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Title: Landscape heterogeneity as a surrogate of biodiversity in mountain systems: what is the most appropriate spatial analytical unit?

Article Type: Research paper

Keywords: habitat diversity; mammals; birds; reptiles; terrestrial vertebrates; watersheds

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Response to Reviewers: Dear Dr. Petina Lesley Pert, Associate Editor of the journal Ecological Indicators:

We are pleased to submit a revised version of the manuscript ECOLIND-7602R1 "Landscape heterogeneity as a surrogate of biodiversity in mountain systems: what is the most appropriate spatial analytical unit?"

by Paula García-Llamas, Leonor Calvo, Marcelino De la Cruz and Susana Suárez-Seoane which has been accepted for publication in *Ecological Indicators*, according to the Editorial Office.

We would like to thank you and the reviewers for valuable comments and insightful suggestions which have allowed to improve an earlier version of our manuscript.

Following the Editorial Office indications, we provide the manuscript source file in .doc format according to the Guide for Authors' instructions. All figures are submitted in TIFF format. We also provide highlights and the graphical abstract in PDF format. Further we have implemented the minor comments provided by the reviewers. Our responses to the comments are appended to this letter.

This manuscript is based on original research carried out by the authors, which is not being considered for publication elsewhere. All authors agree with the contents of the manuscript and its submission to *Ecological Indicators*. No part of the research has been published in any form elsewhere. All appropriate ethics and other approvals were obtained for the research and acknowledged in the manuscript. Funding sources have also been acknowledged in the manuscript. We apply for color version just for online publication.

Sincerely,
The authors

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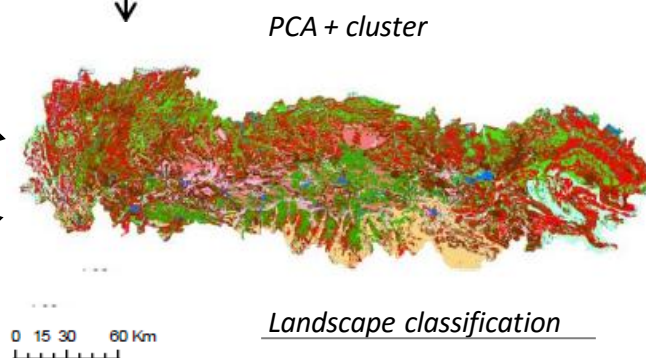
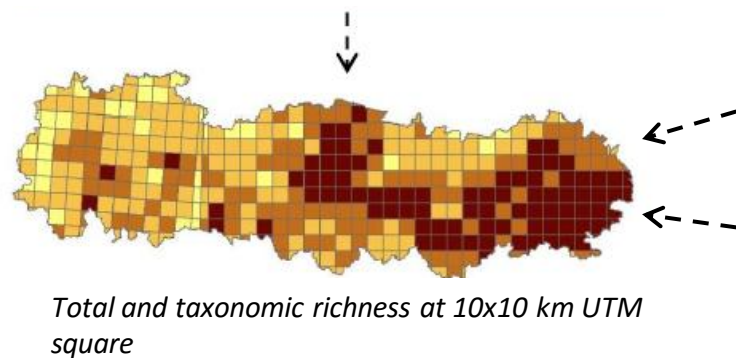
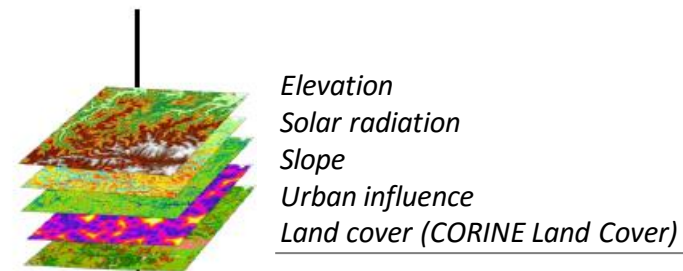
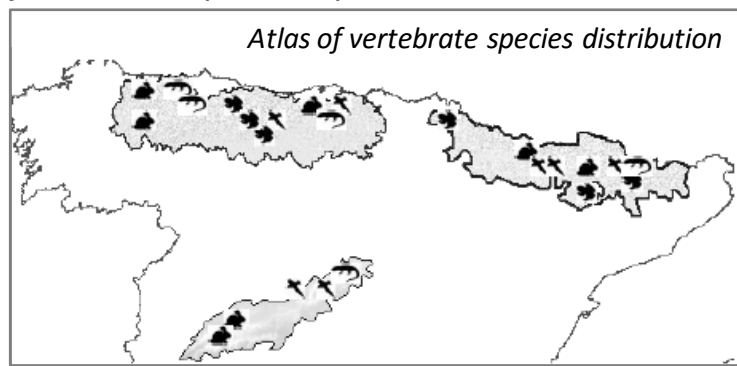
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Graphical Abstract(for review)



Total and taxonomic richness (S) at watershed level

Total and taxonomic richness (S) at square window level

Landscape heterogeneity (H'_β) at square window level

Landscape heterogeneity (H'_β) at watershed level

Square window $\rightarrow S_i \sim H'_\beta + \log(\text{Area}) + \text{mountain system}$

Watershed $\rightarrow S_i \sim H'_\beta + \log(\text{Area}) + \text{mountain system}$

Highlights

- Landscape heterogeneity turned to be a good surrogate of terrestrial vertebrate species richness, although conditioned by the selected analytical approach and landscape data.
- The eco-geographic approach offered a good analytical scheme for assessing relationships between landscape heterogeneity and species richness in biodiversity studies
- The size of arbitrary analytical units affects predictive power of landscape heterogeneity, this increasing when increasing the analytical units' size.

Landscape heterogeneity as a surrogate of biodiversity in mountain systems: what is the most appropriate spatial analytical unit?

Paula García-Llamas · Leonor Calvo · Marcelino De la Cruz · Susana Suárez-Seoane

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Keywords: habitat diversity; mammals; birds; reptiles; terrestrial vertebrates; watersheds.

25 **1. Introduction**

Loss of biodiversity is one of the main impacts of land use change, and is associated with landscape fragmentation and habitat loss over recent decades (Lindenmayer et al 2002; Herrando et al 2014). Knowledge of the factors driving biodiversity patterns has become a priority for researchers and conservation practitioners (Morelli et al 2013).
30 Considerable efforts have been made to develop and improve methods for evaluating components of current biodiversity to enable the identification of priorities for conservation (Priego-Santander et al 2013). Conservation strategies require the quantification of biodiversity, although time and cost limitations of biodiversity data collection make this a challenging task (Ewers et al 2005). Thus, the development of
35 biodiversity indicators that reduce the effort of biodiversity estimation, therefore speeding up the decision-making process, has become a priority for conservation biologists (Rossi and van Halder 2010; Laurila-Pant et al 2015).

There is a large body of literature in which different environmental variables (e.g., climate, land cover (Kivinen et al 2007; Mehr et al 2011), topography (Krömer et al
40 2013; Yu et al 2015), soil properties (Medinski et al 2010), human population density or habitat diversity (Moreno-Rueda and Pizarro 2007) have been used to make spatial predictions of species richness. Currently, there is increasing agreement about the consideration of landscape as the most pertinent level for biodiversity management actions (Walz 2011), since landscape-based evaluations provide a larger-scale
45 perspective of ecological processes than traditional site-based ones (Pino et al 2000). The use of landscape metrics as a proxy of species richness has become a popular approach (Lindenmayer et al 2002; Rossi and van Halder 2010), made easier by the continuous development of remote sensing techniques and Geographic Information Systems (GIS) (Wagner and Fortin 1987). Amongst the large number of landscape

50 metrics used as biodiversity surrogates, landscape heterogeneity is gaining valuable
recognition within conservation strategies (Walz 2011). It is generally accepted that
landscape heterogeneity is positively related to ecological niche diversity (Katayama et
al 2014). Furthermore, landscape heterogeneity plays an important role in population
dynamics, as it may control dispersal rates, movement patterns or foraging strategies
55 (Johnson et al 1992), which suggests some connection between landscape heterogeneity
and species richness. Nevertheless, the estimation of biodiversity from landscape
metrics is often affected by the methods employed to observe, analyse and process
landscape patterns (Walz 2011). Since landscape metrics, including landscape
heterogeneity, describe geometric and spatial properties of landscape (Gimona et al
60 2009), the ecological response emerging from landscape analyses might be conditioned
by the shape (i.e., delineated boundaries; Moser et al 2007; Cushman and Mc Garigal
2008) or size (i.e., spatial scale) of the analytical unit used for landscape quantification
(Weibull et al 2000; Plexida et al 2014; Ye et al 2015).

The landscape is a continuum, but for practical reasons it must be split into spatial
65 analytical units providing a frame for landscape metrics quantification. This is often
rather arbitrary (Verberk et al 2006; Walz 2011). Difficulties arise as differently
delineated spatial analytical units might provide different statistical relationships for the
same ecological process, making the interpretation and applicability of landscape
metrics estimations challenging (Saura and Martínez-Millán 2001). Most studies
70 addressing landscape heterogeneity as a surrogate of species richness (e.g. Atauri and de
Lucio 2001; Moreno-Rueda and Pizarro 2007; Schindler et al 2013) are based on a
systematic partition of the landscape using arbitrarily defined spatial analytical units,
such as UTM grids or circular buffers. However, the use of spatial analytical units with
eco-geographic meaning could also provide a useful approach when predicting

75 biodiversity, as displayed by Priego-Santander et al (2013). This study showed the
potential of landscape heterogeneity as a predictor of plant richness on the basis of land
units defined from geomorphology, geology, relief, climate, soil and land cover
features. Watersheds are increasingly being used in environmental modelling and
management, as they represent integrated socio-ecological (Mayer et al 2014),
80 geomorphological (Montgomery et al 1995) and multifunctional (Karadağ 2013) units
with potential application for analyses at multiple scales (Tinker et al 1998). For
example, watersheds have been considered as operational spatial units to assess the
relationship between soil erosion and regional landscape change (Li and Zhou 2015), to
identify and manage natural resources (Baloch and Tanik 2008) and to analyse land
85 cover change (Mendoza et al 2011; Álvarez-Martínez et al 2014). However, the
application of eco-geographical units, including watersheds, as spatial analytical units in
biodiversity modelling is under-evaluated. There is a clear need to explore the role of
eco-geographical spatial analytical unit approaches as an alternative to traditional
arbitrary ones in biodiversity studies.

90 Similarly, the influence of the size of the spatial analytical unit on the detection of
relationships between landscape heterogeneity and species richness has been
highlighted in different studies (e.g. Tews et al 2004; Morelli et al 2013; Schindler et al
2013). Relationships emerging from the use of a particular spatial analytical-unit size
are not necessarily consistent across different sizes. This is a consequence of the
95 operational scale at which organisms interact with their environment (Tews et al 2004).
Taxa with a higher mobility and a strong demand for space are expected to be more
influenced by larger landscape surface areas than smaller or sedentary species (Suárez-
Seoane and Baudry 2002; Schindler et al 2013). Thus, multiscale analyses are required

to detect the scale at which ecological phenomena leave their biological signal (Lechner
100 et al 2012).

