



Assessing the ecological basis of conservation priority lists for bird species in an island scenario

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ABSTRACT

Proneness to extinction varies naturally and continuously according to the ecological phenomena that compound rarity even before anthropogenic effects may play a role. This is particularly obvious in islands, where populations are often (and naturally) small and fragmented and, consequently, conservation priority lists may have a large number of species clustered unhelpfully in the higher threat categories. In this study we propose a simple model of threat based on natural descriptors of rarity and taxonomic distinctiveness (area of occupancy, population abundance and trend, and endemism), assess its correlation with ecological features of the species (habitat preferences and body size) and check whether the Spanish Red data Book and a normative conservation priority list (the Canary Islands Catalogue of Threatened Species and its administrative revision) includes these ecological bases for birds. We found that a large variation in threat (48.2%) was explained by phylogeny, habitat breadth and preference for urban areas (with a negative effect), and preference for agricultural environments (a positive effect). The Spanish Red data Book and the administrative lists tested are poorly related to descriptors ordering the extinction risk and loss of taxonomic singularity, so some changes would make their categories more coherent. We contend that the ecological bases of rarity should be taken into account to understand why some populations/species are at higher extinction risk whereas others remain relatively safe, as this would provide firmer grounds on which to base conservation priorities.

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Introduction

Extinction proneness varies naturally according to natural phenomena defining ecological rarity and making populations more or less vulnerable to demographic, environmental or genetic stochasticity (Ricklefs & Bermingham 2002; Williamson 1989). For instance, the probability of having conservation problems is higher in species with low population sizes or local abundances (Mace & Kershaw 1997; O'Grady et al. 2004b; Pimm et al. 1993), small range sizes (Cardillo et al. 2008; Gage et al. 2004; Purvis et al. 2000) and those with specialized habitat preferences (Hawkins et al. 2000; Julliard et al. 2004; but see Gage et al. 2004). Therefore, species differ intrinsically in their vulnerability even before taking into account human-related factors that alter the natural scenario through habitat loss, over-exploitation and species introduction (Dulvy et al. 2003; Hawkins et al. 2000; Keane et al. 2005). However, although these anthropic factors typically exacerbate the risk

of extinction, for some species linked to settlements and agricultural areas the effect is positive in terms of a net gain of population and range size (see for example Thiollay 1995; Williamson & Fitter 1996).

Regardless of the specific underlying causes, the Earth is currently experiencing a loss of biodiversity comparable to the great geological extinctions (see a review in Bini et al. 2005). A growing concern regarding this environmental catastrophe has led to the extensive compilation of threatened species lists, whose aim is to identify the risk of extinction of taxa and to promote conservation actions that would help to halt the biodiversity crisis both globally and regionally (Butchart et al. 2004; Miller et al. 2007). These lists are mostly based on the perceived pattern of rarity of the species (and subspecies), typically taking into account population sizes and trends, and the size of the range of distribution (Given & Norton 1993; Mace & Lande 1991). Thus, for example, several internationally acknowledged schemes to rank threatened species have developed criteria based on quantitative thresholds for those components of rarity to assign taxa to categories of threat (BirdLife-International 2004; IUCN 2006; O'Grady et al. 2004a). Such quantitative approaches have sometimes been claimed to be too data-demanding and impractical for use in mega-diverse

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areas with scarce resources available for conservation (Soberón & Medellín 2007).

On the other hand, the assessments of extinction risk are normally designed for and entertained at large scales, being more questionable when applied regionally (Gärdenfors 2001; Miller et al. 2007). This is because population sizes are necessarily lower at these more reduced scales, where they may not represent a relevant fraction of the global number of individuals for the taxa evaluated. Also, regional and local populations may be interlinked by migration in a metapopulation net, where local extinctions may be followed by immigration and recolonization from other areas in a fragmented landscape (Hanski 1998). These issues led the IUCN to define a less known set of guidelines for the application of its criteria (likely the scheme more extensively used) at regional levels, typically countries, which consider both the relevance of the assessed population and the possibility of immigration (Gärdenfors et al. 2001; IUCN 2003). However, IUCN criteria was designed for global assessments and their application to regional levels has been criticized (Mrosovsky 2003; Soberón & Medellín 2007). In summary, although IUCN criteria have been successfully adopted in a number of official national listing efforts, the application of IUCN regional guidelines has been less extensively used (Miller et al. 2007). Islands pose a particularly acute problem, because, being commonly small, disaggregated and isolated territories, a great part of their taxa could potentially qualify as threatened considering IUCN thresholds on the grounds of their small ranges and population sizes with little contact with source continental areas. Martín (2009) has pointed out that if rarity does not always equate with negative population trends, or involve a clear risk of extinction, the use of absolute thresholds as a reference to be applied to all regions is not advisable, since this can lead to an overestimation of the threat in small, isolated, regions. It is common to assume that island species would tend to exhibit the metapopulation dynamics described for fragmented landscapes in general (where in this case the fragmentation is a natural condition) and suffer from the risks of small populations (Caughley 1994; Hanski 1998). Indeed the record of extinctions in islands is outstanding and has traditionally been used to appraise the biodiversity crisis (Caughley & Gunn 1996; Pimm et al. 2006) but, even if this high vulnerability is generally held to be true at the global level, the extent to which island biotas are comparatively more threatened in current years is debatable (Eric 2002; Manne et al. 1999). Moreover, we want to emphasize that, from an applied point of view, clumping together most or all island species in a high risk category blurs any distinction among them and does not help the managers' process of decision-making at regional levels. This drawback is particularly serious when absolute thresholds are used to flag species and have thus led to the proposal of criteria to specific taxa or regions (Báldi et al. 2001; Martín et al. 2005; Sapir et al. 2003).

Formally, assessing threat and setting priorities for conservation action are two different processes. The first, typically entertained by Non Governmental Organizations, leads to the definition of red lists that rank species by their estimated risk of extinction, while the second is performed by state agencies and qualifies information on extinction probability with a set of social and economic values, to generate a normative act (de Grammont & Cuarón 2006). In practice, however, the normative texts commonly state their risk assessment basis but do not acknowledge other inputs to the classification scheme (Farrier et al. 2007).

Here, we propose a conceptual model to assess the relative risk of biodiversity loss at the taxa level in islands (i.e., loss of phylogenetic history or taxonomic distinctiveness), based on a set of easily measurable biological patterns defining rarity and also considering the loss of genetic diversity, thus taking into account both the risk of extinction and the singularity of taxa (see Marsh et al. 2007 for a similar approach to score species). This model is con-

trasted with the Spanish Red Data Book (Madroño et al. 2005), which declared to have been built following the IUCN criteria and the regional guidelines, and the Canary Islands Catalogue of Threatened Species, the official document for the oceanic archipelago of the Canary Islands, which classifies taxa in four categories of threat and regulates the corresponding conservation action to deploy. This catalogue has been recently revised by the Canary Government according to elaborate criteria (Martín et al. 2005, see a summary in Appendix A). This revision explicitly declares that the new proposed categories estimate the risk of extinction and thus is a suitable and generalizable testing scenario to examine the ecological basis underlying the classification scheme, in order to detect any deficiencies and propose amendments that would make it more coherent (see a similar approach in Cardillo et al. 2005; Kotiaho et al. 2005). The Canary Islands are an oceanic archipelago composed of small islands where regional managers have questioned the application of common global thresholds (and regional guidelines) to label species as threatened (Martín 2009), which we feel may also be a common concern in different territories. Overall, we consider the evaluation performed here an example that could be generalizable to other archipelagos and continental areas.

Methods

Study area

We focused on the Canary Islands oceanic archipelago, located at the southwestern limit of the Palaearctic region (27°30'–29°30'N and 13°20'–18°15'O). Among the main seven islands in the archipelago, we chose three that comprise the whole variability of the environmental characteristics in the Canaries: Fuerteventura (1660 km²), Tenerife (2059 km²) and La Palma (708 km²). The minimum distance to the nearest mainland (northwestern Africa) varies from 96 km in Fuerteventura to 417 km in La Palma. Fuerteventura and Tenerife have an ancient geological history (15–20 million years old) while La Palma is among the youngest in the archipelago (1–2 million years). Fuerteventura is arid and low (less than 800 m a.s.l., and 300 mm of precipitation per year) with a predominance of steppe-like landscapes and woodlands almost completely absent, while Tenerife and La Palma reach higher altitudes (more than 2400 m a.s.l., and with many areas with more than 600 mm of precipitation per year) and have an extensive representation of native pine and evergreen ('*laurisilva*') forests. Habitat diversity in these islands is tightly related to their maximum altitude, ranging from the highest in Tenerife and La Palma to the lowest in Fuerteventura. See Juan et al. (2000) and Fernández-Palacios and Martín Esquivel (2001) for more details on island characteristics.

Bird species and data

We obtained data on four main descriptors of rarity (Gaston 1994; density, range size, population trend and endemism) and the degree of threat (according to the Canary Islands Catalogue of Threatened Species) for almost all of the terrestrial bird species that breed in the Canary Islands (44 species, excluding those marine, aquatic, exclusively migrant and swifts, *Apus* spp., of very difficult sampling using line transects; see Appendix B).

