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Original article

Biodiversity and Land uses at a regional scale: Is agriculture the biggest threat for reptile assemblages?

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ABSTRACT

The human exploitation of land resources (land use) has been considered the major factor responsible for changes in biodiversity within terrestrial ecosystems given that it affects directly the distribution of the fauna. Reptiles are known to be particularly sensitive to habitat change due to their ecological constraints. Here, the impact of land use on reptile diversity was analysed, choosing Catalonia (NE Iberia) as a case study. This region provides a suitable scenario for such a biogeographical study since it harbours: 1) a rich reptile fauna; 2) a highly diverse environment showing strong variation in those variables usually shaping reptile distributions; and 3) good species distribution data. Potential species richness was calculated, using ecological modelling techniques (Ecological Niche Factor Analysis – ENFA). The subtraction of the observed from the potential species richness was the dependent variable in a backwards multiple linear regression, using land use variables. Agriculture was the land use with the strongest relation with the non-fulfilment of the potential species richness, indicating a trend towards a deficit of biodiversity. Deciduous forest was the only land use negatively related with the subtracted species richness. Results indicate a clear relationship between land use and biodiversity at a mesoscale. This finding represents an important baseline for conservation guidelines within the habitat change framework because it has been achieved at the same spatial scale of chorological studies and management policies.

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1. Introduction

The human exploitation of land resources (land use) is the major artificial disturbance of the structure and composition of the landscape, affecting directly the distribution of the fauna (e.g. Brotons et al., 2005) and, consequently, species richness (Waltert et al., 2004). Land use impacts directly the entire ecosystem from the properties of the soils (Dupouey

et al., 2002; Latty et al., 2004) and plant community (Verheyen et al., 2003; Schulte et al., 2007), to the structure of invertebrate (e.g. Eyre et al., 2003) and vertebrate fauna (Chamberlain and Fuller, 2000; Heikkinen et al., 2004; Waltert et al., 2004; Gagné and Fahrig, 2007). The dimension of this problem is enormous as over one-fifth of the land area of the globe has been converted to human-dominated uses (Hoekstra et al., 2005). Consequently, a large number of studies have focused their attention on this problem. However, most of the studies are centred on very specific and localised impacts, such as livestock grazing effects (Woinarski and Ash, 2002), impacts of organic farming (e.g. Bengtsson et al., 2005) and deforestation in tropical regions (e.g. Waltert et al., 2004). Therefore, understanding how different land uses may affect biodiversity patterns at a regional scale is an urgent conservation issue (Foley et al., 2005) since it allows determination of general trends. Land use change is considered the factor with the biggest effect on the loss of biodiversity, particularly in Mediterranean ecosystems (Sala et al., 2000). In the Mediterranean Basin anthropogenic activities and their agricultural economies have been impacting the natural environment since the Neolithic and the beginning of the Bronze Age (about 6000 years B.P.; Bottema et al., 1990). Such a long history of human impacts led to the Critical/Endangered conservation status of the "Mediterranean Shrubs" ecoregion due mostly to extensive habitat loss, fragmentation and degradation (Olson et al., 2001). Nevertheless, the Mediterranean Basin is still one of the world's biodiversity hotspots (Myers et al., 2000).

A particular study case is Catalonia (NE of the Iberian Peninsula), which offers a good opportunity for biogeographical studies since its frontiers comprise the Mediterranean and Atlantic biogeographical regions (Sillero, 2006). Although there are other transitional areas in the Mediterranean Basin, Catalonia is singular in suffering drastic habitat transformations due to different, and often opposing, forces (urbanisation, culture abandonment, intensive agriculture and forestry, fires) within small spatial scales and time periods (Lloret et al., 2002). Moreover, Catalonia harbours an extremely rich reptile fauna including almost 80% of the reptile species from the Iberian Peninsula within only slightly more than 5% of the peninsular territory (Llorente et al., 1995). Reptiles are recognised as extremely sensitive to local habitat changes (Jäggi and Baur, 1999; Woinarski and Ash, 2002; Anadón et al., 2006; Castellano and Valone, 2006) due to their ecologic and physiological constraints, low dispersal capacity and small home ranges (Huey, 1982). In fact, White et al. (1997) found that reptiles (together with amphibians) are more prone to the risks associated with landscape change than other vertebrate taxa.

