

## Using local autocorrelation analysis to identify conservation areas: an example considering threatened invertebrate species in Spain

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**Abstract** Local autocorrelation statistics offer new opportunities for the discrimination of important conservation areas since the spatial dependence of local values upon neighbouring ones may assist conservation decisions. We exemplify the use of local autocorrelation statistics for conservation purposes using data on Spanish threatened invertebrates to identify areas composed of similarly species-rich localities (*hot spots*), species-rich “islands”, cold spots and species-poor “islands”. In order to assess the probable causes of the detected patterns differences in environmental, land use and protected area variables were examined between the different regions. Distributional data for threatened invertebrate species in Spain at 100 km<sup>2</sup> UTM cell resolution were used. After defining a neighbouring area, statistically significant local autocorrelation values (both positive and negative) were estimated. Kruskal–Wallis ANOVA by rank test was used to compare the environmental, land use and protected area percentages between the cells of the different regions. Around 11 and 2 % of total cells can be considered *hot spots* and rich “islands”, respectively. *Hot spots* are characterized by a lower percentage of anthropic land uses and a higher percentage of current protected area. However, approximately a third part of these cells possess at least 98 % of their area unprotected. Rich “island” cells are not environmentally different from those considered as cold spots, though experiencing a lower rate of anthropization and higher proportion of protected area. Unfortunately, almost 70 % of these rich “island” cells have <2 % of their areas currently protected. The use of local autocorrelation statistics on species richness values may complement conservation decisions by discriminating interconnected sites facilitating local persistence (*hot spots*) as well as isolated and vulnerable sites (rich “islands”). The study of different variables associated

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with these regions allows us to suggest determinant causal factors. Our results suggest that land use changes due to human activities are the main cause of threats.

**Keywords** Biodiversity important regions · Hot spots · LISA analysis · Local autocorrelation · Spain · Threatened invertebrates

## Introduction

One of the fundamental challenges in conservation biology involves the development of theoretical principles and methods for the selection of nature reserves (Williams et al. 2004). Because the distribution of biodiversity is frequently patchy, past ad hoc procedures to design networks of reserves should be replaced with quantitative and repeatable procedures capable of maximising the representation of biodiversity with the minimum protected area effort (Margules and Pressey 2000; Cabeza and Moilanen 2001; Wiersma and Nudds, 2009). Among the different scientific criteria for reserve selection, complementarity, irreplaceability, and vulnerability have become key concepts that have guided the design of conservation planning tools, avoiding intuitive judgements (Margules et al. 1988; Vane-Wright et al. 1991; Pressey et al. 1993; Pressey 1994; Justus and Sarkar 2002; Wilson et al. 2005; Sarkar et al. 2006). However, although adjacency has been included as a criterion in reserve selection procedures (Nicholls and Margules 1993), application of these concepts and methodologies does not explicitly consider the spatial structure of biodiversity data to propose conservation areas.

Both biological and environmental variables are frequently and inevitably spatially autocorrelated, as their values at adjacent localities are more similar than random. To explore ecological and biogeographical patterns, global measures of spatial autocorrelation devoted to estimating the average spatial dependence of a variable over a study region are now frequently considered in ecological and biogeographical studies (Legendre and Fortin 1989; Leduc et al. 1992; Legendre 1993; Bini et al. 2000; Fox et al. 2000; Rahbek and Graves 2001; Diniz-Filho and De Campos Telles 2002; Niggebrugge et al. 2007). However, when the underlying process that generates the spatial pattern is non-stationary and local patterns shifted in space, the use of these global autocorrelation measures may not be representative. Under these circumstances it is useful to examine the spatial dependence of each local value upon neighbouring ones by calculating local measures of spatial autocorrelation (Anselin 1995; Sokal et al. 1998). Application of local autocorrelation statistics in biology can be considered occasional, being used for example, to study the regional distributional patterns of specific species (Dennis et al. 2002), the genetic structure of populations (Double et al. 2005; Sokal and Thomson 2006), or declines of species populations (Koenig 2001). However, although the possibilities associated with local spatial autocorrelation statistics could present new opportunities for the discrimination of important conservation areas (Nelson and Boots 2008) this statistical technique has not yet been applied for this purpose.