The maximum ecological density was obtained through bird surveys carried out during the breeding season in 2002–2003 (Tenerife), 2005–2006 (Fuerteventura) and 2007 (La Palma), trying in every island to sample the whole diversity of habitats and throughout the altitudinal gradient. Field work was conducted from March to April depending upon the geographical variation in the breeding phenology of the birds in these three islands (beginning earlier at lower altitudes and in the drier Fuerteven-

tura, then at higher altitudes and in the more western, oceanic, islands). The survey method was the line transect, frequently used in extensive assessments of abundance, general distribution patterns and habitat preferences of birds (Bibby et al. 2000). The transects were carried out at a low speed (1–3 km/h approximately) on windless and rainless days, walking cross-country or on little used dirt tracks (just on impenetrable terrain such as young lava fields and some precipitous slopes, assuming that dirt tracks have little effects of bird communities, see Delgado et al. 2008), during the 4 hours after dawn and the 2.5 hours before dusk. A total number of 1715 line transects of 0.5 km (geolocated and measured by means of portable GPSs) were made (Tenerife: $n = 592$, Fuerteventura: $n = 686$, La Palma: $n = 437$), and we consider these as a representative sample of abundance within habitats. Bird transects were stratified *a posteriori* in 37 major habitat types according to overall vegetation and topographic characteristics: 12 in Fuerteventura, 14 in Tenerife and 11 in La Palma (average sample size per habitat type was 45 line transects of 0.5 km, with a minimum of 16 transects in one habitat of Tenerife and 19 in two of La Palma, see Appendix C). We estimated densities in each habitat (i.e., abundance per unit area) correcting for the detectability of the species by using the perpendicular distances to the birds and fitting different models that are commonly used to explain the loss of detectability as a function of the distance from the transect line (i.e., distance sampling: Buckland et al. 2001). Data on relative abundances (birds per 0.5-km transects) and absolute densities (birds per km²) can be consulted in <http://www.vertebradosibericos.org/aves/distribucion/canarybirds.html>. We estimated the maximum density recorded in the major habitat types censused in each island, as a measure of the maximum ecological abundance a species can attain in its most favourable environment in each one of the three studied Canary Islands. We could have used estimates for bird population sizes in the islands, which are more commonly applied as absolute thresholds to assign species to a category of threat. However, we employed maximum densities in the preferred habitats because, from a manager point of view, these estimates are far easier to obtain reliably (reassuringly, densities and areas of occupancy – see below – are surrogates for population sizes in our study area: Carrascal & Alonso 2005; Carrascal et al. 2008a).

Range size in Fuerteventura, Tenerife and La Palma was estimated as the area of occupancy. It was measured with the most recent atlas of Canary Breeding Birds (Lorenzo 2007) as the number of 5 km × 5 km UTM squares occupied in each island where breeding was estimated as confirmed or probable (an evidence for the relationship between range size and area of occupancy is that the number of occupied UTM squares correlates with the frequency of appearance in the line transects: Tenerife: $r = 0.66$, La Palma: $r = 0.74$, Fuerteventura: $r = 0.74$, all $P < 0.0001$). A crude estimate of population trends in the last 25 years was obtained for each island considering the available literature (Carrascal & Alonso 2005; Carrascal & Palomino 2005; Carrascal et al. 2008a; Lorenzo 2007; Martín & Lorenzo 2001; Palacios 2004), assigning –1 to taxa that have clearly declined, +1 to clearly increasing taxa and 0 for stable populations or not clearly decreasing or increasing population trends. The threshold between stable and declining populations is roughly equivalent to IUCN quantitative criterion A2 for threatened (vulnerable category) taxa because it is based on inferred population size reductions of $\geq 30\%$ according to reported declines in extent of occurrences over the last 25 years (10 years or three generation in the IUCN scheme), which either are not well understood or have not ceased (IUCN 2001). The degree of endemism was quantified as an ordinal variable, assigning 0 for naturalized populations coming from old introductions or recent colonizer taxa, 1 for native species and subspecies shared with continental areas (Europe and Africa); 2 for endemic subspecies for the Canary Islands; and 3

for endemic species for the Canary Islands. This index would help managers to set priorities by telling apart rare singular taxa from recent colonizers with small ranges (such as expanding introduced species). In each case the taxonomic status was assessed based on the synthesis by Martín and Lorenzo (2001), and other more recent molecular studies (Baker & Omland 2006; Dietzen et al. 2003, 2006, 2008a, 2008b; García-del-Rey et al. 2007; González et al. 2008; Illera et al. 2007; Kvist et al. 2005; Packert et al. 2006). *Chlamydotis undulata* (Idaghdour et al. 2004; Pitra et al. 2004), *Calandrella rufescens* (Martín et al. 2005, p. 23), *Sylvia atricapilla* (Dietzen et al. 2008b) and *Sylvia melanocephala* (Dietzen et al. 2008b) are considered native species not clearly distinguished at the subspecific level from continental populations. The formerly *Regulus teneriffae* (Perrins 1998) has been considered as *Regulus regulus* with two distinctive subspecies in the Canary Islands (Packert et al. 2006). *Cyanistes teneriffae* is not considered as an endemic species of the Canary Islands because the subspecies *C. teneriffae ultramarinus* (from Fuerteventura and Lanzarote islands) is also present in the nearby mainland (Dietzen et al. 2008a; Kvist et al. 2005); therefore *C. teneriffae* populations of La Palma (*palmensis*) and Tenerife (*teneriffae*) are considered endemic subspecies, but the population of Fuerteventura is considered a native species not clearly distinguished at the subspecific from continental populations. The same argument applies to *Lanius meridionalis koenigi*, because although this taxa significantly differs from its European counterpart, there is no firm evidence to distinguish the Canarian taxa from *L.m. algeriensis* on the African mainland (González et al. 2008).

To explore commonly claimed correlates of threat (as a compound measure of predictors of rarity, see below) we also took into account niche breadth, affinity for anthropogenic habitats and body size. Niche breadth of habitat distribution in each island was calculated following the Levins index divided by the number of habitat categories (n) considered (Levins 1968):

$$HB = \frac{(\sum p_i^2)^{-1}}{n}$$

where p_i is the proportion of the density for each species measured in the habitat i (dividing density in habitat i by the sum of all densities recorded in the n habitat categories of each island). This index ranges between 1 (evenly distributed across the n habitats) and $1/n$ (only present in one habitat).

Several variables characterizing the 0.5 km transects were obtained averaging three visual estimations on 25-m radius circular plots (located at 125, 250 and 375 m within the line transect). Two of them were used in this paper: the coverage of urban areas (buildings, paved streets) and any kind of agricultural land-use. The affinity for these novel environments of anthropogenic origin in the Canary Islands was calculated as the weighted average of the cover of urban and agricultural habitats (weighing by the number of birds recorded in each transect).

Finally, body mass was used to account parsimoniously for several aspects related to life history (Cardillo et al. 2005; Peters 1983). This was obtained from published literature (Perrins 1998) as the mean weight of males and females, or as the average value of body weight range in spring and summer.

The current official Regional Catalogue of the Canary Islands uses five categories that were ordered and coded as follows: “risk of extinction” – 3; “sensitive to habitat alteration” and “vulnerable” – 2; “special interest” – 1; “non-threatened” – 0. This catalogue has been recently revised by the Canary Government (Martín 2004) and uses the following four threat categories, that were ordered and coded in the following sequence: “risk of extinction” – 3, “vulnerable” – 2; “special interest” – 1; “non-threatened” – 0. The categories of the Spanish Red Data Book (Madroño et al. 2005) follows applies IUCN criteria with guidelines for its application to regional level

(Gärdenfors 2001); they were also ordered and coded as follows: “endangered” – 3; “vulnerable” – 2; “near threatened” – 1; “non-threatened” – 0.

Statistical analysis

Each population of the same species inhabiting a particular island was considered as separate data for the analysis (referred loosely to in the following as taxa). This is because the populations are normally sedentary with little or no evidence of current connection among them (some species have reached the point of acquiring recognized subspecific differentiation). Thus, we considered islands as different ecological and evolutionary scenarios where populations of the same species may exhibit particular relationships between rarity, habitat preferences and body size (Juan et al. 2000; Martín & Lorenzo 2001). Original figures of maximum ecological density and number of occupied 5 km × 5 km UTM squares were log-transformed before being rescaled, because these variables are usually related logarithmically in macroecological patterns (Gaston & Blackburn 2000). The other variables were also transformed to linearize their relationships.

We used phylogenetic eigenvector regression (PVR) to quantify the amount of phylogenetic signal among population variation in distribution-abundance patterns (Diniz-Filho et al. 1998; Diniz-Filho & Torres 2002). The phylogenetic hypothesis used was taken from Sibley and Ahlquist (1990), based on DNA–DNA hybridization data, as this work is the only one that provides a topology for all the families and genera used in this study and seems to be well resolved above the subfamily level (Mooers & Cotgreave 1994). This hypothesis was complemented by the above mentioned molecular phylogenetic studies and taxonomic reviews for subspecific differences among islands. Although more recent phylogenies (Barker et al. 2004; Ericson et al. 2006; Fain & Houde 2004) show several regions of the Sibley and Ahlquist (1990) phylogeny to be misleading, these discrepancies do not affect the phylogenetic hypothesis used in this study for the analyzed species. With PVR, we first did a Principal Coordinates Analysis on the matrix of pairwise phylogenetic distances between the 97 populations of 44 species, and then selected a reduced set of eigenvectors to account parsimoniously for the phylogenetic signal (five PVR components obtained by the broken-stick rule).

Other statistical procedures used were one-way ANOVA using a linear contrast of conservation status, one-way ANCOVA, multiple regression, proportional odds regression analysis, k-means clustering and Kendall tau correlations. All analyses were carried out using Statistica (StatSoft 2001).

Results

Synthetic measure of extinction risk and loss of taxonomic distinctiveness

A simple unweighted model of risk of extinction and loss of taxonomic distinctiveness was obtained by selecting the four variables easy to measure: size of the distribution range, maximum ecological density in the preferred habitat, population trend and endemism. The values of these variables were added together after being previously rescaled from 0 to 1 (see Given & Norton 1993 for a similar approach): 0 for largest density and area of occupancy, population increase (trend=1) and not endemic species (endemism=0); 1 for smallest density and area of occupancy, clear population decrease (trend=-1) and endemic status in the Canary Islands (endemism=3). Thus, this threat score ranges from 0 (increasing population trend, very abundant and widely distributed, and recent colonizer or introduced species) to 4 (high

risk of extinction of an undoubtedly declining taxa, extremely scarce, very locally distributed and endemic). Threat scores ranged between 0.33 and 3.16.

