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Extinction trends of threatened invertebrates in peninsular Spain

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Abstract Using information from two recently published atlases of threatened invertebrate species in peninsular Spain, we examined the climatic, land use and geographic characteristics of the 100 km^2 UTM cells with most likelihood of suffering extinctions (extinction cells), as well as the attributes of the species prone to population extinctions. Extinction cells have had significantly (1) lower precipitation values, (2) higher temperatures, (3) higher percentages of anthropic land uses or (4) higher rates of anthropization during the last 20 years than the remaining cells. Nevertheless, probable extinctions may occur under a wide range of climatic and anthropization change rates and these variables can only explain a low proportion ($\sim$5 %) of variability in the occurrence or number of extinction cells. Aquatic species seem to suffer higher local extinction rates than terrestrial species. Interestingly, many invertebrate species with approximately 25 or less occurrence cells are on a clear trajectory towards extinction. These results outline the difficulties and uncertainties in relating probable population extinctions with climatic and land use changes in the case of invertebrate data. However, they also suggest that a third of the considered Spanish threatened species could have lost some of their populations, and that current conservation efforts are insufficient to reverse this tendency.

Keywords Extinction · Threatened species · Arthropods · Molluscs · Species decline · Spain

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Introduction

Extinction is a natural process, but there is growing awareness that it is occurring at an unnaturally rapid rate as a consequence of human activities (Pimm and Raven 2000; Pimm et al. 2006; Laurence 2010). What is more, the vast majority of extinctions could remain ignored because they probably occur within small, greatly neglected organisms (Dunn 2005; Cardoso et al. 2010, 2011). This alarming process is a key issue in conservation biology as it involves an irreversible loss of biological information, with unpredictable consequences (Kerr and Currie 1995; Purvis and Hector 2000; Rao and Larsen 2010). In spite of the importance of estimating extinction rates, there is an important lack of information about trends and causes of extinction of endangered species, especially invertebrates (Dunn 2005). In this sense, although invertebrates constitute approximately 75 % of described global biodiversity in terms of species numbers (IUCN 2010), their conservation is still a long-neglected issue (Samways 2005; Carpaneto et al. 2007; Cardoso et al. 2011). As an example, only 53 % of countries with National Red Lists have assessed taxa within this species group (Zamin et al. 2010) having been recently proposed an adaptation of the IUCN red list criteria in the case of invertebrates (Cardoso et al. 2011, 2012).

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To date, the pool of existing criteria for assessing extinction risks are reduced to the assessment of range size (area of occupancy and extent of occurrence), while the measures of continuing decline are sometimes subjective, and, as a consequence, categorizations made by different authorities differ and may not accurately reflect actual extinction risks (Mace et al. 2008; IUCN 2010). Thus, monitoring of any recent decline in population size or geographical range of a species must be one of the first steps to understanding the mechanisms that promote extinction (Lubchenco et al. 1991; Daily and Ehrlich 1995; Daily et al. 2001).

Entomological knowledge of extinction rates is a long way behind that achieved for assessments of change in vertebrates and plants (see Pimm and Raven 2000; MacKenzie et al. 2003; Jones et al. 2004; Stuart et al. 2004; Berglund and Jonsson 2005; IUCN 2010 and references therein). However, as has been mentioned above, reasonable evidence of population and geographical decline of invertebrate species is elusive because of our ignorance of their life histories and distributions (Moore 1991; Gaston and Mound 1993; Diniz-Filho et al. 2010). Most documented invertebrate extinctions are from well-studied taxa in well-studied regions (Mawdsley and Stork 1995), simply because these are the species whose absences we are capable of noticing (Dunn 2005). Information about invertebrates is scarce and efforts have been directed mainly towards butterflies, bees, dragonflies and beetles of North America and north and central Europe (see, for example, Thomas 1986; Turin and den Boer 1988; Desender and Turin 1989; van Swaay 1990; Cox and Elmqvist 2000; Lobo 2001; Thomas 2005; Sárospataki et al. 2005; Carpaneto et al. 2007; Stokstad 2007). This assessment could be especially interesting in areas of high diversity, such as Spain, which still harbours about 50 % of European plant and terrestrial vertebrate species and more than 30 % of European endemic species (Medail and Quezel 1997; Araújo et al. 2007; Reyjol et al. 2007), although human activities have been going on for centuries.