The spatial dependence of local values upon neighbouring ones may assist conservation decisions by identifying (i) *hot spots*, or areas composed of similarly species-rich localities; and (ii) species-rich localities situated within a species-poor regional matrix (species-rich “islands”). The first type of localities will have a high probability of being connected, thus favouring the persistence of species due to the maintenance of metapopulation dynamics. The second type of localities can be considered to be at risk from a conservational point of view due to their isolation, which may be due to the environmental singularity of these

localities and/or intense human influences (Underwood et al. 2009). Selection of new reserves by the classical methods based in the complementarity criterion may fail to discriminate and detect these two important types of localities when they do not sufficiently complement the species list of the reserves previously selected. Here, we use the best available distributional data for endangered, critically endangered and vulnerable invertebrate species of Spain (Verdú and Galante 2008; Verdú et al. 2011) to exemplify the use of local autocorrelation statistics for conservation purposes. Specifically we aim to (i) identify both concentration (*hot spots*) and abnormal patterns (species-rich “islands”) in the occurrence of threatened invertebrate species using local autocorrelation statistics, (ii) examine whether their presence is related to the occurrence of special environmental conditions and/or human-altered land uses, and (iii) assess the capacity of current protected areas to represent them. Finally, we discuss the possibilities of this approach for the delimitation of protected areas as well as the specific implications of the obtained results for the conservation of the threatened invertebrates in Spain.

## Methods

### Study area

The study area was comprised of Peninsular Spain and the Balearic Islands, extending over 504,645 km<sup>2</sup> and belonging to Mediterranean Region. The Macaronesian Region (Canary Islands) was excluded from this study. This territory includes a large variety of biomes, reliefs, climates, and soil types and includes altitudes ranging from sea level to 3,483 m. Landscapes vary extremely, from some that are almost desert like to others green and fertile. There are also long coasts in the east along Mediterranean Sea and in the west along the Atlantic Ocean and Cantabric Sea. Due to this great variety of reliefs and climates, Spain presents an enormous diversity of vegetation types, from deciduous and coniferous forests and evergreen woodland to scrubland and annual grassland (Rey-Benayas and Scheiner 2002).

### Biological data

The biological data used in this study come from Spanish atlases which compile all available information on the distribution of threatened invertebrates in Spain (Verdú and Galante, 2008; Verdú et al. 2011), including endangered, critically endangered and vulnerable species (IUCN 2003). The data used came from several sources such as literature, scientific collections, as well as new data obtained from specific fieldwork carried out by around 150 taxonomists during thirteen years (from 1997 to 2010). We elaborated a database using all of this information that finally included distributional data for 230 species: 16 critically endangered, 48 endangered and 166 vulnerable species, corresponding to 7, 37 and 101 species of arthropods and 9, 11 and 65 species of molluscs, respectively (see Appendix 1 in the Supplementary Material). In summary, we obtained information about the presence of threatened invertebrates for 2,108 UTM 100 km<sup>2</sup> cells from mainland Spain and the Balearic Islands (approximately 39 % of total cells). Although many of the primary field data is available at a higher resolution (UTM cells of 1 km<sup>2</sup>) we decided to use the occurrences in cells of 100 km<sup>2</sup> in order to limit the effect of survey effort differences. Thus, in this study we assume that the exhaustive survey effort

carried out is enough intensive to minimize the effects of survey biases and incompleteness on detected patterns.