The main goal of this study is to determine the impact of land use on the biodiversity pattern at a regional scale, using reptiles as model group and Catalonia as model area. In order to achieve this objective we have used modern GIS techniques and specific niche requirements to determine habitat suitability for each reptile species, excluding the land use variables. With this habitat suitability, expressed as probability of species occurrence, we used the ROC threshold method to establish the potential distribution of each species and then estimated the potential species richness. The incongruence between the potential and observed species richness was correlated with the main land use types present in Catalonia with the intention of determining the type(s) of land use constraining reptile biodiversity (measured as species richness).

2. Materials and methods

2.1. Observed Species Richness (OSR)

The Catalonian reptile fauna includes 32 autochthonous species (3 chelonians, 1 amphisbaenian, 17 saurians and 11 ophidians). However, 7 of these species present an extremely restricted distribution in Catalonia: 4 reach Catalonia on their global distribution limits and the other 3 correspond to endemic Pyrenean species with an extremely localized distribution. Considering UTM squares, none of these 7 species have ranges greater than 3% of the total squares existent in Catalonia. Consequently, our analysis focuses on the remaining 25 species (see Appendix 1).

Regarding herpetofauna, Catalonia can be considered exhaustively sampled and distribution databases are of high quality. The history of herpetology in Catalonia goes back about one and a half centuries (Llorente et al., 1995) and, therefore, it is expected that the coverage of the species sampling is currently near to saturation, at least at the 10 × 10 km UTM grid scale (Llorente and Montori, 2002). The first atlas is 25-years-old (Vives-Balmaña, 1982), and field sampling for the 1995 Catalonia atlas (Llorente et al., 1995) was addressed to cover the existing gaps, allowing an even coverage of the whole territory. We obtained the reptile distribution data from the Spanish Herpetological Association database for the Catalonia region, including the 1995 Reptile and Amphibian Catalan Atlas (Llorente et al., 1995), the Spanish Herpetological Atlas (Pleguezuelos et al., 2002) and further updates yielding 12,170 records. Significantly, no major changes in the species distribution patterns between 1995 and 2002 were observed indicating the reliability of the distribution data. We inserted the records, represented on the 10 × 10 km Universal Transverse Mercator (UTM) grid (384 squares in Catalonia), in a georeferenced database and displayed it using ArcMap 9.2 GIS (ESRI, Redlands, California, USA). Through the addition of all species present in each 10 × 10 km UTM quadrate, we obtained the Observed Species Richness (hereafter OSR).

2.2. Potential Species Richness (PSR)

We used niche-based models to predict the potential geographic distributions of the species based on climatic and geological variables. The ecogeographical variables (EGV) used to model the distribution (9 climatic, 4 geological and altitude: Appendix 2) were selected based on their significance to the distribution of reptile fauna (Hadden and Westbrooke, 1996; Guisan and Hofer, 2003; Rodríguez et al., 2005). Since geological variables were too complex for the purpose of this analysis, some minor categories were clustered. The spatial resolution of the study is constrained by the data with the coarsest scale (10 × 10 km squares) – the distributional species data. In order to standardise all the EGVs, two procedures

were applied using ArcMap 9.2 GIS (ESRI, Redlands, California, USA): 1) aggregation for the climatic variables, i.e. the medium value of the variable was obtained for each 10 × 10 km UTM squares; and 2) percentage of the total area occupied by each type of geological variable within each 10 km square in relation to the total area of the square, obtaining an independent variable for each type. The land use variables will be considered in the last stage of the analysis (see Section 4).

To create the niche-based models, we selected a modelling technique that uses only presence data: the Ecological-Niche Factor Analysis or ENFA (Hirzel et al., 2002). ENFA is implemented in the Biomapper 3.1 software (Hirzel et al., 2004), and the procedures outlined by Hirzel et al. (2002) were followed. ENFA was already successfully applied for modelling potential distribution with Iberian herpetofauna (e.g. Soares and Brito, 2007; Santos et al., 2006; Sillero, 2006). The ENFA analysis calculates habitat suitability maps by comparing the distributions of eco-geographical variables (EGV) with the distribution of the species. These maps express the suitability of habitat in values ranging from 0 to 1 for every 10 × 10 km square (Hirzel et al., 2002).

