Journal of Environmental Management 79 (2006) 279–289
Journal of Environmental Management
www.elsevier.com/locate/jenvman

Identifying areas of high herpetofauna diversity that are threatened by planned infrastructure projects in Spain

José M. Rey Benayas [a,*], Enrique De La Montaña [a], Josabel Belliure [a], Xavier R. Eekhout [b]

[a] Dpto. de Ecología, Edificio de Ciencias, Universidad de Alcalá, 28871 Alcalá de Henares, Spain
[b] Museo Nacional de Ciencias Naturales (CSIC), C/José Gutiérrez Abascal, 2. 28006 Madrid, Spain

Received 5 January 2005; revised 16 June 2005; accepted 21 July 2005
Available online 25 October 2005

Abstract

A major task related to conservation is to predict if planned infrastructure projects are likely to threaten biodiversity. In this study we investigated the potential impact of planned infrastructure in Spain on amphibian and reptile species, two highly vulnerable groups given their limited dispersal and current situation of population decline. We used distribution data of both groups to identify areas of high herpetofauna diversity, and compared the locations of these areas with the locations of the planned road, high-speed train railway and water reservoir network. Four criteria were used for this identification: species richness, rarity, vulnerability, and a combined index of the three criteria. From a total of 1441 cells of 20×20 km, areas of high diversity were defined as those cells whose ranked values for the different criteria included either all species or all threatened species. The combined index provided the smallest number of cells needed to retain all threatened species (1.7 and 2.6% of the cells for amphibian and reptile species, respectively). Coincidences between these high diversity areas and cells including planned infrastructures—denominated 'alert planning units'—were 35.4% for amphibians and 31.2% for reptiles. Mitigation of the potential impacts would include actions such as barriers to animal access to roads and railways and ecoducts under these constructions. Our approach provides conservation authorities information that can be used to make decisions on habitat protection. A technique that identifies threats to herpetofauna before they occur is also likely to improve the chance of herpetofauna being protected.
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Keywords: Alert planning units; Combined index of biodiversity; Conservation; Herpetofauna; Potential environmental impact

1. Introduction

An intense debate within the scientific community on biodiversity conservation strategies over the last decade has been fed by increasing rates of biodiversity loss (Heywood, 1995; Costanza et al., 1997; Pimentel et al., 1997; Ricketts et al., 1999; Terborgh, 1999; Tilman, 1999; Bininda-Emonds et al., 2000; Cincotta et al., 2000; Myers et al., 2000; Pimm and Raven, 2000; Dietz and Adger, 2003; Mittermeier et al., 2003). Identifying areas with outstanding biodiversity features helps provide information that decision makers can use, together with other information such as cost, to determine priorities for conservation (Pearlstine et al., 2002; Sarkar and Margules, 2002; Williams et al., 2002; Matsuda et al., 2003; Rey Benayas and de la Montaña, 2003; Rodrigues et al., 2004). These features are most frequently pinpointed based on criteria such as species richness, rarity (particularly endemic *taxa*), taxonomic uniqueness, threatened species, representativeness, and indicator *taxa* (Kirkpatrick, 1983; Usher, 1986; Williams et al., 1991; Prendergast et al., 1993; Faith and Walker, 1996; Castro et al., 1997; Reid, 1998; Rey Benayas et al., 1999; Williams and Araujo, 2000; Virolainen et al., 2001; Garson et al., 2002; Margules et al., 2002).

Establishing protected areas seems to be one of the most useful tools for preserving large pools of biodiversity, and constitutes the cornerstone on which regional strategies are built (Margules and Pressey, 2000; Gaston et al., 2002; Williams et al., 2002). Many studies have addressed the issue of identifying priority areas for conservation for gap analysis purposes, i.e. the detection of highly valuable areas that do not include nature reserves. However, reserves alone are not enough for nature conservation. The speed of anthropogenic change is accelerating and has dramatically increased the risk of habitat loss and disturbance (Corbett, 1989; Adams, 1999; Krzysciak, 2000; Kiesecker et al., 2001; Faith and Walker, 2002). Therefore, an important task is foreseeing potential threats to particular species, groups of species, and valuable sites for

* Corresponding author. Tel.: +34 91 8854987; fax: +34 91 8854929.
E-mail address: josem.rey@uah.es (J.M.R. Benayas).

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doi:10.1016/j.jenvman.2005.07.006

conservation such as planned infrastructure. This issue has received little attention in the scientific literature.

Our aims are, first, to determine if planned infrastructure projects are likely to threaten herpetofauna in Spain. Secondly, we asked which species could be most affected by these infrastructures and which areas are needed to maintain all amphibian and reptile species free of infrastructure impacts. One of our specific targets is to achieve a set of areas that contain (i) all species and (ii) all threatened species. Finally, we make the point that areas of overlap of high herpetological diversity and infrastructure should be monitored by conservationists. These identified 'alert planning units' should be considered candidate areas for actions to mitigate environmental impact. Planned infrastructure in Spain includes the construction of about 5000 km of highways, 2000 km of high-speed train railways, and 100 water reservoirs. More than 90% of the Spanish territory is not legally protected and is thus susceptible to damage by such new infrastructure. A technique that identifies threats to herpetofauna before they occur is also likely to improve the chance of herpetofauna being protected.

We used amphibians and reptiles as target groups because they are two *taxa* that are highly vulnerable due to their current population declines and ecological requirements (Blaustein et al., 1997; Lips, 1998; Pounds et al., 1999; Houlahan et al., 2000; Kiesecker et al., 2001; Green, 2003). Most of these species have small home ranges and are sedentary. They are usually poor colonizers, and are often reliant on a brief immature phase for dispersal. With a few exceptions, these species exhibit very limited long-distance movement capabilities for dispersal over a large region. They therefore have little capacity to avoid even temporary threats or changes to their habitats (Corbett, 1989; Adams, 1999; Krzysciak, 2000; Kiesecker et al., 2001; Biek et al., 2002; Green, 2003). Amphibians and reptiles are threatened by habitat loss, land use change, and in many cases human antipathy (Corbett, 1989; Krzysciak, 2000; Semlitsch, 2000; Biek et al., 2002). Reports of declining amphibian populations in many parts of the world are numerous, particularly in the last few decades, and are attributable to factors such as habitat destruction and fragmentation, increased road density and traffic, alien predators, contaminants, emerging infection diseases, and climatic change (Gardner, 2001; Stow et al., 2001; Collins and Storfer, 2003; Kats and Ferrer, 2003). Some of the above-mentioned factors affect local populations, whereas others may have more widespread impact (Davenport, 1997; Richter et al., 1997; Adams, 1999; Kolozsvary and Swihart, 1999; Rouse et al., 1999; Gibbons et al., 2000; Krzysciak, 2000; Semlitsch, 2000; Cohen, 2001).

We used four criteria for identifying areas of high herpetofauna diversity: species richness, rarity, vulnerability, and a combined index of the three criteria. Next, we evaluated the efficiency of the various criteria used to identify these areas. Finally, we compared the locations of these areas with the locations of the infrastructure projects, and identified the coincidences as alert planning units. Our intention is to provide useful information for herpetofauna conservation. 'Alert maps' may be useful to decision-makers because they point to where in the country large pools of amphibian and reptile diversity are particularly under threat and proper actions can be taken. Our analyses are illustrative, not exhaustive. A similar approach can be used either for different species groups, criteria, or threats to biodiversity.