In comparison with other systems, the higher environmental variability found in
mountains provides a large diversity of habitats, and therefore landscape heterogeneity
(Jobbágy et al 1996), which allows for higher species richness (Dufour et al 2006). In
fact, mountains have been largely recognized as important sources of biodiversity
105 worldwide (La Sorte and Jetz 2010). Meanwhile, they are highly prone to biodiversity
loss due to their susceptibility to human and natural disturbances (Martinelli et al 2007).
Therefore, the need to explore potential biodiversity predictors, such as landscape
heterogeneity, becomes particularly relevant in mountain systems.

The main goal of this study was to evaluate the potential of landscape heterogeneity
110 as an indicator of species richness, assessing the effect of different methodological
choices on the detection of significant ecological relationships. Specifically, we
assessed the impact of using analytical units of different shape and size, i.e. following
an eco-geographic (watersheds) versus an arbitrary (square windows of different size)
spatial analytical approach, to quantify species richness and landscape heterogeneity. As
115 a study case, we evaluated the effect of the different choices on the relationship between
landscape heterogeneity and species richness of vertebrates (total and per taxonomic
group) in three mountain systems with different biogeographical influences in Spain.

2. Material and methods

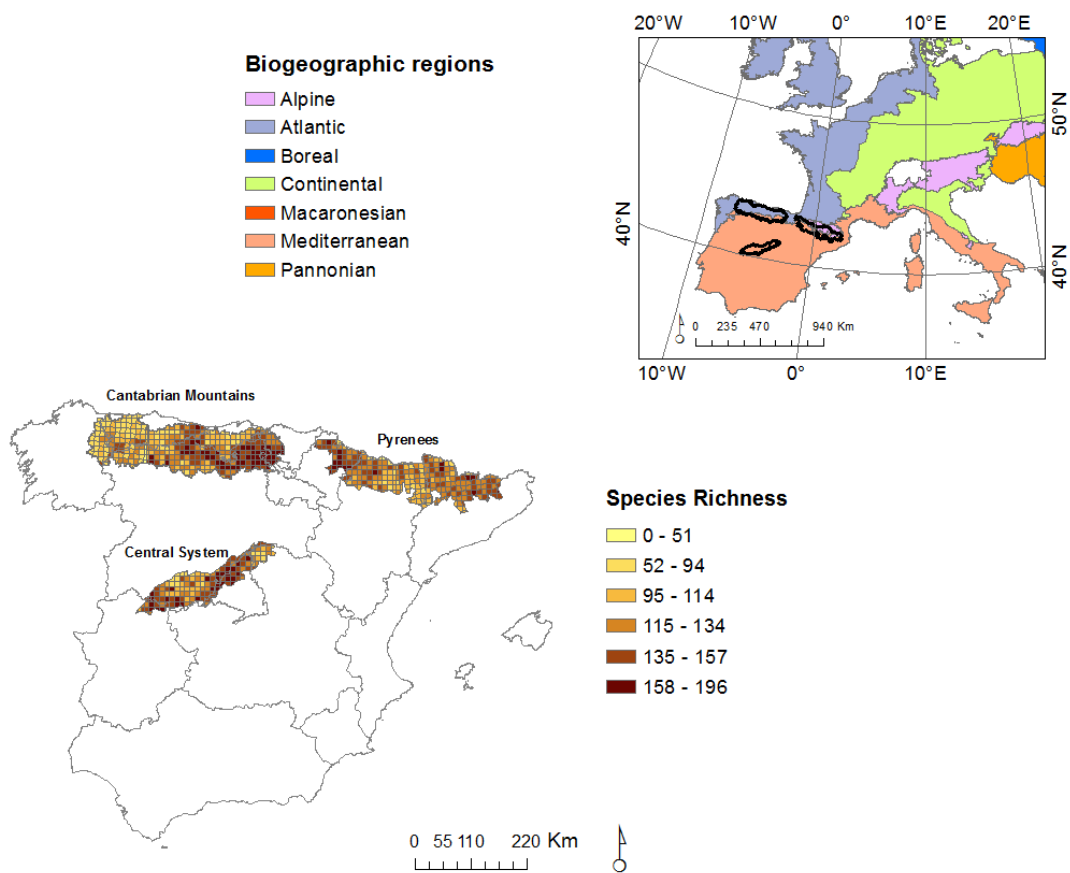
120 *2.1 Study area*

The study area includes three mountain ranges located across Spain (Figure 1): the
Cantabrian Mountains (CM), the Central System (CS) and the Spanish Pyrenees (SP).

CM represent the transition between Atlantic and Mediterranean regions, CS is a typical example of the Mediterranean area and SP are located at the transition between the
125 Mediterranean, Atlantic and Alpine regions (Rivas-Martínez et al 1987). All of them present a great diversity of climatic conditions. Annual rainfall ranges from 700 to 2400 mm in CM, 500 to 1500 mm in CS and 600 to 1400 mm in SP. Mean annual temperature ranges from -2.5 to 22.5 °C in all cases. These mountain systems are of particular interest as they are partially or totally included in the Mediterranean Basin,
130 often considered as a biodiversity hotspot (Myers et al 2000; Maiorano et al 2013), and they house a wide variety of ecosystems, habitats and endemic species. In particular, CM and SP represent the southern limit of a wide range of species of Euro-Siberian origin and the northern-most distribution of species of Iberian and Ibero-African origin (Martínez-Rica and Recoder 1990; Morán-Ordóñez 2012), while CS has been
135 recognized as a migratory route and speciation centre (López-Sáez et al 2014).

In CM, where altitude ranges from sea level up to 2650 m.a.s.l., dominant land covers are crop fields, in lowlands, and natural formations, such as heathlands, scrublands and deciduous forests (dominated by *Fagus sylvatica*, *Betula pubescens*, *Quercus petraea* and *Q. robur*, on northern slopes or by *Q. pyrenaica* and *Q. ilex*, on
140 southern slopes), in mid-highlands (Morán-Ordóñez et al 2011). Natural grasslands mainly cover areas at the highest altitudes. In CS, altitude ranges from 280 to 2592 m.a.s.l. The landscape is dominated by forests of *Q. pyrenaica*, *Q. ilex* and *Q. suber* (especially in the western sector) and *Pinus sylvestris* and *P. nigra* eastwards (López-Sáez et al 2014). Shrublands and heathlands of *Cytisus oromediterraneus*,
145 *Echinopartum ibericum*, *E. barnadesii* or *Erica australis* constitute the main features of the landscape above 1600 m.a.s.l. (Rivas-Martínez et al 1987), while grasslands dominate in the areas at the highest altitudes (López-Sáez et al 2014). In SP, the altitude

ranges from 100 to 3404 m.a.s.l. The landscape mosaic is mainly dominated by natural formations such as *Quercus pubescens* and *Q. ilex* and *Pinus sylvestris* and *Fagus sylvatica* forests, covering, respectively, sunny and shaded slopes, while *P. uncinata* forests appear above 1800 m.a.s.l. (Lasanta-Martínez et al 2005; Roura-Pascual et al 2005), along with scrublands and natural grasslands. Crop fields can be found in valley bottoms.



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Figure 1. Study area: The Cantabrian Mountains, the Central System and the Spanish Pyrenees. Information on biogeographic regions was obtained from the Spanish Ministry of Agriculture, Food and Environment (<http://www.magrama.gob.es/>). Additionally, total species richness in the three corresponding mountain systems.

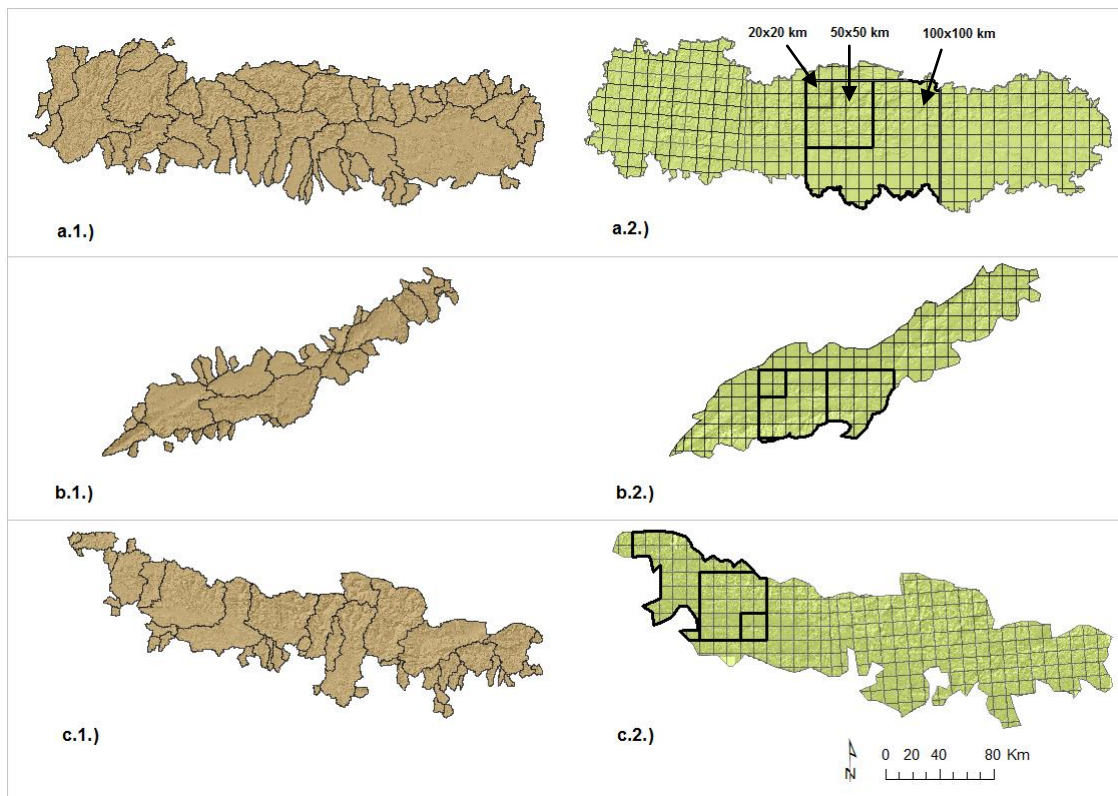
160 Information on species richness was obtained from the official database of vertebrates of Spain (Ministry of Agriculture, Food and Environment 2012; www.magrama.gob.es).

2.2 Spatial analytical unit approaches

To assess the influence of the spatial analytical unit chosen to evaluate the relationship between landscape heterogeneity and species richness, we considered two different approaches that consisted of splitting the study area according to either geographic-ecological or arbitrary criteria (Figure 2). The first approach (the “eco-geographic approach”) involved the delineation of a set of watersheds that were derived from a Digital Elevation Model (DEM) at 25 m resolution (www.ign.es) using ArcGIS 10.2 (Esri, 2014), for each mountain system. Only watersheds of the highest order (i.e., fourth order) were considered, since those of lower order were smaller than the 10x10 km UTM square (the basic resolution of the analyses, constrained by species data availability). Watersheds covering only one 10x10 km UTM square were also excluded from the analyses since they did not allow for landscape heterogeneity quantification. As a result, we selected 76 watersheds for further analyses (39 in CM, 13 in SC and 24 in SP); mean area size was $582 \pm 652 \text{ km}^2$ (range 102.33-4227.79 km^2).

