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1 **TITLE**

2 Using species distributions models for designing conservation strategies of Tropical Andean
3 biodiversity under climate change

4

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ABSTRACT

Biodiversity in the Tropical Andes is under continuous threat from anthropogenic activities. Projected changes in climate will likely exacerbate this situation. Using species distribution models, we assess possible future changes in the diversity and climatic niche size of an unprecedented number of species for the region. We modeled a broad range of taxa (11,012 species of birds and vascular plants), including both endemic and widespread species and provide a comprehensive estimation of climate change impacts on the Andes. We find that if no dispersal is assumed, by 2050s, more than 50% of the species studied are projected to undergo reductions of at least 45% in their climatic niche, whilst 10% of species could be extinct. Even assuming unlimited dispersal, most of the Andean endemics (comprising ~5% of our dataset) would become severely threatened (>50% climatic niche loss). While some areas appear to be climatically stable (e.g. Pichincha and Imbabura in Ecuador; and Nariño, Cauca, Valle del Cauca and Putumayo in Colombia) and hence depict little diversity loss and/or potential species gains, major negative impacts were also observed. Tropical high Andean grasslands (páramos and punas) and evergreen montane forests, two key ecosystems for the provision of environmental services in the region, are projected to experience negative changes in species richness and high rates of species turnover. Adapting to these impacts would require a landscape-network based approach to conservation, including protected areas, their buffer zones and corridors. A central aspect of such network is the implementation of an integrated landscape management approach based on sustainable management and restoration practices covering wider areas than currently contemplated.

Keywords: Andes, biodiversity, conservation, climate change, threats, climatic niche, maxent

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1. Introduction

Despite ambitious goals to significantly reduce the rate of biodiversity loss by 2010 (CBD, 2007), biodiversity continues to be severely threatened (Ramirez-Villegas et al., 2012; Sachs et al., 2009). These threats include over exploitation of natural resources (e.g. water, agricultural soils), habitat loss and degradation, and invasive species (Butchart et al., 2010; Kim and Byrne, 2006). Biodiversity loss has been increasing since the second half of the 20th century, and is likely to continue into the future (Kim and Byrne, 2006; MEA, 2005). With climate change entailing likely increases in temperature and regional and seasonal changes in precipitation (Knutti and Sedlacek, 2013), ecosystems and their services are likely to suffer additional stresses (Chen et al., 2009; Feeley and Silman, 2010; Fuhrer, 2003; IPCC, 2007).

The Tropical Andes tops the list of worldwide hotspots for species diversity and endemism (Fjeldså et al., 1999; Gentry, 1995; Sklenár and Ramsay, 2001). For this reason, the region is considered a key priority for biodiversity conservation (Brooks et al., 2006; Myers et al., 2000). At the same time, the Tropical Andes have been identified as one of the most severely threatened natural areas globally (Jetz et al., 2007; Mittermeier et al., 1997). During the last century, concentration of human population and associated demands for goods and services in the inter-Andean valleys and the inner slopes of the Andean ridges, has transformed a significant portion of the natural landscape causing habitat loss and degradation followed by species extinction and disruption of ecosystem functions (e.g. water-flow regulation), especially in the Northern Andes (Bruinsma, 2003; Wassenaar et al., 2007; Armenteras et al., 2011; Rodriguez et al., 2013).

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71 Resource-base over-exploitation of natural resources has led to a severe land degradation process
72 (Podwojewski et al., 2002; Poulenard et al., 2001, 2004), increasing the pressure on the goods
73 and services provided by these ecosystems (Rundel and Palma, 2000). In addition, the Andes are
74 expected to undergo severe stresses over the next 100 years as a result of climate change
75 (Beaumont et al., 2011; Malcolm et al., 2006).
76
77 Addressing potential impacts from climate change is important because the environmental
78 impacts of human activities (Biesmeijer et al., 2006; MEA, 2005) could be exacerbated by the
79 likely rapid changes in the climate system during the 21st century (IPCC, 2007; Knutti and
80 Sedlacek, 2013). Warren et al. (2013) estimated that, in the absence of any climate change
81 mitigation strategy, large range contractions for ca. 60 % of plants and 35 % of animals could be
82 expected globally. Understanding and quantifying the extent at which climate change could
83 threaten Andean species is therefore critical since many of the species in the region occur in low
84 dense populations with narrow distribution patterns (i.e. endemics) with a high level of
85 replacement within the environmental gradients. These characteristics make the Andean biota
86 particularly sensitive to climate change disruptions.
87
88 Our primary objective was to assess the likely impacts of climate change on the distributions of
89 vascular plant and bird species of the Tropical Andes. Using species distributions modelling
90 techniques, we assessed the potential climatic niche of 11,012 species, and then projected them
91 under the SRES-A2 emission scenario for two periods: 2020 and 2050. Future projected changes
92 in species assemblages, including richness, turnover and range size were assessed. Lastly, the