2. Materials and methods

2.1. The study area

Spain is one of the richest countries in the European Union with respect to amphibian and reptile diversity, with 28 and 58 species, respectively. The study area includes the Spanish fraction of the Iberian Peninsula and the Balearic Islands (Fig. 1). It embraces a variety of biomes, relief, climates, and soil types despite a relatively small area (585,644 km^2 in total). Two major climatic zones, Mediterranean and Atlantic, are present (Font Tullot, 1983). The Mediterranean climate, with its seasonality, warm, dry summers and cool, wet winters, is characteristic of most of Iberia and the Balearic Islands. The Atlantic climate is wetter, cooler and less seasonal and is found in a band ca. 100 km wide along the western and northern coast and also influences the Pyrenean Mountains in the northeast. The driest and warmest areas in the south of the country served as *refuges* during Pleistocene glaciations. In contrast, the northern transition zone between the two climates is substantially younger, having emerged only after glacial retreat. Within regions, the relative extent of different vegetation types, natural landscapes, and diversity patterns depends not only on the environmental status and variation, but also on human impacts. Thus, land management—particularly agriculture—can affect diversity (Leiva et al., 1997).

2.2. Planning units and data sources

Our analyses used cells of 20×20 km, defined by UTM coordinates, as planning units. We examined the presence and absence of amphibian and reptile species in 1441 cells. We built the cell-by-species matrices using the species distribution maps from Pleguezuelos (1997). We considered 28 amphibian species and 48 non-marine reptile species (Appendix A).

2.3. Criteria for identifying priority areas of high herpetofauna diversity

We used four criteria to identify areas of high herpetofauna diversity: species richness, rarity, vulnerability, and a combined index of the three criteria. There are many forms of rarity, responding to different combinations of geographical range, local abundance, habitat specificity, and habitat occupancy (Rabinowitz, 1981; Rey Benayas et al., 1999). In this study, rarity of a species i was defined by its geographical range measured as the inverse of the number of cells where it was present ($1/n_i$). Currently, there are not established criteria in Spain classifying species into rarity categories according to their geographical ranges (Perring and Farrel, 1983; Cameron,

Fig. 1. Map of continental Spain and the Balearic Islands. It illustrates the planned infrastructure network considered in this study. Symbols: solid lines are highways, dashed lines are high-speed railways, and gray squares are reservoirs.

1998). For a cell r, the rarity index was $\sum_{i=1}^{S}(1/n_{ri})/s_r$, where s_r was the number of species found in the cell.

Species vulnerability was quantified using the categories of the *Red Book of Spanish Vertebrates* (Blanco and González, 1992; Appendix 1). The species categories that were considered are the following: endangered, vulnerable, rare, undetermined, insufficiently known, introduced, and non-threatened. These categories were previously defined by the International Union for the Conservation of Nature (IUCN, 1988). Complete definitions can be found in Appendix 1. These categories are now under revision. Vulnerability is actually a surrogate concept of rarity plus rates of habitat loss and other threats. We assigned every category a score related to its degree of vulnerability, ranging from 5 for endangered species to 1 for non-threatened and introduced species. Intermediate categories were assigned 4 (vulnerable and undetermined), 3 (rare) and 2 (insufficiently known). We acknowledge the subjectivity of these scores; they represent a rank and thus a relative value. For a cell, the vulnerability index was $\sum_{i=1}^{S} V_{ri}/s_r$, where V_{ri} was the vulnerability score of the species i present in the cell. Finally, we used a combined index of species richness, rarity, and vulnerability defined by Rey Benayas and de la Montaña (2003): $\sum_{i=1}^{S}(1/n_{ri})V_{ri}$. In this index, species richness is implicit in $\sum_{i=1}^{S}$.

Next, all diversity indices for both *taxa* were ranked. To quantitatively define areas of high herpetofauna diversity, we considered the pool of cells within the upper ranked values for the various criteria that included either all species or all threatened species. For our purposes, 'threatened species' were considered those belonging to the endangered, vulnerable, rare, and undetermined categories of IUCN (1988).

2.4. The planned infrastructure network

We obtained information on the locations of newly planned highways and roads, high-speed train railways, and water reservoirs (Fig. 1) till year 2007 from official public documents available on-line at http://www.mfom.es/home/Infraes/intro.html.

2.5. Data analysis

We examined the relationships between the four criteria across *taxa* by means of correlation analysis using Bonferroni corrections for multiple comparisons. To evaluate the effectiveness of the various criteria used to identify areas of high diversity, we looked at the number of ranked cells that included all species and all threatened species. The congruence between areas of high diversity for both *taxa* was analyzed by means of χ^2. Then, we examined the coincidence between the location of these areas and the location of the planned infrastructure. Those cells that were categorized as areas of high diversity and that included planned infrastructure were considered alert planning units. Next, we ascertained which species were present only in alert planning units and how many of the cells occupied by threatened species within the areas of high diversity were alert planning units. Finally, when necessary, we calculated the number of additional cells without planned infrastructures that should be added to the selected areas of high diversity to ensure the representativeness goal (retention of all species).

3. Results

3.1. Evaluation and distribution of areas of high herpetofauna diversity

The performance of the four indices based on the average number of cells needed to retain all species for both *taxa* were ranked: rarity (75 cells) < vulnerability (122) < combined index (159.5 cells) < species richness (626.5) (Table 1). However, the number of cells based on the combined index drops to 49 if the endemic *Alytes dickhilleni* is removed from the analysis. The performance of the indices based on the average number of cells needed to retain all threatened species for both *taxa* were ranked: combined index (30.5 cells) < rarity (46.5) < vulnerability (57.5) < species richness (626.5) (Table 1). All cells where the five threatened amphibian species were present were encompassed by the combined index. Similarly, all cells where the 13 threatened reptile species appeared were retained by this index, with the exception of 82 out of 109, 5 out of 46, and 7 out of 58 cells for *Emys orbicularis*, *Coluber viridiflavus*, and *Elaphe longissima*, respectively. Thus, the combined index performed better than the other criteria because fewer cells were needed to retain threatened herpetofauna diversity.

Distribution of areas of high diversity for amphibians and reptiles as defined by the combined index is shown in Fig. 2. These areas for amphibians are mainly aggregated in the Atlantic climatic region of Iberia and in the Balearic Islands (Fig. 2(a)). The distribution of areas of high reptile diversity indicates an aggregation in the Balearic Islands, Pyrenean Mountains, and the southern coast (Fig. 2(b)). The interior section of the Iberian Peninsula is less favored for both taxonomic groups.

3.2. Congruence of areas of high amphibian and reptile diversity

The correlation coefficients between each criterion used to identify areas of high diversity between the two *taxa* were 0.73, 0.13, 0.35, and 0.05 for richness, rarity, vulnerability and the combined index, respectively ($n = 1441$, $P < 0.0001$ in all cases except for the combined index which was not significant at $P = 0.05$ after correcting for multiple comparisons). To what extent do areas of high diversity for both *taxa* overlap? Using the results shown in Table 1 and Fig. 2, congruence between these areas for amphibians and reptiles averaged 43.3% for all criteria. The vulnerability criterion produced the highest dispersion of areas of high diversity for both *taxa* (18.9% congruence, $\chi^2 = 15.86$, $P = 0.0012$), whereas richness produced the highest aggregation (87.9% congruence, $\chi^2 = 356.6$, $P < 0.0001$). Rarity (32.7%, $\chi^2 = 64.0$, $P < 0.0001$) and the combined index of biodiversity (33.8%, $\chi^2 = 54.14$, $P < 0.0001$) produced an intermediate level of congruence.