The second spatial analytical unit approach (the “arbitrary approach”) involved the definition of square windows of different size (20x20 km, 50x50 km, 100x100 km), which were obtained by successive aggregation of the original UTM grid system of 10x10 km. The different sizes made it possible to assess the role of scale on the performance of spatial analytical units. The 20x20 km and 50x50 km window sizes were chosen as proximal sizes to the mean \pm SD watershed area. Furthermore, conservation planning in Spain is often conducted at a regional or sub-regional scale, thus we selected the 100x100 km window size as a proximal size to a sub-regional spatial scale. The need of adapting square windows to the biogeographic limits of the mountain systems imply that coverage was not complete for some windows located across the borders of the study area. Incomplete windows are usually excluded from the

analyses (Li et al 2017) in order to avoid potential bias on species richness and landscape heterogeneity. However, aiming to include as much as possible of the area of the mountain systems, we only excluded from further analyses windows with a coverage lower than 75% (at 20x20 km and 50x50 km window size) and 45% (at 100x100km window size). This resulted in 154 windows of 20x20 km (75 in CM, 28 in SC and 51 in SP), 20 of 50x50 km (10 in CM, 4 in SC and 6 in SP) and 12 of 100x100 km (5 in CM, 3 in SC and 4 in SP).



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Figure 2. Spatial analytical units: watersheds on the left (a.1 to c.1) vs. square windows of different size (20x20 km, 50x50 km and 100x100 km) on the right (a.2 to c.2) in the three mountain systems studied: a) the Cantabrian Mountains; b) the Central System; c) the Spanish Pyrenees.

200

2.3 Vertebrate species data

Distribution of terrestrial vertebrate species was obtained from the official database of vertebrates of Spain (Ministry of Agriculture, Food and Environment 2012;

205 www.magrama.gob.es), which combines field records gathered by volunteers with information from published sources. The data consist of species presence and absence locations for the period 1980-2007, collected on the basis of direct observations and indirect methods (pellets, tracks, bed sites) and assembled in a regular UTM grid system of 10x10 km (Pleguezuelos et al 2002; Martí and del Moral 2003; Palomo et al 2007).

210 Species richness was calculated for each 10x10 km UTM square as: (i) “total richness” or total number of species; and (ii) “taxonomic richness” or the number of species per taxonomic group (mammals, breeding birds, reptiles and amphibians). Similarly, we calculated the relative value of species richness (number of species per 100km²), both total richness and species richness per taxonomic group for each
215 mountain system (See Figure 1 and Table A1 from the appendix). Only squares with at least 75% coverage of their area included within the limits of each mountain system were considered for analyses, resulting in a set of 624 squares. Based on these squares, total and species richness per taxonomic group were also computed for both watersheds and square windows of different size, by aggregating the original information from the
220 10x10 km UTM grid system. The 10x10 km squares intersecting boundaries between watersheds were fully considered as part of the watershed if they overlapped at least 50% with the watershed area.

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2.4 Estimators of landscape heterogeneity

Landscape heterogeneity was calculated on the basis of a landscape classification elaborated by integrating variables accounting for topography, urban influence and land cover. Land cover was obtained from the categorical map CORINE Land Cover 2006 (derived from LANDSAT at 30m; <http://land.copernicus.eu/pan-european/corine-land-cover>), which had previously been converted into independent continuous variables by calculating the proportion covered by each class. Topographic variables consisted of elevation, solar radiation and slope. They were derived from a Digital Elevation Model (DEM) (25 m spatial resolution), obtained from the Spanish Geographic Institute (www.ign.es) and resampled at the resolution of the land cover variables (i.e. 30 m), by applying the nearest neighbor method. Urban influence was measured as the Euclidean distance from each 30 m pixel to the nearest settlement, using input data downloaded from the Spanish Geographic Institute site (www.ign.es) at 1:25000 and 1:200000 spatial resolution. Landscape classification consisted of running a Principal Component Analysis (PCA) over the pool of variables, followed by an ISODATA unsupervised classification accomplished by using the Iso Cluster unsupervised classification function in ArcGis 10.2 (Esri, 2014); this aggregated pixels into coherent classes. The number of classes was defined aiming to represent at best the ecological framework of the study area according to our expert knowledge. To do this, classes resulting from the classification process were characterized by using thematic information related to topography, urban influence and land cover and when necessary, they were combined or reclassified. The classification process led to 11 landscape classes in the CM, 13 in the CS and 8 in the SP (Figure A1 and Table A2 from the appendix). The overall accuracy of classifications and user's and producer's accuracy per class were evaluated through confusion error matrices (Congalton, 2001) by using thematic information related to

topography, urban influence, land cover and ortho-photographs (years 2006 to 2009, scale 1:5000 to 1:10000). Overall accuracy was 88 % for the CM and the SP and 84 % for the CS (Table A3 from appendix). See García-Llamas et al (2016) for more

255 technical details on the landscape classification method.

Landscape heterogeneity was estimated as beta diversity (H'_β ; Eq. 1) for each spatial analytical unit (watershed or regular window) using an additive model based on the comparison of both alpha and gamma diversity (Lande 1996; Tárrega et al 1997; Jost 2006, 2007).

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$$H'_\beta = H'_\gamma - \frac{\sum_{i=1}^N H'_{\alpha_i}}{N} \quad (\text{Eq. 1})$$

where H'_γ is gamma diversity (i.e., the diversity of landscape classes at each spatial analytical unit) and H'_{α_i} is alpha diversity (i.e., the diversity of landscape classes at each 10x10 km UTM square within each spatial analytical unit). Both H'_γ and H'_{α_i} were
265 calculated using the Shannon diversity index (Shannon 1948) (Eq. 2),

$$H' = - \sum_{i=1}^R p_i \ln p_i \quad (\text{Eq. 2})$$

where p_i is the proportion of each landscape class within the spatial analytical unit or
270 the 10x10 km UTM square, respectively.

All analyses were done in ArcGIS 10.2 (Esri, 2014) and by using the Patch Analyst extension (Rempel et al 2012).

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2.5 Statistical analyses

To analyse the relationship between landscape heterogeneity and absolute richness (total and species richness per taxonomic group) of terrestrial vertebrates, separate
280 linear models were fitted for each of the considered spatial analytical unit approaches (eco-geographic approach vs. arbitrary approach). In the particular case of mammals and amphibians, richness data were transformed for normality by using the log (x) function.

Species richness was considered in the models as the dependent variable and
285 landscape heterogeneity (H'_{β}) as the predictor. For the arbitrary approach, in order to avoid potential bias due to the incomplete coverage of some windows across borders of mountain systems, we included the logarithm of the area of the square window (i.e. proportion of the square window included within the study area) as an additional predictor in models. For the eco-geographical approach, we included the logarithm of
290 the watershed area as an additional predictor of landscape heterogeneity in order to partial-out the confounding effect of the different areas of individual watersheds (which is expected to affect both species richness and landscape heterogeneity *per se*). To control for intrinsic differences in species richness between the three study areas, "mountain system" was also included in both approaches as a predictor (Gelman and
295 Hill 2006; Seoane 2014). In all cases, we checked model residuals to assess the appropriateness of the model and confirmed the absence of spatial autocorrelation by computing correlograms. Data analyses were carried out with the R 3.1.2 statistical programme (R Development Core Team 2014) using the 'lm' function (R Development Core Team 2014). Correlograms were computed with the 'correlog' function in the 'ncf'
300 R package (Bjornstad 2013).

In addition, the relative effect of the explanatory variables in each model was also assessed by commonality analysis (Legendre and Legendre 2012; Ray-Mukherjee et al 2014). This method, based on the normal R^2 of partial and complete regression models, apportions the variation of the response (species richness) in several fractions: the pure effect of the singular predictors, the shared effect of each pair, triplet or any subset of predictors and the shared effect of the total set of predictors. For this, we used the 'commonalityCoefficients' function in the R package 'Yhat' (Nimon and Oswald 2013).

3. Results

Landscape heterogeneity exerted a positive effect on species richness in all the fitted models. The magnitude and significance of its effects, however, varied according to the spatial analytical unit approach and the taxonomic group (Tables 1 and 2; Tables A4 and A5 from appendix). For the case of the eco-geographic approach, models accounted for more than 53% of the variance of species richness (Table 1). Landscape heterogeneity was a significant predictor of species richness in all cases, except in mammals. It explained between 28% (mammals) and 50% (all taxa together) of the variance of species richness, although between 26% and 40% of this variation was shared with the logarithm of watershed area (Figure 3). Indeed, the logarithm of watershed area had a significant effect on species richness in all taxa considered, explaining between 37 % (reptiles) and 55 % (total richness) of the variance of species richness. The mountain system had a significant effect in all cases, except for both amphibians and for total species richness (Table 1). It explained between 3% (total richness) and 14 % (reptiles) of the total variance (Figure 3).

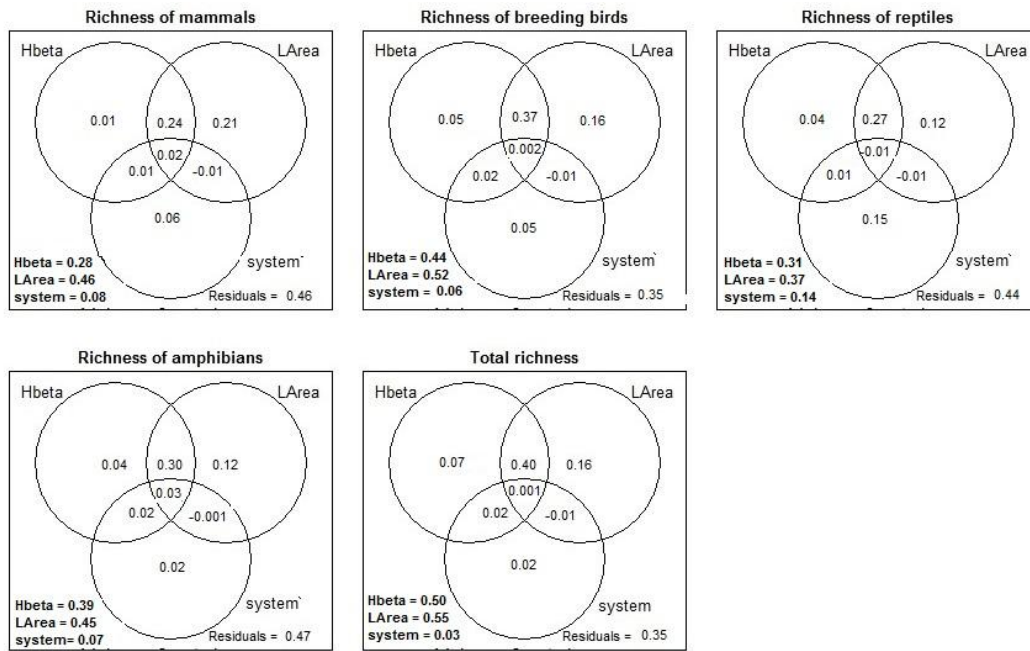
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Table 1. Anova table including: the explained sum of squares (*Sum. Sq.*), *F* value, significance (*Sig.*) and normal coefficients of determination (R^2) of regression linear models achieved when working with watersheds as spatial analytical units and landscape heterogeneity.

	Sum Sq	F value	Sig.	R^2
330	<hr/>			
	Mammal richness			0.54
	H_β	0.09	0.86	n.s.
	log(A)	3.54	32.13	***
	Mountain system	0.95	4.29	*
	<hr/>			
	Breeding bird richness			0.65
335	H_β	1555.90	9.53	**
	log(A)	5167.20	31.64	***
	Mountain system	1782.90	5.46	**
	<hr/>			
	Reptile richness			0.56
	H_β	92.12	5.90	*
	log(A)	301.07	19.28	***
	Mountain system	366.81	11.75	***
340	<hr/>			
	Amphibian richness			0.53
	H_β	33.83	6.55	*
	log(A)	96.45	18.68	***
	Mountain system	14.77	1.43	n.s.
	<hr/>			
	Total richness			0.65
	H_β	7214.00	12.76	***
345	log(A)	17726.00	31.35	***
	Mountain system	1694.00	1.50	n.s.
	<hr/>			

H_β landscape heterogeneity; log(A) logarithm of watershed area; ***significance at level <0.001; **significance at level 0.01; *significance at level 0.05; n.s. no significance.

