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Use of insect distribution across landscape-soil units to assess conservation priorities in a Mediterranean coastal reserve: the tenebrionid beetles of Castelporziano (Central Italy)[1]

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Abstract

We investigated the conservation concern of landscape-soil units within Castelporziano lowland forest (a natural reserve of 6000 hectares facing the Tyrrhenian sea) using tenebrionid species vulnerability as defined by the Kattan index, which is based on species rarity, and IUCN categories. Species rarity was evaluated according to various measures of geographical distribution, habitat specialization and population size on a regional level. Measures of species vulnerability were combined into two indexes of conservation concern for each landscape-soil unit: (1) the Biodiversity Conservation Concern index, BCC, which reflects the average rarity score of the species present in a site, and (2) the Biodiversity Conservation Weight, BCW, which reflects the sum of rarity scores of the same species assemblage. Because the same species was subject to multiple evaluations under different criteria, we obtained various series of BCC and BCW values.

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BCC and BCW provided complementary information and the use of different scoring schemes of species vulnerability produced correlated, but not identical conservation ranking in landscape-soil unit prioritization. Forests, wetlands and beach-dunes were identified as high priority units, whereas dry pastures, traditional crop mosaics, pinewoods and maquis/garrigue shrublands, were identified as medium-low priority units. Tenebrionids were proved to be particularly useful as a bioindicator group in coastal areas because they include both sand-dwelling and saproxylic species, being distributed, with distinct communities, in all coastal biotopes.

Key words: Coleoptera Tenebrionidae; Species vulnerability; Species extinction risk; Area prioritization; Coastal environments

1 Introduction

Coastal environments were among the first to be exploited by Mediterranean peoples. As a consequence, large natural areas along the Mediterranean coasts are virtually absent, and most of coastal landscape can be considered the result of long-lasting management by more than 300 generations (Blondel and Aronson 1999). In particular, continued grazing and fire were important forces in converting the original coastal forests into maquis vegetation. Thus, large patches of sclerophyllous-broadleaf (*Quercus ilex* L.) forest, which is considered the climacic vegetation of Mediterranean coastal ecosystems, are now extremely rare and sparse. An approximation of closed-canopy forest exists locally in some areas where *Quercus ilex* of large tree size is associated with aged *Phillyrea latifolia* L. and well-developed *Pistacia lentiscus* L. These communities are obviously of conservation value, subject to their continued exclusion from cyclic burning.

The wealth of herbaceous perennial plants which follows in the wake of cyclic burning is soon replaced by scrubland formations, generally indicated as maquis. The term maquis usually refers to the first stage of degradation (which is known in Italy as "macchia"), followed by a scrubland with lower height and complexity, usually indicated as garrigue. Thus, in the best preserved sites, the coastal scenario of the Mediterranean region shows a great variety of environments and vegetation types that form a more or less well defined transect from the shoreline inland to dune communities, garrigues, low maquis, high maquis and sclerophyllous-broadleaf forests. Although sclerophyllous forests have long been recognized as of extreme conservation value because of their reduction to small and isolated patches, the importance of preserving sandy shores, dunes and other fragile soils has been recently come into sharp focus. Most Mediterranean dunes have been largely transformed into spoiled flat beaches, tourist settlements and places for second homes, while coastal erosion is now one of the most important environmental threats in the Mediterranean. Erosion data showed that 1500 km of artificial coasts can be found in the EU marine areas (European Environment Agency 1999) and more than 70% of natural dunes have been destroyed (Géhu, 1985; Salman and Strating 1992). In addition to the loss of sclerophyllous-broadleaf forests by grazing and fire and to the destruction of dunes, a further factor of landscape alteration in Mediterranean coastal areas is represented by land reclamation. This is particularly true for land reclamation of former wetlands that hosted ponds, lagoons and lowland forests, now reduced to sparse and highly isolated fragments.

The importance of conserving well preserved coastal areas is intuitive, and their identification has been mainly driven by considerations about the extent and fragmentation level of target biotopes and their possible threats. However, because of the complex landscape mosaic that characterizes Mediterranean areas, it is important to have clear indication about which biotopes, within a certain area, have higher conservation values. In particular, because of the high fragmentation of natural sites along the coasts, it is important to identify the biotopes that may best contribute to preserve species that are most endangered from both biophysical (e.g. climate change, soil degradation, pollution) and socioeconomic (e.g. urban sprawl, land-use changes, tourism) pressures. For this purpose, biotopes should be prioritized according to the conservation status of the species they host. However, this approach is generally biased by the fact that only few target species, typically represented by the vascular plants and charismatic vertebrates, are used for biotope prioritization, whereas a more solid approach should take into account the contribution of entire and representative plant and/or animal communities. Of course, in most cases it is impossible to have extensive species lists, but many arthropod groups are easy to sample and are ecologically so varied to be used as ecological indicator (McGeoch 1998).

The Natural Reserve of Castelporziano (Rome, Central Italy), one of the largest and best preserved coastal forest in Italy, offers the unique opportunity of exploring the contribution that a selected group of arthropods, the tenebrionid beetles (Coleoptera Tenebrionidae), can offer to the identification of biotope priorities in a coastal eco-mosaic including beaches, dunes, low maquis, high maquis, natural forests, wet areas, planted pinewoods and traditional crops. Tenebrionid beetles are an important arthropod group in the Mediterranean basin because they are taxonomically diversified and abundant in a wide array of biotopes (Fattorini 2008). Mediterranean tenebrionids include both soil dwelling species frequenting xeric and dry landscapes and mesophilous species mostly associated with well preserved forest stands (Fattorini 2008). Tenebrionids have been recently demonstrated to be a reliable bio-indicator of increasing climate aridity (Fattorini and Salvati 2014) and may also represent a suitable focus group for an integrated coastal management aimed at preserving landscape, soil and biological diversity. Mediterranean tenebrionids show various degrees of ecological tolerance, including both eurytopic species and stenotopic species, strictly associated with particular types of soil and/or vegetation. Thus, they form distinct communities along an ideal transect from the beach-dune system (where many sand dwelling, highly specialized xero-thermophilous species live) to the broadleaf forests (where several saproxylic species occur). In the present study, we also took advantage of data collected during a long term project of tenebrionid inventorying and monitoring at regional level, in order to rank the biotopes of Castelporziano according to the conservation value of the tenebrionid species they harbour.