Table 1
Number (and proportion in parenthesis) of cells that are needed to retain all species and threatened species of amphibians and reptiles according to four criteria

	All amphibian species	Threatened amphibian species	All reptile species	Threatened reptile species
Richness	850 (59.0%)	850 (59.0%)	505 (35.0%)	505 (35.0%)
Rarity	95 (6.6%)	38 (2.63%)	55 (3.8%)	48 (3.3%)
Vulnerability	115 (8.0%)	47 (3.3%)	128 (8.9%)	67 (4.6%)
Combined index	130 (9.02%)	24 (1.7%)	189 (13.1%)	37 (2.6%)

3.3. Coincidence between areas of high hepetofauna diversity and planned infrastructure

Coincidences between areas of high diversity and cells including planned infrastructure were low and not statistically associated. Forty-six (35.4%) and 59 (31.2%) cells identified as areas of high diversity according to the combined index for amphibians and reptiles, respectively, were affected by planned infrastructures (denominated alert planning units in Fig. 3). Only one amphibian species (the non-native toad *Bufo mauritanicus*) and one reptile species (the non-native turtle *Pseudemys picta*) were exclusive to alert planning units. Since these non-native, introduced species appeared in only one cell, we did not need to examine which cells free of planned infrastructure should be added to the areas of high diversity list to retain the species lost in the alert sites. However, five threatened amphibian species lost a substantial presence in this list if alert planning units were eliminated: *Chioglossa lusitanica* (from 47 to 25 cells), *Triturus alpestris* (41– 24), *Bufo viridis* (21–17), *Rana dalmatina* (12–7), and *Alytes muletensis* (2–1). Planned infrastructure did not affect any cell where five threatened reptile species were present (*Lacerta agilis*, *L. araica*, *L. aureolioi*, *Podarcis lilfordi*, and *P. pityusensis*), while the remaining threatened reptile species were affected by the infrastructure, with the following loss of cells where they were present: *C. viridiflavus* (from 41 to 32 cells), *E. longissima* (51–33), *T. hermanni* (34–21), *E. orbicularis* (27–17), *Chamaleo chamaleon* (30–21), *Testudo graeca* (21–12), *Algyroides marchi* (11–7), and *Lacerta bonnali* (7–5).

4. Discussion

4.1. Distribution of areas of high herpetofauna diversity

The number of species in any region is likely to depend on its location (Davidowitz and Rosenzweig, 1998). In the European context, a clear pattern of species numbers and ratios of amphibians to reptiles emerges from north to south. The northward territories have fewer species, with a higher proportion of amphibians, and this holds true even for islands (Corbett, 1989). Diversity of amphibians and reptiles in our study area supports this pattern.

Three groups of ecological and evolutionary factors important for the distribution of areas of high amphibian and reptile diversity are biogeography, the effects of mountain

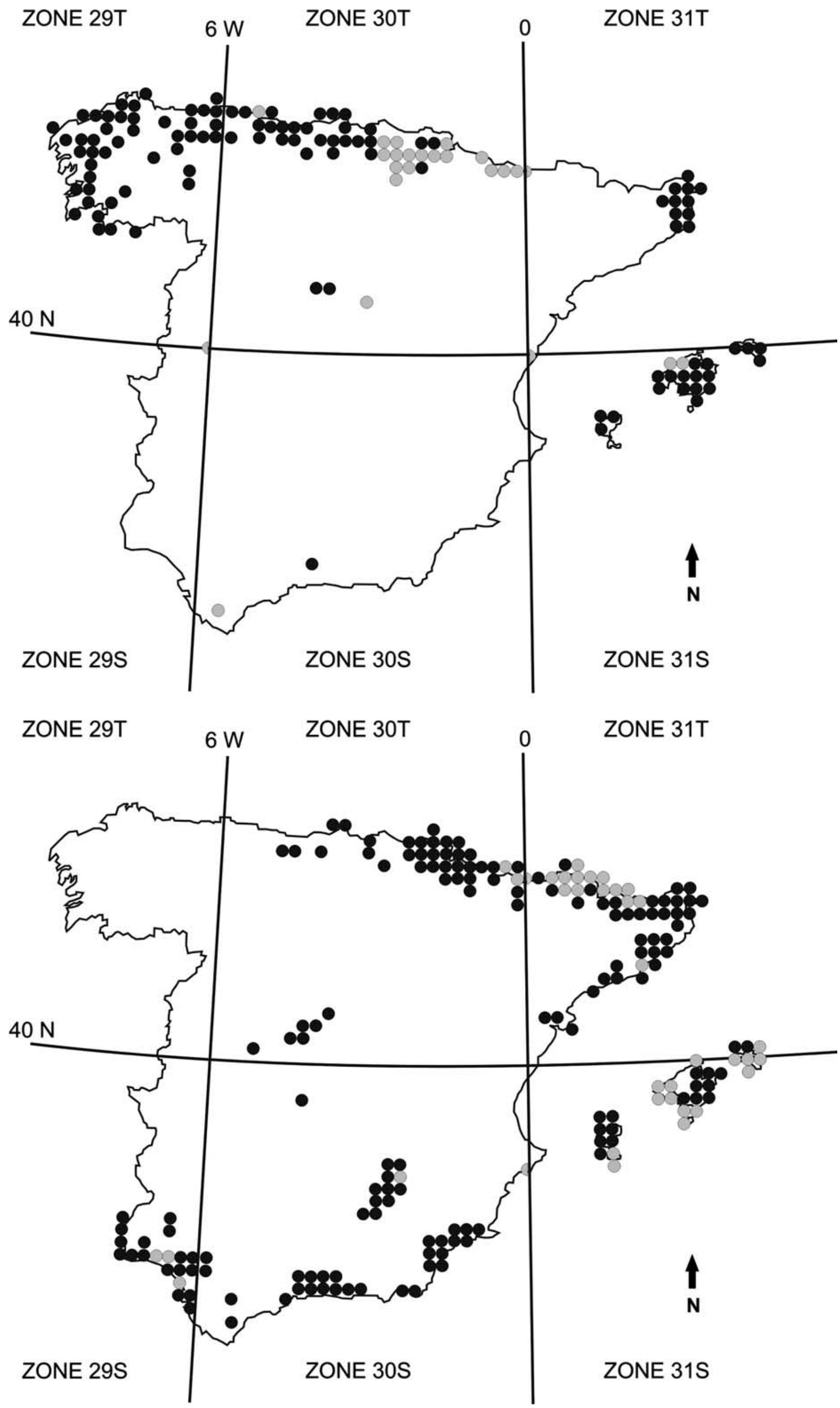

Fig. 2. Maps of areas of high diversity identified according to the combined index of biodiversity for (a) amphibians (top) and (b) reptiles (bottom). Symbols are the following: solid circles retain all species and gray circles retain all threatened species.