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Figure 3. Variance partitioning of species richness in watersheds explained by the pure and combined effect of landscape heterogeneity (Hbeta), logarithm of watershed area (Larea) and mountain system (system). Total variance (pure and combined effect) explained by predictors is in bold. For example, for richness of mammals total variance of Hbeta = 0.01 (pure effect) + (0.24+0.02+0.01; shared effect) = 0.28. Negative effects arise from the presence of suppression among predictors due to correlation among variables (Ray-Mukherjee et al 2014) .

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The performance of the arbitrary approach was poorer than the eco-geographical one, with a clear influence of windows size on the significance of relationships. Models accounted for more than 24 % of the variance of species richness at 20x20 km, more than 46 % at 50x50 km and more than 71% at 100x100 km window size. Significant relationships between landscape heterogeneity and species richness were only found for mammals and breeding birds at the largest window size (100x100 km) (Table 2). The percentage of total variation explained by landscape heterogeneity was between 69%

(mammals) and 15% (breeding birds), although between 42% and 10% of this variance was shared with the logarithm of the area of the square window, and between 27% and 28% with the mountain system (Figure 4). The significance of the effect and predictive power of landscape heterogeneity decreased for all taxa for the smallest window sizes (i.e. 50x50 and 20x20 km) (Table 2; Table A4 and Figures A2 and A3 from the appendix).

Further, we found an influence of the mountain system and the logarithm of the area of the square windows, depending on the window size and taxonomic group. Mountain system and area had a significant effect on total richness and richness of all taxonomic groups, at 20x20 km window size, these variables mainly explaining total variance of models. At 50x50 km, only mountain system significantly influenced results, this accounting for most species richness total variance (Table 2, Table A5 and Figures A2 and A3 from the appendix). At 100x100 km, the mountain system and the area significantly influenced breeding birds, reptiles and total species richness (only the area in this case). The mountain system accounted for between 15% (breeding birds) and 59% (reptiles) and the area between 5% (reptiles) and 75% (total richness) of the total variance of models (Table 2; Figure 4 and Table A5 from the appendix).

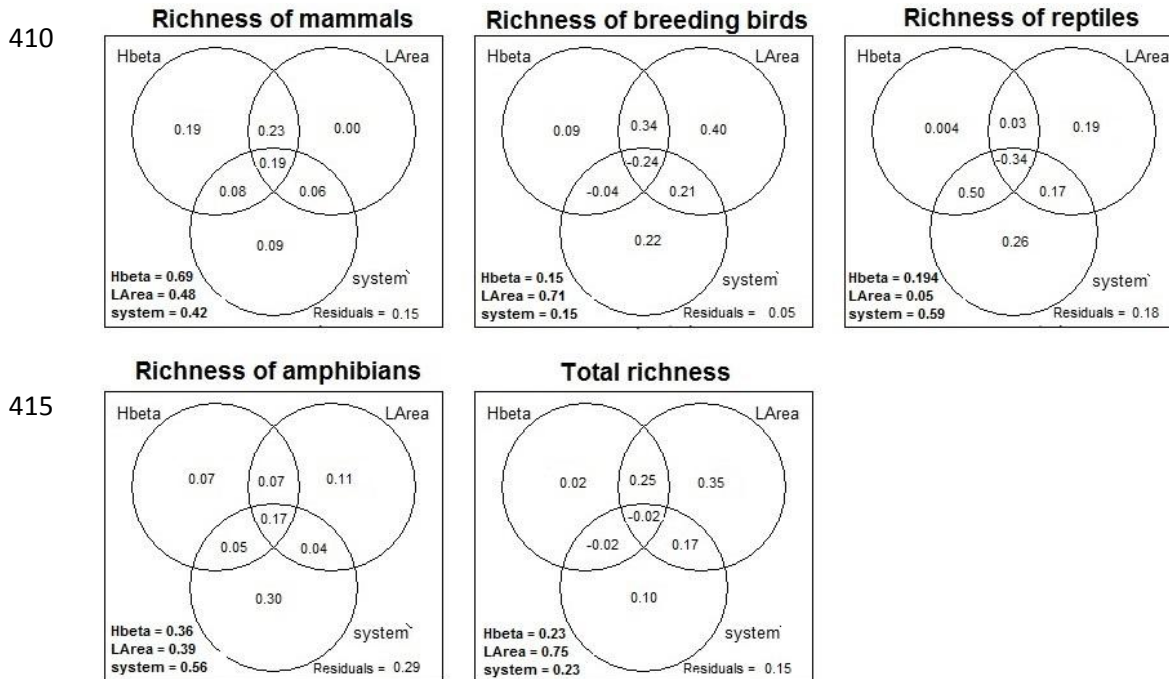
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Table 2. Anova table including: explained sum of squares (*Sum. Sq.*), *F* value, significance (*Sig.*) and normal coefficients of determination (R^2) of linear models based on square windows of different sizes (20x20 km, 50x50 km and 100x100 km).

	20x20			50x50				100x100				
	Sum Sq	F value	Sig.	R^2	Sum Sq	F value	Sig.	R^2	Sum Sq	F value	Sig.	R^2
Mammal richness				0.24				0.46				0.83
H_β	0.16	1.54	n.s.		0.004	0.53	n.s.		0.02	7.38	*	
log(A)	2.08	20.18	***		0.001	0.21	n.s.		0.00	0.001	n.s.	
Mountain system	2.17	10.53	***		0.05	2.72	n.s.		0.01	1.86	n.s.	
Breeding bird richness				0.35				0.47				0.95
H_β	98.50	0.47	n.s.		40.49	0.76	n.s.		163.01	11.14	*	
log(A)	11924.9	56.55	***		142.63	2.68	n.s.		754.46	51.59	***	
Mountain system	3291.1	7.80	***		325.67	3.05	n.s.		410.50	14.04	**	
Reptile richness				0.35				0.85				0.82
H_β	52.77	3.36	n.s.		0.37	0.14	n.s.		0.35	0.16	n.s.	
log(A)	234.07	14.92	***		2.09	0.76	n.s.		16.63	7.40	*	
Mountain system	922.96	2941	***		221.53	40.44	***		22.78	5.07	*	
Amphibian richness				0.27				0.64				0.71
H_β	24.83	3.36	n.s.		15.55	4.42	n.s.		2.79	1.70	n.s.	
log(A)	107.20	7.24	**		4.72	1.98	n.s.		0.45	0.27	n.s.	
Mountain system	139.34	18.83	***		29.21	6.12	*		11.86	3.61	n.s.	
Total richness				0.38				0.60				0.85
H_β	1675.00	2.96	n.s.		190.22	1.99	n.s.		115.82	0.91	n.s.	0
log(A)	37761.00	66.83	***		157.27	1.65	n.s.		2156.07	16.85	**	
Mountain system	5970.00	5.28	**		902.48	4.73	*		584.81	2.28	n.s.	

H_β landscape heterogeneity; ***significance at level <0.001; **significance at level 0.01; *significance at level 0.05; n.s. no significance.



420 **Figure 4.** Variance partitioning of species richness explained by pure and combined
 425 effects of landscape heterogeneity (Hbeta), logarithm of window area (Larea) and
 mountain system (system) for windows of 100x100 km (or less if they intersected the
 limits of the study area). Total variance (pure and combined effect) explained by
 predictors is in bold. For example, for richness of mammals total variance of Hbeta =
 0.19 (pure effect) + (0.23+0.19+0.08; combined effect) = 0.69. Negative effects arise
 from the presence of suppression among predictors due to correlation among variables
 (Ray-Mukherjee et al 2014).

4. Discussion

430 Landscape heterogeneity has been largely recognized as a cost-effective instrument
 to predict biodiversity (Ewers et al 2005), especially in large areas. However, as we
 demonstrated in this study, the predictive power of this indicator depends on the spatial
 analytical unit approach. Thus, the selection of an appropriate analytical framework for
 assessing landscape heterogeneity-species richness relationships requires careful

435 consideration in view of a practical implementation. In this sense, our study advocates
the need to incorporate eco-geographically relevant spatial analytical units, based on
linkages between physical and biological resources and processes (Montgomery et al
1995), rather than arbitrarily delineated ones (typical of traditional approaches) within
the framework of biodiversity studies.

440 The eco-geographic spatial analytical unit approach, based on the use of watersheds,
enabled the detection of significant relationships between landscape heterogeneity and
species richness for almost all taxonomic groups. Generally, species richness depends
on the presence of 'keystone structures' (i.e., distinct spatial landscape structures
providing resources, shelter or goods crucial for species; Tews et al 2004), which are the
445 result of interaction between biotic and abiotic features (e.g. climate, soil type, watering,
human perturbations; Blasi et al 2008) and influence the use of territory by animals
(Mazía et al 2006). In this context, watersheds may better reflect these keystone
structures than arbitrary spatial analytical units, as they represent areas where climate,
hydrology, geomorphology and land use history interact in predictable and repetitive
450 ways, determining the composition and structure of landscapes and their biotic
communities (Karadağ 2013). In fact, for example, watershed characteristics have been
employed to effectively predict the existence of protected habitats (Baattrup-Pedersen et
al 2012).

Although watersheds might offer a good sampling scheme to analyze the
455 relationships between landscape heterogeneity and species richness, variance
partitioning analyses showed that the predictive capacity of the landscape heterogeneity
was mainly associated with the watershed area. Two of the major mechanisms of
increased species richness are the increase in both area (spatial analytical unit size) and
variety of habitat types (here expressed as landscape heterogeneity) (Kohn and Walsh

460 1994). Both mechanisms are mutually complementary, thus one can act as a surrogate
of the effect of the other, making it difficult to discern their direct effect on species
richness patterns (Triantis et al 2003). Consequently, although both area and landscape
heterogeneity partially contributed to explaining species richness separately, their large
combined effect on species richness means that these two variables should be
465 considered in conjunction (Kallimanis et al 2008) when working with eco-geographic
spatial analytical units, in conservation planning.

Despite being widely accepted in landscape analysis (e.g. Nogués-Bravo and
Martínez-Rica 2004; Flick et al 2012) the arbitrary spatial analytical unit approach,
based on the use of square windows, revealed some limitations with regards to
470 modelling landscape heterogeneity-species richness relationships. Only two taxonomic
groups (i.e. mammals and breeding birds) showed significant statistical effects of
landscape heterogeneity. Further, the size of the spatial analytical unit also largely
influenced the predictive capacity of landscape heterogeneity, as demonstrated in other
studies (Morelli et al 2013, Schindler et al 2013; Chambers et al 2016). In our particular
475 case we only found significant relationships between landscape heterogeneity and
species richness at the 100x100 km window size. Such an outcome might be related to
the fact that species attributes (e.g. mobility or dispersal capacity) widely rule the effect
of landscape heterogeneity (Barbaro and Van Halder 2009; Perović et al 2015) and the
spatial scale (i.e. size of the spatial analytical unit) at which this effect emerges (Miguet
480 et al 2016). Consequently, it would be expected that the spatial scale at which landscape
heterogeneity exerts its effect would be larger for taxa with greater mobility or demand
for space (e.g. mammals or birds) than for less mobile ones (e.g. reptiles or
amphibians), because they interact with the landscape over a larger spatial extent
(Schindler et al 2013; Miguet et al 2016). As a result, the lack of explanation of

485 mammal richness within watersheds might suggest that the watersheds in this study are
not large enough to efficiently capture the effect of landscape heterogeneity over species
richness of this taxon. These results emphasise the need to consider the spatial scale
appropriate for both the subject of study and the ecological question posed (Wu 2004),
i.e., it might be associated with the scale at which organisms perceive the landscape.