2. Materials and methods

2.1 Study area

The Natural Reserve of Castelporziano is located in Central Italy, between Rome and the Tyrrhenian cost. With a protected area of some 6000 hectares, this reserve is one of the largest and best preserved coastal areas in Italy. In particular, this reserve hosts large relicts of the mesic and damp forests that, up to the beginning of 1900, dominated the Tyrrhenian coasts and that have been destroyed by land reclamation. At present, Castelporziano is a mosaic of natural habitats, planted forests and agro-ecosystems (see Fattorini and Maltzeff (2001) for a detailed description of the study area). Della Rocca et al. (2001) recognized and mapped 15 vegetation units occurring in the study area (including non vegetated areas) producing a detailed land cover map at 1:10,000 resolution scale. However, since some units occupied a very small fraction of the study area and are based on subtle differences in vegetation structure that are not relevant for tenebrionids, at least on the basis of current information on their ecology, we adopted the following simplified nomenclature (in parenthesis, the original units provided by Della Rocca et al. 2001): macchia and garrigues (garrigues), high macchia shrubland (*Viburno - Quercetum ilicis* vegetation), wetlands (hygrophilous woodland, wet grasslands - wet pasture lands, *Rubo – Ulmion* vegetation, ponds), forests (*Echinopo – Quercetum, Viburno - Quercetum ilicis* with high trees, *Viburno - Quercetum ilicis ericetum, Viburno - Quercetum ilicis suberetosum*), pinewoods (reforestations), pasture (dry grasslands - dry pasture lands characterized by the presence of *Dasypyrum villosum* and *Vulpia ligustica*) and crop (cultural vegetation: fields cultivated with lucerne (*Medicago sativa*) following the procedures of rotation and rest). This nomenclature system was aimed at creating more homogeneous landscape-soil units (see also Salvati et al. 2012), that integrate vegetation information with ancillary information provided by a digital soil map of the study area, and that reflect tenebrionid habitat preference. Because of their importance for sand-dwelling tenebrionids, beach and dunes were added as further landscape-soil unit. Figure 1a illustrates the spatial distribution of landscape-soil categories used in this study.

2.2 Species distribution across landscape units

Primary data were obtained from Fattorini and Maltzeff (2001), where details on sampling methods and information on species distribution and ecology in the study area are provided. Data included materials collected by one of us (PM) from 1994 to 2000, specimens collected in the study area by other people and preserved in public and private collections, as well as literature records. After the publication of the abovementioned study, PM performed additional sampling, which allowed the collection of new and more detailed information on the distribution of tenebrionid beetles in the study area. These additional surveys also allowed the collection of some species cited in Fattorini and Maltzeff (2001) on the basis of literature records or materials collected by other people, such as *Phaleria* spp., *Halammobia pellucida*, *Pseudoseriscius normandi*, *Xanthomus* spp., *Ammobius rufus*, *Trachyscelis aphodioides*, *Corticeus unicolor*. Moreover, *Bolitophagus reticulatu*s, previously found only using light traps, has been collected in polypores in wet woodlands. With respect to the species list provided in Fattorini & Maltzeff (2001), the current list includes an additional species, *Scaphidema metallica*, which was previously overlooked. The presence of this species in the study area has been cited by Rossi and Cesari Rossi (1977) and confirmed by personal findings (Capocotta, 1.II.2000, 5 es., 15.VI.2001, 1 es., P. Maltzeff leg., in coll. Maltzeff).

Species distribution across landscape-soil units in the study area is given in Table 1. In this table, we indicated with the letter P that the target species has been found in a given unit (ascertained presence), and with the symbol * that the target species probably occupies a given landscape unit although it has not been found there (putative presence). This codification allowed us to obtain two matrices of species distribution with different confidence about putative presences: (1) a binary (1/0) matrix of species presence/absence where putative presences were considered as ascertained presences, and (2) a "probabilistic" matrix where ascertained presences were coded as 1, absences as 0, and putative presences as 0.5.

2.3 Species conservation ranking

Species conservation concern was expressed as species vulnerability using the Kattan index (Kattan 1992), which is based on species rarity, and IUCN Red List categories. We used Kattan values already available for the tenebrionids of the study area as given in Fattorini and Di Giulio (2013), Fattorini et al. (2013) and Fattorini (2014). Therefore, we resume here only the basic steps used to calculate the Kattan index of each species, because the primary data and a complete description of both the rationale and the adopted protocols can be found in Fattorini (2013).

First, species rarity was assessed using a multidimensional characterization that takes into account (1) geographical distribution (wide/narrow distribution), (2) habitat specificity (broad/restricted habitat specificity) and (3) abundance (abundant/scarce population). Such a multidimensional characterization of species rarity has been previously applied to bryophytes (Gabriel et al. 2011), vertebrates (e.g. Kattan 1992; Manne and Pimm 2001; Isaac et al. 2009) and arthropods (e.g. Fattorini 2008, 2010a, 2010b, 2014; Fattorini et al. 2012).

Rarity scores were calculated using a database including data for 25,349 specimens directly examined by one of us (SF), plus literature data for 1,394 specimens. Data originated from museum and private collections, publications and unpublished species lists and spanned from 1860 to 2011 (see Fattorini et al. 2013 for details). Sample sites were geo-referenced (latitude and longitude decimal degrees) with the maximum precision allowed by the original record. Then, each point record was assigned to a 10 × 10 km grid cell using the Universal Transverse Mercator (UTM) geographical coordinate system.

The very large sampling effort made over more than a century by hundreds of collectors interested in different insect groups and who used any kind of collecting method (hand searching, pitfall traps, aerial traps, soil examination, etc.) should reduce the impact of possible collector preferences for certain biotopes, sites or species. Also, tenebrionids are frequently collected by amateur entomologists interested in beetles or insects in general, or as a by-product of generalized collecting activities performed by entomologists mostly interested in other groups. Thus, it is unlikely that amateur entomologists not directly interested in tenebrionids under-collected common species and over-collected rare species.

Species geographical rarity was measured using two alternative ways. A first measure (hereafter called 'UTM') was given by the number of 10 × 10 km UTM cells occupied by a species. UTM cells have the advantage of being equally sized and are commonly used to measure insect rarity (see, for example, papers presented in Reemer et al. 2003). However, a given UTM cell can include areas with very different environmental conditions (namely altitude and land-use). Thus, as an alternative way of measuring geographical rarity, the number of communes from which a species is known (COM) was also used. Communes are small administrative units, of varying size but typically smaller than 10 km^2, whose boundaries typically reflect major topographic changes, and which might encompass an environmentally relatively uniform area.

Ecological rarity is the most difficult aspect of rarity to be evaluated because of both the lack of data on species ecology and multiple ways of evaluating species preferences. Thus, ecological rarity was expressed using five different measures: (1) species distribution (presence/absence) across phytoclimatic units (PHY) as defined in Blasi (1994); (2) distribution of species abundances across phytoclimatic units using the Shannon index (HPHY); (3) species distribution along the elevation gradient (ELE), divided into 100 m step belts (0-100 m, 101-200 m, 201-300 m, etc); (4) distribution of species abundances across elevation belts using the Shannon diversity index (HELE), (5) and a combined measure (HTOT) obtained by summing HPHY and HELE (see Hanski 1978). Finally, the number of specimens collected for each species was considered a measure of local rarity (ABU), assuming contactability as a proxy for population size (cf. Strayer 1999). These rarity measures were then combined to obtain an overall index of species vulnerability following Kattan's (1992) method. For this, species were initially dichotomized into two groups (common and rare) according to whether they were above or below the median of each type of rarity measure (cf. Arita et al. 1990; Isaac et al. 2009). Species with 'wide geographical distribution - broad habitat specificity - high abundance' were classified as 'common' in three dimensions (vulnerability index: 1), while those with 'narrow distribution - restricted habitat specificity - low abundance' were classified as 'rare' in three dimensions (vulnerability index: 8). To rank the other six combinations of rarity measures, species with narrow geographical distribution were considered more vulnerable on a regional scale, and species with restricted habitat specificity were considered more vulnerable regardless of their abundance (see Kattan 1992 for details). Thus, species received a vulnerability score as follows: 1: species that are not rare for any dimension (measure); 2: species rare only for abundance; 3: species rare only for habitat; 4: species rare only for range; 5: species rare for both habitat breadth and abundance; 6: species rare for both geographical range and abundance; 7: species rare for both habitat breadth and geographical distribution; 8: species rare for geographical distribution, habitat breadth and abundance. Because of the various ways of measuring each kind of rarity, for each species the following ten Kattan indexes were calculated with the following combinations of measures: K1: UTM, PHY, ABU; K2: UTM, HPHY, ABU; K3: UTM, ELE, ABU; K4: UTM, HELE, ABU; K5: UTM, TOT, ABU; K6: COM, PHY, ABU; K7: COM, HPHY, ABU; K8: COM, ELE, ABU; K9: COM, HELE, ABU; and K10: COM, TOT, ABU (Table 2). For *Tenebrio obscurus*, an anthropophilous species not considered in Fattorini (2014), but found in the study area as a native species, a Kattan value of 1 was used.