The k-means clustering procedure was applied to the 97 populations and the four variables used to build the threat score to obtain four distinct groups, which is the same number of categories in the Canary Islands Catalogue of Threatened Species. Four significant and clearly separated clusters or groups were obtained, all of them differing in the four considered variables describing the risk of extinction and “loss of phylogenetic history” (taxonomic distinctiveness) of each taxa (Table 1 and Appendix B). The most important variable separating the 97 populations of the 44 terrestrial bird species was the maximum ecological density. This objective and mathematically blind way of building threat categories identifies two extreme groups: one of high conservation concern (including 19 population-species according to a parsimonious combination of their rarity, negative population trend and endemism) and another of 24 non-endangered taxa. Between these two opposite clusters there are other two groups of lower and higher conservation concern.

Table 2 shows the pattern of variation of the threat score related to islands (factor), and phylogenetic eigenvectors, body mass, habitat breadth, and coverages of urban areas and agricultural land-use (covariates). A significant model was obtained ($F_{11,85} = 7.20, P \ll 0.001$) accounting for 48.2% of the variance. Threat score showed a marked pure phylogenetic effect accounting for 24.6% of the variation in observed inter-taxa variation (the second, third and fifth phylogenetic components were highly significant). Threat score was significantly and negatively associated with habitat breadth and preferences for urban areas, and positively related with preferences for agricultural environments. Habitat breadth and preferences for novel environments of anthropogenic origin accounted for 16.8% of the interspecific variation in threat score. Body mass of terrestrial bird species did not significantly affect threat score. There were significant differences among islands in the threat score: the score was higher for those bird populations inhabiting La Palma island.

Test of regional catalogues of conservation status

The association between the threat score of the species according to the descriptors of rarity and the categories of conservation status assigned by the official Regional Catalogue of the Canary Islands was positive and significant (linear contrast of conservation status: $F_{1,93} = 9.10, P = 0.003$; Fig. 1), although the amount of explained variance by the model was low (8.7%). The revised Canary Islands Catalogue of Threatened Species was also significantly related to the threat score (linear contrast of conservation status: $F_{1,93} = 11.73, P < 0.001$) and accounted for a similar amount of variance in rarity (10.6%). However, the threat score varies greatly within the categories of conservation status, which have broad overlap (Fig. 2).

The association between the threat score of the species according to the descriptors of rarity and the four categories of conservation status assigned by the Spanish Red Data Book for the terrestrial birds of the Canary Islands was not significant (linear contrast of conservation status: $F_{1,93} = 1.76, P = 0.187$), and the amount of explained variance by the model was very low (1.8%).

The clustering of species according to the k-means procedure is significantly related to both the official normative (Kendall correlation, $\tau = 0.348, P < 0.001$) and revised ($\tau = 0.311, P < 0.001$) Canary Islands Catalogue of Threatened Species, although the probability that the categories of the lists are in the same order is low (i.e., low values of Kendall tau). The Kendall correlation between the k-means classes and the Spanish Red Data Book for the terres-

Table 1

Results of the k-means clustering applied to 44 terrestrial bird species in three islands (final sample size is 97 populations) to define four groups, using the population trend, endemism index, maximum ecological density ($\log[\text{birds}/\text{km}^2]$) and area of occupancy ($\log[\text{number of occupied } 5 \text{ km} \times 5 \text{ km UTM squares}]$) as clustering variables. Proportional Odds Regression model predicting the conservation status assigned by the revision of the Canary Islands Catalogue of Threatened Species (Martín Esquivel et al. 2004; the status is treated as a multinomial ordered response variable) to the same 97 populations, using the same predictor variables. *F*: statistic of the one-way ANOVA comparing the four clusters defined by the k-means procedure (d.f. = 3, 93). %var: percentage of variance accounted for each variable separating the four groups. regr. coeff.: regression coefficient; deviance: deviance accounted for by that effect (amount of original deviance = 151.4); *P*: significance.

| | k-Means clustering | | | Proportional odds regression | | |
|----------------------------|--------------------|----------|----------|------------------------------|----------|----------|
| | %var | <i>F</i> | <i>P</i> | regr. coef. | Deviance | <i>P</i> |
| Endemism index | 7.2 | 2.42 | 0.071 | 0.62 | 3.3 | 0.056 |
| Population trend | 10.6 | 3.67 | 0.015 | 0.47 | 1.3 | 0.239 |
| Maximum density (in log) | 91.0 | 312.95 | <0.001 | −0.40 | 9.0 | 0.002 |
| Area of occupancy (in log) | 22.4 | 8.95 | <0.001 | −0.48 | 1.8 | 0.156 |

trial birds of the Canary Islands, was also significant ($\tau = 0.326$, $P < 0.001$).

The proportional odds regression analysis relating the conservation status assigned by the revised Canary Islands Catalogue of Threatened Species to the descriptors of rarity was globally significant ($\chi^2 = 19.80$, d.f. = 4, $P < 0.001$) and explained 13.1% of the observed deviance (Table 1). The test assigned increasing categories of threat according to the singularity of taxa (endemism) and decreasing ones according to their maximum densities attained in the most preferred habitat. Maximum population density was, by far, the most important predictor of the conservation status of the revised Canary Islands Catalogue of Threatened Species (Table 1). The Catalogue does not include two key parameters related to conservation status or extinction proneness: area of occupancy and the recent population trends of the species (surprisingly, its categories relate positively with trend). Similar results are obtained when analyzing the normative official Canary Islands Catalogue, so we omit the results for the sake of brevity.

The proportional odds regression analysis applied to the Spanish Red Data Book using the descriptors of rarity was also globally significant ($\chi^2 = 23.63$, d.f. = 4, $P < 0.001$) and explained 12.2% of the observed deviance. The scoring of the Spanish Red Data Book assigned increasing categories of threat according to the singularity of taxa (endemism; but marginally significant, $P = 0.079$) and decreasing ones according to their maximum densities attained in the most preferred habitat (the most important predictor, $P < 0.001$). Nevertheless, the Spanish Red Data Book does not relate to the area of occupancy of species ($P = 0.982$) and surprisingly is positively associated with the recent population trends of the species (regression coefficient = 0.87, $P = 0.042$).

Therefore, the official and revised Canary Islands Catalogue of Threatened Species and the Spanish Red Data Book are significantly, but not tightly related to the estimated risk of extinction and the taxonomic distinctiveness of the species, and they fail to include some relevant correlates of extinction risk.

Table 2

ANCOVA model analyzing the variation of the threat score in three different islands of the Canary archipelago according to several covariates: five phylogenetic components (PVR1–PVR5), body mass, habitat breadth and two measures of the affinity for novel environments of anthropogenic origin (urban and agricultural areas). d.f. are 2, 85 for the factor ISLAND and 1, 85 for the covariates. It is shown the standardized regression coefficients (beta), the corresponding significance (*F* and *P* values) and the percentage of variance (% var.) accounted for by each predictor variable.

| | beta | <i>F</i> | <i>P</i> | % var. |
|-----------------------------------|-------|----------|----------|--------|
| ISLAND (factor) | – | 3.56 | 0.033 | 4.3 |
| Body mass (in ln) | 0.18 | 2.02 | 0.159 | 1.2 |
| Habitat breadth | −0.34 | 14.97 | <0.001 | 9.1 |
| Urban cover (in log) | −0.24 | 7.08 | 0.009 | 4.3 |
| Agricultural areas cover (in log) | 0.21 | 5.52 | 0.021 | 3.4 |
| Phylogenetic effects | | | | |
| PVR1 | −0.17 | 2.11 | 0.150 | 1.3 |
| PVR2 | −0.27 | 10.06 | 0.002 | 6.1 |
| PVR3 | 0.30 | 12.42 | 0.001 | 7.6 |
| PVR4 | 0.19 | 3.87 | 0.052 | 2.4 |
| PVR5 | 0.32 | 11.85 | 0.001 | 7.2 |

Discussion

Ecological determinants of rarity and threat

Our data show that the landbirds of the Canary Islands can be ordered according to their conservation status – and grouped in a few objectively separate subsets if so desired – through a model based on key ecological determinants of the risk of extinction and loss of taxonomic distinctiveness. Our model, intuitive and comprehensible, is based on simple, easy to obtain variables that include the same concepts considered by IUCN criteria (IUCN 2001; area of occupancy, local density and population trend), plus the taxonomic distinctiveness or evolutionary singularity of the species (endemism). A weighting scheme might be devised to relate the four components of our model according to its presumed relevance in an island scenario, for example weighing endemism more if one should feel inclined to give preeminence to singularity. However, we do not find clear theoretical grounds to base such a weighting and we think that a raw scheme is likely more generalizable. Indeed, other similar approaches to the analysis of the risk of extinction do not apply weights (Given & Norton 1993; Mace & Lande 1991). We included endemism to account for the risk of loss of phylogenetic history though we acknowledge that this inclusion is debatable because it focuses on species prioritization rather than on extinction likelihood. However, we felt inclined to do so because, first, the catalogues we compare with are normally used to prioritize conservation actions and, second, managers and conservation advocates (and general public opinion) consider the consequences of extinction (Báldi et al. 2001; Marsh et al. 2007). Finally, our approach results in a relative ordering of taxa that may help other schemes based on categories by distinguishing among species in the same (possibly supernumerary) class.

Overall, we found that the likelihood of showing conservation problems is in a great part deterministically based on ecological and life history patterns that cannot be easily modified by man (if at all).