The resulting habitat suitability grid maps were converted into presence/absence maps through the ROC (Receiver Operating Characteristic) technique (Zweig and Campbell, 1993) as suggested by Hirzel et al. (2002), using the SPSS version 15.0 for Windows (SPSS Inc, Chicago, IL). A potential species richness (hereafter PSR) was then calculated by the addition of all the presence/absence maps.

2.3. Subtracted Species Richness (SSR)

The observed species richness was subtracted from the potential one resulting in a subtracted species richness (PSR − OSR = SSR), which is the keystone variable of this study. High values of SSR (i.e. a number of potential species superior to the value of observed ones indicating a possible deficit of biodiversity) are expected to give information about the areas where the species richness is being constrained by factors different from the EGVs. On the other hand, low values of SSR, i.e. observed species richness similar to the potential one indicates fulfilment of potential species richness.

2.4. Correlation of land use variables with Subtracted Species Richness

The SSR was correlated with variables of land use (Appendix 3), using a backwards multiple linear regression, implemented in SPSS. The land use variables were transformed in the same way the geological variables were (percentage of each kind of land use per 10 km square). In order to prevent a border effect, the percentages of each land use type were calculated only with the usable area of Catalonia, excluding the area of the sea and administrative territories (namely, France, Andorra and Aragon). Furthermore, all 10 km squares that included land areas less than 50% of the total 100 km^2 (65 squares) were not included in this methodological step (e.g. coastal 10 km squares with more than 50% of sea area). Because the land use variables were expressed as the percent of each land use type

in each 10 square, these variables were transformed by means of an arcsine function applied on the square-root, to achieve normality (Zar, 1999).

2.5. Comparative analysis of the areas with different values of diversity (deficit, neutral and surplus)

We compared the maps of observed distribution with the potential distribution of each species to better understand the patterns of diversity obtained in the latter procedure. The objective was to identify the contribution of each species to the different values of diversity allowing a more precise assessment of the diversity patterns. We performed a chi-square test comparing the number of squares with observed presence of a species versus the number of squares with predicted presence. The test was performed for each species separately comparing the areas of deficit values of diversity (SSR > 0) with areas of neutral values (SSR ≡ 0). The same procedure was repeated for the areas of surplus values of diversity (SSR<0). Although every species was individually evaluated, corrections for multiple tests, namely sequential Bonferroni (Rice, 1989) and False Discovery Rate (FDS; Benjamini and Hochberg, 1995) procedures were applied. The second one is usually considered more accurate than the first because it does not inflate the Type II error rate (Moran, 2003).

3. Results

The observed (a) and potential (b) species richness values are graphically represented in Fig. 1. In both maps an east–west decreasing pattern in the species richness can be observed, with extreme north-eastern Catalonia being the area with the highest values (located in the Atlantic-Mediterranean transition).

Subtracted Species Richness ranged from −10 to 15 species (Fig. 2). As previously commented, negative values indicate those 10 km squares where the observed species richness (OSR) is greater than the potential species richness (PSR) – a surplus of biodiversity. In contrast, positive values appeared in those squares where the potential distribution models estimated more species than that observed – a deficit of biodiversity.