### Data analysis

LISA analysis (Local Indicators of Spatial Association; Anselin 1995) was used to determine the degree of spatial association among local richness values of threatened invertebrate species. LISA allows the detection of local statistically significant patterns by examining the change in the intensity of the spatial autocorrelation of the species richness for each individual locality (UTM 100 km<sup>2</sup> cells) within the study area. We first estimate the greatest distance with significant global spatial autocorrelation using Moran's  $I$  values (50 km in our case) to subsequently use this distance to define a neighbouring area, where the same autocorrelation index is computed separately for each locality. Thus, we discriminate those localities with statistically significant local autocorrelation values (both positive and negative). For this purpose, observed local Moran's  $I$  values are compared against those provided by a Monte Carlo permutation procedure ( $n = 999$ ) in which data are spatially randomised, and localities with Moran's  $I$  values statistically different at a 5 % level are selected. Subsequently, localities with statistically significant local autocorrelation values are divided into those showing positive and negative autocorrelations. Positive local autocorrelations reflect clusters of localities with similar species richness values, while negative ones allow us to discriminate outliers (Anselin 1995). The distribution of species richness values showed that most of the 100 km<sup>2</sup> UTM cells of mainland Spain and the Balearic Islands (61.2 %) do not harbour any threatened invertebrate species and that more than half (56.3 %) of the remaining cells with observations only contain one species (mean  $\pm$  95 % confidence interval;  $0.71 \pm 0.03$  species per grid cell). Taking into account the upper quartile of the species richness values (UQ = 1), we consider as species-rich cells those that include more than one threatened invertebrate species and as species-poor cells those that contain one or any species. Thus, clusters of cells with positive and significant local autocorrelation values and more than one species are considered as *hot spots* ( $H_S$ ), i.e., cells with high species richness values surrounded by other highly species-rich cells. Similarly, cells with low species richness values ( $\leq 1$  species) surrounded by other similarly species-poor cells have been designated 'cold spots' ( $C_S$ ). Additionally, outliers with negative significant local autocorrelations and high species richness values ( $> 1$  species) will be identified as rich "islands" ( $R_I$ ), i.e., cells that show high species richness values and are surrounded by neighbours with low species richness values. Cells with low species richness values that are surrounded by cells with higher species richness values are designated poor "islands" ( $P_I$ ). Finally, the remaining cells ( $R$ ) with non-significant local autocorrelation values and/or no species data were also discriminated.

All the calculations were carried out using Spatial Analysis in Macroecology software (SAM; Rangel et al. 2010).

### Climatic, land use and protected surface data

A number of environmental, land use and protected area variables were calculated for each one of the 100 km<sup>2</sup> UTM grid cells. Climatic data were obtained from WORLDCLIM (version 1.3, <http://www.worldclim.org>; see Hijmans et al. 2005), which includes data on 19 bioclimatic variables at an approximate resolution of 1 km<sup>2</sup> obtained by the interpolation of climate station records from 1950 to 2000. After calculating the average values in each 100 km<sup>2</sup> UTM grid cell they were standardised to a 0 mean and 1 standard deviation.

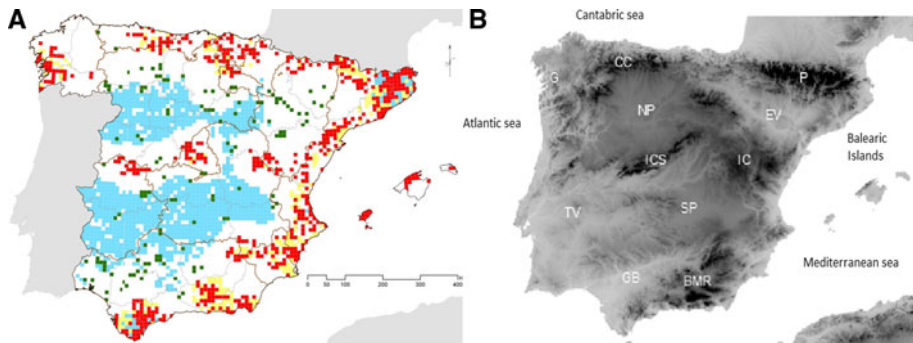
Subsequently, the values of all of these bioclimatic variables together with the mean altitude and altitude range obtained from a Digital Elevation Model (Clark-Labs 2000) were subjected to a principal component analysis (PCA) to obtain uncorrelated environmental factors (Varimax rotation). Four factors were selected with eigenvalues above 1, which jointly accounted for 90.2 % of the total variability (45.5 %, 29.1 %, 10.5 % and 5.1 %, respectively). The two main factors represent a ‘mountain’ and an ‘aridity’ gradient, respectively. Factor 1 is positively related to altitudinal variables and negatively related to the mean annual temperature and the minimum temperature of the coldest month (absolute factor loadings higher than 0.80). Factor 2 is positively related to the maximum temperature of the warmest month and negatively to the precipitation of the warmest quarter. The values of these four PCA factors were used to estimate environmental differences between the five types of cells previously defined.

Similarly, we reclassified the 44 categories (level 3) established in the land use map provided by Corine Land Cover 2006 (see [www.eea.europa.eu](http://www.eea.europa.eu)) to obtain two main types of land uses: anthropic (21 categories) and natural (23 categories), with semi-natural traditional low intensity land uses being considered as natural areas here. Next, we calculated the percentage of the area covered by these two land use types in each UTM cell. We also estimated the rate of anthropisation as the percentage of each cell area that changed from natural to anthropic land uses considering Corine Land Cover maps for 1990 and 2006. Finally, we estimated the percentage of each UTM cell area covered by current protected areas (PAs) and the Natura 2000 network ( $N_{2000}$ ), which was obtained from [www.redeuroparc.org](http://www.redeuroparc.org) (Europarc-España 2008).