In this study we used two recently published atlases of threatened invertebrate species of Spain (Verdú and Galante 2008; Verdú et al. 2011), which provide information not only about current geographical distributions of threatened invertebrates in this country, but also on the localities (in our case Universal Transverse Mercator or UTM 100 km^2 coordinate system cells) where a species has probably become extinct. This exhaustive information allowed us to undertake one of the most comprehensive analyses to date on the extinction of threatened invertebrate species in Europe and specifically in the Mediterranean countries. We aim here to study extinction trends of threatened invertebrates in Spain, trying to assess their probable causes. Specifically, we aim (1) to identify the spatial distribution of the localities (UTM 100 km^2 cells) that have suffered possible extinctions, (2) to examine whether these probable extinction cells are located in singular geographical and environmental conditions, (3) to estimate if the changes in the climatic conditions (1970–2007) and/or land use (1990–2006) are associated with the occurrence of probable extinction cells, (4) to identify some easily measurable traits of species that are at major risk and lastly, (5) to asses if protected areas are effective in preventing these probable extinctions.

Materials and methods

Data origin

Biological data came from Spanish atlases for threatened invertebrates: endangered and critically endangered species (Verdú and Galante 2008), as well as vulnerable species (Verdú et al. 2011). These atlases compiled all the information on the distribution of the considered species in Spain (Peninsular Spain, Balearic and Canary Islands) that was available from several sources such as literature, scientific collections, as well as new data obtained from specific fieldwork conducted for the elaboration of these atlases. In this study, we only used the information that came from peninsular Spain, and excluded island data, due to its special historical, environmental and geographical conditions. A database was elaborated with all this information, which finally included distribution data for 220 species: 16 of which are critically endangered (CR), 47 endangered (EN) and 157 vulnerable (VU), corresponding to 7, 37, and 96 species of arthropods and 9, 10, and 61 species of molluscs, respectively. In summary, we obtained information about the presence or absence of these species in 2011 UTM cells with a resolution of 100 km^2 from mainland Spain.

The data used came from exhaustive fieldwork carried out by 150 taxonomists during 13 years (from 1997 to 2010). From 2006 to 2010 these experts visited the localities in which previous occurrences were known according to bibliography, natural history collections and taxonomist experience (Verdú et al. 2011). Once surveying was over, experts were asked to qualify the localities visited according to the current probability of species presence. They assigned a score of zero to localities with a high probability of local extinction due to habitat disappearance, habitat alteration and/or continued absences after repeated surveys. When the different surveys carried out by the experts did not allow us to confirm the occurrence of a species in a locality, we assumed that this population had more probabilities of becoming locally extinct, or at least suffering a significant decline. For the sake of simplicity, the UTM cells in which the occurrence of a species was not confirmed will be denominated extinction cells from now on.

Climatic, land use and protected area data

Spatial and environmental variables were calculated for each UTM 100 km^2 cell in order to examine if the probable extinction cells were located under singular geographical and environmental conditions. Spatial variables correspond to the longitude and latitude of each cell centroid. Climatic data for each cell was obtained from Worldclim (version 1.3, http://www.worldclim.org; see Hijmans et al. 2005), which contains data on 19 bioclimatic variables at a spatial resolution of 30 arc-s (c. 1 × 1-km resolution) obtained by the interpolation of climate-station records from 1950 to 2000. Using this information we extracted the average values of five variables for each UTM cell capable of reflecting the main variation in climatic conditions: annual mean temperature, maximum temperature of the warmest month, minimum temperature of the coldest month, annual precipitation and precipitation of the driest month. The mean altitude of each UTM cell was also considered and was extracted using a Digital Elevation Model (Clark-Labs 2000).