490 The arbitrariness of the spatial analytical unit influences both the size and shape of
the selected spatial analytical units. Any change in these settings will provide a different
description of the area of analysis, which is linked to the modifiable areal unit problem
(MAUP); and which might affect results of statistical models (Dungan et al 2002; Dark
and Bram 2007; Nouri et al 2017). We tried to minimize the usual bias associated to the
495 scale problem of the MAUP (inflated correlation at higher levels of aggregation; Wong
2009), by estimating both the response (species richness) and the explanatory variables
(landscape heterogeneity) as cumulative figures obtained from the 10 x 10 km raw data,
rather than as averages or any other measure of central tendency of smaller units.

However, contrary to our results with watershed units, for arbitrary windows with a size
500 close to the mean \pm SD watershed area (i.e., 20x20 and 50x50 km window), no
significant effects of landscape heterogeneity on vertebrate species richness were
detected. This inconsistency when changing boundary delineation could be the result of
the different description of the region leading to different analytical results, which is
related to the zoning problem of the MAUP (Jelinski and Wu 1996). Further, when
505 boundaries of spatial analytical units are arbitrarily set, they could not reflect the spatial
structure of the environmental and biological components of the landscape (Wagner and
Fortin 2005). Thus, arbitrary boundaries may mask relationships between landscape
heterogeneity and species richness. As an example, amphibians are usually favored by
landscape heterogeneity, as they use a complex landscape matrix of terrestrial and

510 aquatic habitats during different stages of their life cycles (Mawangi 2010). Further,
different species of amphibians could require different aquatic environments from the
headwater to the lower reaches. However, arbitrary spatial analytical units might not
necessarily encompass both terrestrial and aquatic habitat patches, or all aquatic
environments. Consequently, watershed might be a proper landscape approach to
515 include amphibians breeding, foraging and overwintering habitat patches (Maxell
2009). In the case of reptiles, some studies (e.g., Shipam et al 2004) have highlighted
the importance of preserving watershed-level heterogeneous landscape conditions for
preserving their diversity. In fact, the physiography of watersheds (i.e. slope, elevation
or naturally-occurring aquatic habitats) determines the diversity of canopy covers and
520 aquatic habitats that ultimately influence the variation of humidity and soil mixture, on
which reptiles are highly dependent (Shipam et al 2004). Therefore, analyzing
watersheds, rather than arbitrary spatial analytical units, might give ground for a better
understanding of landscape heterogeneity-reptile richness relationships.

Results on the performance of eco-geographical and arbitrary spatial analytical unit
525 approaches have important implications from a practical perspective. Developing an
eco-geographical approach based on watersheds is not a new approximation in analysis
and conservation management. For example, the unit plan used by the U.S. Forest
service until the mid-1970s was based on watershed delineation (Montgomery et al
1995). Also in the mid-nineties, ecosystem management based on watershed analysis
530 was implemented in the Pacific Northwest federal lands and, on a volunteer basis, on
forested watersheds in Washington (WFPB 1992; 1993). In the case of Spain,
conservation management is generally addressed independently by different regional
autonomous administrations (Morillo and Gómez-Campo 2000), except for some
protected areas (i.e. national parks covering different regions) for which collaborative

535 networks do exist. Consequently, management actions have usually been carried out according to varying arbitrary management units at regional or sub-regional scale, such as administrative boundaries. As our results show, the performance of arbitrary units could not be considered satisfactory. Even for the larger unit size considered (100 x 100 km window), richness for some taxa (and total richness also) appeared unrelated to
540 landscape heterogeneity so, at least with the sizes considered in our study, the arbitrary approach could not be considered an appropriate approximation to a sub-regional scale of management on the ground. On the contrary, our results showed that watersheds, i.e., eco-geographical rather than regular windows, or political or administrative boundaries, might provide a more rational basis for the management of biodiversity, based on
545 potential indicators such as landscape heterogeneity. Thereby, this study revealed the major importance of adopting a trans-bordering and inter-regional management framework that advocates continuous and integrated engagement of all entities involved in decision-making.

Notwithstanding the important role of landscape heterogeneity as an indicator of
550 species richness, caution is urged as high landscape heterogeneity might lead to fragmentation and, may thus have negative effects on biodiversity (Dufлот et al 2014). Further, it should be noted that the detected relationships between landscape heterogeneity and species richness depend on the landscape metrics used for analyses (Cale and Hobbs 1994). Although there is no consensus regarding the most appropriate
555 and informative index for landscape heterogeneity, the Shannon diversity index has been successfully used in several studies (e.g. Pino et al 2000; Oindo et al 2003; Priego-Santander et al 2013; Lee and Martin 2017), along with other metrics such as patch richness, Simpson's diversity index or Simpson's evenness (Schindler et al 2013). We are unaware of the use of a beta-diversity metric in landscape analyses, but its

560 performance in our study gives ground for further consideration in this field. Further,
we claim for the convenience of using a landscape classification adapted to the study
site (i.e. the mountain system), thus allowing the ecological meaning and reliability of
landscape classification to be increased and therefore, the reliability of landscape
indices (Shao and Wu 2008). Additionally, the influence of landscape heterogeneity on
565 species richness may depend not only on the spatial analytical unit approach or the used
landscape metrics, but also on the study site considered (Amano et al 2008; Oliver et al
2010). In this context, the study site might condition the existing pool of species, due to
differences in historical land-uses (Devictor et al 2010), environmental conditions and
biogeographical history (Tamm et al 2016). Therefore, differences in the existing pool
570 of species might determine disparities in the emerging ecological response among study
sites. This might stem from the different ways in which species perceive the landscape
(Farina 2001). In this sense, our analyses were performed in mountain areas, where high
abiotic heterogeneity (i.e. climate or topography) would increase heterogeneity on the
landscape scale, hence resulting in higher site species richness compared to flat areas
575 (Jobbágy et al 1996). In mountain regions, the watershed is a basic eco-geographic unit
that has been used for ecological management (Zhang et al 2014). However, as
watersheds integrate biotic and abiotic processes determining landscape (Karadağ
2013), we could expect the use of watersheds to also be useful for species richness
modeling from landscape heterogeneity, in more homogeneous topographic areas.

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5. Conclusions

Our study has shown how landscape heterogeneity, measured by a beta-diversity
metric, could predict terrestrial vertebrate richness in mountain systems, although
conditioned by the spatial analytical unit approach used for evaluation. This study

585 indicates the high potential of eco-geographical spatial analytical unit approaches, based
on watersheds, in biodiversity studies. The arbitrary spatial analytical units approach
reflects how limiting it could be to apply spatial analytical units unrelated to the
phenomena under study. In connection with this, we corroborate the effect of the size of
arbitrary spatial analytical units on predictive power of landscape heterogeneity, which
590 is generally better at larger sizes. These findings have important practical implications
as they underline the need to consider landscape heterogeneity in biodiversity
conservation strategies. Furthermore, this study offers a valuable cost-effective
framework for environmental management and spatial modeling, with potential to be
adapted for national and global applications. Simultaneously, it makes visible important
595 methodological issues that may affect biodiversity estimations and that should be
considered in decision-making.

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Figure1

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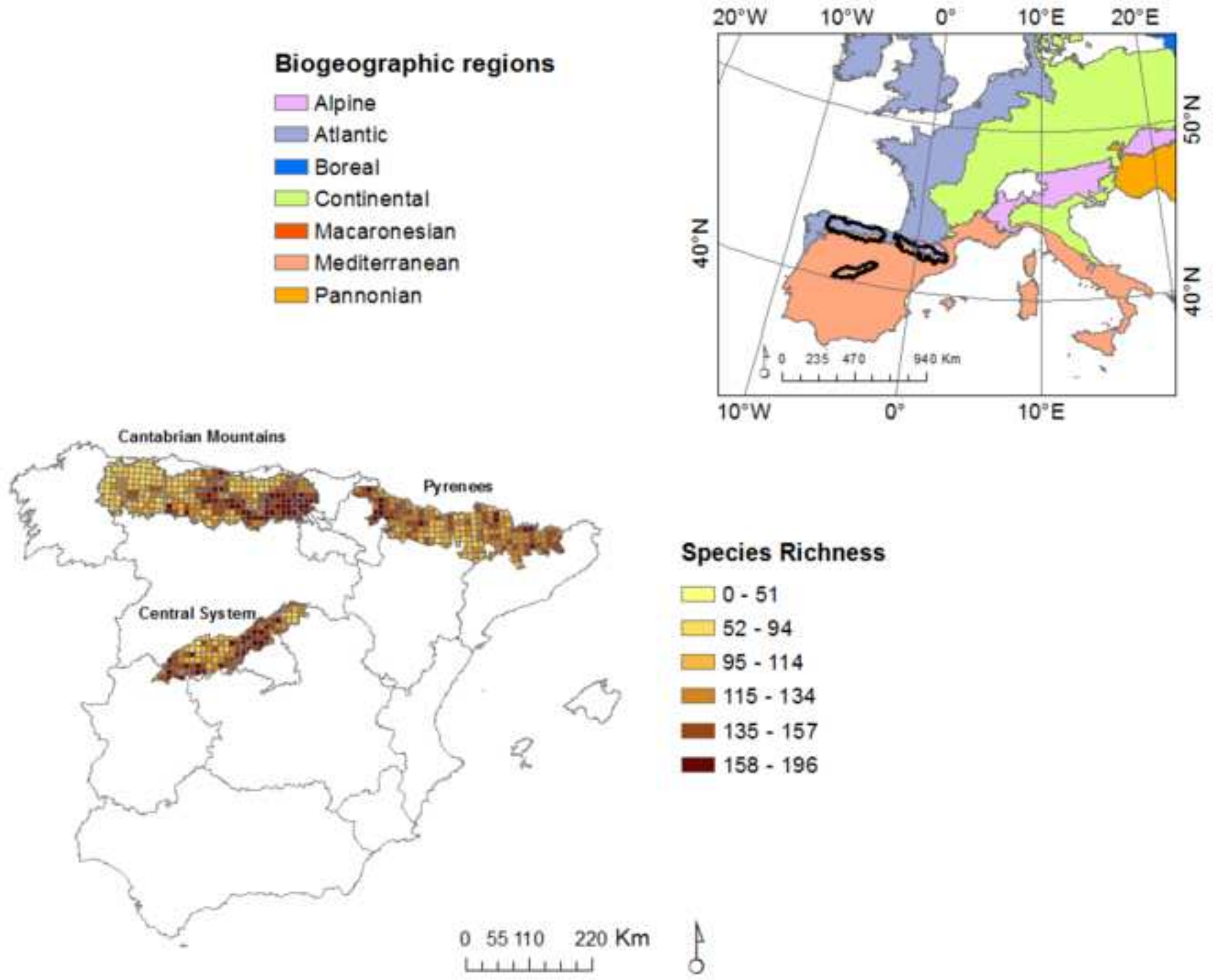