A Kruskal–Wallis ANOVA by rank tests was used to compare the environmental, land use and protected area percentages between those cells of the five categories identified by local autocorrelation analysis. Multiple post hoc comparisons of mean ranks were also estimated to detect statistically significant differences for all pairs of groups (Siegel and Castellan 1988).

## Results

Approximately 11 % of total Spanish cells (UTM 100 km<sup>2</sup>) can be considered as belonging to *hot spots*, and within these cells, we find data belonging to 86.1 % of the considered threatened species. These *hot spots* are mainly located along the Mediterranean coast from the southern corner near the Gibraltar strait to the boundary with France as well as in the Baetic and Sub-baetic Mountain ranges, the Balearic Islands, the Pyrenees, the Cantabrian Cordillera, the Galician northwestern region, the Iberian Cordillera and the Iberian Central System, which separates northern and southern inner plateaus in where cold spot areas are located (Fig. 1). The values of the four environmental PCA factors in the *hot spot* cells are statistically different from those of all the remaining cell categories (Table 1), except in the case of poor “island” cells, which seem to be climatically and topographically similar to these *hot spots*. *Hot spot* cells are also characterised by a statistically significantly lower percentage of anthropic land use according to the 2006 Corine Land Cover data (mean = 42.5 %; minimum = 0, maximum = 99.4 %;  $\pm 95$  % confidence interval = 2.2%). In approximately 38 % of the *hot spot* cells, more than half of the area is anthropised, and in 17 % of these cells, the percentage of anthropised area is higher than 75 %. Interestingly, *hot spot* cells do not show statistically different anthropisation rates from 1990 to 2006 compared to any of the other considered cell categories (Table 1). *Hot spot* cells also include a significantly higher percentage of current



**Fig. 1** **a** Hot spots (red), cold spots (blue), rich “islands” (green) and poor “islands” (yellow) for threatened Spanish terrestrial invertebrates according to LISA analysis results. **b** In the right, main physiographic regions of the Iberian Peninsula: *BMR* Baetic and Subbaetic Mountain ranges, *CC* Cantabrian Cordillera, *EV* Ebro Valley, *P* Pyrenees, *GB* Guadalquivir Basin, *ICS* Iberian Central System, *IC* Iberian Cordillera, *SP* South Plateau, *NP* North Plateau, *TV* Tajo Valley, *G* Galician north-western region. (Color figure online)

protected area (mean = 24.7 %; minimum = 0, maximum = 100 %;  $\pm 95$  % confidence interval = 2.5 %) than the other cell categories, though this percentage is similar to those richness “island” cells in the case of the Nature 2000 network (Table 1). However, for almost 36 % of  $H_S$  cells,  $< 2$  % of their area is currently protected, and for 59 % of these cells,  $< 15$  % of the area is protected. Including as protected area the Nature 2000 network improves the situation, but 36 % of  $H_S$  cells still have  $< 15$  % of their area protected.

**Table 1** Number of 100 km<sup>2</sup> UTM cells ( $C_{UTM}$ ), mean species richness of threatened invertebrate species ( $S \pm 95$  % confidence interval) and statistically significant differences in the considered environmental, land use and protected area variables between the five UTM cell categories according to local autocorrelation analysis: hot spots ( $H_S$ ), cold spots ( $C_S$ ), rich “islands” ( $R_I$ ), poor “islands” ( $P_I$ ) and remaining cells ( $R$ )

	$H_S$	$R_I$	$C_S$	$P_I$	$R$
$C_{UTM}$	588	130	1254	302	3159
$S$	$3.16 \pm 0.12$	$2.72 \pm 0.17$	$0.04 \pm 0.01$	$0.00 \pm 0.00$	$0.52 \pm 0.03$
$PCA_{ENV}$ F1	a	b, c	b, d	a	c, d
$PCA_{ENV}$ F2	a			a	
$PCA_{ENV}$ F3	a	b, c, d	b	a, c, e	d, e
$PCA_{ENV}$ F4	a	b, c	b	a	c
% anthropic land use	(42.5 %)	a, b (56.6 %)	(65.4 %)	a, c (52.5 %)	b, c (52.7 %)
% anthropisation	a	a	a	a	a
%PA	(24.7 %)	a, b (13.7 %)	(6.6 %)	a (12.1 %)	b (9.4 %)
%N <sub>2000</sub>	a (37.8 %)	a, b (32.2 %)	c (21.7 %)	c, d (20.0 %)	b, d (25.4 %)