We used interpolated climatic maps representing annual average values for two decades (1970–1979 and 1998–2007), downloaded from the SECAD Geonetwork (http://secad.unex.es/portal/). Using raw data provided by the Meteorological State Agency of Spain (see www.aemet.es/) this platform allowed us to obtain the following climatic data: July maximum temperature, February minimum temperature, July precipitation and October precipitation. These climate maps were produced in two stages. First, monthly maps were generated by means of a kriging interpolation procedure using meteorological station raw data. Second, empirical local gradients were calculated for both monthly maximum and minimum temperatures by means of moving windows over the study area. The gradients were selectively added to the maps in areas where there were no mountain meteorological stations, thus representing mountain climates more accurately, as their temperatures were strongly overestimated in the kriging maps due to lack of meteorological data. The monthly maps were combined later to generate new maps with longer periods for use in this study. We calculated the differences in these four variables between the two decades in order to estimate whether the change in the climatic conditions was associated with the occurrence of probable extinction cells.

Land use variables were extracted in order to assess if the location of extinction cells was associated with the human transformation of the territory. We reclassified the 44 categories (label 3) established in the land use map provided by Corine Land Cover 2006 (see www.eea.europa.eu), obtaining two main types of land uses: anthropic (21 categories) and natural (23 categories), considering seminatural traditional low intensity land uses as natural areas. Next, we calculated the percentage of the area covered by these two basic land use types in each UTM cell. We also estimated the rates of anthropization and naturalization as percentages of each cell area that changed from natural to anthropic land uses and vice versa according to the Corine Land Cover maps of 1986 and 2006.

Lastly, we estimated the percentage of each UTM cell area covered by current protected areas (PAs) and Natura 2000 network (N_{2000}) obtained from www.redeuroparc.org (Europarc 2008), with the purpose of examining if the occurrence of extinction cells was associated with lack of protected areas.

Different species attributes were also delimited to assess if they were associated with the variation in the level of population extinction across species. All species considered were assigned to three binary qualitative variables representing the high level taxonomic group to which each species belonged (Arthropoda and Mollusca), their IUCN category ("endangered/critically-endangered" and "vulnerable") and the general ecosystem type in which they inhabited (aquatic or terrestrial). After calculating the percentage of extinction cells over the total number of occurrence cells for each species, we examined whether these values differed significantly between the two states of each of these three characteristics. Two additional quantitative variables, reflecting the body length of the species (body size) and the total number of UTM cells with occurrence data for each species (a surrogate for distribution range size) were also used to estimate its relationships with the species percentage of extinction cells.

Nonparametric Mann–Whitney *U*-tests were used to estimate whether the values of all the formerly mentioned variables differed significantly between extinction cells and remaining cells with threatened invertebrate information, as well as whether percentages of extinction cells differ for each one of the binary species characteristics. We also provide mean values with a 95 % confidence interval. Spearman rank correlation coefficients were used in the case of continuous variables (distribution range size and body size). Generalized Linear Models were additionally used to estimate the explanatory capacity of the variables selected to measure the climatic and land use changes on the occurrence of extinction cells (binomial distribution and a logit link function), as well as on the variation in the number of probable extinctions in the cells (Poisson distribution and a log link function).

Results

Spatial and environmental patterns

Approximately 39 % (2,107 squares) of the total cells considered include records of threatened invertebrate species. A total of 167 of these cells (around 8 %) were identified as extinction cells.

Both longitude and latitude values differed significantly between the extinction cells and all remaining cells with threatened invertebrate records (Table 1), so extinctions tended to occur in southern and eastern locations (Fig. 1). This spatial structure had an environmental correspondence as extinction cells had significantly higher temperature and lower precipitation values (Table 1). As expected, the percentage of current anthropic land use was also significantly higher in the extinction cells.