Figure2
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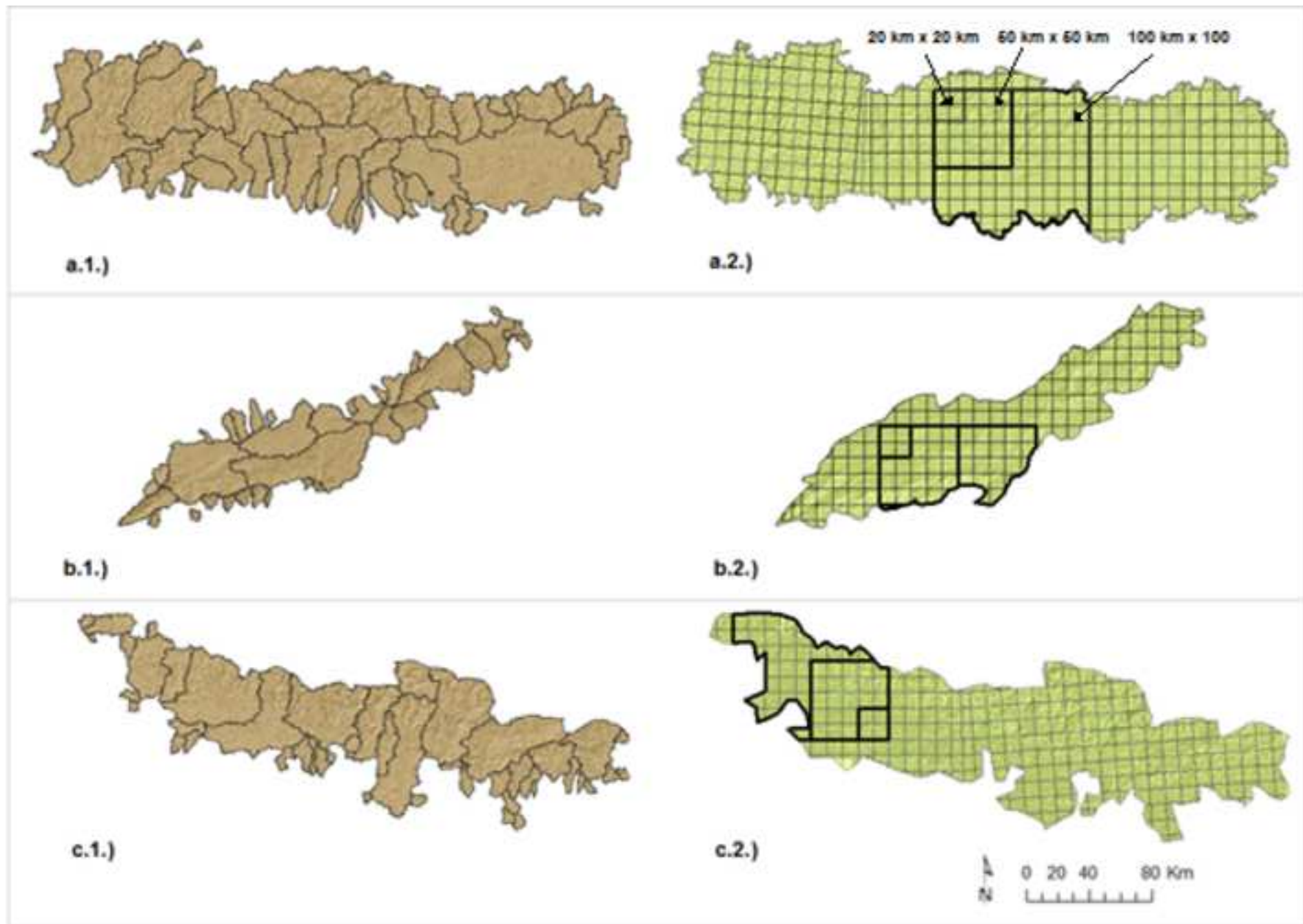


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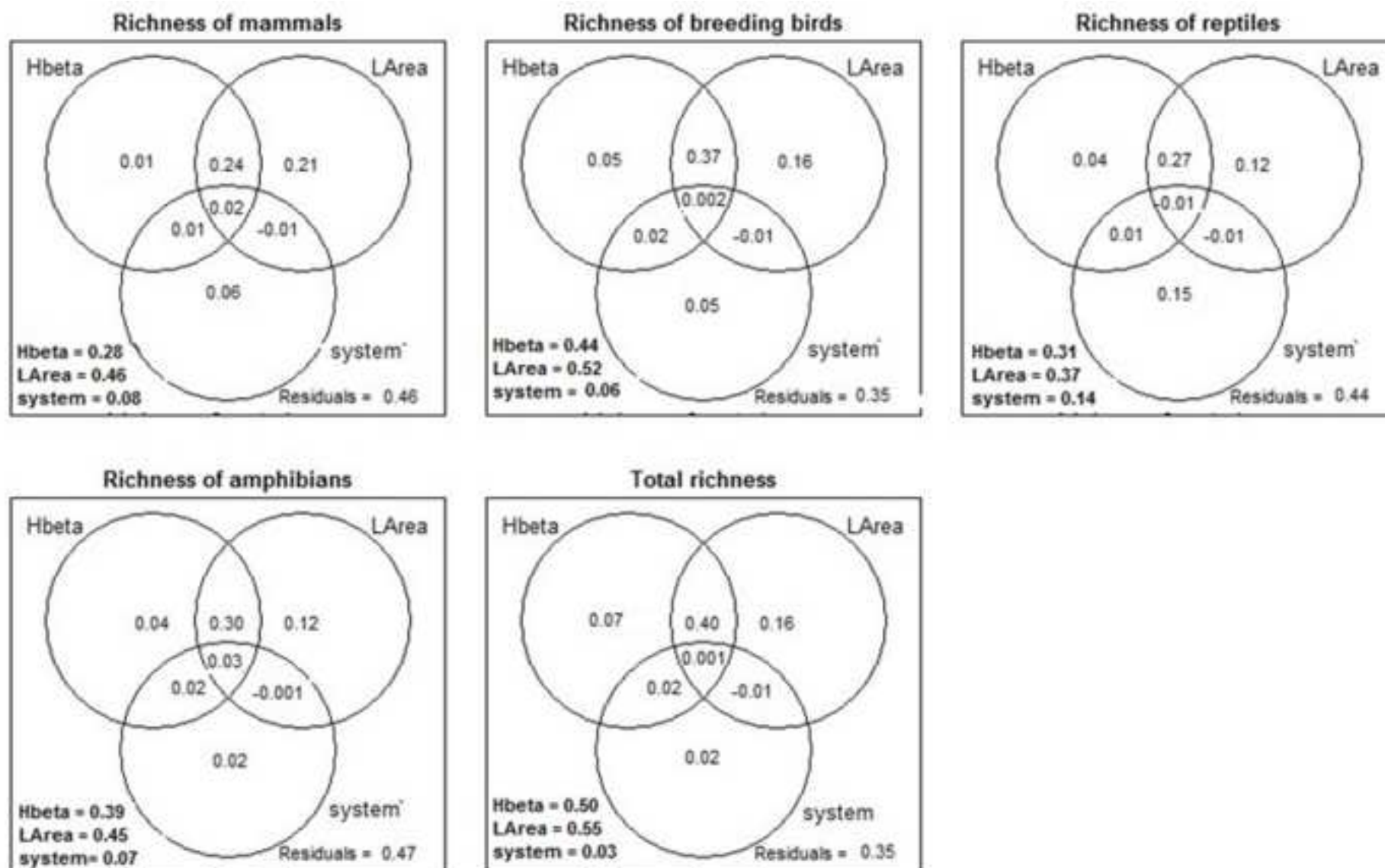


Figure 4

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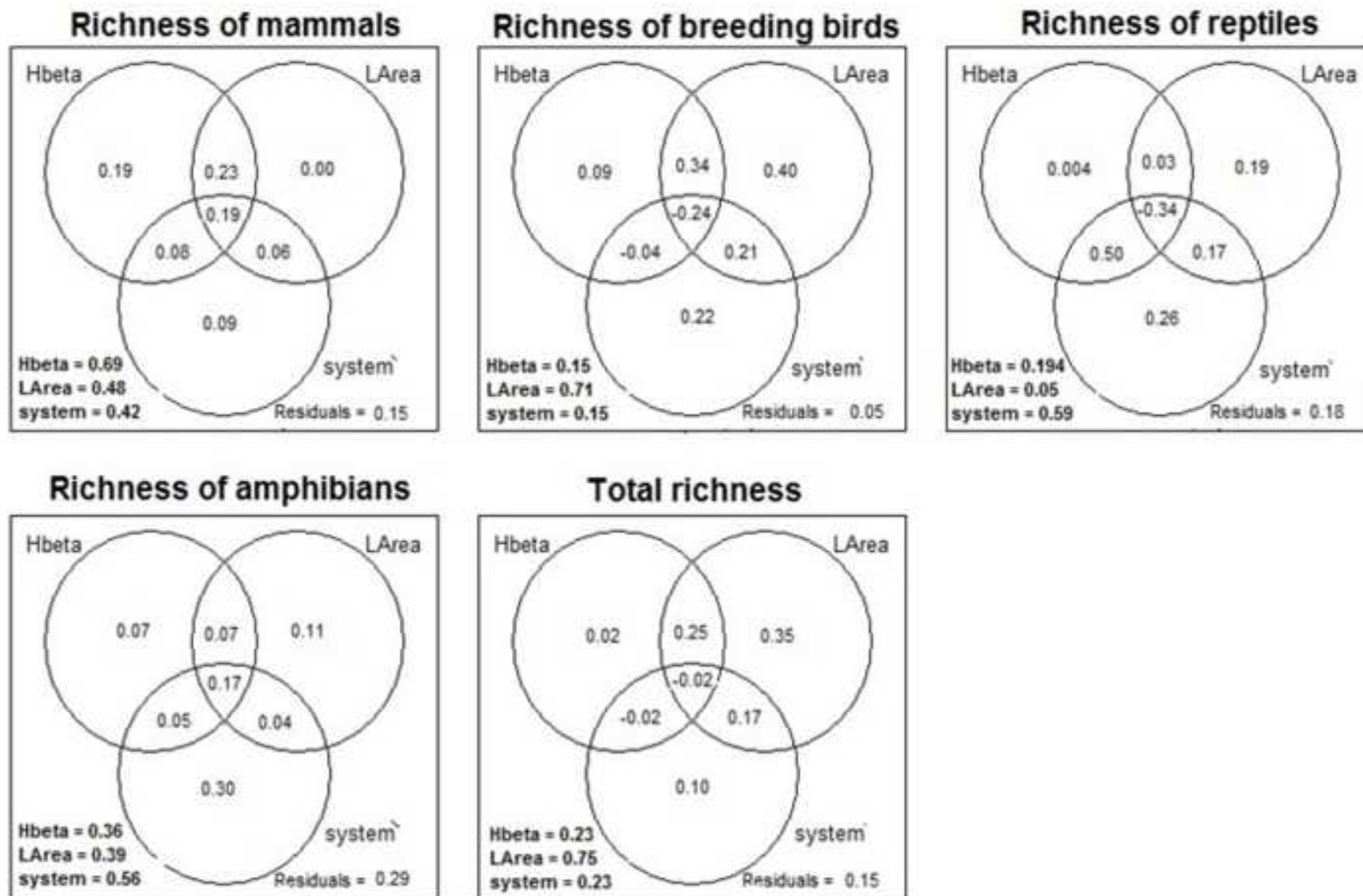


Table A1. Absolute (S_a ; number of species) and relative (S_r ; number of species per 100 km²) values of species richness: total and taxonomic (mammals, breeding birds, reptiles and amphibians) richness for each mountain system (Cantabrian Mountains, Central System and the Spanish Pyrenees).