Urbanised land use was excluded from the final model of the multiple linear regression analysis. Deciduous forest land use was the only independent variable showing a negative relationship with SSR (Table 1). For all the other variables, the model resulting from the multiple linear regression indicated a positive relationship. Since the dependent variable is a subtraction (subtracted species richness – SSR) the interpretation of these results is not intuitive. Positive relationships of the SSR with one land use variable indicate that 10 km squares with high values of SSR (i.e. deficit of biodiversity), also present high percentages of that land use; thus its effect on the reptile diversity is negative. Agriculture is the land use variable with the deepest negative effect (Table 1) followed by degraded forest, pine and holm oak forests. On the other hand, a negative relationship means that the same 10 km squares join low values of SSR (surplus of biodiversity) and high percentages of land uses, which could be indicative of the
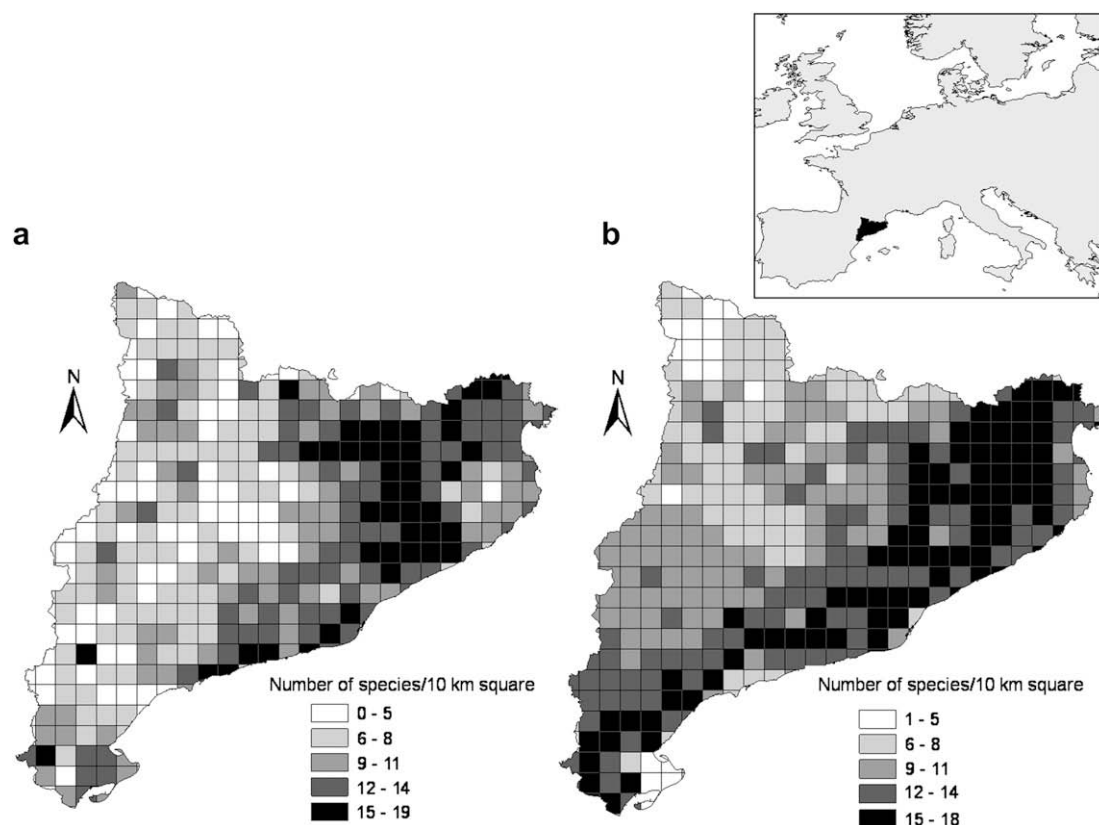
Fig. 1 – Location of Catalonia in the European context and Species Richness maps: a) Observed Species Richness and b) Potential Species Richness derived from the Ecological Niche Factor Analysis (ENFA) models.

land uses allowing the fulfilment of the potential "natural" species richness. Deciduous forest was the only land use variable presenting this relationship.

The results of the chi-square test are presented in Appendix 4. The species that contributed most to the biodiversity surplus, in order of significance (unadjusted), were *Psammodromus algirus*, *Timon lepidus*, *Rhinechis scalaris*, *Mauremys leprosa*, *Tarentola mauritanica*, *Malpolon monspessulanus* and *Emys orbicularis*. These species fall in two groups: 1) those that depend on habitat features at a microhabitat scale (the two terrapins – *M. leprosa* and *E. orbicularis* – and the gecko *T. mauritanica*); and 2) widespread Mediterranean reptiles (*P. algirus*, *T. lepidus*, *R. scalaris* and *M. monspessulanus*).