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93 projected impacts in selected groups of species of Andean origin were analysed. Finally, we
94 discuss future strategies to reduce expected biodiversity loss.
95
96 **2. Study area**
97 The study area (Tropical Andes hereafter) comprises all interconnected areas above altitudes of
98 500 m within the countries of Venezuela, Colombia, Ecuador, Peru and Bolivia, plus the Sierra
99 Nevada de Santa Marta in Colombia, delimited using data from the SRTM digital elevation
100 model (Farr et al. 2007). Extending over 1.5 million km^2 from 11º N to 23º S, the Tropical Andes
101 are the longest and widest mountain region in the tropics (Figure 1) (Clapperton, 1993; Fjeldså
102 and Krabbe, 1990). The morphological and bioclimatic heterogeneity of the Andes have led to
103 the formation of an enormous diversity of microhabitats favouring speciation (Mittermeier et al.,
104 1997; Young et al., 2002). Moreover, their location between the lowlands of the Amazon, La
105 Chiquitanía and El Chaco to the east and the Chocó, Tumbes-Guayaquil and the arid systems of
106 the Sechura Desert to the west, has created complex dynamics of species exchange and isolation
107 (Bass et al., 2010; Young et al., 2002). The Tropical Andes harbours more than 45,000 vascular
108 plant (20,000 endemics) and 3,400 vertebrate species (1,567 endemics) in just 1 percent of the
109 Earth's land mass (Lamoreux et al., 2006; Olson et al., 2001).
110
111 **3. Methods**
112 We modelled the climatic niches of 11,012 species (1,555 birds and 9,457 plants) using species
113 distributions models. We modelled the climate-constrained present-day distributions of all
114 species, and projected them onto two different future periods (2020s, 2050s) and two contrasting
115 dispersal scenarios. The approach implemented here aims to evaluate the likely impacts of

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climate change on the widest array possible of Andean plant and bird species by mid 2020s and mid 2050s and comprises the following six steps:

1. Assembling of species occurrence data
2. Generation of climate surfaces
3. Maximum entropy species distribution modeling
4. Analysis of projected climate change impacts on species assemblages
5. Delineation of conservation recommendations for the 2020s and 2050s

3.1 Species datasets

Presence data for 11,012 species (1,555 birds and 9,457 plants) were sourced from three databases. CONDESAN, the Centro de Datos para la Conservación de la Universidad Nacional Agraria La Molina (CDC-UNALM), and a previous global study (Warren et al., 2013) (W2013). From the three sources, we extracted all occurrences in the five tropical Andean countries (i.e. Venezuela, Colombia, Ecuador, Peru and Bolivia) of all vascular plant clades (*Magnoliophyta*, *Pteridophyta*, *Pinophyta*, *Psilophyta*, *Cycadophyta*, *Gnetophyta*, *Lycopodiophyta*) and bird (class *Aves*, phylum *Chordata*) species with at least one record within the study area (Figure 1B). By including these three sources of data we ensured the inclusion of common and widespread species (see Warren et al. 2013) as well as narrow-range Andean endemics and imperil species (also see Sect. 4.1 for details).

CONDESAN's database consisted of data from multiple sources. Vascular plant specimen data were obtained from the Missouri Botanical Garden's Vascular Tropicos (VAST) nomenclatural database (Garden, 2004), the Herbarium of the National Science Institute in Colombia (ISN) and

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the Catholic University Herbarium (QCA) in Ecuador. Bird species data were obtained from databases belonging to the Chicago Field Museum of Natural History, Academy of Natural Sciences of Philadelphia, California Academy of Sciences and the Berkeley Museum of Natural History and cross-checked with BirdLife International database (version 2012). Additional data were obtained from private databases (Juan Fernando Freile for Antpittas, Paul Hamec for *Dendroica cerulea*; Cal Dodson-Lorena Endara for orchid's records and James Luteyn's database stored at the New York Botanical Garden site for *Ericaceae*) and published literature (Casares et al., 2003; Renjifo et al., 2002; Schuchmann et al., 2001). The CDC-UNALM database was produced from the review of papers and reports during the last 25 years. It also comprises field reports obtained by its own research as well as data provided by other national (i.e. Peruvian) researchers. The W2013 database was originally sourced from the Global Biodiversity Information Facility (GBIF, available at http://data.gbif.org). Warren et al. (2013) thoroughly checked the GBIF plant and animal database for location errors following the methodology of Ramirez-Villegas et al. (2012), whereby the consistency of the location data is verified at both geographic (using coastal and country borders) and environmental (using outlier-removal tests) levels. We carefully checked bird species names using BirdLife's taxonomy database as a reference. Plant taxonomy was verified using The Plant List (http://www.theplantlist.org, see Warren et al., 2013).

3.2 Climate data

Current climate data were derived from WorldClim (Hijmans et al., 2005). WorldClim is a global gridded dataset of monthly climatological means of maximum, minimum and mean temperature and total precipitation developed through Thin Plate Spline interpolation of long-

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term (i.e. 1950-2000) weather station records (Figure 1A). There is a generally dense distribution of weather stations across the core of our geographic analysis domain (Hijmans et al., 2005). Using the monthly WorldClim data we derived 10 'bioclimatic' indices (Busby, 1991; Rivas-Martinez, 2004) (Table 1). These indices describe annual and seasonal trends and allow for an adequate characterization of the species bioclimatic niches. These indices are important limiting factors for growth and development of species, and have been used extensively for predicting species distributions using presence-only data (Elith et al., 2006; Graham et al., 2008; Warren et al., 2013). For the Andes, the 10 bioclimatic indices chosen cover aspects of both average and extreme conditions of a year. In addition, the use of the ombrothermic index allows for differentiating climate conditions between and across ecosystems (Rivas-Martinez, 2004).