	Cantabrian Mountains		Central System		Spanish Pyrenees	
	S_a	S_r	S_a	S_r	S_a	S_r
Mammals	88	0.27	85	0.07	64	0.29
Breeding birds	198	0.62	188	0.10	166	0.74
Reptiles	29	0.09	30	0.23	20	0.09
Amphibians	19	0.06	17	0.13	15	0.07
Total species	334	1.06	320	0.20	265	1.18

Figure A1. Landscape classifications based on CORINE, for a.1) the Cantabrian Mountains; b.1) the Central System; c.1) the Spanish Pyrenees.

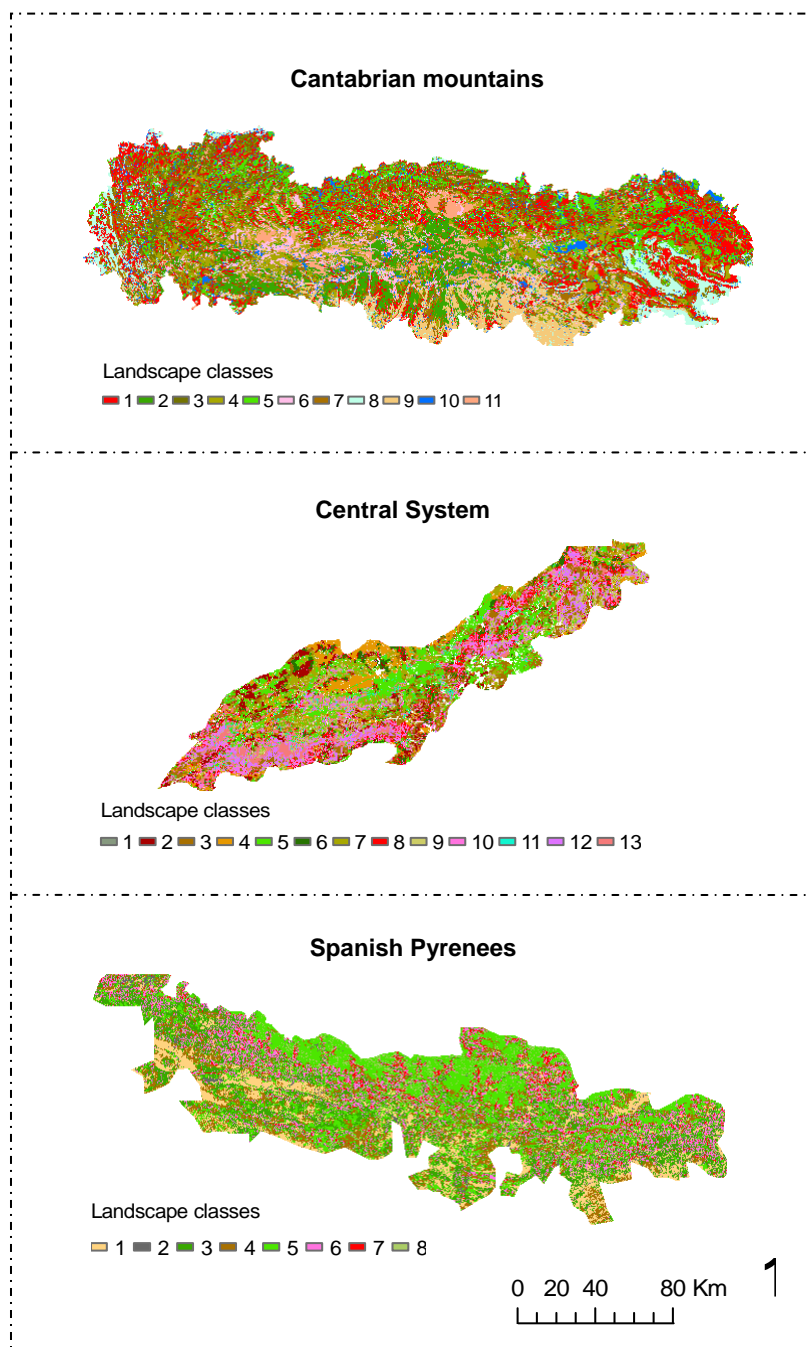


Table A2. Description of landscape classes obtained from a set of variables accounting for topography, urban influence and land cover (CORINE).

Class	Description
	Cantabrian Mountains
1	Forests covering coastal and middle-mountain areas at an average altitude of 688.16 ± 299.69 m.a.s.l., in areas with moderate to steep slope (mean value $20.73^\circ \pm 10.72^\circ$) and at mid-distance to settlements (mean value 3.65 ± 4.05 km)
2	Forests covering central mountains and piedmont areas at an average altitude of 1267.14 ± 240.47 m.a.s.l. with moderate slope (mean value $17.56 \pm 9.77^\circ$), those being at mid-distance to settlements (mean distance 2.67 ± 2.48 km)
3	Transitional to woodlands with low urban influence (mean distance to settlements 5.24 ± 4.92 km), covering mainly Atlantic and Sub-Atlantic mountains located at low altitudes (mean value 624 ± 213.42 m.a.s.l.), with moderate to steep slope (mean value $24.08^\circ \pm 9.86^\circ$)
4	Transitional woodlands from central and southern areas of the Cantabrian Mountains, with relatively low urban influence (mean distance to settlements 3.9 ± 4.11 km), those being located at an average altitude of 1207.40 ± 290.56 m.a.s.l., in areas with moderate slope (mean value $18.05^\circ \pm 10.30^\circ$)
5	Pastures from mid-low Atlantic mountains and coastal areas (mean altitude 520.22 ± 263.10 m.a.s.l.), with moderate slope (mean value $15.87^\circ \pm 9.68^\circ$) and at mid-distance to settlements (mean value 3.66 ± 4.69 km)
6	Pastures covering bottom valleys and hillsides of the central Cantabrian Mountains, lying at an average altitude of 1216.65 ± 290.14 m.a.s.l., in areas with moderate slope (mean value $16.22^\circ \pm 10.51^\circ$) at mid-distance to settlements (average distance 2.8 ± 3.54 km)
7	Shrub-herbaceous associations lying at an average altitude of 816.66 ± 378.43 m.a.s.l., in hillsides with moderate slope (mean value $19.27^\circ \pm 10.88^\circ$) and at mid-distance to settlements (mean distance 3.63 ± 3.83 km)
8	Croplands from depressions and coastal plains (average slope $8.84^\circ \pm 8.18^\circ$), lying at the lowest altitudes (mean value 605.05 ± 241.81 m.a.s.l.) and those being close to settlements (mean distance 1.64 ± 1.80 km)
9	Croplands (non-irrigated arable lands) from paramos and countryside (mean slope $5.70^\circ \pm 5.79^\circ$) lying at an average altitude of 977.27 ± 99.30 m.a.s.l. and those being the closest class to settlements (mean distance 1.26 ± 0.99 km)
10	Water surfaces and artificial surfaces lying at an average altitude of 654.15 ± 393.13 m.a.s.l., in areas with moderate slope (mean slope $14.58^\circ \pm 11.05^\circ$) and at mid-distance to settlements (mean value 2.56 ± 3.54 km)
11	Rocks and areas with little or no vegetation covering the highest altitudinal ranges (mean value 1352.71 ± 485.47 m.a.s.l.), in areas with steep slope (mean value $25.94^\circ \pm 12.54^\circ$), those being far from settlements (mean distance 4 ± 3.58 km)

Table A2. (cont.)

Central System	
1	Forests covering lowlands and bottom valleys (mean slope $5.28^\circ \pm 2.47^\circ$), at an average altitude of 1131.37 ± 253.73 m.a.s.l. and at mid-distance to settlements (mean value 2.63 ± 1.38 km)
2	Forests lying at an average altitude of 1226.79 ± 305.41 m.a.s.l. on hillsides with moderate slope (mean value $13.32^\circ \pm 3^\circ$) and at mid-distance to settlements (mean value 28.55 ± 15.36 km)
3	Forests lying at an average altitude of 1223.95 ± 336.03 m.a.s.l., on hillsides with moderate to steep slope (mean value $22.60^\circ \pm 6.25^\circ$) and at a mid-distance to settlements (mean value 28.62 ± 15.68 km)
4	Transitional to woodlands covering lowlands (mean altitude 1252.28 ± 275.44 m.a.s.l.) and areas with slight slope (mean value $5.31^\circ \pm 2.37^\circ$), those being at mid-distance to settlements (mean value 2.50 ± 14.51 km)
5	Transitional to woodlands covering middle-hillsides (mean altitude 1352.24 ± 332.77 m.a.s.l.) with moderate slope (mean value $12.80^\circ \pm 2.79^\circ$), those being at mid-distance to settlements (mean value 2.86 ± 1.66 km)
6	Transitional to woodlands covering the highest altitudes (mean value 1436.55 ± 339.83 m.a.s.l.) with moderate to steep slope (mean value $24.69^\circ \pm 6.04^\circ$), those being at mid-distance to settlements (mean value 3.55 ± 1.84 km)
7	Pastures covering valley bottoms (mean altitude 1181.78 ± 246.51 m.a.s.l.) and hillsides with a slight slope (mean value $5.16^\circ \pm 3.22^\circ$) and at mid-distance to settlements (mean 2.29 ± 1.70 km)
8	Pastures covering middle mountain areas (mean value 1348.88 ± 314.95 m.a.s.l.) with moderate slope (mean value $14.62^\circ \pm 3.25^\circ$) and at mid-distance to settlement (mean value 2.60 ± 1.72 km)
9	Shrub-herbaceous associations located at low altitudinal ranges (mean value 1177.51 ± 301.89 m.a.s.l.) on hillsides with moderate slope (mean value $13.17^\circ \pm 8.63^\circ$) and at mid-distance to settlements (mean value 2.75 ± 1.64 km)
10	Croplands (herbaceous) covering valley bottoms and plains (mean slope $5.01^\circ \pm 4.70^\circ$) at an average altitude of 1028.33 ± 201.01 m.a.s.l., those being close to settlements (mean value 1.46 ± 0.9 km)
11	Woody croplands covering valley bottoms and plains (mean slope $6.75^\circ \pm 5.40^\circ$) at an average altitudinal range of 962.96 ± 233.65 m.a.s.l., those being close to settlements (mean value 2.06 ± 1.34 km)
12	Water surfaces and artificial surfaces located at an average altitude of 1002.28 ± 229.45 m.a.s.l., in areas of slight slope (mean value $4.87^\circ \pm 4.22^\circ$), those being close to settlements (mean value 1.31 ± 1.25 km)
13	Rocks and areas with little or no vegetation covering the highest altitudes (mean value 1481.65 ± 371.04 m.a.s.l.), in areas with moderate to steep slope ($20.08^\circ \pm 10.21^\circ$) at mid-distance to settlements (mean distance 3.54 ± 2.17 km)
Spanish Pyrenees	
1	Forests lying at an average altitude of 1000.49 ± 323.21 m.a.s.l., covering hillsides with moderate slope (mean value $14.28^\circ \pm 5.29^\circ$) at mid-distance to settlements (mean distance 2.95 ± 1.74 km)
2	Forests lying at an average altitude of 1163.56 ± 368.78 m.a.s.l., covering hillsides with steep slope (mean value $27.97^\circ \pm 3.57^\circ$), those being at mid-distance to settlements (mean value 2.97 ± 1.77 km)
3	Forests lying at an average altitude of 1313.67 ± 397.56 m.a.s.l., in areas with very steep slope (mean value $39.39^\circ \pm 4.23^\circ$), those being at mid-distance to settlements (mean value 2.98 ± 1.83 km)
4	Mosaic of grasslands, rocks and areas with little or no vegetation and transitional to woodlands, covering the highest altitudinal ranges (mean value 1547.19 ± 623.41 m.a.s.l.), in areas with moderate slopes (mean value $20.09^\circ \pm 9.55^\circ$) and at mid-distance to settlements (mean value 3.69 ± 2.61 km)
5	Mosaic of grasslands, rocks and areas with little or no vegetation, covering the highest altitudinal ranges (mean value 1588.50 ± 653.16 m.a.s.l.), in areas with very steep slopes (mean value $44.02^\circ \pm 9.07^\circ$) and relatively far from settlements (mean value 3.83 ± 2.64 km)
6	Shrub-herbaceous associations lying at an average altitude of 1080.83 ± 414.27 m.a.s.l., covering hillsides with moderate slope (mean value $19.17^\circ \pm 9.36^\circ$) and at mid-distance to settlements (mean value 2.53 ± 1.58 km)
7	Croplands covering valley bottoms lying at the lowest altitudinal ranges (mean value 775.08 ± 219.38 m.a.s.l.) in areas with slight slope (mean value $10.56^\circ \pm 8.04^\circ$), those being close to settlements (mean value 1.89 ± 1.51 km)
8	Water surfaces and artificial surfaces lying at an average altitude of 803.85 ± 395.78 m.a.s.l., in areas with moderate slope (mean value $14.21^\circ \pm 10.07^\circ$), those being close to settlements (mean distance 2.05 ± 1.91 km)