IUCN red list categories were assigned using IUCN criteria at the regional level (IUCN 2003). Here, we referred to the Italian territory as the region of interest. Species were classified into IUCN categories using Criterion B2 (estimated area of occupancy) and at least two of the following sub-criteria: (a) severely fragmented or few locations, (b) continuing decline, (c) extreme fluctuations. Area of occupancy was evaluated by overlapping known species occurrence data (from literature and unpublished data) on a 10 × 10 km grid and counting the number of occupied cells. Criteria met by threatened species are given in Table 3. All other species resulted to be LC (last concern).

2.4 Landscape-soil unit ranking

Landscape-soil units were ranked on the basis of the vulnerability of their tenebrionid communities. For this, two different measures of prioritization, the Biodiversity Conservation Concern (BCC) index (Fattorini 2006), and the Biodiversity Conservation Weight index (Fattorini et al. 2012), were used. In the BCC index, species occurring in a given area unit are classified into categories of endangerment and weighted by the respective vulnerability. The BCC index also combines the vulnerability of each species with total richness to obtain a measure of relative conservation. The BCC can be calculated as:

$$\text{BCC} = \frac{\sum_{i=1}^{L}(\alpha_i - \alpha_{\min})}{L(\alpha_{\max} - \alpha_{\min})} \qquad (1)$$

where L is the local species richness, α_i is the weight assigned to the ith category of vulnerability, α_{min} is the minimum weight among all species; and α_{max} is the maximum weight among all species. This formulation ensures the index ranges from 0 (all species belonging to the lower conservation category) to 1 (all species belonging to the highest endangerment category). The BCC index has been previously applied to identify priority areas or biotopes for butterflies in Mediterranean islands and European countries (Fattorini 2006, 2009; Dapporto and Dennis 2008), fish in France (Bergerot et al. 2008; Laffaille et al. 2011), tenebrionids, butterflies, birds and mammals in the Central Apennines (Fattorini 2010a, 2010b), arthropods in Azorean forest fragments (Fattorini et al. 2012), and tenebrionids in urban biotopes (Fattorini, 2014).

The BCC index is a ‘relative measure’, which means that it is not sensitive to species richness. This may be an advantage to compare species assemblages with different species richness (Fattorini 2006, 2010b), but poses some problems. For example, an assemblage with a single species, having this species α_{max}, would receive the same score as an assemblage with 10 species, all with α_{max}. Or worse, an assemblage with a single species with α_{max} would receive a higher score than an assemblage with 10 species, 9 with α_{max} and one with $\alpha_i < \alpha_{max}$. The BCW index was thus introduced to overcome these problems, but it is dependent on species richness. The BCW can be calculated as:

$$\text{BCW} = \frac{\sum_{i=1}^{L}(\alpha_i - \alpha_{\min})}{\sum_{i=1}^{S}(\alpha_i - \alpha_{\min})} \qquad (2)$$

where S is the total species richness for all sites (other symbols are as in the BCC). In general, an absolute index like the BCW might be preferable to prioritize the areas with the highest numbers of vulnerable species, but a relative index like the BCC may help the identification of areas with few, but highly imperiled species. Thus, the BCC and the BCW should be used jointly to provide a more comprehensive overview of conservation priorities (Fattorini et al. 2012).

BCC and BCW indexes were calculated using both species vulnerability expressed by Kattan index and species extinction risk expressed by IUCN categories. In particular, for each spatial unit two sets of BCC values were calculated using Kattan index: (1) a set of ten values obtained using the respective Kattan index scores (K1 to K10) for the matrix of species presence/absence, and (2) a set of ten values obtained using the respective Kattan scores for the matrix of occurrence probabilities. In both sets α values correspond to Kattan scores, ranging from 1 (α_{min}) to 8 (α_{max}). For each unit four BCC values were also calculated using the IUCN categories. Because of the presence of NT species, two alternative ranking schemes were adopted. In one scheme, we considered NT species as if they were LC species and adopted the scale: LC=NT=1, VU=2, EN=4. In another scheme, we assigned to the NT species an intermediate score, thus obtaining the scale: LC=1, NT=1.5, VU=2, EN=4. Because both scales were used for the presence/absence matrix and the matrix of occurrence probabilities, four different measures were obtained. Analogous calculations were done for BCW.

2.5 Statistical analysis

To highlight the conservation values of the various landscape-soil units in Castelporziano, we illustrated separately BCC and BCW results. For each unit, we obtained a series of BCC and BCW because of the multiplicity of the way each species was ranked. To test correlation between values of BCC (or BCW) obtained using the various Kattan and IUCN scores we used Spearman non-parametric rank inference. Because the use of multiple ways of calculating BCC and BCW index led to multiple series of values, for each landscape-soil unit we calculated average values for BCC and BCW indexes based on the various Kattan and IUCN scores and then illustrated the spatial distribution of both indexes through maps derived from the abovementioned land cover map (see 'Study area' section). On the basis of their statistical distribution, three classes of Kattan index values were identified to provide a simplified and easy-to-interpret graphical representation: Low (< 0.1), Medium (between 0.1 and 0.2) and High (> 0.2). A Principal Components Analysis was run to provide a comprehensive analysis of the conservation value of the landscape-soil units considered in the present study using both the BCC and BCW indexes calculated for each unit. Components with eigenvalues > 1 were selected. To illustrate the most relevant results of the analysis, plots of loadings and scores were used for, respectively, landscape-soil classes and conservation indexes.