[Table 1 here]

We obtained future climate projections from the CMIP3 (Coupled Model Inter-comparison Project phase 3) web data portal (https://esg.llnl.gov:8443/index.jsp) (Meehl et al., 2007). We downloaded monthly time series of temperature and precipitation data for the baseline period (20^{th} century) and projections of future climate for the 21^{st} century for the SRES-A2 emission scenario for 24 different Intergovernmental Panel on Climate Change (IPCC) coupled GCMs (Table 2). We chose SRES-A2 because we considered the full-mitigation SRES-B1 unlikely, and because differences between SRES-A2 and SRES-A1B and SRES-A1FI by 2050s are negligible (Hawkins and Sutton, 2009). Based on the availability of maximum and minimum temperature data, we further selected a subset of nine GCMs (Table 2).

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185 **[Table 2 here]**

186

187 Using the complete GCM time series, for each of the GCMs, months and variables, we
188 calculated the 30 year running average over the baseline period (1961-1990) and two future
189 periods: 2020s (2010-2039) and 2050s (2040-2069), representing the early and mid- 21st century.
190 We then calculated the anomalies (deltas) of each GCM future scenario with respect to the
191 baseline period (average 1961-1990 climate) for each month, variable and period.

192

193 Given the significant heterogeneity in Andean climates, coarse scale GCM grids fail to represent
194 the diversity of niches where species are distributed, hence we increased the resolution of the
195 GCM data by means of empirical downscaling with the delta method (Ramirez-Villegas and
196 Jarvis, 2010). For each month, variable, and period, the respective set of GCM deltas was
197 averaged (i.e. ensemble mean). Temperature anomalies were directly added, whilst precipitation
198 anomalies were added as a relative factor to the value in WorldClim in order to avoid
199 precipitation values below zero due to the differences between the GCM simulated and
200 WorldClim observed baseline. For each of the future periods, we calculated the same bioclimatic
201 indices as for current climate data (Table 1). This yielded climate scenarios for each of the future
202 periods as an average trend of the set of available GCMs on the SRES-A2 emission scenario.

203

204 We used the ensemble mean (rather than individual GCMs) owing to processing and storage
205 needs, and given the considerable number of species being modelled and the resolution at which
206 the models were projected (2.5 arc-min).

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208 3.3 Species distribution models (SDMs)

209 Species distributions were modelled using Maxent (Phillips et al., 2006; Phillips and Dudík,
210 2008), a robust bioclimatic envelope modelling techniques (Smith et al., 2013). We modelled
211 only species with at least 10 distinct locations (Ramírez-Villegas et al., 2010; Wisz et al., 2008),
212 as a compromise between model quality and sufficient coverage of limited-range species.
213 Maxent models the climate-constrained distribution of a species using presence-only data and a
214 set of environmental descriptors (Elith et al., 2010; Phillips et al., 2006). Maxent has been tested
215 extensively and has been found to suitably perform as a state-of-the-art modelling technique both
216 under current and future conditions (Costa et al., 2010; Phillips, 2008; Smith et al., 2013).

217

218 Here, we followed a similar methodology to that employed by Warren et al. (2013), whereby
219 default features optimised to broad species groups were used to construct Maxent models for
220 each species (Phillips, 2008; Phillips et al., 2006; Phillips and Dudík, 2008). For each species we
221 drew 10,000 pseudo-absences from the countries where the species was reported (according to
222 our database). This was done to avoid over-fitting of the models whilst maintaining a good
223 discrimination between presence and absence of the species (Isaac et al., 2009; VanDerWal et
224 al., 2009).

225

226 Most niche modeling techniques are sensitive to the number of predictors used and Maxent is no
227 exception (Braunisch et al., 2013; Dormann, 2007; Phillips, 2008). Excess predictors in a Maxent
228 model can cause over-fitting and hence bias the responses under future scenarios by over-
229 weighting certain drivers over others (Warren and Seifert, 2010). Hence, following Warren et al.
230 (2013), we reduced the number of predictors in the Maxent model for species with low numbers