refuges, and the ecological requirements of the species. Biogeographic effects in the Iberian Peninsula include: (1) climate differences in the northern fringe as compared to the rest of the Peninsula, particularly the transition between the Atlantic and Mediterranean climates, and (2) the insular effect in the Balearic Islands. Increased regional species diversity in transition zones is consistent with the analyses for plant species of Rey Benayas and Scheiner (2002). The Iberian Peninsula includes a high proportion of amphibian (25%) and reptile (20%) endemics. Many species that are characteristic of the Atlantic climate are only present at the northern part of Iberia, and its origin is presumably the refuge effect of Iberia for

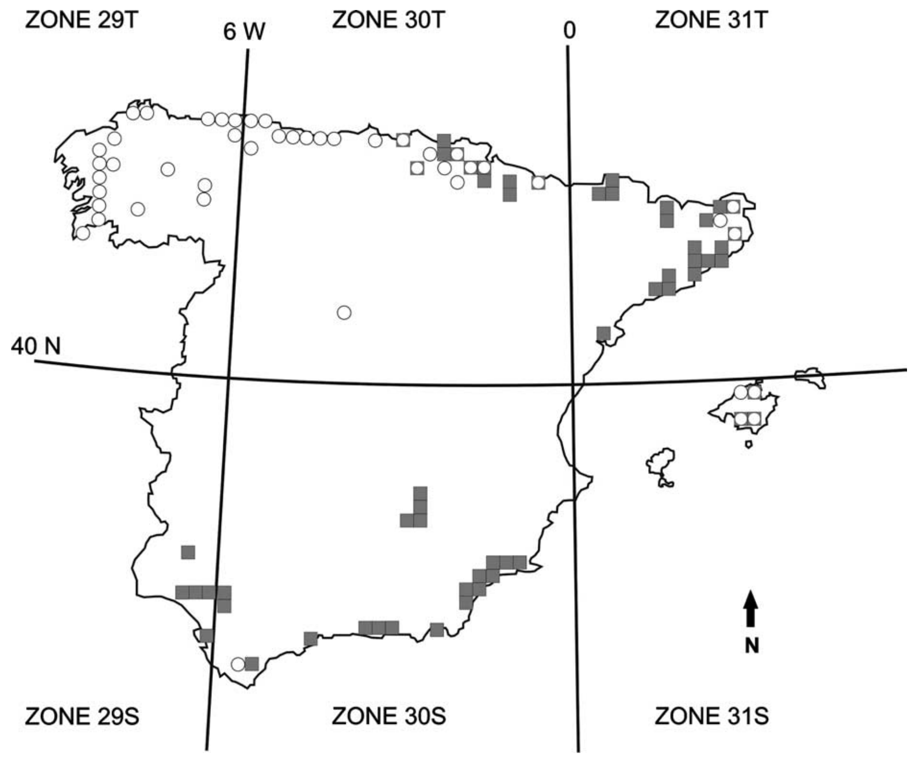

Fig. 3. Map of alert planning units (i.e. coincidences between areas of high diversity identified according to the combined index of biodiversity and planned infrastructure). Symbols: empty circles are alert planning units for amphibian species and gray squares are alert planning units for reptile species.

the European fauna during the Quaternary (Vargas and Real, 1997). By contrast, the Strait of Gibraltar notably exerted a barrier effect (Busack, 1986). As a consequence, the Iberian herpetofauna has higher similarity with northwestern Europe than with Northern Africa (Oosterbroek and Arntzen, 1992). The peaks in species richness of amphibian and reptile species found in the central-western mountain ranges and other areas such as the Cantabrian and Pyrenean mountains may also be due to refuge in a land with over a millennium of varied agricultural, silvicultural, and pastoral practices. This explanation was favored by Castro et al. (1997) for the distribution of terrestrial vascular plants in the same study area. Island biogeographic effects are also evident in our maps. Most of the territory of the Balearic Islands shows areas of high diversity. The explanations for these patterns may lie in the differences in dispersion capabilities and speciation and extinction rates of the *taxa* (Blondel and Aronson, 1999).

Apart from biogeographic and *refugia* effects, differences in the ecological requirements of the *taxa* also contribute to the patterns of the distribution of areas of high diversity (Meliadou and Troumbis, 1997; James and Shine, 2000). A large proportion of these areas for amphibian species are concentrated in northern Spain as higher precipitation and lower evaporation rates increase moisture in air and soil as well as flooded habitats suitable for reproduction. Conversely, there is an aggregation of areas of high reptile diversity in the dry southern part of the Iberian Peninsula. The difference in ecological requirements for both taxonomic groups leads to a moderate to low congruence between their respective areas of high diversity.

Anthropogenic factors influencing the patterns of species diversity should be considered as well. A study on biogeographical regions of the Iberian Peninsula, based on the distribution of freshwater fish and amphibians, assessed the influence of humans based upon data of native and well-established introduced species (Vargas et al., 1998). The effect of species introductions by humans is clearer in islands. The presence of 13 amphibian and reptile species non-endemic to the Balearic Islands is the result of anthropogenic introductions during the nearly 8000 years of sea traffic between the islands and the continent (Mayol, 1997).

4.2. Efficiency of the criteria used to define areas of high diversity

Our results show differences in the effectiveness of the different criteria used to define areas of high diversity. Rarity and the combined index showed the highest efficiency since fewer cells were needed to retain all species or all threatened species. The combined index of Rey Benayas and de la Montaña (2003) has the additional value of simultaneously taking into account species richness, geographical rarity, and vulnerability and allowed the retention of most of the cells where the threatened species were present. Richness showed the lowest efficiency as many cells were needed to retain significant pools of diversity. This fact is important, as species richness constitutes one of the most utilized criterions in conservation decisions (Caldecott et al., 1996; Rossi and Kuitunen, 1996; Médail and Quézel, 1997; Reyers et al., 2000; Pearlstine et al., 2002; Rodrigues et al., 2004). The greater efficiency of the rarity criterion, as compared to the richness criterion, is supported by other studies (Williams et al., 1996; Margules et al., 1988; Haeupler and Vogel, 1999).

4.3. Areas of high diversity and planned infrastructure

Areas harboring high levels of species diversity and that are also under severe threat are usually defined in the literature as hotspots (Myers, 1988; 1990; Prendergarst and Eversham, 1995; Beissinger et al., 1996; Harcourt, 2000; Myers et al., 2000). We used diversity of two ecologically contrasting *taxa* that are highly vulnerable and that have historically been under-considered in conservation plans. Indeed, effective conservation measures remain inadequate as compared to other vertebrates and very few action plans for conservation of endangered species are currently being implemented in Spain (Márquez, 2004). The moderate to low congruence of areas of high diversity of both *taxa* makes decisions on conservation strategies more difficult.

The coincidences between areas of high diversity and the newly planned infrastructure are highest for road construction due to the higher spatial extent of this as compared to other types of infrastructure considered in our study. Road construction is a serious threat to biodiversity due to a variety of effects such as restricted movement between populations, increased mortality (particularly as the ecological requirements of ectotherms make roads an optimal site for basking), habitat fragmentation, greater edge effects, increased human access to wildlife habitats, and increased accessibility for exotic predators (May and Norton, 1996; Findlay and Bourdages, 2000). Populations of susceptible species are expected to decline gradually after road construction, with local extinction occurring sometime later. Thus, the full effects of road construction on these *taxa* may be undetectable for decades. Direct mortality (road kills) has been documented for both groups, representing 23 and 89% of the total vertebrate individuals killed in the study area to date (Barbadillo and García-Paris, 1991).