Cell categories with the same letter do not present statistically significant differences based on multiple post hoc comparisons with Kruskal–Wallis ANOVA by rank tests. A Bonferroni correction for multiple comparisons ( $p < 0.005$ ) was applied.  $PCA_{ENV}$  F1 to  $PCA_{ENV}$  F4 are the four environmental factors with eigenvalues above 1 derived from a principal component analysis using the 19 WORLDCLIM bioclimatic variables and mean altitude and altitude range. The percentage of cell area with anthropic land use corresponds to Corine Land Cover 2006, while the percentage of anthropised cell area represents the change from natural to anthropic land uses considering Corine Land Cover data for 1990 and 2006 (mean percentages in brackets). %PA and %N<sub>2000</sub> are the percentages of cell area covered by current protected areas and the Natura 2000 network, respectively

Additionally, roughly 2 % of all Spanish UTM cells (Table 1) can be considered to be rich “island” cells ( $R_I$ ), harbouring 29.6 % of the total considered species.  $R_I$  cells are scattered across the margins of the north and south plateaus and the Ebro valley (Fig. 1) and are not statistically different from an environmental perspective from those considered as cold spots or even from those cells without threatened invertebrate species or with non-significant local autocorrelation values (Table 1). The percentage of anthropic land uses in  $R_I$  cells (mean = 56.6 %; minimum = 1 %, maximum = 99.4 %;  $\pm 95$  % confidence interval = 5.0 %), although significantly lower than that in cold spot cells (mean = 65.4), is not significantly different from the percentages experienced by the poor “island” cells or the remaining cells without significant local autocorrelation values (Table 1). Approximately 35 % of  $R_I$  cells possess more than 75 % of their area anthropised, and in 59 % of these cells at least half of the area is currently anthropised. The percentage of current protected area (mean = 13.7 %; minimum = 0 %, maximum = 100 %;  $\pm 95$  % confidence interval = 5.0 %) follows the same pattern;  $R_I$  cells do not exhibit statistically different percentages of protected area compared to poor “islands” or the remaining Spanish cells without significant local autocorrelation values. Almost 70 % of  $R_I$  cells have <2 % of their areas currently protected, and almost 47 % of these cells possess <15 % of their area protected under the Nature 2000 network.

## Discussion

Although the number of species included in the Spanish atlases of continental threatened invertebrates (Verdú and Galante, 2008) constitutes a modest representation of the true richness of invertebrates of this country (Ramos et al. 2001), the taxonomic and survey effort provided by this collaborative project can allow us to discriminate important areas for the future conservation of these species. The use of invertebrates in conservation planning should be prioritised (Leather et al. 2008; Zamin et al. 2010), though information on these species has rarely been used for delimitation of candidate Spanish reserves (Martín-Piera 2001; Romo et al. 2007; Sánchez-Fernández et al. 2008), which are frequently based on plant and vertebrate data (Araújo et al. 2007).

Delimitation of protected areas to conserve biological diversity, although delicate and risky, can be considered one of the most important decisions involved in systematic conservation planning. When an entire territory is not integrally and sustainably managed, selection of reserves may imply the abandonment of a large portion of the territory to resource exploitation by human actions and free market opportunism. Reserve selection should be based on scientific and empirical data and clear, reliable and testable methods that are able to represent biological diversity. For this purpose, the use of scientific criteria based on species or habitat occurrences has become a standard, although it has rarely been put into practice (Knight et al. 2008). This is the case of Europe, where only 26 % of forest protected areas (from a total of 101 different protected areas) have been selected according to complementarity, spatial design and site suitability criteria, and therefore, from a network viewpoint, biodiversity conservation is far from guaranteed over the long term (Branquart et al. 2008). However, site selection based on complementarity or irreplaceability concepts does not take into account the biodiversity values of the neighbouring selected sites and this deficiency may decrease species persistence probabilities (Cabeza and Moilanen 2003). As one of the main goals of these standard approaches is avoidance of redundancy, they frequently select a minimum set of protected sites evenly distributed across the considered region in order to maximise compositional differences. Furthermore,