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231 of occurrences. For those species with < 40 unique data points, a set of six climate predictors was
232 used (i.e. P1, P4, P12, P15, Io and Iod2), whilst for taxa with > 40 unique data points, the
233 complete set of 10 predictors (i.e. P1, P4, P5, P6, P12, P15, P16, P17, Io and Iod2) was used.
234 This choice was a compromise between having overly-complex Maxent models for species with
235 low numbers of occurrences and having overly-simplistic models for species with very large
236 numbers of occurrences.
237
238 Maxent models were fitted using cross-validation (10 iterations), each one randomly dropping
239 10-20% input points. We then assessed the model skill using the Area under the ROC (Receiver
240 Operating Characteristic) Curve of the test data (AUC_{Test}), calculated as the average AUC_{Test} of
241 the 10 runs. Despite known limitations (Lobo et al., 2008; Warren et al., 2013), AUC_{Test} is a
242 useful metric for selecting Maxent models of appropriate complexity (Warren and Seifert, 2010)
243 and is a widely used model accuracy and selection criterion (Braunisch et al., 2013; Graham et
244 al., 2008; VanDerWal et al., 2009). The procedure applied here allowed us to discard species
245 with models showing low predictive skill: only models with 10-fold average test $AUC_{Test} \geq 0.7$
246 were projected onto the future climatic periods.
247
248 We then projected the fitted models onto both the continuous WorldClim current climate
249 surfaces and the downscaled surfaces of future climate conditions (2020s and 2050s). We then
250 binned the probability distributions using the 'prevalence threshold' (Liu et al., 2005; 2013). This
251 threshold is defined as the average probability over all input data points used to fit the model (i.e.
252 training presence points). To reduce commission (i.e. straying too far from the actual niche of a
253 taxon) or omission (i.e. missing major species populations due to lack of observations), the

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254 current climate distributions of each species were further clipped within a 300 km buffer around
255 the respective input occurrence points (also see Warren et al. 2013).
256
257 For future climatic scenarios, species distribution maps were first binned using the prevalence
258 threshold, and then further limited using two assumptions about species' dispersion mechanisms
259 (Jarvis et al., 2008; Thomas et al., 2004; Thuiller et al., 2005): (1) no dispersal and (2) unlimited
260 dispersal. For the no dispersal scenario, the projected future distributions were not allowed to
261 stray away from the current-climate distribution. For the unlimited dispersal scenario, all future
262 suitable areas outside the current-climate distribution were considered of the future distribution.
263 This implies that a species can migrate and occupy any new site that becomes suitable under
264 future climatic conditions. We acknowledge that unlimited dispersal is unrealistic (particularly
265 for plants), but we use this scenario to illustrate the likely impacts of climate change on diversity
266 even when the best possible conditions are assumed (e.g. through use of assisted migration, also
267 see Sect. 5.3).
268
269 3.4 Assessment of climate change impacts in species assemblages
270 Species richness was calculated using the binned species distributions as the total number of
271 species in a given site (i.e. pixel) and then used to calculate changes in species richness as the
272 difference between future species richness and current species richness divided by current
273 species richness. Additionally, we calculated the species turnover for the unlimited dispersal
274 scenario (Broennimann et al., 2006). This index arises from a modification of the 'classical'
275 species turnover (beta-diversity) indicators (Lennon et al., 2001; Whittaker, 1960) which are

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276 computed in geographic space using a defined spatial neighbourhood (Broennimann et al., 2006)
277 (Eq. 1).
278
279 $$\text{species turnover} = 100 * \frac{\text{species gain} + \text{species loss}}{\text{initial species richness} + \text{species gain}}$$ [Equation 1]
280
281 This turnover index has a lower limit of zero when the 'species gain' and the 'species loss' are
282 zero (both of which are very unlikely to happen with a large set of species), and an upper limit of
283 100, when the whole set of species changes from one time period to the other (i.e. either the
284 species gain or loss equals the initial species richness and there is no loss or gain respectively).
285
286 3.5 Assessment of individual species responses to climate change
287 To estimate the sensitivity to climate change at the species level for both migration scenarios and
288 periods, we intersected the current and future climatic niches and calculated the *climatic niche*
289 *persistence*. This is defined as the percentage of area that remains suitable in relation to the total
290 area in the current climatic niche (Loehle and LeBlanc, 1996; Peterson et al., 2001). *Climatic*
291 *niche loss* and *gain* were first calculated as the percentage area predicted to become unsuitable or
292 suitable respectively in the future climatic niche in relation to the total area in the current
293 climatic niche (Broennimann et al., 2006). The *species range change* was then calculated as the
294 difference between *climatic niche gain* and *loss*. This represents the percentage of range
295 expansion or contraction in relation to the current climatic niche for each species under the future
296 scenarios.
297
298 **4. Results**

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299 4.1 Species datasets

300 Our final modelling dataset comprised 478,301 vascular plant occurrences for 9,457 species and
301 88,636 bird occurrences for 1,555 species (Figure 1B). The W2013 dataset provided the greatest
302 proportion of occurrences, with 93% of all locality points used, and holding data for 9,371
303 vascular plants species and 1,429 birds. The database from CDC-UNALM provided 4.14% of the
304 occurrence points used for 186 vascular plant and 1,316 bird species. CONDESAN's dataset
305 contributed 2.9% of the occurrences representing 501 birds and 237 vascular plants. Despite the
306 majority of records were from the W2013, the CDC-UNALM and CONDESAN datasets
307 provided critical occurrence data for rare, endemic and narrow-range species that were poorly (if
308 at all) represented in the W2013 database (see e.g. Supplementary Figure S1 in Warren et al.
309 2013).