3 Results

Pair-wise correlations between series of Kattan values obtained with different combinations of alternative measures of rarity showed strong correlations ($0.752 < r_s < 0.988$, $p < 0.001$, $n = 39$ for all pair-wise comparisons). Moreover, correlation coefficients between series of BCC values (combination 1 to 10) obtained with probabilistic and binary approach were always very high and significant ($r_s > 0.95$, $p < 0.001$, $n = 10$ for all pair-wise comparisons), thus indicating that the different BCC series of values produce similar results. At the same time, BCW indicators were identical using both probabilistic and binary approaches. Maps of the selected indicators were reported in Figures 1-3. As clearly illustrated in the maps, the ten selected Kattan BCC indexes (using both 'binary' and 'probabilistic' formulation, Fig. 1b and 1c) candidate dunes, deciduous forests, macchia and pinewoods as the landscape-soil units with the highest conservation value. These classes are also characterized by the lowest observed coefficient of variation, which indicates that estimates are affected by moderate variability. To the contrary, Kattan values for the low-value classes, such as shrubland, crop and pastures, were affected by a higher variability. Maps also identified a certain degree of spatial fragmentation in the distribution of BCC indexes due to habitat heterogeneity. Use of the ten Kattan BCW indexes (which produced identical results for the 'binary' and 'probabilistic' approaches, Fig. 1c) produced conservation ranking similar to that obtained from the BCC, with dunes and deciduous forests (together with wetlands) classified as high-value units, pinewoods and macchia classified as medium conservation value units and crop, shrubland and pastures classified as low conservation value units. Interestingly, coefficients of variation for Kattan BCW indexes revealed the same pattern already observed for BCC indexes, discriminating between classes characterized by low-variability and high conservation value and classes characterized by moderate to high variability and rather low conservation value. Taken together, these results stress the conservation importance of forest, wetland and beach-dune units, which received the highest scores for both the presence/absence and the probabilistic matrix, and independently from which Kattan index scheme was used.

BCC and BCW indexes formulated according with the IUCN scheme provided similar results (Figure 2). Using IUCN categories, BCC scores calculated with the two ways of considering NT species (Figure 2a and 2c) were strongly correlated ($r_s = 0.973$, $p < 0.001$, for the presence/absence matrix; $r_s = 0.952$, $p < 0.001$, for the probabilistic matrix; $n = 8$ in both cases). As already observed for Kattan BCW scores, the BCW index using IUCN categories produced identical scores independently from the type of matrix used (Figure 3). Taken together, maps indicate that using the IUCN categories, both the BCC and the BCW indexes stressed the importance of wetlands and forests. However, the BCW index (Figure 3) recovered also the beach-dune system as an additional important landscape unit, thus confirming results obtained using Kattan index scores. According to the abovementioned maps, a spatial heterogeneity in conservation priority was also recorded for indexes based on IUCN categories.

Principal Components Analysis (Figure 4a) confirms the results derived from descriptive statistics by separating high conservation value classes (forest, wetland, beach-dunes) from medium-low conservation value classes (pastures, macchia, shrubland, crop and pinewood). The analysis also clearly discriminates between BCC and BCW indicators (Figure 4b), thus indicating that the two indexes provide complementary, not redundant information.

4. Discussion

The Mediterranean basin has been inhabited by man since prehistoric times and most of its ecosystems have been moulded by people through millennia. As a result, the current coastal landscape is an eco-mosaic of a variety of biotopes, with few and scattered remnants of the original vegetation. At a scale of about 1 km from the seashore landwards, the following main biotope types can be typically recognized in Tyrrhenian Italy: (1) beach, (2) dunes, (3) low maquis, (4) high maquis, (5) wetlands, (6) forests, (7) steppes and (8) cultivated plots (Fattorini and Maltzeff 2001, Fattorini 2002, 2005, Fattorini and Vigna Taglianti 2002). From this respect, Castelporziano estate is paradigmatic, because this area hosts all these biotopes, with large forest patches. The biotopes that host the highest number of tenebrionid species in Castelporziano were the natural forests, the wet zones and the beach-dune system. Wet zones and forests have a high similarity in species composition, because they are spatially and ecologically interrelated, with ponds typically located within patches of mesophilous woodlands, which form a continuous with sclerophyllous-broadleaf forests.

Because of extensive deforestation, mature angiosperm woodlands, especially of *Quercus ilex* L., are precious habitats for several species everywhere along the Mediterranean basin. There is a huge number of species associated with mature angiosperm woodlands and most of them have scattered distributions. Although most of them are relatively eurytopic, some are relatively stenotopic for oaks (e.g., *Neatus noctivagus*) or oaks and beeches (*Diaclina testudinea*, *Corticeus unicolor*, *Iphthiminus italicus*, *Accanthopus welikensis*) (Fattorini and Maltzeff 2001; Fattorini 2002, 2005). Some of these species may have rather large vertical distributions, but most of them are confined to, or more common in, lowland areas. Large deforestation in lowland areas can be responsible for the jeopardized distribution of some of these species (Fattorini and Maltzeff 2001; Fattorini 2002, 2005), and for the recent retreat of *Iphthiminus italicus* to higher altitudes (about 1400 m) in Sicily, where extensive woodlands are still present (Aliquò and Leo 1996). Mesophilous woodlands host rich tenebrionid communities, including several stenotopic species. For example, Schawaller (1996) reports the following species inhabiting poplar woodlands in coastal areas of Northern Greece: *Bolitophagus reticulatus, Diaperis boleti Palorus depressus, Uloma culinaris, D. testudinea*, *Neatus picipes* and *Nalassus plebejus*. Notably, substantially the same assortment of species was found in the wet areas and forests of Castelporziano (with *Neatus noctivagus* replacing *N. picipes* and *Nalassus dryadophilus* replacing *N. plebejus*).

Both descriptive statistics and multivariate analysis (PCA) performed with conservation indexes (using different Kattan and IUCN formulation) clearly separated forests, wetlands and dunes from the other types of landscape units, indicating the former classes as the most interesting for conservation purposes. The analysis also pointed out the differences in the two conservation indexes BCC and BCW and the need for a comprehensive assessment of conservation priorities based on pieces of information provided by the two indexes and their various formulations. In the present study, such a comprehensive assessment was carried out using a principal components analysis, which showed that the two indexes BCC and BCW provide complementary information and that the use of different scoring schemes of species vulnerability produced correlated, but not necessarily identical conservation ranking in area prioritization. Notably, the alternative use of a binary and a probabilistic matrix produced similar results, thus suggesting that the BCC and BCW are quite insensitive to uncertainties in species distribution data. In-depth investigation on this key issue through the use of data covering larger areas is particularly needed.

Most coastal sites in Tyrrhenian Italy have been (re)forested with pine trees. In fact, pines are autochthonous species in the Mediterranean (albeit not everywhere), and in some circumstances it is not clear if the present pine forests are native or planted. Also, invasive pines have accompanied evergreen oaks in the process of secondary succession (Blondel and Aronson 1999). Thus, although the oak forest is considered the climax vegetation of the Mediterranean basin, deforested sites, occupied by garrigues and maquis vegetation, tend sometimes to be colonized by pines more than by oaks (cf. Marcuzzi 1965)]. Marcuzzi (1965) observed that in Apulia tenebrionids are absent from oak forests, while all tenebrionid species found in the pinewoods are also found in garrigues and maquis. Although there are some tenebrionid species that are more commonly found on conifers than deciduous trees (such as *U. culinaris* and *Menephylus cylindricus*), the majority of the species are more commonly associated with angiosperm woodlands, being rarely found on gymnosperms (e.g. *Colpotus strigosus*, *Scaphidema metallicum*, *Helops caeruleus*, *Accanthopus welikensis*, *Nalassus dryadophilous*) (Fattorini and Maltzeff 2001; Fattorini 2002, 2005) and the pinewood of Castelporziano do not rank high for species richness. As a matter of fact, principal components analysis classified pinewoods as a unit with poor conservation value together with pastures, crop, macchia and shrubland.