these procedures allow us to optimise conservation resources, but their implementation generally produces ties that may be solved by incorporating other variables, such as land holdings, accessibility, species characteristics or costs of protection (Sarkar et al. 2006). In this regard, incorporation of local autocorrelation statistics may complement decisions directed at the selection of protected areas based on complementarity or irreplaceability by recognising both interconnected and isolated sites that are relevant from a biodiversity perspective. Spatial delimitation of *hot spots* and rich “islands” and examination of their comparative magnitude in a region may assist complementarity based decisions in making an appropriate choice when the biodiversity benefit of including an additional reserve is similar among different sites. In these circumstances, *hot spot* localities detected by using local autocorrelation analysis may be privileged because the local persistence of species can be favoured in a connected region (Briers 2002; Carroll et al. 2003). However, when *hot spot* areas are not at risk, it is probably more convenient to select threatened areas, such as rich “islands” to diminish their vulnerability. Nevertheless, local autocorrelation statistics could represent an especially useful tool when biodiversity conservation strategies aim to favouring species persistence. Local autocorrelation results may provide guidelines for sustainable and strategic territorial planning aimed to achieve sustainable socio-economic development safeguarding natural resources and improving human quality of life (Costanza and Daly 1992). Under this scenario, rich “islands” would be highly protected areas that, in our case, should be treated as micro-reserves (Laguna et al. 2004); while *hot spots* could be priority areas subjected to sustainable management in accord with conservation measures for protected areas and the agricultural practices of surrounding areas.

Our results confirm that current Spanish protected areas and even the Nature 2000 network do not adequately represent threatened terrestrial invertebrates (Verdú et al. 2011, Hernández-Manrique et al. 2012). Although most of these species inhabit in *hot spot* cells, which present a comparative significantly lower proportion of anthropised area and a higher percentage of protected area, only 35 % of these cells have more than 2 % of their area currently protected, and more than half of these *hot spot* cells would have <15% of their area protected. The wider and still not completely implemented Natura 2000 network of protected areas clearly improves this situation, but even under this scenario, more than a third of *hot spot* cells would have <15 % of their area protected. In the case of rich “island” cells, the situation is even worse; in more than 58 % of these cells, at least half of the area is anthropised, while in 35 % of them, the proportion of anthropised area is greater than 75 %, and in 70 % of these cells, <2 % of the area is protected. These, results highlight the inadequacy of current and foreseen area reserves to represent Spanish endangered invertebrates.

The particularity of our study group (threatened species) and how this factor could be affecting our results are worth noting here. In this regard, it is probable that the richest areas are associated with some human influence simply because if this were not the case, such species would be not considered threatened. This effect should be more important with respect to the % anthropic land use than the % anthropisation. In any case, as occurs at a global extent (Lenzen et al. 2009), our results suggest that land use changes due to human activities are the main cause of threats to Iberian biodiversity as well as the origin of the observed autocorrelation patterns. The poor “island” cells detected, which represent approximately half the number of *hot spot* cells, are environmentally indistinguishable from *hot spots* but characterised by a significantly higher degree of human alteration. Furthermore, the detected cold spots correspond to the two Iberian plateaus that have been subjected to the most intense and secular human disturbance since Mid-to Late Holocene times (Carrión et al. 2010), especially during the last 2500 years (Franco-Múgica et al.



2005). Thus, if Iberian *hot spots* were partially located in areas subjected to less intense anthropogenic action, the rich “island” cells would correspond to relatively unaltered but isolated localities included within a severely transformed regional matrix that reflects a severe fragmentation process during the last centuries. Conservation efforts in Europe should frequently be oriented towards seminatural areas that are traditionally managed (Pullin et al. 2009), and the present results emphasise that important areas for threatened invertebrates are currently relatively anthropised. Unfortunately, the rates of land use changes occurring during the period from 1990 to 2006 are similar in all of the considered categories (even those categories having statistically different percentages of protected area), which is a result demonstrating the vulnerability of the localities harbouring these threatened species, even during this period in which the social relevance of protected areas seems to be highest. Consequently, existing conservation strategies based primarily on the protection of certain areas and vertebrate species, are clearly insufficient to ensure the conservation of invertebrate species, and the conservation of Spanish invertebrates requires sustainable, alternative strategies.

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