Other factors complicate assessment of the environmental impacts of roads, such as their distance to target populations. The density of paved roads on lands up to 2 km away from the habitat occupied by different species has been shown to influence species richness (Findlay and Houlahan, 1997). This suggests that most existing policies, which focus almost exclusively on actions within the habitat itself and/or a narrow buffer zone around the perimeter, are unlikely to provide adequate protection for biodiversity.

The lower level of coincidence of high-speed train railways with areas of high diversity responds to the reduced area potentially affected by this type of infrastructure in the study region. Their negative effects, however, can be very important in terms of fragmentation, and with the exception of the risk of mortality due to road-kill, largely coincide with those related to road construction.

The potential effects of newly planned reservoirs on the species have received little study. Water availability in the habitat is one of the most important factors affecting the temporal distribution of reproductive activities in amphibian species. However, reservoirs often imply canalization of small waterways and the loss of water levels in related water complexes that negatively affect their use by amphibians. Moreover, the walls of reservoirs and canals are often too steep and high to allow individuals access as an alternative habitat for reproduction (Barbadillo et al., 1997). The importance of lakeshore development on amphibian abundance has been studied by Woodford and Meyer (2003). Usually, far from being an advantage, these constructions represent a risk for species conservation.

There were several limitations of our study that should be considered in evaluating our results. We evaluated diversity from distribution maps that may have false absences. Given the available data, the definitions of areas of high diversity are to some extent arbitrary, and our results are obviously scale dependent. Some authors such as Pressey and Nicholls (1989) have criticized scoring approaches, but the results that we obtained are actually promising. Our approach would have benefited from adding other groups of species, additional human threats such as land-use or land-cover change, as well as land ownership and value to the geographical analysis of biodiversity and planned infrastructure (Dobson et al., 2001; Scott et al., 2001). It can also be argued there was a mismatch of scales and a 'knowledge-action gap' (Pfeffer and Sutton, 1999), since we analyzed diversity at the grain of 20×20 km and most environmental impact assessment (EIA) and mitigation decisions address areas within 100's of meters of a proposed development. Smaller cells will increase efficiency (Pressey and Logan, 1998), and might also reduce the number of alert planning units. This study has addressed biodiversity retention (*sensu* Cowling, 1999). The inclusion of some measure of biodiversity persistence (Araújo and Williams, 2000), or designing the network of areas of high diversity to incorporate environmental processes (e.g. Cowling et al., 2003a,b) would also considerably improve this assessment.

However, we believe that the common application of EIA at the project level fails to ensure adequate consideration of potentially serious trans-boundary, widespread, indirect, cumulative, and synergistic ecological effects (Treweek et al., 1998). Maps of areas of high diversity have been suggested as useful tools for environmental impact assessment and mitigation (Ayensu et al., 1999). Clearly, some form of strategic ecological assessment is required to ensure that the development of new infrastructure is compatible with the conservation of habitats and species. Our study highlights areas where planned infrastructures are likely to impact herpetofauna. Mitigation of such a potential impact would include actions such as barriers to animal access to roads and railways and ecoducts under these constructions (Joly et al., 2003; Kats and Ferrer, 2003; Rossel, 2003; Semlitsch and Bodie, 2003; Willson and Dorcas, 2003). Collaboration with stakeholders, especially those organizations proposing and undertaking these new infrastructures, and the development of an implementation strategy would greatly facilitate conservation interventions in alert planning units (Driver et al., 2003).

5. Conclusions

We produced a map that combines the locations of areas of high herpetofauna diversity and the locations of the planned increases in public infrastructure in Spain. Portions of the territory as small as 1.7 and 6.6% of the total area were found to include all threatened species and all species, respectively, with the additional value that these areas retain most of the cells occupied by the threatened species. The map highlighted a number of alert planning units that can be used in subsequent analysis by decision makers. We were able to identify which species will likely be the most affected by this infrastructure. Fortunately, we found that there is no need to identify additional sites free of planned infrastructure that would retain some lost species. This approach can be used to favor the conservation of other taxonomic groups anywhere in the world.

Acknowledgements

This study was funded by the projects REN2000/745 and CGL2004-00355/BOS (Ministerio de Educación y Ciencia, Spain) and GR/AMB/0757/2004 (Comunidad de Madrid). We are indebted to Salvador Rebollo, Rafael Márquez and Robert Manson for their comments on an earlier version of this manuscript. Four anonymous reviewers improved the contents and presentation of the current manuscript.

Appendix A

List of species used in the identification of areas of high diversity. The original raw data were extracted from Pleguezuelos (1997). The number in parenthesis refers to the number of 20×20 km cells where the species was present. The vulnerability status according to the *Red Book of Spanish Vertebrates* (Blanco and González, 1992) is indicated. Categories are defined as follows. Endangered: species in danger of extinction and unlikely to survive if causal factors persist; Vulnerable: species that would soon belong to the endangered category if causal factors persist; Rare: species with small populations at risk mostly because they extend on small geographical ranges or habitats or because their populations are sparse; Undetermined: species that do belong to the endangered, vulnerable or rare categories, but the current knowledge does not allow a certain assignment; Insufficiently known: species that are suspected to belong to the former categories, but there is no certainty about that; Non-threatened: species with no evident threats. Abbreviations: E, endangered; V, vulnerable; R, rare; U. undetermined; I-K, insufficiently known; I, introduced; N-T, non-threatened.

A.1. Amphibians

Alytes cisternasii (139) (N-T), *A. dickhilleni* (43), *A. muletensis* (2) (E), *Alytes obstetricans* (409) (N-T), *Bufo bufo* (786) (N-T), *Bufo calamita* (617) (N-T), *B. mauritanicus* (1), *B. viridis* (21) (R), *C. lusitanica* (48) (R), *Discoglossus galganoi* (331) (N-T), *Discoglossus pictus* (11) (I), *Euproctus asper* (69) (N-T), *Hyla arborea* (216) (N-T), *Hyla meridionalis* (150) (N-T), *Pelobates cultripes* (430) (N-T), *Pelodytes punctatus* (303) (N-T), *Pleurodeles waltl* (270) (N-T), *Rana catesbeiana* (1), *R. dalmatina* (12) (V), *Rana iberica* (142) (N-T), *Rana perezi* (881) (N-T), *Rana pyrenaica* (5), *Rana temporaria* (161) (N-T), *Salamandra salamandra* (376) (N-T), *T. alpestris* (41) (R), *Triturus boscai* (237) (N-T), *Triturus helveticus* (172) (N-T), *and Triturus marmoratus* (364) (N-T).

Note: *A. dickhilleni* and *R. pyrenaica* have recently been catalogued; *B. mauritanicus* and *R. cartesbeiana* are introduced species and are not included in the Red Book.