Temporal changes in climate and land use

Extinction cells seemed to have experienced a significant increase in summer temperatures (around 0.5 °C) over the last 30 years, but a lower decrease in winter temperatures (Table 1). While summer precipitation changes did not significantly differ between extinction cells and the remaining cells with invertebrate threatened data (all the considered area have experienced an aridification process), the increase in autumn precipitations was significantly lower in the considered extinction cells (Table 1). With regard to land use changes, the rate of naturalization in extinction cells was not significantly different from that found in the remaining cells, but the extinction cells would have experienced a significantly higher rate of anthropization (Table 1).

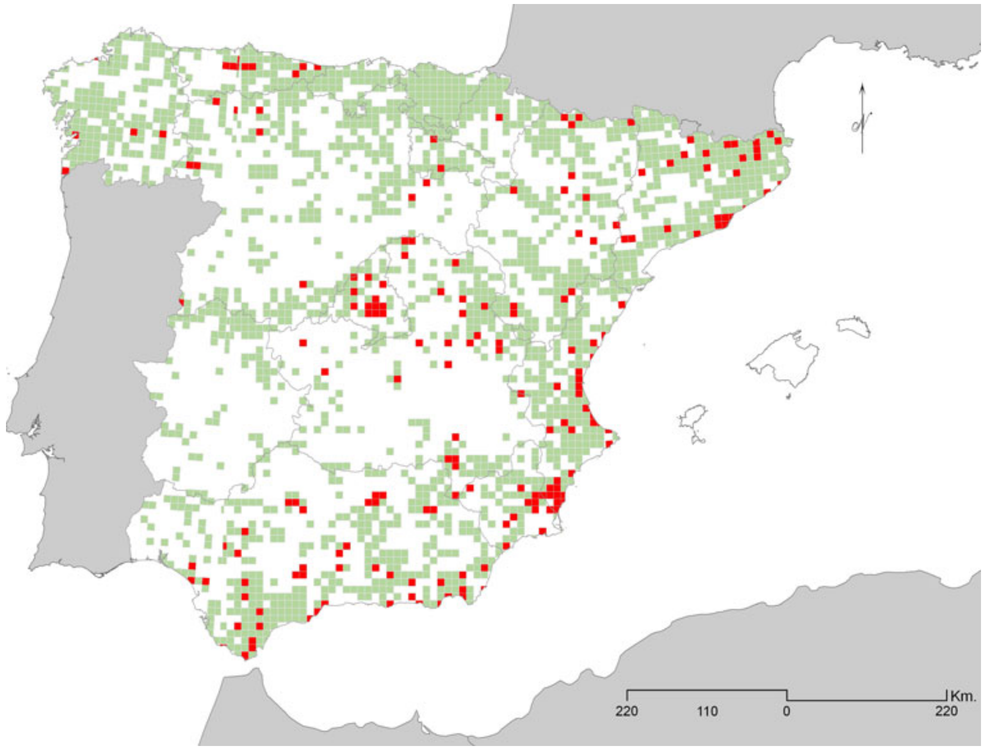
Each of the four variables considered to measure climatic changes in the two decades (1970–1979 and 1998–2007) were hardly able to explain 1 % of deviance in the occurrence of extinction cells, while a full model that incorporated all these climatic variables together only accounted for 1.6 % of total deviance. The change in the anthropization rate was not a variable with great explanatory capacity (1.0 %) either, but the addition of the current percentage of anthropic land uses allowed us to explain 5 % of total deviance. Similar results were found if the number of probable extinctions was used as a dependent variable; although in this case the current percentage of anthropic land uses accounted for 5.6 % of total variability. As we observed, probable extinctions could have occurred under a wide range of temperature changes and anthropization rates (Fig. 2).