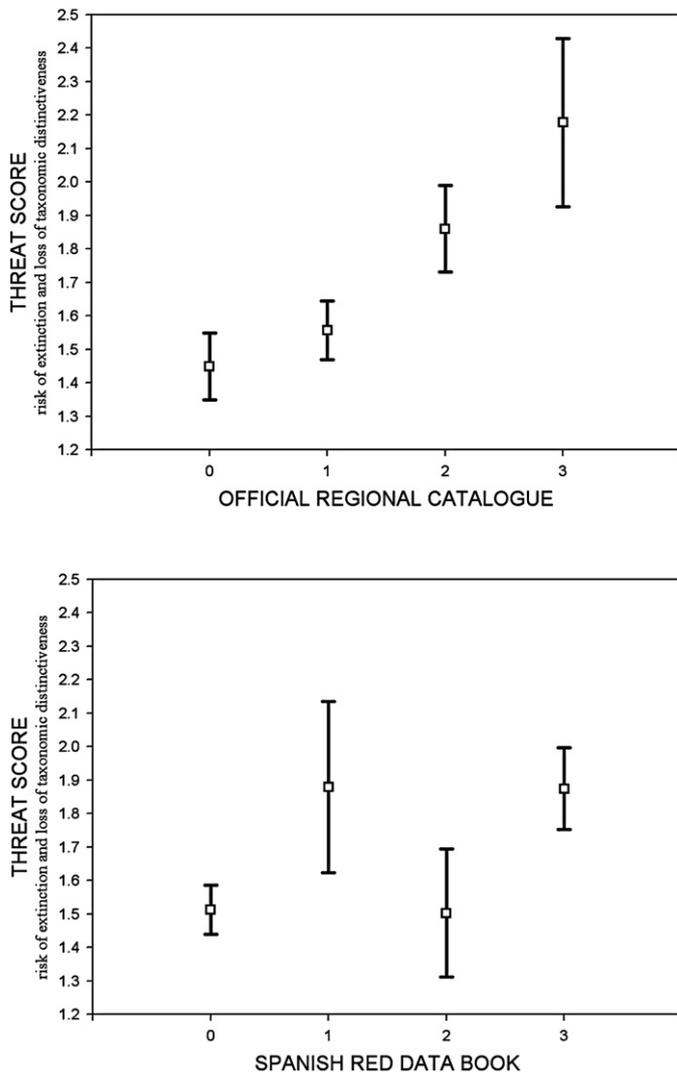


Fig. 1. Average threat scores of 44 terrestrial bird species in three islands of the Canary Islands (97 populations) in four categories of the Spanish Red Data Book (Madroño et al. 2005) and the official Canary Islands Catalogue of Threatened Species (Martín Esquivel et al. 2004). For the SCORE, denoting the risk of extinction and loss of taxonomic distinctiveness in the Canary Islands, see Appendix B. 0 – Not threatened, 1 – special interest or near threatened, 2 – vulnerable, and 3 – endangered. Vertical bars denote \pm one standard error.

First, density relates negatively to body size, with large-sized taxa frequently showing lower local abundances and population sizes, and thus, presumably, higher extinction risks (Cardillo et al. 2005; Carrascal & Tellería 1991; Gaston & Blackburn 2000; Marquet et al. 1995). This size-related pattern is, from a manager's perspective, a natural constraint to the maximum number of individuals that a given place may harbour, regardless of the conservation efforts invested in that area. Thus, the manager should not simply focus on how scarce a species is, but on whether its population density in the preferred habitat is lower than that expected by their size. Second, range size correlates positively with habitat breadth in numerous taxa and habitats (for the Canary Islands: Carrascal et al. 2008c; and elsewhere: Gaston 2003; Hurlbert & White 2007). Stenotopic species tend to show reduced distributions and are more sensitive to human alterations (Swihart et al. 2003), but the habitat preferences patterns exhibited by a particular species can be considered unmodifiable (although anthropic-related perturbations may expel a species from its preferred habitat under direct degradation or fragmentation). Third, the degree of endemism, as a measure of taxonomic singularity attained in evolutionary time,

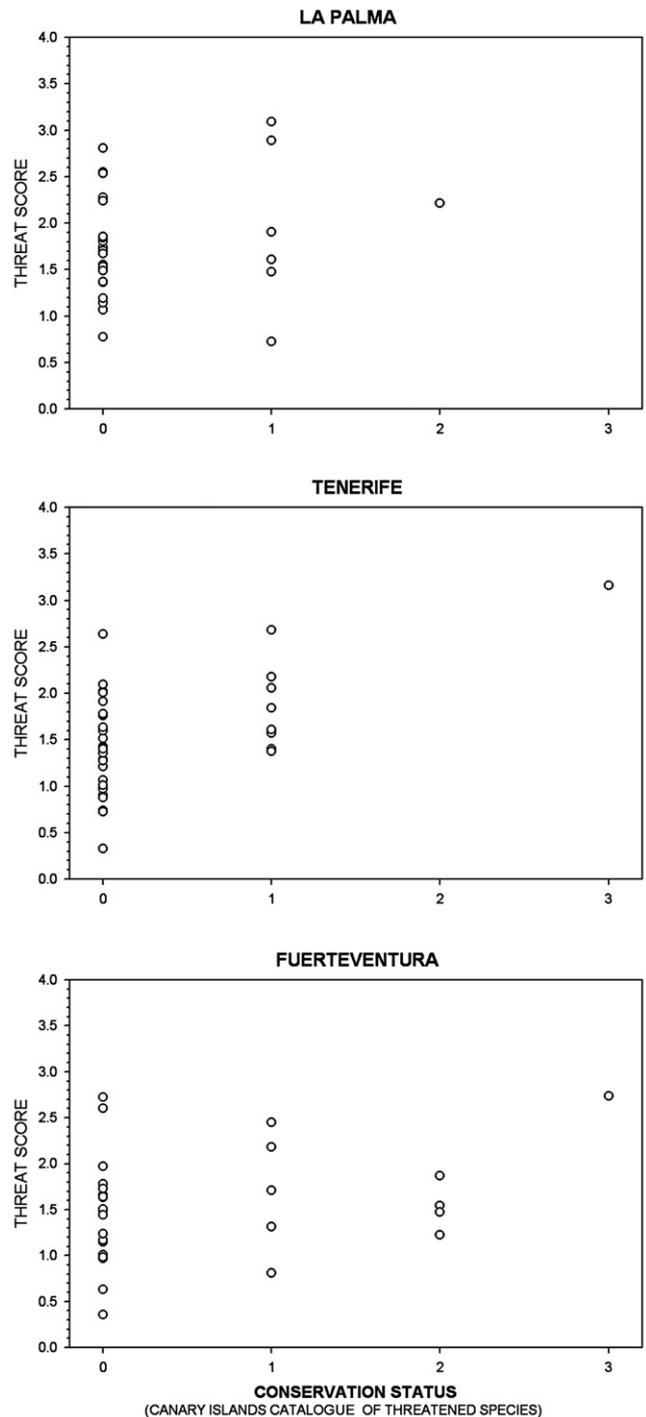


Fig. 2. Relationship between the threat scores of 44 terrestrial bird species of the Canary Islands and the categories defined in the revised Canary Islands Catalogue of Threatened Species (Martín Esquivel et al. 2004), in three islands representative of the environmental characteristics of the archipelago. 0 – Not threatened, 1 – special interest, 2 – vulnerable, and 3 – endangered species.

is unchangeable. Fourth, rarity and threat risk is not randomly distributed throughout the evolutionary history of species, but it is more concentrated in some clades than in others (Gage et al. 2004; Gittleman & Purvis 1998; McKinney 2001). This is clearly shown by the results of this paper, as phylogenetic effects account for 24.3% of the variation within taxa in the threat score. Finally, interactions between life history and human-caused threats, such as habitat loss and direct exploitation, may result in different effects of the threats on different taxa (Owens & Bennett 2000). If fore-

seen, these provide the opportunity to tailor conservation actions for specific groups of taxa. Nevertheless, and because population decline in several species is strongly linked with alteration of agricultural habitats (Martín & Lorenzo 2001; e.g., *Burhinus oedicnemus*, *Streptopelia turtur*, *Carduelis cannabina*, *Carduelis carduelis*, *Petronia petronia*; see the positive association between preferences for agricultural cover and the threat score in Table 2), changing agricultural practice is the primary driver of population declines, which is certainly something that can be mitigated by conservation actions and policy intervention.

We did not find a significant relationship between body size and threat score. Large body size is apparently a common correlate of present extinction susceptibility in many vertebrate groups (Brook & Bowman 2005; Purvis et al. 2003; Reynolds et al. 2005), and especially so in birds (Gaston & Blackburn 1995; Owens & Bennett 2000). Nevertheless, several authors have empirically showed that large-bodied species of island avifaunas do not suffer from a higher extinction proneness when compared with small-sized ones (e.g., Faaborg 1982; Gottelli & Graves 1990; Thiollay 1998). It must also be noted that some large-bodied species in the Canary Islands hold elevated population densities (*C. undulata*: Carrascal et al. 2006; *C. corax*: Nogales 1994; *Cursorius cursor*: Palomino et al. 2008). On the other hand, among several endemic taxa now extinct there are small and big species, which illustrate the lack of a clear pattern of body size and extinction in the Canary Islands (Gangoso et al. 2006; Martín & Lorenzo 2001; Rando & Alcover 2008; Rando et al. 1999): *Puffinus holeae* (c. 700 g), *Haematopus meadewaldoi* (perhaps 600–800 g), *P. olsoni* (c. 350 g), *Coturnix gomerae* (probably c. 150 g), *Emberiza alcoveri* (c. 37 g), *Carduelis triasi* (<50 g). Similarly, several terrestrial breeding bird species varying greatly in size have experienced severe conservation problems or local extinctions in the past 50 years (Gangoso et al. 2006; Martín & Lorenzo 2001): *C. undulata* (1200–2400 g), two birds of prey (*Milvus milvus* and *N. percnopterus*; 550–2200 g), *C. rufescens* (23 g) and the endemic *Fringilla teydea polatzeki* (30 g).

Therefore, managers seem to have limited room to fight conservation threat because some species naturally tend to be rare considering habitat preferences, ecological width and the allometric relationship of body mass with population density. These traits are inherent to each species and cannot be altered by any conservation program. This situation is particularly striking in islands, where it is obvious that small land masses harbour spatially restricted, small populations. But the key question is not how scarce the species are, but to ascertain the extinction proneness or which species are threatened and which will be next (Manne & Pimm 2001).