A.2. Reptiles

Acanthodactylus erythrurus (187) (N-T), *A. marchi* (11) (R), *Anguis fragilis* (301) (N-T), *Anolis carolinensis* (1), *Blanus cinereus* (238) (N-T), *Coluber hippocrepis* (305) (N-T), *C. viridiflavus* (46) (R), *Coronella austriaca* (222) (N-T), *Coronella girondica* (495) (N-T), *Chalcides bedriagai* (199) (N-T), *Chalcides striatus* (402) (N-T), *Chamaeleo chamaeleon* (30) (E), *E. longissima* (58) (R), *Elaphe scalaris* (664) (N-T), *E. orbicularis* (109) (V), *Hemidactylus turcicus* (152) (N-T), *L. agilis* (4) (V), *Lacerta araica* (2) (E), *Lacerta aurelioi* (3) (E), *L. bonnali* (7) (U), *Lacerta lepida* (904) (N-T), *Lacerta monticola* (55) (N-T), *Lacerta perspicillata* (5) (I), *Lacerta schreiberi* (177) (N-T), *Lacerta viridis* (129) (N-T), *Lacerta vivipara* (77) (N-T), *Macroprotodon cucullatus* (172) (N-T), *Malpolon monspessulanus* (752) (N-T), *Mauremys leprosa* (257) (N-T), *Natrix maura* (841) (N-T), *Natrix natrix* (462) (N-T), *Podarcis bocagei* (128) (N-T), *Podarcis hispanica* (752) (N-T), *P. lilfordi* (12) (V), *Podarcis muralis* (192) (N-T), *Podarcis pityusensis* (12) (R), *Podarcis sicula* (8) (I), *Psammodromus algirus* (771) (N-T), *Psammodromus hispanicus* (389) (N-T), *P. picta* (1), *Tarentola mauritanica* (389) (N-T), *T. graeca* (21) (E), *Testudo hermanni* (34) (V), *Trachemys scripta* (46), *Trionyx spiniferus* (5), *Vipera aspis* (129) (N-T), *Vipera latastei* (263) (N-T), *and Vipera seoanei* (136) (N-T).

Note: *A. carolinensis*, *P. Picta*, *T. scripta* and *T. spiniferus* are introduced species that are not included in the Red Book.

References

Adams, M.J., 1999. Correlated factors in amphibian decline: exotic species and habitat change in western Washington. Journal of Wildlife Management 63, 1162–1171.

Araújo, M.B., Williams, P.H., 2000. Selecting areas for species persistence using occurrence data. Biological Conservation 96, 331–345.

Ayensu, E., Claasen, D., van, R., Collins, M., Dearing, A., Fresco, L., Gadgil, M., Gitay, H., Glaser, G., Juma, C., Krebs, J., Lenton, R., Lubchenco, J., McNeely, J.A., Mooney, H.A., Pinstrup-Andersen, P., Ramos, M., Raven, P., Reid, W.V., Samper, C., Sarukhán, J., Schei, P., Galízia Tundisi, J., Watson, R.T., Guanhua, X., Zakri, A.H., 1999. International ecosystem assessment. Science 286, 685–686.

Barbadillo, L.J., Garcia-Paris, M., 1991. Problemas de conservación de los anfibios en España. Quercus 62, 20–25.

Barbadillo, L.J., Garcia-Paris, M., Sanchíz, B., 1997. Orígenes y relaciones evolutivas de la herpetofauna ibérica. In: Pleguezuelos, J.M. (Ed.), Distribución y biogeografía de los anfibios y reptiles en España y Portugal, Monografías de Herpetología 3. Universidad de Granada, Asociación Herpetológica Española, pp. 47–100.

Beissinger, S.R., Steadman, E.C., Wohlgenant, T., Blate, G., Zack, S., 1996. Null models for assessing ecosystem conservation priorities: Threatened birds as titers of threatened ecosystems in South America. Conservation Biology 10, 1343–1352.

Biek, R., Funk, W.C., Maxell, B.A., Mills, L.S., 2002. What is missing in amphibian decline research: Insights from ecological sensitivity analysis. Conservation Biology 16, 728–734.

Bininda-Emonds, O.R.P., Vazquez, D.P., Manne, L.L., 2000. The calculus of biodiversity: integrating phylogeny and conservation. Trends in Ecology and Evolution 15, 92–94.

Blanco, J.C., González, J.L., 1992. Libro Rojo de los Vertebrados de España. ICONA-Ministerio de Agricultura, Pesca y Alimentación. Madrid, Spain.

Blaustein, A.R., Kiesecker, J.M., Chivers, D.P., Anthony, R.G., 1997. Ambient UV-B radiation causes deformities in amphibian embryos. Proceedings of the Natural Academy of Science USA 94, 13735–13737.

Blondel, J., Aronson, J., 1999. Biology and Wildlife of the Mediterranean Region. Oxford University Press, Oxford, UK.

Busack, S.D., 1986. Biogeographic analysis of the herpetofauna separated by the formation of the Strait of Gibraltar. National Geographic Research 2, 17–36.

Caldecott, J.O., Jenkins, M.D., Johnson, T.H., Groombridge, B., 1996. Priorities for conserving global species richness and endemism. Biodiversity and Conservation 5, 699–727.

Cameron, R.A.D., 1998. Dilemmas of rarity: biogeographic insights and conservation priorities for land mollusca. Journal of Conchology Special Publication 2, 51–60.

Castro, I., Moreno, J.C., Humphries, C.J., Williams, P.H., 1997. Strengthening the Natural and National Park system of Iberia to conserve vascular plants. Botanical Journal of the Linnean Society 121, 189–206.

Cincotta, R.P., Wisnewski, J., Engelman, R., 2000. Human population in the biodiversity hotspots. Nature 404, 990–992.

Cohen Jr.., M.M., 2001. Frog decline, frog malformations, and a comparison of frog and human health. American Journal of Medical Genetics 104, 101–109.

Collins, J.P., Storfer, A., 2003. Global amphibian declines: sorting the hypotheses. Diversity and Distributions 9 (2), 89–98.

Corbett, K., 1989. Conservation of European Reptiles and Amphibians. Christopher Helm Publishers, Bromley, Kent, UK.

Costanza, R., Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

Cowling, R.M., 1999. Planning for persistence — systematic reserve design in Africa's Succulent Karoo desert. Parks 9, 17–30.

Cowling, R.M., Lombard, A.T., Rouget, M., Kerley, G.I.H., Wolf, T., Sims-Castley, R., Knight, A.T., Vlok, J.H.J., Pierce, S.M., Boshoff, A.F., Wilson, S.L. 2003a. A Conservation Assessment for the Subtropical Thicket Biome. Terrestrial Ecology Research Unit Report No. 43, University of Port Elizabeth, South Africa.

Cowling, R.M., Pressey, R.L., Rouget, M., Lombart, A.T., 2003b. A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa. Biological Conservation 112, 191–216.

Davenport, J., 1997. Temperature and the life-history strategies of turtles. Journal of Thermal Biology 22, 479–488.

Davidowitz, G., Rosenzweig, M.L., 1998. The latitudinal gradient of species diversity among North American grasshoppers (Acrididae) within a single habitat: a test of the spatial heterogeneity hypothesis. Journal of Biogeography 25, 553–560.

Dietz, S., Adger, W.N., 2003. Economic growth, biodiversity loss and conservation effort. Journal of Environmental Management 68, 23–35.

Dobson, A.P., Rodríguez, J.P., Roberts, W.M., 2001. Synoptic tinkering: integrating strategies for large-scale conservation. Ecological Applications 11, 411–420.