Table 1 Results of Mann–Whitney *U* tests (U) comparing the values of the different considered variables between extinction cells (EC) and the remaining cells with information on threatened invertebrate species (RC), and the percentages of extinction cells for each one of the binary species characteristics

	U	r_s	EC	RC
Cell characteristics				
Longitude	135,245***		577 ± 38	505 ± 11
Latitude	121,986***		4,406 ± 35	4503 ± 10
Mean altitude	137,874***		589 ± 67	700 ± 20
Annual mean temperature	127,187***		13.9 ± 0.5	12.7 ± 0.2
Maximum temperature of the warmest month	131,596***		28.5 ± 0.5	27.4 ± 0.2
Minimum temperature of the coldest month	141,518***		2.4 ± 0.5	1.5 ± 0.1
Annual precipitation	126,319***		594 ± 40	696 ± 13
Precipitation of the driest month	141,812***		19 ± 2	25 ± 1
Anthropic land use	115,721***		43.2 ± 4.2	28.1 ± 1.2
Change in July maximum temperature	131,569***		0.53 ± 0.18	0.26 ± 0.05
Change in February minimum temperature	140,500**		0.03 ± 0.06	−0.12 ± 0.03
Change in July precipitation	$150{,}902^{ns}$		−5.9 ± 1.2	−5.0 ± 0.3
Change in October precipitation	133,359***		8.4 ± 4.2	17.0 ± 1.2
Rate of anthropization	133,375***		0.86 ± 0.30	0.43 ± 0.09
Rate of naturalization	$157{,}909^{ns}$		0.04 ± 0.23	0.23 ± 0.07
% PAs	$295{,}101^{ns}$		1.9 ± 1.6	1.7 ± 0.5
$\%N_{2000}$	$287{,}966^{ns}$		15.4 ± 3.9	16.9 ± 1.2
Species characteristics				
Taxonomic group	$5{,}360^{ns}$			
IUCN category	4,209*			
Ecosystem type	3,062***			
Distribution size		0.28***		
Body size		0.03^{ns}		

In the case of continuous variables Spearman rank correlation coefficients (r_s) were provided. Mean values (±95 % interval confidence) were also included for all variables. * $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$

Fig. 1 Probable extinction UTM cells (in *red*) of Peninsular Spain and all cells with information on threatened invertebrate species (in *green*) according to the data of (Verdú and Galante 2008; Verdú et al. 2011)



Species characteristics

The mean percentage of extinction cells by species was 8.4 % (mean ± 95 % confidence interval; ±2.4). We found that 72 species (~33 %) had suffered probable extinction in some cells, with 14 of these (~6 %) extinct in at least 50 % of their previously known cells (Table 2) and three [*Buprestis splendens* Fabricius, 1775; *Alzoniella galaica* (Boeters & Rolán, 1988) and *Lindenia tetraphylla* (Van der Linden, 1825)] can be considered as potentially extinct in peninsular Spain since none of their former populations can be confirmed at this moment.

The percentage of species extinction cells did not significantly differ between Arthropoda and Mollusca (mean ± 95 % confidence interval; 7.6 ± 3.1 and 9.8 ± 4.0, respectively). However, this percentage was significantly higher in the endangered species (14.7 ± 4.5) than in vulnerable ones (5.9 ± 2.8), and in aquatic species (16.8 ± 4.6) than in terrestrial (5.3 ± 2.7) ones (Table 1). The body size of species did not seem to be related with the proportion of extinction cells of species, but the total number of occurrence cells was clearly related (Table 1). Consequently, a clear trajectory towards extinction can be visualized depending on the detected range size of species (Fig. 3).

Protected sites

Lastly, the percentage of area included as PAs and N_{2000} did not differ significantly between extinction cells and the remaining cells with threatened species records (Table 1).

Discussion

The clear geographical pattern in the distribution of probable extinction cells shows that invertebrate species living in the south-eastern corner of the Spanish Mediterranean are probably the most threatened. This spatial structure has an environmental correspondence (as climate also shows a spatial structure). Thus, the probable extinction cells harbour significantly higher temperatures and lower precipitation values than the remaining cells with a presence of threatened invertebrates. These results suggest that some semiarid Mediterranean ecosystems such as sand dunes, saline aquatic environments and xerophilous scrublands may be specially inclined to suffer local extinctions or species declines. These particular and interesting habitats are being deeply altered and destroyed by human activities, mainly as a consequence of urbanisation, road constructions, recreation, cropping and the effects of grazing animals (van der Meulen and Salman 1996; Didham et al. 2007; Martínez et al. 2008; Millán et al. 2011).