Association between administrative catalogues and measures of extinction risk

The administrative lists and the Spanish Red Data Book tested are poorly related to descriptors ordering the extinction risk and loss of taxonomic singularity, so some changes would make their categories more coherent. Moreover, although the Spanish Red Data Book and the revision of the official Catalogue explicitly take into account raw estimates of density, endemism and area of occupancy, they do not correlate with population trends (see Table 1). Population changes are taken into account to estimate extinction risk and general biodiversity loss (Gregory et al. 2005; IUCN 2001). Trends help to discover the causes of threat and may be used as an early warning signal of conservation problems, so they are crucial pieces of information (Newton 2004; Seoane & Carrascal 2008; Van Turnhout et al. 2007). We found that even a crude measure of population trend helps to characterize the degree of threat.

The 97 taxa (population-island) of bird species cluster together according to the descriptors of rarity and taxonomic distinctiveness

after applying an objective mathematical clustering approach. There are two extreme groups, one composed mainly of non-endemic taxa with large within-island ranges, high densities and positive or stable trends (around half of the total studied taxa), and another of rare species that are currently restricted to few places where they attain low abundances and/or have undergone severe reductions in the last 25 years (although they were likely more common in the near past; Martín & Lorenzo 2001). There exist many coincidences between the official normative list of the Canary Islands Catalogue of Threatened Species and the cluster of species of high conservation concern: 12 out of 21 species in this cluster are considered as endangered or vulnerable in the normative catalogue (e.g., *N. percnopterus* and *C. undulata* in Fuerteventura, *B. oedicnemus* and *Upupa epops* in Tenerife and La Palma, *Falco pelegrinoides* and *C. corax*). Conversely, the administrative revision of the Canary Islands Catalogue of Threatened Species (Martín et al. 2005) downgrades many species to categories of low conservation concern that have high threat scores in our classification: only four out of 21 species included in the cluster of highest risk are considered as endangered or vulnerable by this administrative revision.

There are also many points of agreement in the set of non-endangered species between our ordination and clustering of species and the position of the species in the normative or administrative revision of the Canary Islands Catalogue of Threatened Species. Thus, none of the 52 populations included in our two groups of least concern (codes 0 and 1 in Appendix B) is considered vulnerable by the administrative revision of the Canary Islands Catalogue. On the contrary, the older normative catalogue shows more discrepancies: seven of those taxa are considered vulnerable (*C. bollii*, *C. junoniae* in Tenerife and La Palma, *Pyrhacorax pyrrhacorax*, *Saxicola dacotiae* and *F. teydea* in Tenerife). However, these species have undergone positive population trends in the last 25 years and have large populations in the islands they inhabit (Carrascal et al. 2008a; Lorenzo 2007; Martín & Lorenzo 2001; Martín Esquivel et al. 2004), thus it is questionable whether they should raise immediate conservation concerns.

We found a larger disagreement with the Spanish Red Data Book. This is built by first applying global IUCN criteria designed for the whole distribution area of species to a local geographic context. Then, the categories of threat are claimed to be elaborated by using regional guidelines that take into account the contribution of Canary populations to the world population of each species. However, a meticulous reading of the species accounts reveals that this regional correction has not been done, at least quantitatively. Indeed, there is little reference to the guidelines in the text of the species accounts and when sizes of island populations are estimated these are apparently based on expert opinion rather than on factual data. Reported population sizes (generally not attributed) are much lower than recent estimates based on standardized sampling design and methods (see for example Carrascal et al. 2008a, 2008b; Palomino et al. 2008). Moreover, to our view the species accounts overstate the likely (or just presumed) negative effect of human impacts and the fact that the species/populations are endangered because they live in areas <5000 km², thus making them more prone to enter a category of threat. Dubiously, the categories of this red book correlate positively with recent population trend (more positive trends for higher threat categories). All in all, we think the Spanish Red Data Book tend to err too much on the side of caution for the Canary taxa, as might be commonplace in local lists of endangered species (see Craig 2006). This overly cautionary view can overwhelm policy makers and managers and policy, which could have difficulties to prioritize among species.

We find two general dilemmas when confronting the extinction risk predicted by our model to the actual process of assigning a threat category. First, there are several species that were widely

distributed and very abundant in the first half of the XX century but are currently on the brink of extinction likely due to the abandonment of traditional agricultural activity. So, what should managers and legislators do with this set of species that show a high degree of ecological rarity (and thus a high likelihood of extinction) but are not taxonomically singular (i.e., they are not endemic)? Managers would probably not be willing to invest part of their scarce resources on the conservation of these seemingly anthropic species. What category of threat should be assigned to them? Rarity alone would assign them to the maximum category, though those species benefited from human alteration of natural habitats (deforestation for pasturelands and agriculture) and once this human transformation of land ceased, they became scarce and are now about to disappear. This is the case of several autochthonous species in La Palma and Tenerife (*Coturnix coturnix*, *U. epops*, *P. petronia*, *C. carduelis*, *C. cannabina*, *Miliaria calandra*). Also, from the point of view of sheer species richness they amount close to 20% of the terrestrial birds in La Palma, and thus their disappearance would mean a considerable loss of biodiversity. The disappearance would likely be irreversible for practical terms, because those species occupy oceanic islands at the very southwestern limit of their Palaearctic distribution range, which would make it very difficult for them to immigrate and establish again. They are not recent invaders and conservation advocates consider them to be part of the natural fauna in their own right. Indeed, some of these non-endemic species reach high threat scores and are included in the cluster of highest conservation concern in our ordination/classification of proneness to extinction risk (see code 3 for variable KM-4 in Appendix B).

Second, five species are overvalued in the normative Canary Catalogue according to our model of rarity (*C. bollii*, *C. junoniae*, *P. pyrrhocorax*, *S. dacotiae* and *F. teydea* are considered 'vulnerable' in the Catalogue). Although four of them are endemic species, they are abundant in their preferred habitats. The administrative revision of the Canary Catalogue downgrades these five species to the categories of 'non-threatened' or 'special interest'. However, given that *S. dacotiae* and *P. pyrrhocorax* currently occupy just one island (Martín & Lorenzo 2001; there is some evidence that they could have occupied other islands as well in the past, Rando et al. 1997) and they raise social interest, they could be left, debatably, in a middle category. We contend that, when building a list of threat, managers, legislators and conservation advocates would frequently find these dissonances between any general model of extinction risk and the social interest a particular species raises, and that the solution to these dilemmas calls for the consideration of a set of social and economic values external to the building of any objective model of rarity (Farrier et al. 2007; Leonard 2008).

Finally, two endemic subspecies (*Accipiter nisus granti* and *Buteo buteo insularum*) have high threat scores and are included in the highest cluster of conservation concern of our analyses, but they are not included in the high threat categories of the catalogues (considered of 'special interest' by the normative catalogue, and as 'non-threatened' by its administrative revision). These raptor species have 'slow' life cycles and live at low densities (Gaston & Blackburn 2000; Reynolds 2003). Therefore, if they have restricted geographical ranges, it is probable that declining abundances and range sizes mediated by habitat loss or mortality render them more susceptible to local extinctions that would result in their extirpation from small islands. In insular volcanic environments the probability of catastrophic natural phenomena is high (e.g., volcanic eruptions, storms) and the influence of human impacts are proportionally higher due to the small area (Fernández-Palacios & Whittaker 2008; Whittaker & Fernández-Palacios 2007). Therefore, in our opinion it would be sensible to upgrade them to higher categories of conservation concern, especially considering that their local extinction implies an irreparable loss of taxonomic singular-

ity (because they are endemic taxa), in spite of their currently safe conservation status.

In conclusion, simple ecological models similar to the one presented here could be built to quantify the threat status of avifaunas elsewhere. Different bird populations may suffer from different environmental pressures across the distribution range of this taxa (changes in climate, patchiness, isolation, human impacts, etc.), which determine local variations in extinction proneness and threat status (Isaac & Cowlshaw 2004). In our opinion, the ecological bases of rarity (Gaston 1994; Manne & Pimm 2001) should be taken into account at the regional scale to understand why some populations/species are at higher extinction risk whereas others remain relatively safe. This approach would provide firmer grounds on which to base conservation priorities, complementary to other current alternatives (Gärdenfors 2001).

Acknowledgements

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Appendix A.

Synthesis of the criteria used by Martín et al. (2005) to revise the Canary Islands Catalogue of Threatened Species, the official document for the oceanic archipelago of the Canary Islands.

Martín et al. (2005) state that while working for the Canary regional government they were compelled to develop alternative criteria to the IUCN scheme for the Canary Catalogue of Threatened Species. They argued that this was so mainly because they did not find the IUCN scheme – which is designed for global assessments – to suit adequately their insular territorial context, since it tends to overvalue threat status in islands due to their small areas (see an elaboration of this argument in Martín 2009).

Martín et al. (2005) revision focused on evolutionary significant units, defined as those natural populations whose extinction would imply the disappearance of an endemic taxa. Also, they extend the focus to closed populations, even though these populations may not be morphologically or genetically different from others in continental areas or neighboring islands.

Their scheme proceeds according to two main steps. First, they evaluate whether a given taxa shows either a decline in population size, a deleterious change in distribution (decreasing occupancy and increasing fragmentation), or an increase in extinction probability. They assign three categories of threat as follows (for the complete set of criteria, please, see pp. 86–87 in the original reference):

Endangered (from the Spanish “en peligro de extinción”):

If population size or distribution have decreased by 75% in the last 10 years or three generations. This threshold is lowered to 20% if the number of mature individuals is less than 250. This category also applies if occupancy is below 20 km² (for native taxa present in more than one island), 5 km² (for native taxa present in just one island) or 2.5 km² (for native taxa occupying less than 1% of an island).

Also, the same applies if range size is lower than 200 km² and the ratio between the number of occupied localities and the number of occupied islands is equal or greater than 4.

Populations with fewer than 25 mature individuals qualify as Endangered if have negative trends or fluctuations in size.

If the threat affects mainly to habitat, the category is *Sensitive to habitat alteration* (from the Spanish “sensible a la alteración del hábitat”).