310

311 **[Figure 1 here]**

312

313 4.2 Performance of species distribution models

314 Almost half of the plant (48%) and bird species (44%) had an average test AUC > 0.9,
315 suggesting a good aptitude of the models to discriminate the species' fundamental climatic niche.
316 The average test AUC of all plant species was 0.874 (median = 0.894, SD = 0.088), while that of
317 bird species was 0.872 (median = 0.889, SD = 0.076) (Figure 2). Cross-validated runs indicated
318 that variability of AUC ranged from 0 to 13.7% for training-sets and from 0 to 38.8% for
319 evaluation sets. Relatively unstable test statistics were found for species with very low number of
320 data points (high variability in AUC across repetitions), both in training and test sets.

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[Figure 2 here]

Maxent models performance as measured by the average AUC was relatively similar for birds (BD) and vascular plants (VP), on average (Figure 2). Average training VP AUC ranged from 0.433 to 0.999, whilst test AUC varied from 0.28 to 0.999. In a few cases (< 500 for plants and < 50 for birds) the AUC statistic fell below the 0.7 threshold for model quality, probably owing to a combination of a limited number of species records and an asymmetric spatial distribution (i.e. high spatial autocorrelation). Less than 1 % of the whole set of plant and bird species had an AUC value equal to or worse than random discrimination of presences and absences (AUC ≤ 0.5). All species with average test AUC below 0.7 were removed from any further analyses (see Sect. 3.3.1). Based on a sufficiently high AUC (i.e. > 0.7), a total of 9,062 vascular plant and 1,456 bird species (95.7 and 96.6% respectively) were used in all following analyses.

4.3 Shifts in species richness and community turnover

Current species richness ranged from 0 to 452 species for birds and from 0 to 1,535 species for vascular plants per pixel of 25 km^2 (Figure 3). The highest concentration of plants is located on the outer slopes of the Western and Eastern Andean chain, between 1,500 to 3,000 m in altitude, primarily in the Andes of Colombia, Ecuador and Venezuela as well as on the inner slopes of the Central Chain of Colombia (upper Magdalena river basin) (Figure 3A). Diversity of birds is particularly high throughout the Peruvian Andes, in the montane forests along the Eastern ridge (Range = 141-452), and in the montane forests of the north-western chain of Ecuador (Figure 3B).

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[Figure 3 here]

Patterns of changes in species richness show important differences depending on the dispersal thresholds and the period analysed (2020 or 2050). The unlimited dispersal scenario projects an upslope migration of both plant and bird species suggesting important changes in the configuration of the diversity patterns of Andean biota. On the other hand, the no-dispersal scenarios show a significant reduction in species richness for both plant and bird species with major changes by 2050. The maximum richness values in the no dispersal scenario by 2050 period are 1,244 for plant species (mean = 163 ± 178) and 295 for birds (mean = 29 ± 36) per 25 km^2 pixel (Figure 4). Areas showing the largest decreases in species richness are located along the montane forests of the Eastern Andes of Bolivia and Peru between 500 and 1,200 m, on the outer slopes of the Eastern Andean foothills in Colombia and Ecuador, and on the Pacific slope of Northern Ecuador and southern Colombia (Figure 4). Conversely, the areas with minor changes are the highlands of Peru and Bolivia (Altiplano) and the pacific slope of the Peruvian Andes.

Negative changes in species richness are also observed even when unlimited dispersal is considered. Loss of diversity is observed from north to south of the Andes, although some particular areas are worthy of more attention; areas below altitudes of 1,500 m in the east Peruvian Andean mountains (i.e. central and eastern Huanuco, Pasco and Junin) seem to be severely impacted (>60% loss in species richness), and the same pattern is observed in the border between Ecuador and Peru, and in Nariño, Valle del Cauca, and Putumayo in Colombia. These

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367 changes may be attributed to the eastern margins of the mountain chain being less climatically
368 suitable in warmer climates.
369
370 **[Figure 4 here]**
371
372 The projected changes in community turnover are concentrated to a large extent in the High
373 Andes of Bolivia and Peru, as well as in the foothills of the Sierra de la Macarena, Sierra Nevada
374 de Santa Marta and around the Magdalena river basin in Colombia. Significant shifts are also
375 evident in the Venezuelan Andes along the Merida chain (Figure 5).
376
377 **[Figure 5 here]**
378
379 4.4 Individual species responses
380 Increases are projected in average climatic niche size for all species under the unlimited dispersal
381 assumptions for the 2020s period (Figure 6A). As expected, more severe impacts are projected
382 for the 2050s, and this is reflected in a less pronounced increase of range size in the unlimited
383 dispersal scenario and a stronger decrease in the non-dispersal scenario (Figure 6A, B).
384 Considering an unlimited dispersal scenario, the rates of climatic niche expansion seem to be
385 high, with most of the species being highly favoured or barely affected by climate change if
386 migration in fact occurs and other non-abiotic factors remain stable (e.g. land-use patterns, pests
387 and diseases), particularly for birds. Some 45% (n=655) of bird and 41% (n=3,715) of vascular
388 plant species modelled are likely to experience an increase in their climatic niches of 100% or
389 more by 2050s (Figure 6A). By contrast, only a limited proportion of species (< 10 %) is