According to our results, the probability of extinction of invertebrates in peninsular Spain would be slightly but significantly associated with changes in climatic and land use conditions. Localities experiencing an increase in their summer temperatures and with a comparative lower

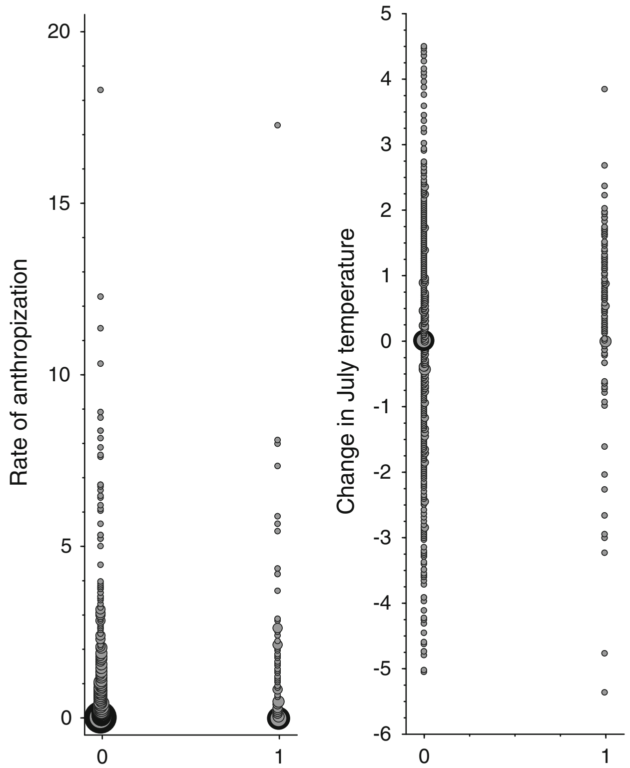


Fig. 2 Variations in the rate of anthropization and change in July temperature between probable extinction cells (1) and the remaining cells with information on threatened invertebrate species (0). Corine Land Cover maps of 1990 and 2006 were used to estimate the rate on anthropization, while temperature values represent the change in interpolated annual average values between the decade 1970–1979 and the decade 2000–2009 (see Sect. "Methods"). The size of the points is proportional to the number of observations

increase in autumn rainfall during the last 40 years may be the ones that have more probabilities of suffering invertebrate extinctions. However, the percentage of variability in the variation of extinctions that can be explained by means of these climatic changes is almost negligible, outlining the necessity of obtaining better empirical data capable of demonstrating the effect of recent climatic changes on species extinctions (Maclean and Wilson 2011). According to our results, the effect of the land use changes that have occurred during the last 20 years is also negligible (~2 %), being the percentage of current anthropic land use the variable with the highest explanatory capacity. In our opinion, these results may suggest that the probable effect of human actions on the decline or extinction of invertebrate populations should be considered on a larger temporal scale. Very recent land use changes may negatively influence endangered populations in the future (the so called "extinction debt"; see Kuussaari et al. 2009), because extinction should be viewed as a process that extends over time and whose consequences are manifested when we consider the total area transformed over time.