Vulnerable (from the Spanish “vulnerable”):

If population size or distribution have decreased by 50% in the last 10 years or three generations. The population threshold is lowered to 10% if the number of mature individuals is less than 1000. This category also applies if occupancy is below 80 km² (for native taxa present in more than one island), 20 km² (for native taxa present in just one island) or 10 km² (for native taxa occupying less than 1% of an island).

Also, the same applies if range size is lower than 200 km² and the ratio between the number of occupied localities and the number of occupied islands is equal or greater than 3.

Populations with fewer than 100 mature individuals qualify as Vulnerable if have negative trends or fluctuations in size.

Special interest (from the Spanish “de interés especial”):

Not necessarily declining but with cultural, ecological or scientific interest. These are not considered threatened species as they are not declining – as opposed to the other three categories.

Appendix B.

Species data for the terrestrial birds in the Canary Islands. For each taxa and island (ISLAND: LP, La Palma; TF, Tenerife; FV, Fuerteventura) the following are given: the degree of endemicity (ENDEM: 0 – recently introduced taxa; 1, autochthonous taxa

shared with continental areas; 2, endemic subspecies for the Canary Islands; 3, endemic species for the Canary Islands), the population trend (TREND: –1, clearly declining; +1, clearly increasing, and 0 for stable or cyclic populations or unknown trend), ecological density (DENS, in birds/km²) estimated for the most preferred habitat, the area of occupancy (nUTM or number of 5 km × 5 km UTM squares with sure or probable breeding), the value given by the model of threat built with these variables (SCORE), body mass (MASS, in g), habitat breadth (HB), and coverage of urban areas (URBAN) and agricultural environments (AGRIC) in those 0.5 km transects where the species were observed. CTS: categories of the official Canary Islands Catalogue of Threatened Species (“risk of extinction” – 3; “sensible to habitat alteration” and “vulnerable” – 2; “special interest” – 1; “non-threatened” – 0); CTSr gives the revised category of threat in the Canary Islands Catalogue of Threatened Species (Martín Esquivel et al. 2004): 0 – not threatened, 1 – special interest, 2 – vulnerable, and 3 – endangered. SRDB: categories of the Spanish Red Data Book; “endangered” – 3; “vulnerable” – 2; “near threatened” – 1; “non-threatened” – 0. KM-4 assigns each population-island to any of the four groups obtained after applying the k-means procedure with variables ENDEMIC, TREND, DENS and nUTM (ordered from 0 – not threatened to 3 – highest conservation concern). Neither *Miliaria calandra* nor *Petronia petronia* were found in the surveys on La Palma, but the current Breeding Bird Atlas for the Canaries report these species as very scarce breeders in the period 2000–2003. We considered it was best to act prudently and include them with the lower density we registered for any species, assigned to the preferred habitat according to the Breeding Bird Atlas (0.5 birds/km²).

| Species | ISLAND | ENDEM | TREND | DENS | nUTM | SCORE | MASS | HB | URBAN | AGRIC | CTS | CTSr | SRDB | KM-4 |
|------------------------------|--------|-------|-------|-------|------|-------|-------|------|-------|-------|-----|------|------|------|
| <i>Accipiter nisus</i> | LP | 2 | 0 | 0.4 | 19 | 2.28 | 204 | 0.41 | 1.0 | 20.4 | 1 | 0 | 2 | 3 |
| <i>Accipiter nisus</i> | TF | 2 | 0 | 0.4 | 37 | 2.02 | 204 | 0.14 | 0.0 | 0.0 | 1 | 0 | 2 | 3 |
| <i>Alectoris barbara</i> | FV | 0 | 0 | 9.4 | 34 | 1.01 | 480 | 0.41 | 0.0 | 8.7 | 0 | 0 | 0 | 1 |
| <i>Alectoris barbara</i> | LP | 0 | 1 | 3.5 | 23 | 0.78 | 480 | 0.31 | 0.5 | 11.3 | 0 | 0 | 0 | 1 |
| <i>Alectoris barbara</i> | TF | 0 | -1 | 14.1 | 64 | 1.21 | 480 | 0.17 | 1.6 | 13.5 | 0 | 0 | 0 | 2 |
| <i>Anthus berthelotii</i> | FV | 3 | 0 | 50.4 | 75 | 1.50 | 16.5 | 0.77 | 0.3 | 15.9 | 1 | 0 | 0 | 0 |
| <i>Anthus berthelotii</i> | LP | 3 | 0 | 64.5 | 37 | 1.74 | 16.5 | 0.20 | 0.7 | 6.0 | 1 | 0 | 0 | 0 |
| <i>Anthus berthelotii</i> | TF | 3 | 0 | 53.3 | 88 | 1.43 | 16.5 | 0.45 | 0.8 | 37.3 | 1 | 0 | 0 | 0 |
| <i>Bucanetes githagineus</i> | FV | 1 | 0 | 46.0 | 54 | 0.97 | 18.1 | 0.62 | 0.5 | 22.3 | 1 | 0 | 3 | 1 |
| <i>Bucanetes githagineus</i> | TF | 1 | -1 | 6.1 | 5 | 2.64 | 18.1 | 0.11 | 0.8 | 21.1 | 1 | 0 | 3 | 2 |
| <i>Burhinus oedicnemus</i> | FV | 2 | 0 | 3.5 | 35 | 1.78 | 461 | 0.51 | 0.0 | 35.7 | 1 | 0 | 3 | 2 |
| <i>Burhinus oedicnemus</i> | LP | 2 | 0 | 0.4 | 4 | 2.89 | 461 | 0.18 | 2.5 | 54.5 | 2 | 1 | 3 | 3 |
| <i>Burhinus oedicnemus</i> | TF | 2 | -1 | 1.6 | 16 | 2.68 | 461 | 0.09 | 0.5 | 13.2 | 2 | 1 | 3 | 3 |
| <i>Buteo buteo</i> | FV | 2 | 0 | 0.8 | 34 | 1.97 | 806.5 | 0.44 | 0.0 | 21.0 | 1 | 0 | 1 | 3 |
| <i>Buteo buteo</i> | LP | 2 | 0 | 0.4 | 21 | 2.24 | 806.5 | 0.53 | 3.1 | 25.7 | 1 | 0 | 1 | 3 |
| <i>Buteo buteo</i> | TF | 2 | 0 | 1.1 | 28 | 2.01 | 806.5 | 0.27 | 1.4 | 19.2 | 1 | 0 | 1 | 3 |
| <i>Calandrella rufescens</i> | FV | 1 | 0 | 94.0 | 65 | 0.81 | 23.3 | 0.54 | 0.0 | 35.4 | 1 | 1 | 3 | 0 |
| <i>Carduelis cannabina</i> | FV | 2 | 0 | 22.5 | 29 | 1.63 | 17.6 | 0.55 | 8.7 | 9.8 | 0 | 0 | 0 | 1 |
| <i>Carduelis cannabina</i> | LP | 2 | -1 | 16.3 | 11 | 2.55 | 17.6 | 0.26 | 6.1 | 84.4 | 0 | 0 | 0 | 1 |
| <i>Carduelis cannabina</i> | TF | 2 | 0 | 63.1 | 38 | 1.40 | 17.6 | 0.18 | 1.7 | 86.9 | 0 | 0 | 0 | 0 |
| <i>Carduelis carduelis</i> | FV | 1 | -1 | 2.7 | 7 | 2.61 | 16 | 0.19 | 1.3 | 57.5 | 0 | 0 | 0 | 2 |
| <i>Carduelis carduelis</i> | TF | 1 | -1 | 1.9 | 29 | 2.09 | 16 | 0.20 | 27.0 | 46.2 | 0 | 0 | 0 | 2 |
| <i>Carduelis chloris</i> | TF | 1 | 0 | 3.7 | 29 | 1.51 | 26.5 | 0.16 | 36.9 | 54.2 | 0 | 0 | 0 | 2 |
| <i>Chlamydotis undulata</i> | FV | 1 | -1 | 1.0 | 28 | 2.18 | 1245 | 0.23 | 0.0 | 29.6 | 3 | 1 | 3 | 3 |
| <i>Columba bolli</i> | LP | 3 | 1 | 58.3 | 15 | 1.61 | 286 | 0.14 | 0.0 | 0.0 | 2 | 1 | 1 | 1 |
| <i>Columba bolli</i> | TF | 3 | 1 | 35.9 | 19 | 1.57 | 286 | 0.14 | 0.1 | 0.6 | 2 | 1 | 1 | 1 |
| <i>Columba junoniae</i> | LP | 3 | 1 | 42.9 | 23 | 1.48 | 328.7 | 0.27 | 0.3 | 6.5 | 2 | 1 | 3 | 1 |
| <i>Columba junoniae</i> | TF | 3 | 1 | 13.0 | 13 | 1.84 | 328.7 | 0.07 | 0.0 | 0.0 | 2 | 1 | 3 | 1 |
| <i>Columba livia</i> | FV | 1 | 0 | 130.7 | 38 | 0.98 | 216 | 0.21 | 33.8 | 13.0 | 0 | 0 | 0 | 0 |
| <i>Columba livia</i> | LP | 1 | 0 | 117.9 | 31 | 1.07 | 216 | 0.45 | 13.4 | 74.9 | 0 | 0 | 0 | 0 |
| <i>Columba livia</i> | TF | 1 | 0 | 121.2 | 72 | 0.74 | 216 | 0.31 | 36.6 | 16.0 | 0 | 0 | 0 | 0 |
| <i>Corvus corax</i> | FV | 2 | 0 | 0.7 | 46 | 1.87 | 1250 | 0.81 | 0.0 | 18.9 | 2 | 2 | 3 | 3 |
| <i>Corvus corax</i> | LP | 2 | 0 | 1.2 | 16 | 2.21 | 1250 | 0.31 | 0.4 | 7.2 | 2 | 2 | 3 | 3 |
| <i>Corvus corax</i> | TF | 2 | -1 | 0.1 | 11 | 3.16 | 1250 | 0.07 | 0.0 | 0.0 | 2 | 3 | 3 | 3 |
| <i>Coturnix coturnix</i> | FV | 1 | 0 | 13.4 | 20 | 1.50 | 98.4 | 0.24 | 0.0 | 72.8 | 0 | 0 | 0 | 1 |
| <i>Coturnix coturnix</i> | LP | 1 | -1 | 0.4 | 15 | 2.54 | 98.4 | 0.18 | 2.5 | 54.5 | 0 | 0 | 0 | 3 |
| <i>Coturnix coturnix</i> | TF | 1 | 0 | 4.0 | 15 | 1.76 | 98.4 | 0.07 | 3.6 | 100.0 | 0 | 0 | 0 | 2 |
| <i>Cursorius cursor</i> | FV | 1 | 0 | 5.4 | 24 | 1.54 | 108 | 0.20 | 0.0 | 0.0 | 2 | 2 | 3 | 2 |
| <i>Cyanistes teneriffae</i> | FV | 1 | 0 | 16.8 | 20 | 1.48 | 11.3 | 0.34 | 4.1 | 33.0 | 2 | 2 | 3 | 1 |
| <i>Cyanistes teneriffae</i> | LP | 2 | 0 | 29.2 | 22 | 1.71 | 11.3 | 0.46 | 1.5 | 12.1 | 1 | 0 | 0 | 1 |
| <i>Cyanistes teneriffae</i> | TF | 2 | 0 | 71.5 | 86 | 1.07 | 11.3 | 0.56 | 4.1 | 5.7 | 1 | 0 | 3 | 0 |