The species that contributed most to the biodiversity deficit, in order of significance (unadjusted), were *M. leprosa*, *Vipera latastei*, *Coronella girondica*, *Psammodromus hispanicus*, *Hemidactylus turcicus*, *T. mauritanica*, *Natrix natrix*, *Lacerta bilineata*, *Natrix maura*, *Zamenis longissimus*, *E. orbicularis*, *Chalcides striatus*, *T. lepidus* and *Anguis fragilis*. This is a heterogeneous assemblage, but their contribution to the biodiversity deficit cannot be attributed to their rarity (except maybe in *V. latastei*).

4. Discussion

In spite of the intrinsic limitations posed by the coarse resolution scale of the distribution data and the aggregation of land uses, our results still provide a good idea of the response of the reptile biodiversity to land use at a regional scale. We have quantified how the observed and the potential reptile species richness differ in a Mediterranean area and examined how land uses can explain these differences. At the regional scale the distribution of the reptile species is mainly explained by climatic factors (Guisan and Hofer, 2003; Rodríguez et al., 2005; Anadón et al., 2006). Potential species richness obtained by modelling the species distributions based mainly on climatic variables should represent species richness free of human influence. This seems a rather strong assumption, since there are other factors (e.g. historical, biotic interactions) constraining the climatic potential distribution of any species (Huston, 1994). Nevertheless, this problem does not represent a real drawback considering our study scale (about 32,000 km^2) and our resolution, 10 × 10 km squares.

The multiple linear regression analysis revealed a positive relationship of the SSR with agriculture, degraded forest and pine and holm oak forest and a negative relationship with deciduous forest. The agricultural land use is the variable most strongly related with the diversity deficit (i.e., with the deepest impact on the non-fulfilment of the PSR). Consequently, this factor could be restraining the distribution of some reptiles in Catalonia and affecting negatively the region's diversity. In fact, Spain has been considered one the European regions considered most vulnerable to biodiversity loss due to agricultural practices (Reidsma et al., 2006). The large land area occupied by agriculture and the high degree of physical and chemical manipulation made this land use one
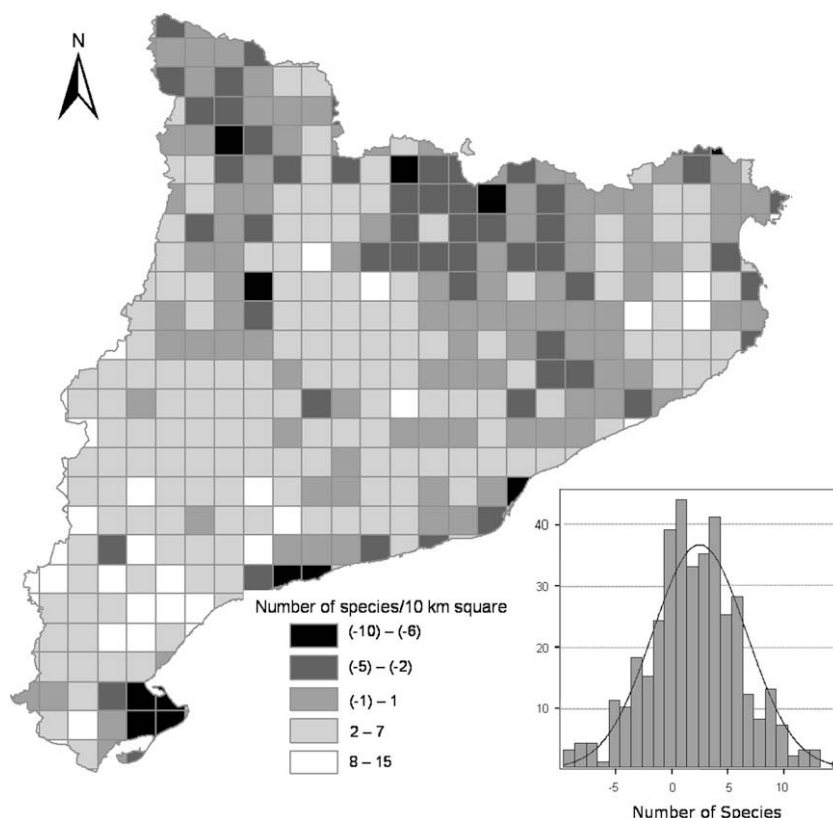
Fig. 2 – Geographical representation and frequency histogram of the subtraction between Potential (Fig. 1b) and Observed Species Richness (Fig. 1a) – Subtracted Species Richness. The classification of the grey categories represented on the map was done manually: the first category assigned was the [−1, 1] and intended to represent the area with neutral values with one species error; the other categories were assigned by roughly dividing the remaining values [−10, −2] and [2,15] in two categories.