Driver, A., Cowling, R.M., Maze, K. 2003. Planning for Living Landscapes: Perspectives and Lessons from South Africa. Center for Applied Biodiversity Science at Conservation International and the Botanical Society of South Africa, Washington, DC, USA and Cape Town, South Africa.

Faith, D.P., Walker, P.A., 1996. How do indicator groups provide information about the relative biodiversity of different sets of areas? On hotspots, complementarity and pattern-based approaches. Biodiversity Letters 3, 18–25.

Faith, D.P., Walker, P.A., 2002. The role of trade-offs in biodiversity conservation planning: linking local management, regional planning and global conservation efforts. Journal of Bioscience 27, 393–407.

Findlay, C.S., Bourdages, J., 2000. Response time on wetland biodiversity to road construction on adjacent lands. Conservation Biology 14, 86–94.

Findlay, C.S., Houlahan, J., 1997. Anthropogenic correlates of species richness in southeastern Ontario wetlands. Conservation Biology 11, 1000–1009.

Font Tullot, I., 1983. Atlas Climático de España. Instituto Nacional de Meteorología, Madrid, Spain.

Gardner, T., 2001. Declining amphibian populations: A global phenomenon in conservation biology. Animal Biodiversity and Conservation 24 (2), 25–44.

Garson, J., Aggarwal, A., Sarkar, S., 2002. Birds as surrogates for biodiversity: an analysis of a data set from southern Quebec. Journal of Bioscience 27, 347–360.

Gaston, K.J., Pressey, R.L., Margules, C.R., 2002. Persistence and vulnerability: retaining biodiversity in the landscape and in protected areas. Journal of Bioscience 27, 361–384.

Gibbons, J.W., Scott, D.E., Ryan, T.J., Buhlmann, K.A., Tuberville, T.D., Metts, B.S., Greene, J.L., Mills, T., Leiden, Y., Poppy, S., Winne, C.T., 2000. The global decline of reptiles, deja vu amphibians. Bioscience 50, 653–666.

Green, D.M., 2003. The ecology of extinction: population fluctuation and decline in amphibians. Biological Conservation 111, 331–343.

Haeupler, H., Vogel, A., 1999. Plant diversity in Germany: A second review. Acta Botanica Fennica 162, 55–59.

Harcourt, A.H., 2000. Coincidence and mismatch of biodiversity hotspots: A global survey for the order, primates. Biological Conservation 93, 163–175.

Heywood, V.H., 1995. Global Biodiversity Assessment. Cambridge University Press, Cambridge, UK.

Houlahan, J.E., Findlay, C.S., Schmidt, B.R., Meyer, A.H., Kuzmin, S.L., 2000. Quantitative evidence for global amphibian population declines. Nature 404, 752–755.

IUCN, 1988. List of Threatened Animals. Gland, Switzerland.

James, C.D., Shine, R., 2000. Why are there so many coexisting species of lizards in Australian deserts? Oecologia 125, 127–141.

Joly, P., Morand, C., Cohas, A., 2003. Habitat fragmentation and amphibian conservation: building a tool for assessing landscape matrix connectivity. Comptes Rendus Biologies 326 (S1), 32–39.

Kats, L.B., Ferrer, R.P., 2003. Alien predators and amphibian declines: review of two decades of science and the transition to conservation. Diversity and Distributions 9 (2), 99–110.

Kiesecker, J.M., Blaustein, A.R., Belden, L.K., 2001. Complex causes of amphibian population declines. Nature 410, 681–684.

Kirkpatrick, J.B., 1983. An iterative method for establishing priorities for the selection of nature reserves: an example from Tasmania. Biological Conservation 25, 127–134.

Kolozsvary, M.B., Swihart, R.K., 1999. Habitat fragmentation and the distribution of amphibians: patch and landscape correlates in farmland. Canadian Journal of Zoology 77, 1288–1299.

Krzysciak, K.R., 2000. What threatens amphibians at dawn of the new millennium? Wiadomosci Ekologiczne 46, 115–126.

Leiva, M.J., Chapin III., F.S., Fernández Alés, R., 1997. Differences in species composition and diversity among Mediterranean grasslands with different history — the case of California and Spain. Ecography 20, 97–106.

Lips, K.R., 1998. Decline of a tropical montane amphibian fauna. Conservation Biology 12, 106–117.

Margules, C.R., Pressey, R.L., 2000. Systematic conservation planning. Nature 405, 243–253.

Margules, C.R., Nicholls, A.O., Pressey, R.L., 1988. Selecting networks of reserves to maximize biological diversity. Biological Conservation 43, 63–76.

Margules, C.R., Pressey, R.L., Williams, P.H., 2002. Representing biodiversity: data and procedures for identifying priority areas for conservation. Journal of Bioscience 27, 309–326.

Márquez, R., 2004. La conservación de los anfibios y reptiles en la España de las autonomías. Quercus 221, 28–35.

Matsuda, H., Serizawa, S., Ueda, K., Kato, T., Yahara, T., 2003. Assessing the impact of the Japanese 2005 World Exposition Project on vascular plants' risk of extinction. Chemosphere 53, 325–336.

May, S.A., Norton, T.W., 1996. Influence of fragmentation disturbance on the potential impact of feral predators on native fauna in Australian forest ecosystems. Wildlife Research 23, 387–400.

Mayol, J., 1997. Biogeografía de los anfibios y reptiles de las islas baleares. In: Pleguezuelos, J.M. (Ed.), Distribución y biogeografía de los anfibios y reptiles en España y Portugal, Monografías de Herpetología 3. Universidad de Granada, Asociación Herpetológica Española, pp. 371–379.

Medail, F., Quézel, P., 1997. Hot-Spots analysis for Conservation of plant biodiversity in the Mediterranean Basin. Annales Missouri Botanical Garden 84, 112–127.

Meliadou, A., Troumbis, A.Y., 1997. Aspects of heterogeneity in the distribution of diversity of the European herpetofauna. Acta Oecologica 18, 393–412.

Mittermeier, R.A., Mittermeier, C.G., Brooks, T.M., Pilgrim, J.D., Konstant, W.R., da-Fonseca, G.A., Kormos, C., 2003. Wilderness and biodiversity conservation. Proceedings of the National Academy of Science USA 100, 10309–10313.

Myers, N., 1988. Threatened biotas: hotspots in tropical forests. Environmentalist 8, 187–208.

Myers, N., 1990. The biodiversity challenge: expanded hotspot analysis. Environmentalist 10, 243–256.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.

Oosterbroek, P., Arntzen, J.W., 1992. Area-Cladograms of Circum-Medoterranean taxa in relation to Mediterranean palaeogeography. Journal of Biogeography 19, 3–20.

Pearlstine, L.G., Smith, S.E., Brandt, L.A., Allen, C.R., Kitchens, W.M., Stenberg, J., 2002. Assessing state-wide biodiversity in the Florida Gap analysis project. Journal of Environmental Management 66 (2), 127–144.

Perring, F.H., Farrel, L., 1983. British Red Data Books: 1, Vascular Plants. RSNC, Nenleham.

Pfeffer, J., Sutton, R.I., 1999. Knowing What to do is not enough: turning knowledge into action. California Management Review 42, 83–107.

Pimentel, D., Wilson, C., McCullum, C., Huang, R., Dwen, P., Flack, J., Tran, Q., Saltman, T., Cliff, B., 1997. Economic and environmental benefits of biodiversity. BioScience 47, 747–757.