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390 expected to experience no increase or a net loss in their climatic niche size. Our estimates
391 indicate that even assuming unlimited dispersal some species are expected to undergo range
392 contraction (even to the extent of extinction), thus highlighting specific sensitivities to climate
393 change.
394
395 **[Figure 6 here]**
396
397 In a no dispersal scenario, the differences between periods become more evident (Figure 6B).
398 Whilst by 2020s the maximum changes in range size are reductions of 50% and 80% for birds
399 and vascular plants, respectively, by the 2050s, species within both groups are projected to
400 experience 100% range reduction, indicating likely extinctions for a vast number of species.
401
402 To illustrate species-specific responses under future climate, we further selected and analysed
403 two contrasting genera for each species group (plants and birds). These genera were selected
404 because they are of relatively recent origin (during the Pleistocene, ca. 1 to 3 million years ago),
405 include species that are endemic to the Andes, and are classified vulnerable or critically
406 endangered by IUCN (Table 3 and 4). Many of the species of the genera *Grallaria* and
407 *Eriocnemis* (class: *Aves*) are projected to expand their niche by more than 100 % if dispersal was
408 assumed. In particular, the species *E. cupreoventris* and *E. nigrivestis* were found to increase
409 their niche considerably by 2020 and 2050. In the case of no-dispersal, however, these species
410 depict range contractions of 69 and 65 % (respectively) by 2050. Similar responses were found
411 for most species of the genus *Grallaria*, notably *G. alleni*, *G. aplotona*, *G. gigantea*, and *G.*

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412 *hypoleuca*, for which range contractions of 59, 83, 54, and 63 % are projected by 2050s (no
413 dispersal), respectively (Table 3).
414
415 Similar responses are reported for the plant genera *Polylepis* and *Gynoxis*. Species such as *P.*
416 *lanuginosa* and *P. tomentela* showed significant increases in range size in both future scenarios
417 (unlimited migration), but rather large decreases in range size under no-migration assumptions.
418 By contrast, some species of these genera (e.g. *P. incana*, *P. reticulate*, *G. buxifolia*, and *G.*
419 *caracensis*) report range contractions for both dispersal scenarios and periods (Table 4). These
420 species that respond negatively even under when unlimited dispersal is allowed can be
421 considered of very high sensitivity, and perhaps also be prioritised for further research to
422 understand such sensitivities.
423
424 **5. Discussion**
425 **5.1 Changes in species distribution patterns**
426 Our results suggest that impacts of climate change over the Andean biota could be extremely
427 severe. This finding is in agreement with previous studies for the Andean region (Feeley and
428 Silman, 2010; Feeley et al., 2011ab; Tovar et al., 2013), other tropical areas (Hole et al., 2009;
429 Miles et al., 2004; Still et al., 1999), or globally (Warren et al., 2013). The effects of climate
430 change on the Tropical Andes can be synthesized at two different levels: the extent of the whole
431 Tropical Andes (regional level), and at the species level. At the regional level, the inner and
432 outer Andean foothills (800 – 1,500 meters) are likely to be the most affected due to a high
433 amount of species loss. In addition, the spatial patterns of species turnover demonstrate a
434 bimodal response. First, an upslope shift of several species from mid elevations to the high

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435 Andes is expected. Second, a large west and southward displacement of species from the upper
436 areas of the northern portion of the study area (i.e. Merida, Perijá and Santa Marta) towards
437 lower latitudes and a significant climatic niche reduction of mountain-top endemics is also
438 projected.
439
440 The areas that would be most affected by high absolute species turnover rates and the subsequent
441 change in the composition of communities are the montane dry forest, the Santa Marta massif,
442 the Mérida ridge, the inner slopes of the Central and Eastern ridges of the Colombian Andes and
443 the Altiplano of Peru and Bolivia (> 3,800 meters).
444
445 At the species level, the biophysical impacts of exposure to climate change are projected to be
446 highly variable. In this study, the two contrasting dispersal scenarios show extremes of a
447 spectrum of projected responses by species to climate change. For plants, it is likely that the true
448 response lies nearer the no-dispersion scenario (see also Feeley et al., 2011a), whereas for birds
449 the response may in some cases resemble that of the full-dispersion scenario. Overall, we report
450 that plant species may be more negatively affected in both magnitude and direction of range
451 change impacts than birds in both periods. The same pattern holds for both migration scenarios,
452 probably due to a greater proportion of endemic and narrow-range plant species and/or the
453 presence of isolated (meta) populations (Figure 6A) (also see Ramirez-Villegas et al. 2012), and
454 perhaps to some extent also due to incompleteness of samples for some species. Yet species
455 interactions might have a prominent role in this point. For example, species interactions can slow
456 climate tracking and produce more extinctions than predicted by climatic niche models only
457 (Urban et al. 2013); or on the contrary, broad-ranging animals might transport seeds enabling