of the largest contributors to the loss of biodiversity world-wide (McLaughlin and Mineau, 1995). Nevertheless, this association is catalysed by the intensity factor (Reidsma et al., 2006; Paggetti et al., 2006; Hendrickx et al., 2007) being the non-intensive farming considered indispensable to biodiversity conservation (Bignal and McCracken, 1996). The number of studies on the responses of the biodiversity to the effects of agricultural intensification is vast but largely taxa-biased, birds being the most studied vertebrate group (e.g. Donald et al., 2001; Robinson and Sutherland, 2002; Wretenberg et al., 2006). However, studies of reptiles report the same trend we

found here, although mainly focused on the species level (Biaggini et al., 2006; Graziani et al., 2006; Anadón et al., 2007; Wisler et al., 2008). Degraded forest, also tending towards a deficit of diversity, is a complex variable because it includes all the forest land use which has lost most of its tree strata. Therefore, the established relationship is less informative since the origins for lacking tree strata are vast including wild-fire and deforestation, both phenomena recognised as preju-dicial for the reptile fauna. (e.g. Greenberg et al., 1994; Findlay and Houlahan, 1997). Two types of forest also showed positive relation with the SSR: pine and holm oak. Densely forested areas have a negative effect on reptiles (Ballinger and Watts, 1995; Jäggi and Baur, 1999) because closed canopy forest could reduce both prey abundance and availability of basking sites (Webb et al., 2005). During the 20th century, the forest area in Catalonia increased more than 130%, from 600,000 ha to 1,400,000 ha (Peix i Massip, 1999). In the medium and low altitude areas, forests are mainly composed of intensive pine plantations, resulting in a vast extension of highly transformed and oversimplified landscapes. The massive spreading of the forest land use, due either to natural succession of abandon rural land or to intensification of forest exploitation, could be responsible for the reptile diversity deficit as the artificial alteration of the forest has a negative effect on the Iberian reptiles (Amo et al., 2007; Moreira and

Table 1 – Final model derived from Multiple Linear Regression analysis using the Subtracted Species Richness as dependent variable. Variables are rated in descendent order.

Independent Variable	Standardised Coefficients	Significance (p)
Agricultural	0.690	<0.001
Degraded Forest	0.366	<0.001
Pine Forest	0.252	<0.001
Holm Oak Forest	0.232	<0.001
Deciduous Forest	−0.167	0.005

Russo, 2007). However, the deciduous forest showed the inverse relationship, suggesting that this land use retains most of its original reptile diversity. In Catalonia, deciduous forests grow in humid regions where forest degradation is low; such regions are hilly and cool (but not cold or mountainous) thus providing sufficient niches not only for Atlantic species but also to Mediterranean ones. We tentatively attribute this high richness to this edge (and microscale) effect.

The only land use variable found non-significant by the multiple linear regression was urbanised land. The total urbanised land does not exceed 5% of the total area of Catalonia. In fact, about 89% of the 10 km squares (340) contain less than 10% of their area urbanised, suggesting that our resolution scale (10 × 10 km) is not adequate to establish a relationship between reptile diversity and urbanisation. Furthermore, the distribution of the urbanised land use is extremely biased towards the coast and, as mentioned above; many squares of this region were eliminated from the analysis to prevent a border effect.