| Species | ISLAND | ENDEM | TREND | DENS | nUTM | SCORE | MASS | HB | URBAN | AGRIC | CTS | CTSr | SRDB | KM-4 |
|---------------------------------|--------|-------|-------|-------|------|-------|-------|------|-------|-------|-----|------|------|------|
| <i>Dendrocopos major</i> | TF | 2 | 1 | 6.7 | 21 | 1.40 | 80.7 | 0.14 | 0.0 | 0.0 | 2 | 1 | 2 | 2 |
| <i>Erithacus rubecula</i> | LP | 1 | 0 | 60.6 | 18 | 1.36 | 16.7 | 0.30 | 0.3 | 3.3 | 1 | 0 | 0 | 1 |
| <i>Erithacus rubecula</i> | TF | 2 | 0 | 88.3 | 46 | 1.29 | 16.7 | 0.33 | 0.3 | 4.2 | 1 | 0 | 0 | 0 |
| <i>Falco pelegrinoides</i> | FV | 1 | 0 | 0.1 | 8 | 2.45 | 469.8 | 0.16 | 0.0 | 0.0 | 3 | 1 | 3 | 3 |
| <i>Falco pelegrinoides</i> | LP | 1 | 1 | 0.1 | 9 | 1.90 | 469.8 | 0.09 | 0.0 | 4.9 | 3 | 1 | 3 | 3 |
| <i>Falco pelegrinoides</i> | TF | 1 | 1 | 0.1 | 19 | 1.61 | 469.8 | 0.14 | 0.0 | 0.0 | 3 | 1 | 3 | 3 |
| <i>Falco tinnunculus</i> | FV | 2 | 0 | 2.9 | 51 | 1.66 | 174.5 | 0.32 | 0.0 | 42.5 | 1 | 0 | 2 | 2 |
| <i>Falco tinnunculus</i> | LP | 2 | 0 | 3.6 | 31 | 1.82 | 174.5 | 0.73 | 5.9 | 39.8 | 1 | 0 | 0 | 2 |
| <i>Falco tinnunculus</i> | TF | 2 | 0 | 2.7 | 97 | 1.41 | 174.5 | 0.64 | 2.5 | 28.1 | 1 | 0 | 0 | 2 |
| <i>Fringilla coelebs</i> | LP | 2 | 0 | 112.0 | 25 | 1.50 | 23 | 0.29 | 0.3 | 2.2 | 1 | 0 | 3 | 0 |
| <i>Fringilla coelebs</i> | TF | 2 | 0 | 33.4 | 28 | 1.60 | 23 | 0.22 | 0.5 | 6.7 | 1 | 0 | 0 | 1 |
| <i>Fringilla teydea</i> | TF | 3 | 1 | 37.7 | 31 | 1.38 | 30.1 | 0.16 | 0.0 | 0.0 | 2 | 1 | 2 | 1 |
| <i>Lanius meridionalis</i> | FV | 1 | 0 | 6.8 | 61 | 1.15 | 63.5 | 0.80 | 1.4 | 15.7 | 1 | 0 | 0 | 2 |
| <i>Lanius meridionalis</i> | TF | 1 | 0 | 2.5 | 24 | 1.63 | 63.5 | 0.30 | 0.5 | 13.4 | 1 | 0 | 0 | 2 |
| <i>Miliaria calandra</i> | FV | 1 | 0 | 4.3 | 20 | 1.64 | 43 | 0.21 | 0.0 | 77.2 | 0 | 0 | 0 | 2 |
| <i>Miliaria calandra</i> | LP | 1 | -1 | 0.5 | 7 | 2.81 | 43 | 0.14 | 4.9 | 100.0 | 0 | 0 | 0 | 3 |
| <i>Miliaria calandra</i> | TF | 1 | -1 | 49.9 | 17 | 1.91 | 43 | 0.09 | 0.4 | 100.0 | 0 | 0 | 0 | 1 |
| <i>Motacilla cinerea</i> | LP | 2 | 0 | 12.5 | 31 | 1.67 | 18 | 0.29 | 18.5 | 84.1 | 1 | 0 | 0 | 1 |
| <i>Motacilla cinerea</i> | TF | 2 | 0 | 7.4 | 74 | 1.40 | 18 | 0.31 | 87.3 | 0.0 | 1 | 0 | 0 | 2 |
| <i>Neophron percnopterus</i> | FV | 2 | -1 | 0.3 | 23 | 2.74 | 2035 | 0.39 | 0.0 | 24.3 | 3 | 3 | 3 | 3 |
| <i>Passer hispaniolensis</i> | FV | 1 | 0 | 406.6 | 65 | 0.63 | 27 | 0.24 | 29.8 | 24.6 | 0 | 0 | 0 | 0 |
| <i>Passer hispaniolensis</i> | LP | 1 | 0 | 14.5 | 9 | 1.81 | 27 | 0.10 | 37.5 | 65.4 | 0 | 0 | 0 | 1 |
| <i>Passer hispaniolensis</i> | TF | 1 | 0 | 153.1 | 70 | 0.72 | 27 | 0.08 | 83.1 | 3.7 | 0 | 0 | 0 | 0 |
| <i>Petronia petronia</i> | TF | 1 | -1 | 165.2 | 8 | 2.06 | 31 | 0.08 | 0.2 | 100.0 | 1 | 1 | 0 | 0 |
| <i>Phylloscopus canariensis</i> | LP | 3 | 0 | 248.1 | 40 | 1.55 | 7.7 | 0.74 | 4.0 | 29.2 | 1 | 0 | 0 | 0 |
| <i>Phylloscopus canariensis</i> | TF | 3 | 0 | 122.3 | 101 | 1.27 | 7.7 | 0.65 | 5.2 | 16.9 | 1 | 0 | 0 | 0 |
| <i>Pterocles orientalis</i> | FV | 1 | 1 | 6.7 | 14 | 1.23 | 474 | 0.43 | 0.0 | 14.6 | 2 | 2 | 2 | 2 |
| <i>Pyrrhocorax pyrrhocorax</i> | LP | 1 | 1 | 21.6 | 35 | 0.73 | 321.5 | 0.40 | 4.2 | 22.4 | 2 | 1 | 3 | 1 |
| <i>Regulus regulus</i> | LP | 2 | 0 | 146.5 | 21 | 1.53 | 5.8 | 0.34 | 0.1 | 0.6 | 1 | 0 | 0 | 0 |
| <i>Regulus regulus</i> | TF | 2 | 0 | 210.1 | 36 | 1.28 | 5.8 | 0.24 | 0.3 | 0.0 | 1 | 0 | 0 | 0 |
| <i>Saxicola dacotiae</i> | FV | 3 | 0 | 26.4 | 53 | 1.71 | 16.5 | 0.39 | 0.0 | 7.3 | 2 | 1 | 3 | 1 |
| <i>Scolopax rusticola</i> | LP | 1 | 0 | 2.0 | 15 | 1.85 | 305 | 0.20 | 0.0 | 0.0 | 1 | 0 | 0 | 2 |
| <i>Scolopax rusticola</i> | TF | 1 | 0 | 2.3 | 17 | 1.78 | 305 | 0.07 | 0.0 | 0.0 | 1 | 0 | 0 | 2 |
| <i>Serinus canarius</i> | FV | 3 | 0 | 4.0 | 7 | 2.73 | 15.3 | 0.33 | 4.0 | 55.4 | 0 | 0 | 0 | 2 |
| <i>Serinus canarius</i> | LP | 3 | 0 | 124.8 | 36 | 1.67 | 15.3 | 0.52 | 10.6 | 15.6 | 0 | 0 | 0 | 0 |
| <i>Serinus canarius</i> | TF | 3 | 0 | 128.2 | 81 | 1.35 | 15.3 | 0.38 | 9.0 | 67.9 | 0 | 0 | 0 | 0 |
| <i>Streptopelia decaocto</i> | FV | 1 | 1 | 240.2 | 43 | 0.36 | 196 | 0.12 | 45.6 | 9.0 | 0 | 0 | 0 | 0 |
| <i>Streptopelia decaocto</i> | LP | 1 | 1 | 54.3 | 11 | 1.07 | 196 | 0.17 | 29.0 | 73.6 | 0 | 0 | 0 | 1 |
| <i>Streptopelia decaocto</i> | TF | 1 | 1 | 115.5 | 58 | 0.33 | 196 | 0.07 | 85.6 | 0.9 | 0 | 0 | 0 | 0 |
| <i>Streptopelia turtur</i> | FV | 1 | 0 | 39.6 | 34 | 1.17 | 125 | 0.18 | 1.4 | 32.6 | 0 | 0 | 2 | 1 |
| <i>Streptopelia turtur</i> | LP | 1 | 0 | 6.4 | 26 | 1.49 | 125 | 0.52 | 6.3 | 61.7 | 0 | 0 | 2 | 2 |
| <i>Streptopelia turtur</i> | TF | 1 | 0 | 18.7 | 84 | 0.90 | 125 | 0.45 | 6.5 | 21.2 | 0 | 0 | 2 | 1 |
| <i>Sylvia atricapilla</i> | FV | 1 | 1 | 2.9 | 5 | 1.73 | 22.3 | 0.17 | 27.6 | 0.0 | 1 | 0 | 0 | 2 |
| <i>Sylvia atricapilla</i> | LP | 1 | 0 | 48.4 | 34 | 1.14 | 22.3 | 0.51 | 11.3 | 70.6 | 1 | 0 | 0 | 1 |
| <i>Sylvia atricapilla</i> | TF | 1 | 0 | 30.3 | 78 | 0.88 | 22.3 | 0.40 | 10.4 | 26.8 | 1 | 0 | 0 | 1 |
| <i>Sylvia conspicillata</i> | FV | 2 | 0 | 37.7 | 68 | 1.24 | 9.5 | 0.57 | 0.2 | 16.9 | 1 | 0 | 0 | 1 |
| <i>Sylvia conspicillata</i> | LP | 2 | 0 | 11.6 | 20 | 1.85 | 9.5 | 0.22 | 1.3 | 14.9 | 1 | 0 | 0 | 1 |
| <i>Sylvia conspicillata</i> | TF | 2 | 0 | 8.7 | 70 | 1.40 | 9.5 | 0.45 | 1.3 | 36.1 | 1 | 0 | 0 | 2 |
| <i>Sylvia melanocephala</i> | FV | 1 | 0 | 46.8 | 16 | 1.44 | 11.2 | 0.21 | 0.6 | 23.2 | 1 | 0 | 0 | 1 |
| <i>Sylvia melanocephala</i> | LP | 1 | 0 | 52.5 | 29 | 1.20 | 11.2 | 0.27 | 2.7 | 44.3 | 1 | 0 | 0 | 1 |
| <i>Sylvia melanocephala</i> | TF | 1 | 0 | 23.0 | 66 | 0.97 | 11.2 | 0.41 | 1.7 | 23.4 | 1 | 0 | 0 | 1 |
| <i>Turdus merula</i> | LP | 2 | 0 | 130.2 | 33 | 1.37 | 86.1 | 0.55 | 4.3 | 34.1 | 0 | 0 | 0 | 0 |
| <i>Turdus merula</i> | TF | 2 | 0 | 127.6 | 83 | 1.01 | 86.1 | 0.37 | 5.3 | 14.3 | 0 | 0 | 0 | 0 |
| <i>Upupa epops</i> | FV | 1 | 0 | 3.8 | 48 | 1.31 | 59.8 | 0.81 | 1.3 | 21.2 | 2 | 1 | 0 | 2 |
| <i>Upupa epops</i> | LP | 1 | -1 | 0.3 | 4 | 3.09 | 59.8 | 0.18 | 4.9 | 79.0 | 2 | 1 | 0 | 3 |
| <i>Upupa epops</i> | TF | 1 | -1 | 0.7 | 32 | 2.18 | 59.8 | 0.27 | 26.0 | 18.6 | 2 | 1 | 0 | 3 |