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458 long-distance dispersal, as documented before during the last de-glaciation period, in which trees
459 dispersed at rates of 100-1000 m year^{-1} (Clark, 1998).
460
461 The projected alteration of the spatial distribution patterns of Andean assemblages (Feeley and
462 Silman, 2010; Feeley et al., 2011a; Jetz et al., 2007) suggest the appearance of novel
463 communities adapted to non-analogous climatic conditions, which could affect the functioning of
464 Andean ecosystems (Williams and Jackson, 2007). Many shrubby and epiphyte species (e.g.
465 *Solanaceae*, *Bromeliaceae*) depend on their specialized symbiotic interactions with animals for
466 seed dispersion and pollination. Climate change effects on these organisms could cause spatial,
467 temporal, or physiological asynchronies between mutualistic species, producing changes in
468 community composition and structure (Zavaleta et al., 2003).
469
470 Our estimates are thus useful in gauging general trends and possible impacts, although it is very
471 likely that individual responses at the species or community level will be determined by species'
472 ecological traits (i.e. dispersal capacity), species interactions (i.e. competition) and/or by their
473 physiological response to stresses, leading (in some cases) to different outcomes. If species are
474 sufficiently mobile they may be able to track the geographic displacement of their climatic
475 niches, or if species are capable of rapid evolutionary change or have a wide range of abiotic
476 tolerances, they may adjust to changing ecological conditions and landscapes (Broennimann et
477 al., 2006). According to Travis (2003) and Opdam and Wascher (2004), the exact nature of a
478 species' response to different rates of climate change depends upon colonization ability and how
479 much of a generalist the species is. For species with lower colonization ability and for specialist
480 species, the threshold occurs at a lower climate change signal. In a human dominated world,

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481 however, natural or semi-natural ecosystems are embedded in tracts of unsuitable landscape, and
482 populations of species restricted to those habitat types are spatially dissected. By consequence,
483 what is ascribed as a shifting species range is in fact the complex result of extinction of (meta)
484 populations at the warm range limit (that surpasses thresholds of species adaptability), and
485 colonization and growth of (meta) populations into regions that newly came within the cold
486 range limit (that enters the range of species adaptability). Hence, for understanding the potential
487 risks of climate change to a species, we must consider the dynamics of the populations
488 constituting the geographical range in connection to the spatial features of the landscapes across
489 the range (also see Sect. 5.3). Human land-use may be especially important in the Andes where
490 anthropogenic activities above tree line and in the piedmont may create a hard barrier to upward
491 migrations, imperilling Andean biodiversity (Feeley et al. 2010; 2011a); therefore, the
492 incorporation of a coupled model that integrates climate change scenarios together with land
493 cover change dynamics is a priority task to analyse specific responses of the Andean biota to
494 these drivers of change.
495
496 **5.2 Species extinction risks**
497 Climatic fluctuations during the Pliocene-Pleistocene period strongly influenced the origin and
498 spatial arrangement of the majority of Andean species used in this study (Luteyn, 2002; Young et
499 al., 2002; Garcia-Moreno et al. 1999). During periods of intense climatic change in the
500 Pleistocene, epiphyte-laden evergreen vegetation remained only where conditions remained
501 stable, suggesting that ecologically stable areas may have existed during the glaciations as small
502 pockets within surrounding drier pieces of montane forest (Fjeldså, 1995; Roy et al., 1997;
503 Arctander and Fjeldså, 1997). As a consequence, many of these surviving species present in

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504 these ecosystems are endemic, with narrow habitat tolerances in conjunction with a restricted
505 distribution range (Kattan et al., 2004). These patterns and conditions constitute a perfect
506 scenario to promote higher rates of species loss and turnover under projected climate anomalies
507 such as those projected in the present study.
508
509 In this context, reductions in the size of the climatic niche such as those herein projected imply
510 that a number of species may become restricted to a few sites. Species with small range sizes are
511 vulnerable to smaller stochastic events as these could affect a larger proportion of the species'
512 total population, especially in fragmented landscapes (With and King, 1999). As a result of this,
513 extinction risks will likely intensify for a large portion of the taxa analysed here, particularly at
514 long lead times (2050s in this study). Our study, as many others, assumes that species will die
515 out within regions that are predicted to become climatically unsuitable for them (Ohlemüller et
516 al., 2006), and takes no account of species- or population-level adaptive responses that may
517 reduce negative effects (see e.g. Harte et al., 2004). Despite that, our results may be conservative
518 given that we (1) did not include habitat loss data for the Tropical Andes in the analysis (Leisher
519 et al., 2013; Ramirez-Villegas et al., 2012), (2) did not consider potential impacts of changing
520 interannual variability (e.g. frequency or intensity of drought or heat waves) in our models, and
521 (3) did not model any secondary effects such as pests, diseases or important species-level
522 interactions required for survival. Furthermore, the rather low generation times of many vascular
523 plants and some bird species will probably preclude adaptation rates from keeping pace with
524 human induced climate change.
525

526 **5.3 Management and conservation implications**

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In conservation planning, irreplaceability (commonly measured as singularity) and vulnerability (measured through threat processes) are among the most important dimensions to analyse (Brooks et al., 2006). Several authors have depicted the Tropical Andes as being within the most vulnerable regions with high irreplaceability (Brooks et al., 2006; Kattan et al., 2004; Mittermeier et al., 1997), placing the region extremely important for conservation action.