The number of observed species was higher than predicted along the coast and in the Pyrenees. However, the first can be an artefact because most coastal squares comprised less land area that those inland, which could produce inaccuracies on the niche-based models. Regarding the Pyrenees, the biodiversity surplus could be due to a spatial scale problem inherent to this study: by constraints due to the database available, the study scale was 10 × 10 km. This resolution, a standard in chorological studies especially at a regional/national scale (e.g. Sindaco et al., 2006), might be too coarse to model the habitat suitability for some species. This might be particularly true in areas highly diverse environmentally, since the model was built based on average values of climatic and altitudinal variables for each 10 km square (see Llorente et al., 1995). This reclassification of the variables used on the calculation of the niche-based model led to a simplification of the environmental diversity turning the squares with extreme values into squares with median values. In fact, the region presenting the highest environmental diversity in Catalonia corresponds to the medium/low Pyrenees where the mountains that slope south and east present typical Mediterranean habitat, whereas those facing north and west harbour typical Atlantic habitat (Tenorio et al., 2001). In such cases a 10 km square comprising both habitats could include more observed species than predicted by the models. This situation matches perfectly with the species found to have a bigger contribution to the diversity surplus. On the one hand, species associated with microhabitat features are independent from the surrounding habitats (terrapins and geckoes). Such species are more dependent on very local factors (ponds and buildings, respectively) different from those used to build the models. On the other hand, there is another group of widespread Mediterranean species, all presenting a similar observed distribution with their northern range limit at the medium/low Pyrenees (Llorente et al., 1995). This region holds the highest environmental diversity providing conditions for the Mediterranean species, probably at the microhabitat scale, leading to negative values of SSR. Of the 25 reptile species analysed, 14 displayed observed/potential presences accounting for the pattern of diversity deficit (see Appendix 4). Interestingly, 3 out of the 4 species categorised of conservation concern in this study (Vulnerable and Near Threatened) by the current Spanish Red Book (Pleguezuelos et al., 2002) fell within this group.

5. Conclusions

Land use has detectable effects on the reptile diversity patterns at a regional scale, even using the coarse resolution of 10 × 10 km. This result contrasts with the study of Busack and Jaksic (1982) who failed to find any pattern of species richness distribution either for the whole Iberian Peninsula or for Cadiz province (S Spain). Probably, the intermediate scale of this study covering a wide variety of climates, habitats and agricultural intensities but with a common palaeographic history, together with a more reliable estimation of distributions through ecological modelling, have allowed us to detect these subtle differences in species richness. Nevertheless, the impact of land use is now well documented for other taxa, mainly invertebrates and birds (see McLaughlin and Mineau, 1995 for a review).

In Catalonia agricultural land use shows the greatest effect on the reptile diversity. Reconciling agriculture and conservation is possible if agricultural practices are developed in sustainable ways (Matson et al., 1997), namely, maintaining the heterogeneity of the landscapes with important relevance to the semi-natural habitat (Billeter et al., 2008). The impact of forests was also identified as negative, except the deciduous forest. Although in other regions, forest conservation policies are mainly addressed to prevent deforestation, there is an increasing awareness that an excess of forest cover may be also problematic for some taxa, especially in human altered landscapes in temperate regions (Greenberg et al., 1994). Reptiles are particularly sensitive to this effect (Ballinger and Watts, 1995; Jäggi and Baur, 1999). The recommended management guidelines advised for adequate silvicultural practices not oriented to economic exploitation of tree products which origins excessive forestation (Greenberg et al., 1994; Webb et al., 2005). Such bad practices are particularly serious in Catalonia, where forestry management is currently leading to an increase of the forested area. This trend is due to both rural abandonment and the generalisation of forest monocultures (mainly pines) for intensive exploitation.

Finally, outcomes from this work, namely, a clear link between land use and biodiversity and a working methodology used, provide helpful directions for detecting the tendencies in other areas and taxa. However, further studies should be conducted at a finer resolution scale, with more accurate distribution data, and using more informative measures (e.g. abundances) of community composition.

Acknowledgments

Appendix.
Supplementary material text

Supplementary data associated with this article can be found in the online version, at doi:10.1016/j.actao.2008.12.003.

R E F E R E N C E S

Amo, L., Lopéz, P., Martín, J., 2007. Natural oak forest vs. ancient pine plantations: lizard microhabitat use may explain the effects of ancient reforestations on distribution and conservation of Iberian lizards. Biodivers. Conserv. 16, 3409–3422.

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