Pimm, S.L., Raven, P., 2000. Extinction by numbers. Nature 403, 843–844.

Pleguezuelos, J.M., 1997. Distribución y biogeografía de los anfibios y reptiles en España y Portugal. Universidad de Granada, Granada, España.

Pounds, J.A., Fogden, M.P.L., Campbell, J.H., 1999. Biological response to climate change on a tropical mountain. Nature 398, 611–615.

Prendergast, J.R., Eversham, B.C., 1995. Butterfly diversity in southern Britain: hotspot losses since 1930. Biological Conservation 72, 109–114.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C., Gibbons, D.W., 1993. Rare species, the coincidence of diversity hotspots and conservation strategies. Nature 365, 335–337.

Pressey, R.L., Logan, V.S., 1998. Size of selection units for future reserves and its influence on actual vs targeted representation of features: a case study in Western New South Wales. Biological Conservation 85, 305–319.

Pressey, R.L., Nicholls, A.O., 1989. Efficiency in conservation evaluation: scoring vs. iterative approaches. Biological Conservation 50, 199–218.

Rabinowitz, D., 1981. Seven forms of rarity. In: Synge, H. (Ed.), The Biological Aspects of Rare Plant Conservation. Wiley, Chichester, pp. 205–217.

Reid, W.V., 1998. Biodiversity hotspots. Trends in Ecology and Evolution 13, 275–280.

Rey Benayas, J.M., de la Montaña, E., 2003. Identifying high-value vertebrate diversity areas for strengthening nature conservation. Biological Conservation 114, 357–370.

Rey Benayas, J.M., Scheiner, S.M., 2002. Plant diversity, biogeography, and environment in Iberia: patterns and inferred mechanisms. Journal of Vegetation Science 13, 245–258.

Rey Benayas, J.M., Scheiner, S.M., García Sánchez-Colomer, M., Levassor, C., 1999. Commonness and rarity: theory and application of a new model to Mediterranean montane grasslands. Conservation Ecology 3(5) (online) .

Reyers, B., van Jaarsveld, A.S., Kruger, M., 2000. Complementarity as a biodiversity indicator strategy. Proceedings of the Royal Society of Biological Science B 267, 505–513.

Richter, B.D., Braun, D.P., Mendelson, M.A., Master, L.L., 1997. Threats to imperiled freshwater fauna. Conservation Biology 11, 1081–1093.

Ricketts, T.H., Dinerstein, E., Olson, D.M., Loucks, C.J., Eichbaum, W., DellaSala, D., Kavanagh, K., Hedao, P., Hurley, P.T., Carney, K.M., Abell, R., Walters, S., 1999. Terrestrial Ecoregions of North America: A Conservation Assessment. Island Press, Washington, USA.

Rodrigues, A.S., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Cowling, R.M., Fishpool, L.D., Da Fonseca, G.A., Gaston, K.J., Hoffmann, M., Long, J.S., Marquet, P.A., Pilgrim, J.D., Pressey, R.L., Schipper, J., Sechrest, W., Stuart, S.N., Underhill, L.G., Waller, R.W., Watts, M.E., Yan, X., 2004. Effectiveness of the global protected area network in representing species diversity. Nature 428 (6983), 640–643.

Rossel, C., 2003. Recuperación de hábitats para la fauna y prevención de impactos. In: Rey Benayas, J.M., Espigares, T., Nicolau, J.M. (Eds.), Restauración de ecosistemas mediterráneos. Servicio de Publicaciones de la Universidad de Alcalá, Alcalá de Henares, Spain.

Rossi, E., Kuitunen, M., 1996. Ranking of habitats for the assessment of ecological impact in land use planning. Biological Conservation 77, 227–234.

Rouse, J.D., Bishop, C.A., Struger, J., 1999. Nitrogen pollution: an assessment of its threat to amphibian survival. Environmental Health Perspectives 107, 799–803.

Sarkar, S., Margules, C., 2002. Operationalizing biodiversity for conservation planning. Journal of Bioscience 27, 299–308.

Scott, J.M., Davis, F.W., McGhie, R.G., Wright, R.G., Groves, C., Estes, J., 2001. Nature reserves: do they capture the full range of America's biological diversity? Ecological Applications 11, 999–1007.

Semlitsch, R.D., 2000. Principles for management of aquatic-breeding amphibians. Journal of Wildlife Management 64, 615–631.

Semlitsch, R.D., Bodie, J.R., 2003. Biological criteria for buffer zones around wetlands and riparian habitats for amphibians and reptiles. Conservation Biology 17 (5), 1219–1228.

Stow, A.J., Sunnucks, P., Briscoe, D.A., Gardner, M.G., 2001. The impact of habitat fragmentation on dispersal of Cunningham's skink (*Egernia cunninghami*): evidence from allelic and genotypic analyses of microsatellites. Molecular ecology 10 (4), 867–878.

Terborgh, J., 1999. Requiem for Nature. Island Press, Washington, USA.

Tilman, D., 1999. The ecological consequences of changes in biodiversity: a search for general principles. Ecology 80, 1455–1474.

Treweek, J.R., Hankard, P., Roy, D.B., Thompson, S., 1998. Scope for strategic ecological assessment of trunk-road development in England with respect to potential impacts on lowland heathland, the Darford warbler (*Sylvia undata*) and the sand lizard (*Lacerta agilis*). Journal of Environmental Management 53, 147–163.

Usher, M.B., 1986. Wildlife Conservation Evaluation. Chapman & Hall, London, UK.

Vargas, J.M., Real, R., 1997. Biogeografía de los anfibios y reptiles de la península ibérica. In: Pleguezuelos, J.M. (Ed.), Distribución y biogeografía de los anfibios y reptiles en España y Portugal Monografías de Herpetología 3. Universidad de Granada, Asociación Herpetológica Española, pp. 309–320.

Vargas, J.M., Real, R., Guerrero, J.C., 1998. Biogeographical regions of the Iberian Peninsula based on freshwater fish and amphibian distributions. Ecography 21, 371–382.

Virolainen, K.M., Nättinen, K., Suhonen, J., Kuitunen, M., 2001. Selecting herb-rich forest networks to protect different measures of biodiversity. Ecological Applications 11, 411–420.

Williams, P.H., Araujo, M.B., 2000. Using probability of persistence to identify important areas for biodiversity conservation. Proceedings of the Royal Society London B 267, 1959–1966.

Williams, P.H., Humphries, C.J., Vane-Wright, R.L., 1991. Measuring biodiversity: taxonomic relatedness for conservation priorities. Australian Systematic Botany 4, 665–679.

Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C., Pressey, R., 1996. A comparison of richness hotspots, rarity hotspots and complementarity areas for conserving diversity using British birds. Conservation Biology 10, 155–174.

Williams, P.H., Margules, C.R., Hilbert, D.W., 2002. Data requirements and data sources for biodiversity priority area selection. Journal of Bioscience 27, 327–338.

Willson, J.D., Dorcas, M.E., 2003. Effects of habitat disturbance on stream salamanders: implications for buffer zones and watershed management. Conservation Biology 17 (3), 763–771.

Woodford, J.E., Meyer, M.W., 2003. Impact of lakeshore development on green frog abundance. Biological Conservation 110 (2), 277–284.