In addition to pines, as a result of land reclamation of wet areas, extensive plantations of *Eucalyptus* have been planted on Italian coasts. Although these trees are alien plants, they now play an important role in insect conservation by providing refuge for many species which find shelter under their bark. Eucalypts shed their bark regularly. If the outer part of the bark is completely shed but not completely detached from the trunk (a condition found in a great number of eucalypt species), the shedding process provides suitable resting places for a number of tenebrionid species. In Castelporziano, several tenebrionid species are regularly found under eucalyptus bark, including both arboreal species, such as *Nalassus planipennis* (an Italian endemic species, found at Castelporziano only under barks of *Eucalyptus*, but also know to live under *Castanea sativa* Mill. barks, possibly its original habitat), *Nalassus dryadophilus* (another xylophilous species, associated with *Castanea sativa* Mill., *Quercus ilex* L., *Populus* sp., *Platanus* sp., *Pinus nigra* Arnold, and the introduced *Robinia* sp., found at Castelporziano under the bark of *Quercus* sp., *Populus* sp., and *Eucalyptus* sp.), and *Catomus rotundicollis* (a more eurytopic species, found at Castelporziano under the bark of *Salix* sp., *Quercus* sp. and *Eucalyptus* sp., but known to be found also under the bark of *Arbutus unedo* L. and *Euphorbia dendroides* L.), as well as more typically soil dwelling species, which can be found under bark as overwintering individuals, such as *Stenosis sardoa ardoini* (found in winter under the bark of *Populus*, *Quercus*, *Salix*, *Pinus*, and especially *Eucalyptus*) (Fattorini and Maltzeff 2001).

Although ecologically very different from forests and wetlands, the beach-dune system ranked very high in terms of species richness and conservation value, as expressed by the BCC and BCW indexes. High tenebrionid richness and conservation values associated with the beach-dune system can be related to the fact that many tenebrionid groups are stenotopic species well adapted to desert habitats (psammophilous species). Sandy shores and dunes are widely recognized as priority biotopes in conservation planning because they are now highly impacted and fragmented habitats. The species inhabiting the beach-dune system are completely different from those of forests (with only one species, *Catomus rotundicollis*, occurring in both biotopes). Yet, the tenebrionid community of the beach-dune system included some species that have high Kattan values because they are rare for the habitat (being strictly associated with coastal sandy soils) and a scattered distribution.

In an ideal transect from the seashore to the inland, the beach-dune system and the forests are placed at the extremes, whereas garrigues and maquis are placed between them and represent a transitional landscape that originated from the degradation of woodlands (e.g. due to clearcutting or fires). Thus, they tend to host generalist species also occurring in the adjacent biotopes (Fattorini 2005). It is therefore not surprising that they scored low for the BCC and BCW indexes. However, this does not imply that garrigues and maquis do not deserve conservation efforts. The high number of species recorded on the dunes, as well as in the natural forests and wetlands of the study area, may be related to the well preservation status of these habitats. But in most Mediterranean areas these biotopes have been destroyed and alternative biotopes, like garrigues and maquis, may represent the only available chance for tenebrionids that, in natural conditions, are more strictly associated with sand dunes and mesophilous woods.

5. Conclusions

The present study proved the importance of species habitat distribution in determining conservation priorities in a highly composed ecomosaic characterized by habitat discontinuities (such as the strong contrasts between patches occupied by sand soils and other with abundant litter) and landscape fragmentation (as in the cases of sparse wetlands). In a relatively small area subjected to remarkable anthropogenic pressure on the fringe, such as Castelporziano estate (60 km^2), the spatial distribution of tenebrionids, based on extensive, long-term surveys, revealed a marked polarization of conservation values, with forests, wetlands, dunes identified as high-quality land from one side, and traditional soil-landscape units of the Mediterranean region, such as dry pastures, traditional crop mosaics, pinewoods and macchia/garrigue shrublands, identified as medium-low quality land from the other side. Interestingly, landscape-soil units identified as having high conservation priorities for tenebrionids formed a very heterogeneous set of biotopes. This is a reflection of the mosaic distribution of the various landscape-soil units, which do not form extensive units but small patches, each of one with a peculiar tenebrionid community.

In our study, we combined all types of natural forests into a single unit, because there is no sufficient information to associate tenebrionid species to different form of forest vegetation. It is however possible that tenebrionid species may have difference preferences for different types of forests. In the future, an in depth survey might provide more detailed information on tenebrionid association with trees and hence a more refined evaluation of the conservation value of different deciduous forest types. However, even with the use on broadly defined landscape units, our analysis illustrated how the spatial analysis of the conservation value of forest stands, based on reliable bio-indicators, may inform sustainable land management and effective conservation measures in an ecologically-fragile Mediterranean landscape.

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Table 1 Tenebrionid distribution across landscape-soil units in Castelporziano. P indicates ascertained presence, * indicates probable presence. Nomenclature follows Löbl and Smetana (2008).

Species	Dunes	Macchia	Shrubland	Wetl and	Forests (mainly deciduous)	Pinewoods	Dry pastures	Crop
Accanthopus velikensis	-	-	-	-	P	-	-	-
Ammobius rufus	P	-	-	-	-	-	-	-
Asida luigionii luigionii	-	P	*	-	-	-	P	P
Blaps gibba	-	-	-	P	-	-	-	-
Bolitophagus reticulatus	-	-	-	P	-	-	-	-
Catomus rotundicollis	P	*	-	P	P	P	*	-
Colpotus strigosus strigosus	-	-	-	P	P	-	-	-
Corticeus unicolor	-	-	-	*	P	-	-	-
Dendarus lugens	-	*	-	-	-	-	*	-
Diaclina testudinea	-	-	-	*	P	-	-	-
Diaperis boleti	-	-	-	P	*	-	-	-
Erodius siculus neapolitanus	P	*	-	-	-	-	-	-
Gonocephalum granulatum nigrum	P	-	-	-	-	-	-	-
Gunarus parvulus	P	-	-	-	-	-	-	-
Halammobia pellucida	P	-	-	-	-	-	-	-
Helops caeruleus	-	-	-	P	P	-	-	-
Iphthiminus italicus italicus	-	-	-	-	P	*	-	-
Menephilus cylindricus cylindricus	-	-	-	P	P	*	-	-
Nalassus aemulus aemulus	P	-	-	-	-	-	-	-
Nalassus dryadophilus	-	-	-	P	P	P	-	-
Nalassus planipennis	-	-	-	P	P	-	-	-
Neatus noctivagus	-	-	-	-	P	-	-	-
Oochrotus unicolor	-	-	-	P	-	-	-	-
Pedinus meridianus	-	P	-	-	-	P	P	*
Phaleria acuminata	P	-	-	-	-	-	-	-
Phaleria provincialis ghidinii	P	-	-	-	-	-	-	-
Pimelia bipunctata	P	*	*	-	-	-	P	-
Platydema violacea	-	-	-	P	-	-	-	-
Pseudoseriscius normandi pacificii	P	-	-	-	-	-	-	-
Scaphidema metallica	-	-	-	P	-	-	-	-
Scaurus striatus	-	P	P	-	-	-	P	P
Stenosis intermedia	P	-	-	-	-	-	-	-
Stenosis sardoa	-	-	P	P	P	P	P	-
Tenebrio obscurus	-	-	-	-	P	P	-	-
Tentyria grossa grossa	P	*	-	-	-	-	*	-
Trachyscelis aphodioides	P	-	-	-	-	-	-	-
Uloma culinaris	-	-	-	P	*	-	-	-
Xanthomus pallidus	P	-	-	-	-	-	-	-
Xanthomus pellucidus	P	-	-	-	-	-	-	-