Previous studies show that habitat destruction is the leading cause of species extinction (Lawton and May 1995; Murcia 1995; McIntyre and Hobbs 1999; Brooks et al. 2002). Our results fail to detect an unequivocal sign of land use changes on extinctions, although they do suggest an association between the increase of anthropic land uses and the probability of invertebrate extinctions. Around 41 % of the considered extinction cells have not suffered any anthropization change, and 21 % have currently less than 10 % of their total area anthropized. Several methodological drawbacks could have conditioned this result. The comparison between the so-called extinction cells and the remaining ones could have been influenced by the particular considered species pool and the used resolution. Furthermore, the estimation of extinction is subject to survey bias, detectability errors and interannual variations that require a high survey effort in order to be accurate. It is worth noting here the particularity of our study group (threatened species) and how this factor could be affecting our results. In this sense, it is probable that the areas with presence of these species have some human influence simply because otherwise, they would not be considered as threatened. So, this could explain the greater importance of the percentage of current anthropic land use and the lack of relevance of the anthropization rate experienced during the last 20 years in explaining the deviance in the occurrence of extinction cells. Two localities with similar anthropization rates may differ in their extinctions depending on the previous impact of human actions; altered area populations may be more on the edge and any small impact might lead to their extinction. Uncertainties in the build of land use estimations and climatic interpolations may contribute to adding noise to the used primary data. Without playing down the influence of these sources of error, we are inclined to suspect that our general inability to find a clear effect of climate or land use changes can also be partially explained by other unconsidered processes related to the influence of demographic and genetic factors (Lande 1998), chemical pollution (Kendall et al. 2010) and the synergic effect of several factors on a specific population (Leidner and Neel 2011).

The estimated differences in local extinctions between species agree with the conservation status proposed previously by experts. Endangered and critically endangered species seem to have suffered significantly higher rates of local extinctions (~15 %) than vulnerable species (~6 %). Our results also indicate that aquatic species are more inclined towards extinction than terrestrial species. These results agree with previous studies showing that rates of biodiversity loss are greater in freshwater systems than in other ecosystems (Ricciardi and Rasmussen 1999; Saunders et al. 2002; Darwall and Vié 2005). Furthermore, human pressures on freshwater resources are likely to

Table 2 Species that have probably suffered extinction in at least 50 % of their previously known 100 km^2 UTM cells

Species	IUCN category	E	HD	%
Arthropoda				
Lindenia tetraphylla (Van der Linden, 1825)	CR			100.0
Buprestis splendens Fabricius, 1775	VU			100.0
Scarabaeus (Scarabaeus) pius (Illiger, 1803)	EN		X	71.4
Ocladius grandii Osella & Meregalli, 1986	VU		X	66.7
Meloe (Taphromeloe) foveolatus Guérin de Méneville, 1842	EN			50.0
Steropleurus politus (Bolívar, 1901)	VU			50.0
Molusca				
Alzoniella (Alzoniella) galaica (Boeters y Rolán, 1988)	CR	X		100.0
Vertigo (Vertigo) moulinsiana (Dupuy, 1849)	CR		X	71.4
Alzoniella (Alzoniella) asturica (Boeters y Rolán, 1988)	VU	X		66.7
Vertigo (Vertilla) angustior Jeffreys, 1830	CR		X	60.0
Islamia lagari (Altimira, 1960)	EN	X		50.0
Theodoxus baeticus (Lamarck, 1822)	EN	X		50.0
Theodoxus valentinus (Graëlls, 1846)	CR	X		50.0
Theodoxus velascoi (Graëlls, 1846)	CR	X		50.0

E: Endemic species of peninsular Spain; HD: Species included in the annexes (II, IV) of the Habitats Directive; %: percentage of extinction cells over total

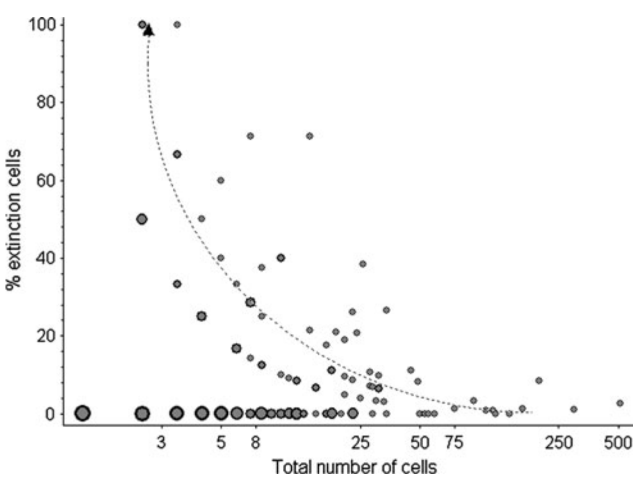


Fig. 3 Relationship between the total number of UTM cells with occurrences for each one of the considered threatened species and the percentage of extinction cells over the total. The *arrow* tries to represent the trajectory to extinction suffered by the invertebrate species as their geographical rarity increases