Appendix C.

Description of the strata in which samples were grouped for the analyses. Values are means. It is also given the sample

size (n : number of line transects per strata) and the estimated area of the strata in the islands. Data was gathered in the field and estimations of cover and height were done by eye.

| | n | Area (km ²) | Altitude (m) | Urban cover (%) | Grass/Forb cover (%) | Shrub cover (%) | Shrub height (m) | Tree cover (%) | Tree height (m) | Agricultural cover (%) |
|---|-----|-------------------------|--------------|-----------------|----------------------|-----------------|------------------|----------------|-----------------|------------------------|
| Fuerteventura | | | | | | | | | | |
| Urban areas | 31 | 22 | 111 | 56 | 10 | 4 | 0.6 | 6 | 5.3 | 0 |
| Agricultural areas | 25 | 76 | 166 | 0 | 26 | 15 | 0.9 | 6 | 2.7 | 100 |
| Pastureland | 41 | 46 | 258 | 0 | 41 | 6 | 0.3 | 0 | 0.0 | 100 |
| Rock, unvegetated | 68 | 188 | 118 | 0 | 7 | 4 | 0.2 | 0 | 0.0 | 0 |
| Plain, steppic areas (scarcely vegetated) | 106 | 293 | 100 | 0 | 6 | 4 | 0.1 | 0 | 0.0 | 0 |
| Plain, steppic areas (shrub cover > 10%) | 115 | 318 | 118 | 0 | 16 | 16 | 0.3 | 0 | 0.0 | 0 |

| | <i>n</i> | Area (km ²) | Altitude (m) | Urban cover (%) | Grass/Forb cover (%) | Shrub cover (%) | Shrub height (m) | Tree cover (%) | Tree height (m) | Agricultural cover (%) |
|--|----------|-------------------------|--------------|-----------------|----------------------|-----------------|------------------|----------------|-----------------|------------------------|
| Sandy areas covered with grasses, forbs and small shrubs ('jable') | 81 | 84 | 71 | 0 | 10 | 7 | 0.3 | 0 | 0.0 | 0 |
| Hilly, steppic areas (scarcely vegetated) | 70 | 301 | 268 | 0 | 15 | 4 | 0.2 | 0 | 0.1 | 0 |
| Hilly, steppic areas (shrub cover > 10%) | 71 | 306 | 260 | 0 | 22 | 18 | 0.4 | 0 | 0.0 | 0 |
| <i>Euphorbia</i> shrubland (on plains) | 21 | 19 | 41 | 0 | 8 | 16 | 0.8 | 0 | 0.1 | 0 |
| <i>Euphorbia</i> shrubland (on hills) | 33 | 2 | 386 | 0 | 20 | 20 | 0.6 | 0 | 0.1 | 0 |
| Riparian dwarf woodlands of <i>Tamarix canariensis</i> | 24 | 4 | 131 | 0 | 13 | 16 | 1.2 | 9 | 5.1 | 0 |
| Tenerife | | | | | | | | | | |
| Urban areas | 57 | 71 | 277 | 87 | 3 | 6 | 0.4 | 3 | 6.0 | 0 |
| Agricultural areas | 89 | 490 | 596 | 4 | 43 | 21 | 0.8 | 2 | 2.4 | 100 |
| Pastureland | 27 | 37 | 705 | 0 | 91 | 6 | 0.7 | 0 | 0.9 | 100 |
| Low-altitude, arid and scarcely vegetated areas | 38 | 3 | 73 | 0 | 3 | 11 | 0.4 | 0 | 0.0 | 0 |
| <i>Euphorbia</i> shrubland (sparse stands on more arid locations) | 27 | 102 | 352 | 0 | 0 | 40 | 0.7 | 0 | 0.0 | 0 |
| <i>Euphorbia</i> shrubland (dense stands with forbs) | 27 | 204 | 403 | 2 | 29 | 58 | 0.8 | 1 | 1.1 | 0 |
| Gullies with well developed vegetation cover | 33 | 8 | 396 | 1 | 11 | 62 | 0.9 | 4 | 3.5 | 0 |
| Mid-altitude shrubland | 35 | 33 | 1091 | 0 | 13 | 41 | 0.7 | 2 | 3.4 | 0 |
| Subalpine shrubland | 40 | 126 | 2202 | 0 | 0 | 37 | 1.1 | 0 | 0.0 | 0 |
| Poorly vegetated subalpine areas | 16 | 23 | 2499 | 0 | 0 | 2 | 0.2 | 0 | 0.0 | 0 |
| Tall heathlands ('fayal-brezal') | 23 | 261 | 905 | 0 | 5 | 47 | 2.2 | 45 | 4.8 | 0 |
| Evergreen forest ('laurisilva') | 36 | 269 | 926 | 0 | 0 | 19 | 1.8 | 83 | 10.0 | 0 |
| Pine forest (below 1500 m) | 45 | 294 | 1451 | 0 | 6 | 22 | 1.5 | 61 | 15.8 | 0 |
| Pine forest (above 1500 m) | 49 | 113 | 1805 | 0 | 1 | 10 | 1.0 | 41 | 11.0 | 0 |
| La Palma | | | | | | | | | | |
| Urban areas | 56 | 4 | 401 | 39 | 17 | 13 | 1.0 | 5 | 3.9 | 64 |
| Agricultural areas ('medianias') | 56 | 132 | 539 | 5 | 43 | 15 | 1.1 | 7 | 4.4 | 100 |
| Agricultural areas ('medianias') interspersed with shrubland | 19 | 41 | 432 | 4 | 27 | 44 | 1.4 | 4 | 3.5 | 100 |
| Banana groves | 22 | 48 | 121 | 4 | 4 | 7 | 0.7 | 66 | 3.4 | 100 |
| Lava fields | 20 | 108 | 524 | 1 | 0 | 10 | 0.6 | 0 | 0.5 | 0 |
| <i>Euphorbia</i> shrubland | 61 | 50 | 327 | 0 | 16 | 57 | 1.1 | 0 | 0.7 | 5 |
| Tall heathlands ('fayal-brezal') | 19 | 40 | 1042 | 0 | 13 | 43 | 1.9 | 45 | 6.2 | 0 |
| Subalpine shrubland | 20 | 18 | 2274 | 1 | 0 | 61 | 0.8 | 0 | 0.0 | 0 |
| Evergreen forest ('laurisilva') | 24 | 69 | 842 | 0 | 18 | 33 | 1.8 | 77 | 13.7 | 0 |
| Pine forests | 100 | 178 | 1348 | 0 | 8 | 22 | 0.9 | 36 | 12.2 | 0 |
| Pine forests interspersed with evergreen forest | 40 | 16 | 1172 | 0 | 11 | 35 | 1.9 | 47 | 14.1 | 0 |

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