Table 2 Values of vulnerability scores calculated using Kattan index in ten different versions (K1 to K10, see text for details). In all cases, Kattan index varied from K=1 (lowest vulnerability) to K=8 (highest vulnerability).

Species	K1	K2	K3	K4	K5	K6	K7	K8	K9	K10
Accanthopus velikensis	1	1	1	1	1	1	1	1	1	1
Ammobius rufus	1	3	3	3	3	1	3	3	3	3
Asida luigionii luigionii	1	1	1	1	1	1	1	1	1	1
Blaps gibba	1	1	1	1	1	1	1	1	1	1
Bolitophagus reticulatus	1	1	1	1	1	1	1	1	1	1
Catomus rotundicollis	1	1	1	3	1	1	1	1	3	1
Colpotus strigosus strigosus	1	1	1	1	1	1	1	1	1	1
Corticeus unicolor	1	1	1	1	1	1	1	1	1	1
Dendarus lugens	2	2	2	2	2	2	2	2	2	2
Diaclina testudinea	6	6	8	8	8	6	6	8	8	8
Diaperis boleti	1	1	1	1	1	1	1	1	1	1
Erodius siculus neapolitanus	1	3	3	3	3	1	3	3	3	3
Gonocephalum granulatum nigrum	1	3	1	1	3	1	3	1	1	3
Gunarus parvulus	8	8	8	8	8	8	8	8	8	8
Halammobia pellucida	7	7	7	7	7	3	3	3	3	3
Helops caeruleus	1	1	1	1	1	1	1	1	1	1
Iphthiminus italicus italicus	1	1	1	1	1	1	1	1	1	1
Menephilus cylindricus cylindricus	8	8	8	8	8	8	8	8	8	8
Nalassus aemulus aemulus	8	8	8	8	8	8	8	8	8	8
Nalassus dryadophilus	1	1	1	1	1	1	1	1	1	1
Nalassus planipennis	1	1	1	1	1	1	1	1	1	1
Neatus noctivagus	6	8	8	8	8	6	8	8	8	8
Oochrotus unicolor	6	6	6	6	6	6	6	6	6	6
Pedinus meridianus	1	1	1	1	1	1	1	1	1	1
Phaleria acuminata	1	3	3	3	3	1	3	3	3	3
Phaleria provincialis ghidinii	1	1	3	3	3	1	1	3	3	3
Pimelia bipunctata	1	1	3	3	1	1	1	3	3	1
Platydema violacea	1	1	1	1	1	1	1	1	1	1
Pseudoseriscius normandi pacificii	3	3	3	3	3	7	7	7	7	7
Scaphidema metallica	2	2	2	2	2	2	2	2	2	2
Scaurus striatus	1	1	1	1	3	1	1	1	1	3
Stenosis intermedia	1	1	3	3	3	1	1	3	3	3
Stenosis sardoa	1	1	1	3	1	1	1	1	3	1
Tenebrio obscurus	1	1	1	1	1	1	1	1	1	1
Tentyria grossa grossa	1	3	1	3	3	1	3	1	3	3
Trachyscelis aphodioides	7	7	7	7	7	3	3	3	3	3
Uloma culinaris	1	1	1	1	1	1	1	1	1	1
Xanthomus pallidus	6	6	8	8	8	6	6	8	8	8
Xanthomus pellucidus	4	7	7	7	7	4	7	7	7	7

Table 3 Categories of extinction risk at National level established following IUCN (2003) criteria. Criteria are explained only for threatened species.

Species	Estimated area of occupancy	Severely fragmented or number of locations < a specified threshold	Continuing decline in	Extreme fluctuation in	IUCN Category
Bolitophagus reticulatus	< 2000 km^2	Severely fragmented	Extent of occurrence and area of occupancy; area, extent and quality of habitat (mature woodlands)		VU
Diaclina testudinea	< 500 km^2	Severely fragmented, number of locations < 10	Area, extent and quality of habitat (mature woodlands)	Area of occupancy, number of locations or subpopulations	EN
Gunarus parvulus	< 2000 km^2	Severely fragmented, number of locations < 10	Extent and quality of habitat (sand dunes)		VU
Iphthiminus italicus	< 1000 km^2	Severely fragmented, number of locations < 10	Extent of occurrence and area of occupancy; area, extent and quality of habitat (mature woodlands), number of locations or subpopulations		VU
Menephilus cylindricus	< 1000 km^2	Severely fragmented, number of locations < 10	Area, extent and quality of habitat (mature woodlands)		NT
Nalassus aemulus	< 2000 km^2	Severely fragmented, number of locations < 10	Extent and quality of habitat (sand dunes)		VU
Neatus noctivagus	< 1500 km^2	Severely fragmented, number of locations < 15	Area of occupancy; area, extent and quality of habitat (mature woodlands), number of locations or subpopulations		VU
Platydema violacea	< 1500 km^2	Severely fragmented, number of locations < 15	Area, extent and quality of habitat (mature woodlands, especially in wet zones)		NT
Pseudoseriscius normandi	< 2000 km^2	Severely fragmented, number of locations < 10	Extent and quality of habitat (sand dunes)		VU

Table 4 Average values and Coefficient of Variation (CV) for the conservation indexes BCC and BCW based on ten different applications of Kattan vulnerability index for each landscape-soil unit using both a probabilistic and a binary matrix for species distribution across landscape-soil units.

	BCC (probabilistic matrix)		BCC (binary matrix)		BCW	
Landscape-soil unit	Average	CV	Average	CV	Average	CV
Dune	0.43	0.17	0.43	0.17	0.63	0.04
Macchia	0.21	0.35	0.10	0.52	0.07	0.46
Shrubland	0.12	0.61	0.06	0.99	0.02	0.97
Wetland	0.20	0.11	0.18	0.12	0.27	0.11
Deciduous forest	0.22	0.13	0.20	0.14	0.27	0.07
Pinewood	0.21	0.19	0.16	0.22	0.10	0.18
Pasture	0.13	0.50	0.08	0.68	0.05	0.60
Crop	0.05	0.94	0.02	2.11	0.00	2.11

Table 5 Average values for the conservation indexes BCC and BCW based on two different applications of IUCN extinction risk categories (NT1: Near Threaded species scored as Least Concern species; NT1.5: Near Threaded species scored higher than Least Concern species) using both a probabilistic and a binary matrix for species distribution across landscape-soil units. For BCW, the two matrices gave identical results.