Table A3. Overall accuracy of landscape classifications and user's and producer's accuracy per class, obtained from topography, urban influence and CORINE (as a proxy of land cover) data.

	Class	User's accuracy (%)	Producer's accuracy (%)	Overall accuracy (%)
Cantabrian Mountains	1	93.33	88.89	88.33
	2	85.29	78.38	
	3	83.33	68.97	
	4	86.49	96.97	
	5	100.00	91.18	
	6	94.12	94.12	
	7	96.15	100.00	
	8	89.47	85.00	
	9	85.00	89.47	
	10	50.00	100.00	
	11	93.33	100.00	
Central System	1	50.00	100.00	84.39
	2	90.00	81.81	
	3	100.00	84.62	
	4	92.30	80.00	
	5	69.23	75.00	
	6	77.78	80.00	
	7	88.00	91.66	
	8	100.00	37.05	
	9	92.30	100.00	
	10	100.00	84.62	
	11	100.00	100.00	
	12	100.00	100.00	
	13	88.00	91.67	
Spanish Pyrenees	1	85.29	87.87	88.00
	2	95.23	66.67	
	3	85.00	71.51	
	4	80.49	97.05	
	5	100.00	66.67	
	6	100.00	100.00	
	7	89.47	100.00	
	8	50.00	100.00	

Table A4. Estimate, standard error (*Std. error*) and significance (*Sig.*) of regression linear models, achieved when working with watersheds as spatial analytical units and landscape heterogeneity calculated from landscape classifications. The intercept represents the Cantabrian Mountains.

	Estimate	Std. Error	Sig.
Mammal richness			
Intercept	2.03	0.26	***
H _β	0.43	0.47	n.s.
log(A)	0.23	0.05	***
Central System	-0.08	0.11	n.s.
Spanish Pyrenees	-0.26	0.09	**
Breeding bird richness			
Intercept	37.18	9.93	***
H _β	56.66	18.36	**
log(A)	11.18	1.99	***
Central System	12.74	4.23	*
Spanish Pyrenees	7.20	3.47	**
Reptile richness			
Intercept	-6.73	2.93	*
H _β	13.49	5.55	*
log(A)	2.56	0.58	***
Central System	4.64	1.30	***
Spanish Pyrenees	4.31	1.05	***
Amphibian richness			
Intercept	-0.22	1.69	n.s.
H _β	8.21	3.21	*
log(A)	1.45	0.34	***
Central System	-0.27	0.61	n.s.
Spanish Pyrenees	-1.04	0.75	n.s.
Total richness			
Intercept	34.41	19.06	n.s.
H _β	120.18	33.65	***
log(A)	20.89	3.73	***
Central System	13.09	5.98	n.s.
Spanish Pyrenees	5.98	13.09	n.s.

log(A) logarithm of watershed area; H_β landscape heterogeneity; ***significance at level <0.001; **significance at level 0.01; *significance at level 0.05; n.s. no significance.

Table A5. Estimate, standard error (*Std. error*) and significance (*Sig.*) of regression linear models, achieved when working with regular squared windows as spatial analytical units and landscape heterogeneity calculated from landscape classifications. The intercept represents the Cantabrian Mountains.

	20x20 km			50x50km			100x100km		
	Estimate	Std. Error	Sig.	Estimate	Std. Error	Sig.	Estimate	Std. Error	Sig.
Mammal richness									
Intercept	1.98	0.40	***	3.71	0.88	***	3.52	0.31	***
H _β	0.22	0.17	n.s.	0.23	0.32	n.s.	1.50	0.55	*
log(A)	0.31	0.07	***	0.06	0.12	n.s.	0.001	0.05	n.s.
Central System	-0.14	0.07	n.s.	-0.08	0.06	n.s.	0.16	0.1	n.s.
Spanish Pyrenees	-0.26	0.06	***	-0.14	0.07	*	0.21	0.10	n.s.
Breeding bird richness									
Intercept	-31.34	18.11	n.s.	20.76	68.22	n.s.	-81.54	23.43	*
H _β	5.40	7.90	n.s.	21.58	24.77	n.s.	114.42	34.47	*
log(A)	23.16	3.08	***	15.28	9.34	n.s.	20.90	2.91	***
Central System	5.42	2.64	*	9.84	4.32	*	36.14	7.05	**
Spanish Pyrenees	12.57	3.26	***	-1.54	5.10	n.s.	29.14	7.07	**
Reptile richness									
Intercept	-9.99	4.94	*	28.08	15.46	n.s.	-4.32	8.39	n.s.
H _β	3.95	2.15	n.s.	2.07	5.61	n.s.	-5.22	13.18	n.s.
log(A)	3.24	0.84	***	-1.85	2.11	n.s.	3.04	1.12	*
Central System	6.27	0.89	***	8.49	0.98	***	5.32	2.59	n.s.
Spanish Pyrenees	3.69	0.72	***	5.44	1.16	***	2.03	2.58	n.s.
Amphibian richness									
Intercept	-7.97	3.59	*	29.11	14.57	n.s.	3.22	7.16	n.s.
H _β	5.96	3.25	n.s.	11.10	5.28	n.s.	14.70	11.26	n.s.
log(A)	2.65	0.61	***	-2.79	1.99	n.s.	0.50	0.95	n.s.
Central System	2.04	0.63	**	0.05	0.91	**	2.82	2.21	n.s.
Spanish Pyrenees	-0.63	0.51	n.s.	3.10	1.13	n.s.	0.25	2.20	n.s.
Total richness									
Intercept	-75.40	29.65	*	100.99	91.30	n.s.	-75.24	63.26	n.s.
H _β	22.26	12.93	n.s.	46.79	33.15	n.s.	94.61	99.46	n.s.
log(A)	41.21	5.04	***	16.05	12.51	n.s.	34.56	8.42	**
Central System	14.61	5.34	**	14.71	5.79	*	29.82	0.72	n.s.
Spanish Pyrenees	-3.40	4.32	n.s.	-6.62	6.83	n.s.	13.98	1.52	n.s.

H_β landscape heterogeneity; log(A) logarithm of the area of the square window; ***significance at level <0.001;

**significance at level 0.01; *significance at level 0.05; n.s. no significance.

Figure A2. Variance partitioning among pure and combined effects of landscape heterogeneity (Hbeta), logarithm of watershed area (LArea) and mountain system (system), as explanatory variables explaining species richness of mammals, breeding birds, reptiles and amphibians and total species richness, based on squared windows of 20x20 km as spatial units of analysis. Total effect of predictors (shared and pure) is in bold.

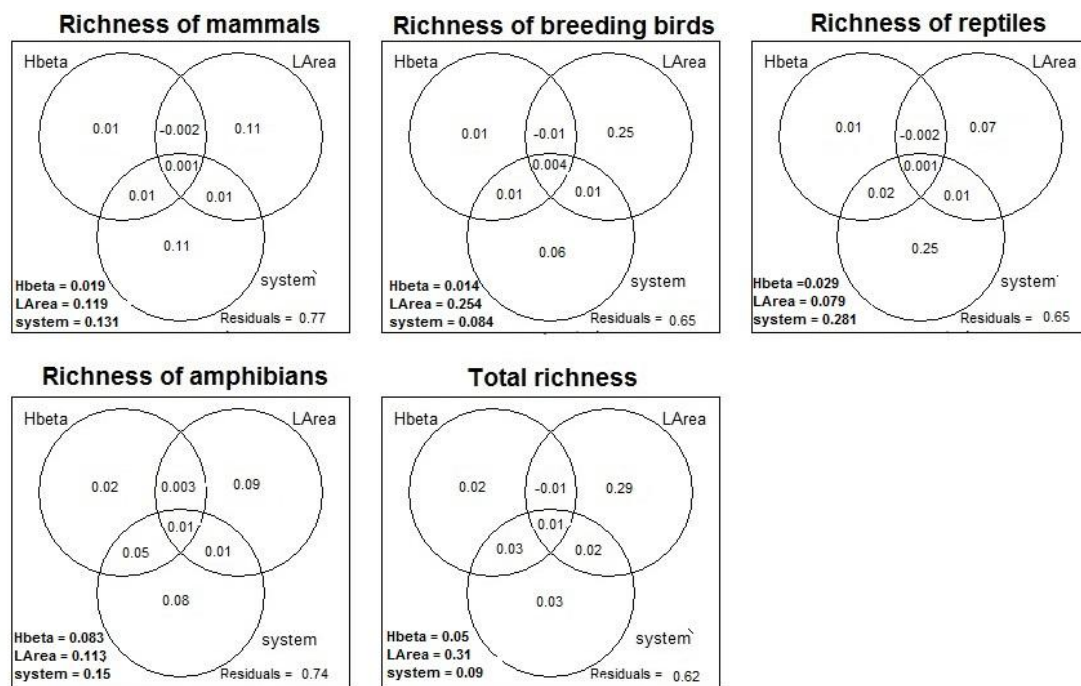


Figure A3. Variance partitioning among pure and combined effects of landscape heterogeneity (Hbeta), logarithm of watershed area (LArea) and mountain system (system), as explanatory variables explaining species richness of mammals, breeding birds, reptiles and amphibians and total species richness, based on squared windows of 50x50 km as spatial units of analysis. Total effect of predictors (shared and pure) is in bold.