increase in the coming decades, putting yet more species at risk (Strayer 2006). The most severe recognized threats to freshwater species is habitat loss, followed by pollution and invasive species (Bräutigam and Jenkins 2001; Sánchez-Fernández et al. 2008; Williams 2009). Spain has lost more than 60 % of all inland freshwater wetlands since 1970; freshwater quality and quantity are both threatened by human activities that cause pressures on the environment, including urbanization, tourism, industry and agriculture (UNEP and DEWA 2004). In this sense, it is necessary to apply specific management measures to protect this aquatic biota even when these species are present in protected areas (Herbert et al. 2010). Many activities, such as dam building, water diversion, changes in land-use or introduction of alien species (Saunders et al. 2002), may occur outside park boundaries but have negative consequences for its freshwater habitats. Thus, the whole-catchment management and natural-flow maintenance are indispensable strategies for freshwater biodiversity conservation (Abellán et al. 2007).

On the other hand, whatever the process that generates extinction, its effects will depend on the size and degree of the genetic connectedness of populations (Moilanen and Nieminen 2002). As expected, we found that the proportion of extinction cells of species was clearly related with the total number of occurrence cells. This result may be suggesting that range extent could be a key factor to explain invertebrate extinction risk. Our study shows that some invertebrate species, with approximately 25 or less occurrence cells are on a clear trajectory towards extinction. However, 64 % of total species (140 species) occur in less than 25 cells and do not seem to follow this decline tendency highlighting the species specific character of extinction risks. Taking into account that having a narrow range is a key criterion for the selection of these endangered species, their decline cannot be considered a surprise (Lande 1993; Leidner and Neel 2011). Very small populations are susceptible to demographic stochasticity, whereby random variations in birth and death rates can lead to extinction even when the average population growth rate is positive (Goodman 1987).

A finding that is no less relevant is the lack of significant differences in supposed extinction cells between protected and unprotected areas. Both current Protected Reserves and the Natura 2000 network should theoretically provide an appropriate mechanism to minimize deterioration of natural habitats, so avoiding extinctions. Specifically, in the Habitats Directive, maintenance or restoration of natural habitats and populations of wild species of Community interest to a favourable conservation status is defined as an overall objective of conservation measures (Mehtälä and Vuorisalo 2007). However, the current coverage of protected areas does not seem able to avoid these probable local extinctions, as they can occur indistinctly both within and outside protected areas. This result casts doubts on the viability of non-charismatic species, even inside protected areas and shows that the occurrence of a species within a protected area (even with multiple capture records) may not guarantee their long-term survival (see Hernández-Manrique et al. 2012).

Our rate of extinction is very limited but the vast majority of species belong to understudied taxa of invertebrates (Dunn 2005; González-Oreja 2008; Evans 1993). Our results showed that 36 % of the threatened species considered in this study (80 species) could have lost some of their populations, and that 17 of these could have become extinct in at least 50 % of their previously known populations. The causes of these probable extinctions are complex and climatic or land use changes cannot be recognized clearly as the main responsible factors. Thus, a precautionary principle should guide our conservation rules in the case of invertebrates by designing new micro-reserves, favouring the connectivity of populations, diminishing their isolation and, indeed, promoting the translocation of specimens. Interestingly, almost 30 % of threatened species considered in this study have been described during the last 30 years, and half of them (39 species) in the last 10 years. Our results indicate that many of the populations of these species could have disappeared in a short period of time after their discovery. In this sense, if we take into account that we are only considering a part of the most charismatic invertebrate species, it is also likely that these rates of extinction would be higher in the rest of inconspicuous, non-charismatic or even unknown invertebrates (Dunn 2005; González-Oreja 2008), which constitute the vast majority of biodiversity.

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