Landscape-soil unit	BCC				BCW	
	NT1		NT1.5		NT1	NT1.5
	Probabilistic matrix	Binary matrix	Probabilistic matrix	Binary matrix		
Dune	0.06	0.06	0.06	0.06	0.33	0.30
Macchia	0.15	0.00	0.15	0.00	0.00	0.00
Shrubland	0.11	0.00	0.11	0.00	0.00	0.00
Wetland	0.11	0.08	0.13	0.10	0.44	0.50
Deciduous forest	0.14	0.11	0.15	0.12	0.56	0.55
Pinewood	0.11	0.05	0.14	0.07	0.11	0.15
Pasture	0.08	0.00	0.08	0.00	0.00	0.00
Crop	0.07	0.00	0.07	0.00	0.00	0.00

Figure 1 Spatial distribution of conservation values of landscape-soil units based on tenebrionid communities in Castelporziano reserve. (a) Classification of landscape-soil units. (b) Conservation values obtained by using the BCC index, Kattan vulnerability scores and a probability matrix of tenebrionid distribution across units. (b) Conservation values obtained by using the BCC index, Kattan vulnerability scores and a binary matrix of tenebrionid distribution across units. (c) Conservation values obtained by using the BCW index and Kattan vulnerability scores (probabilistic and binary matrices gave identical outcomes).

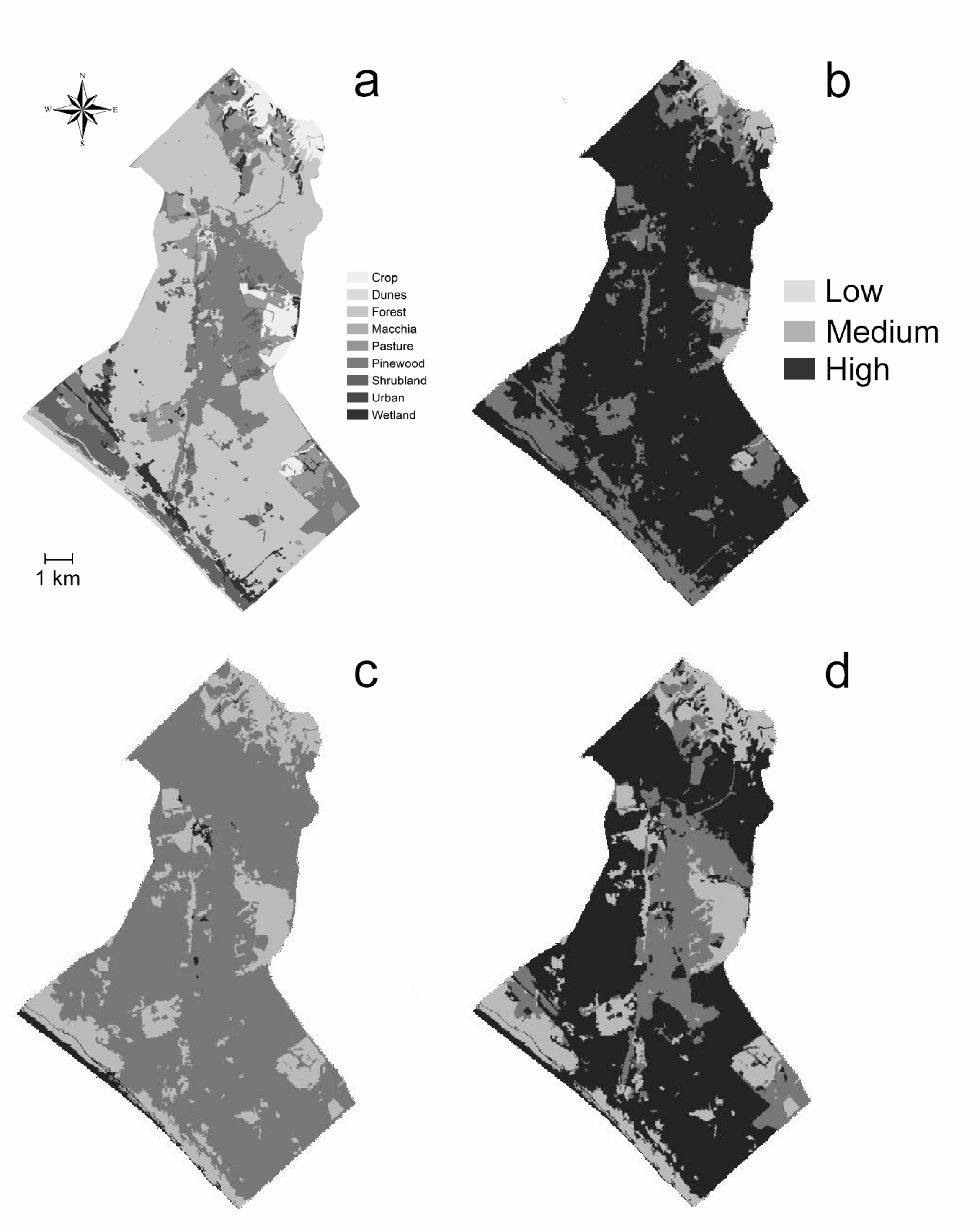

Figure 2 Spatial distribution of conservation values of landscape-soil units based on tenebrionids communities in Castelporziano reserve by using the BCC index and IUCN extinction categories. (a) Conservation values obtained with Near Threatened species scored as Least Concern species and a probability matrix of tenebrionid distribution across units. (b) Conservation values obtained with Near Threatened species scored as Least Concern species and a binary matrix. (c) Conservation values obtained with Near Threatened species scored higher than Least Concern species and a probabilistic matrix. (d) Conservation values obtained with Near Threatened species scored higher than Least Concern species and a binary matrix.

