28.

Phillips, S.J.; Dudik, M., Schapire, R.E., 2004. A Maximum Entropy Approach to Species Distribution Modeling. Appearing in Proceedings of the Twenty-First International Conference on Machine Learning, pp. 655-662.

Phillips, S.J., Anderson, R.P., Schapire, R.E., 2006. Maximum entropy modeling of species geographic distributions. Ecol. Model. 190, 231-259.

Podani J., Schmera, D., 2006. On dendrogram-based methods of functional diversity. Oikos 115, 179–185.

R Development Core Team, 2010. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/

Ripley, B., 2009. Tree: Classification and regression trees. R package version 1.0-27.—[www document] URL http://CRAN.R-project.org/package=tree.

Samways, M.J., 2005. Insect Diversity Conservation. Cambridge University Press, Cambridge, 342 pp.

Severns, P.M., Moldenke A.R., 2010. Management tradeoffs between focal species and biodiversity: endemic plant conservation and solitary bee extinction. Biodivers. Conserv. 19(12), 3605-3609.

Spray, S.L., McGlothlin, K.L., Anderson, D. 2004. Loss Of Biodiversity (Exploring Environmental Challenges: A Multidisciplinary Approach). Rowman & Littlefield Publishers, Lanham 176 pp.

Stockwell, D.R.B., Noble I.R., 1992. Introduction of sets of rules for animal distribution data: a robust and informative method of data analysis. Math. Comp. Simul. 33, 385-390.

Stockwell, D., Peters, D., 1999. The GARP modelling system: problems and solutions to automated spatial prediction. Int. J. Geogr. Inf. Sci. 13(2), 143-158.

Stockwell, D.R.B.; Townsend Peterson, A.T., 2002. Effects of sample size on accuracy of species distribution models. Ecol. Model. 148, 1-13.

Sturani M., 1947. Notizie ecologiche ed etologiche sul *Carabus olympiae* Sella (Coleoptera: Carabidae). Boll. Ist. Entomol. Univ., Bologna 16, 23–84.

Tasser, E., Tappeiner, U., 2002. Impact of land use changes on mountain vegetation. Appl. Veg. Sci. 5(2), 173–184.

Tasser, E., Walde, J., Tappeiner, U., Teutsch, A., Noggler, W., 2007. Land-use changes and natural reforestation in the Eastern Central Alps. Agr. Ecosyst. Environ. 118(1–4), 115–129.

Tilman, D., Knops, J., Wedin, D., Reich, P., Ritchie, M., Siemann, E., 1997. The influence of functional diversity and composition on ecosystem processes. Science 277, 1300–1302.

Tikkanen, O.-P., Heinonen, T., Kouki, J., Matero, J., 2007. Habitat suitability models of saproxylic red-listed boreal forest species in long-term matrix management: cost-effective measures for multi-species conservation. Biol. Conserv. 140(3-4), 359-372.

Tocco, C., Negro, M., Rolando, A., Palestrini, C., 2012. Does natural reforestation represent a potential threat to dung beetle diversity in the Alps? J. Insect Conserv. *In press*

van den Berghe E.P., 1992. On pitfall trapping invertebrates. Entomol. News 103, 149-158.

Vance, C.C., Nol E., 2003. Temporal effects of selection logging on ground beetle communities in northern hardwood forests of eastern Canada. Ecoscience 10(1), 49-56.

Venables, W.N., Ripley, B.D., 2002. Modern Applied Statistics with S. Fourth Edition. Springer, New York.

Whittaker, R.M., 1960. Vegetation of the Siskiyou Mountains, Oregon and California. Ecol. Monogr. 30, 279-338.

Zuur, A.F., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. Mixed effects models and extensions in ecology with R. Springer, pp. 574.