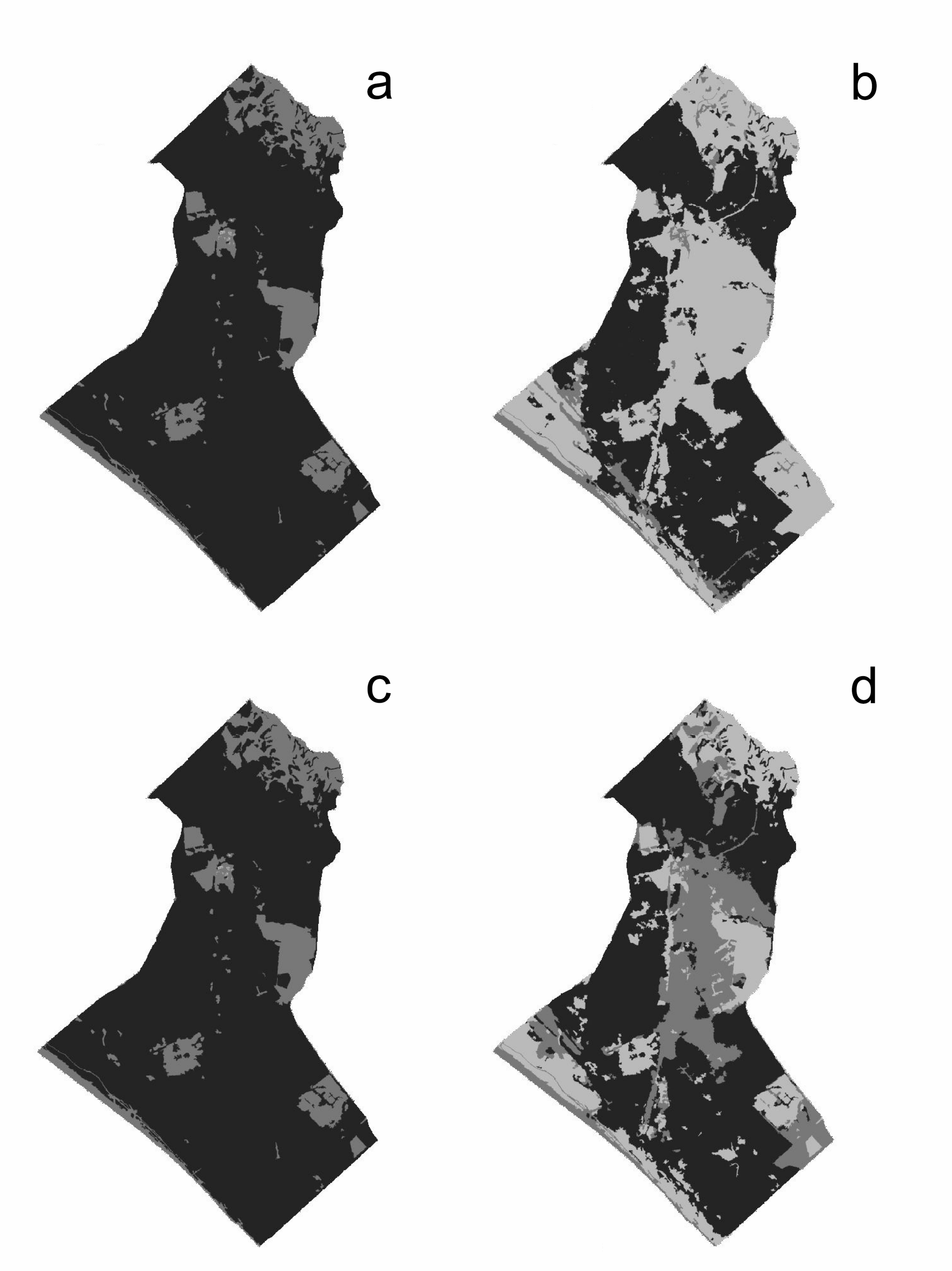

Figure 3 Spatial distribution of conservation values of landscape-soil units based on tenebrionids communities in Castelporziano reserve by using the BCW index and IUCN extinction categories. (a) Near Threatened species scored as Least Concern species. (d) Near Threatened species scored higher than Least Concern species.

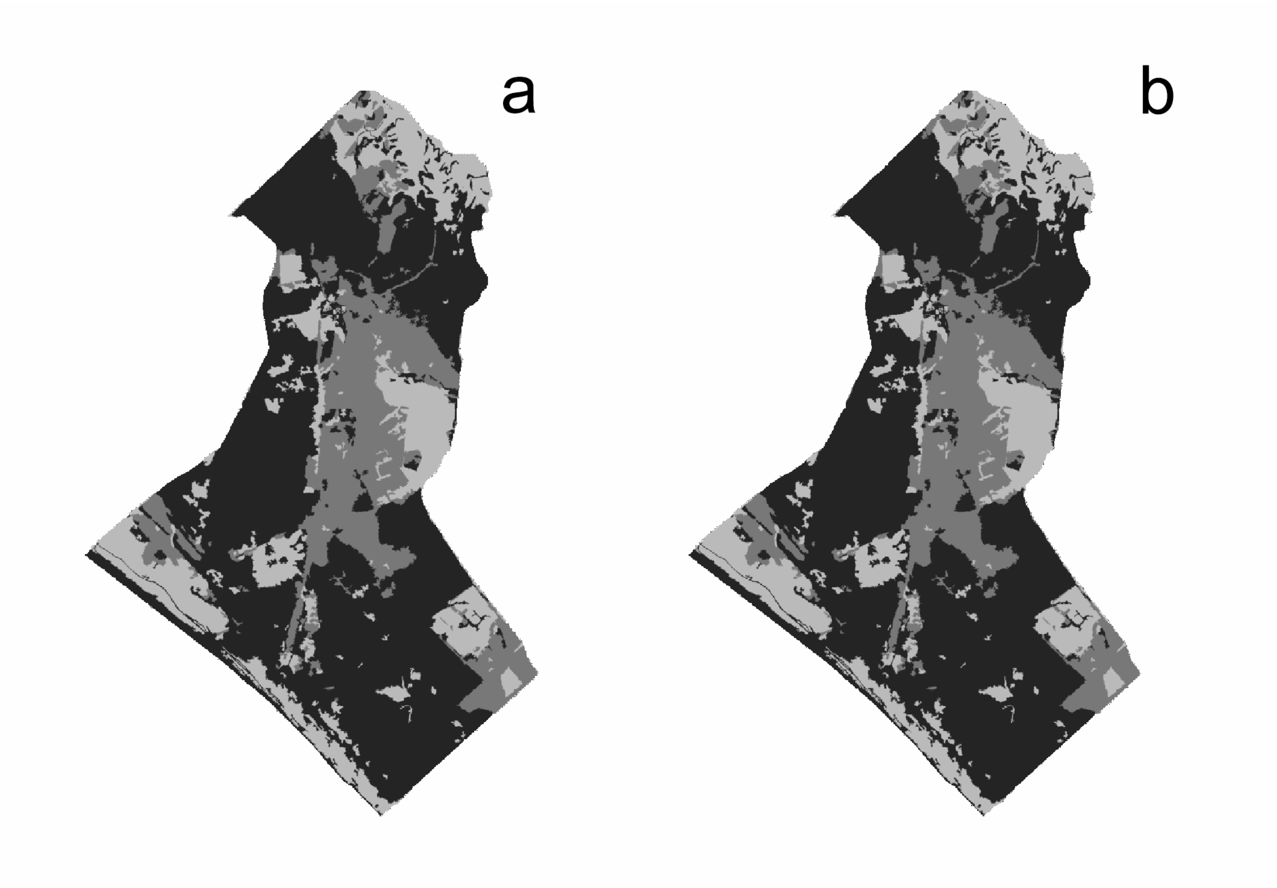

Figure 4 Results of Principal Components Analysis (PCA). (a) PCA loadings. (b) PCA scores. Continuous and dotted lines encompass BCW and BCC values, respectively. In both panels, the first two factors (F1 and F2) are reported. F1 explained 49.0% of variance, F2 explained 28.1% of variance.