The question of whether the current protected area system is sufficient given the challenges of climate change is a critical one. A regional analysis by Ramirez-Villegas et al. (2012) showed that 8 out of 16 conservation areas in South America are in the Andean highlands. According to the present study, negatively impacted areas (orange to red areas in Figure 4) could lose up to 60% of species richness and suffer up to 100% changes in community makeup, thus, affecting ecosystem functioning as well as ecosystem services to human society (Gamfeldt et al., 2008). There is no question that these projected impacts will affect conservation planning during the 21st century, and hence further research should focus on developing a better understanding of conservation effectiveness under future climates for the Andes (Araujo et al., 2004). Tropical mountain systems such as the Andes are highly variable in climate, and therefore, offer a wide range of adaptation pathways for species, further increasing their value for conservation. The herein projected changes in range sizes, species richness and community composition are useful metrics in evaluating tools for conservation, such as for adjusting extinction risk assessments, delimitation of priority conservation areas and conservation targets within protected areas.

Using these results to identify priority areas at a medium to large scale could be particularly useful, given that diversity cannot always be easily captured in a single site-specific targeting of

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conservation in the Andes, requiring instead, conservation actions spread throughout entire biomes (Fjeldså et al., 2005; Ramirez-Villegas et al., 2012). In this context, based on Opdam and Wascher (2004) we propose three major components for a conservation strategy in a warmer Tropical Andes. Firstly, a focus on landscape conditions for biodiversity, where populations potentially can respond to large-scale changes and disturbances. These conditions should allow populations to respond to large-scale disturbances. If species distributions patterns change more dynamically in space and time, local conservation management for single species will be less effective. Secondly, we propose to shift in strategy from protected areas towards landscape networks including protected areas, connecting zones and intermediate landscapes. Thirdly, we propose a shift from a defensive conservation strategy towards a landscape development strategy. A static approach of establishing isolated reserves surrounded by a highly unnatural landscape is not an effective strategy under a climate change scenario. Given the intense land use changes in the Andes, the sensitivity of Andean species to climatic changes, and the fact we are globally already committed to at least +2 °C warming, we must accept that conservation of biodiversity is only effective if we dynamically integrate it in the development of the entire landscape, based on coalitions with other functions such as the identification of key areas for provision of ecosystem services, heterogeneity, and landscape permeability (Brooks et al., 2006).

Regional policy and planning should aim at improving landscape connectivity. Amongst the most evident conservation planning strategies is the establishment of reserves. Particularly under climate change, the inclusion of new areas seems to be a relevant, albeit challenging, task (Hannah et al., 2007). Land tenure issues, poverty, development gaps between rural and urban areas, the demand for natural resources, and an economic model oriented toward extraction (e.g.

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573 mining) make the establishment of new conservation areas difficult in the Andes. In the absence
574 of such possibilities, the appropriate articulation of national reserves with other conservation
575 sub-systems such as protective forests, indigenous territories, civil society reserves, and sub-
576 national protected areas could be an appropriate mechanism of action. In addition, significant
577 attention should be paid to the design (or adjustment) of the Andean protected area system. We
578 recommend the following criteria be taken into account:

579 • Maintain the connectivity across the elevation, moisture and edaphic gradient (Killeen
580 and Solórzano, 2008). These gradients are critical for maintaining beta diversity and
581 response capacity (Thuiller et al., 2008).

582 • Incorporate ecotone diversity in the design of conservation areas. The landscapes within
583 these areas are characterized by habitat mosaics that reflect differences in soil humidity,
584 productivity, among others. These mosaics are occupied by species assembled in
585 communities that reflect the presence of micro-environmental constraints in an area
586 where climate stress is the overriding macro-environmental characteristic. These
587 populations may have genetic traits distinct from core populations pre-adapting them to
588 the physiological stress of climate change (Killen and Solórzano 20008). In the Tropical
589 Andes the preservation of the ecotone between the montane forest and grasslands
590 ecoystems is a fundamental adaptation measure to buffer the massive upward
591 displacement of species ranges in response to increased warming (Feeley et al. 2011b).

592 • The identification of climatically stable areas as potential biological refugia through
593 bioclimatic envelope model (see e.g. dark green areas in Figure 4 combined with dark
594 areas in Figure 3) which could act as connectors and/or corridors between current and
595 future areas of high biodiversity (Vos et al., 2008).

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596

597 Improvement of landscape connectivity through the creation of biological corridors is probably
598 the most frequent recommendation in the scientific literature (Heller and Zavaleta, 2009). We
599 suggest an optimisation of spatial configuration of such corridors and an assessment of the risks
600 of these turning into channels for disease transmission and/or movement of invasive species. In
601 addition to these, a better land use planning through better and targeted government-level
602 policies is warranted in order to reduce the risks of deforestation, loss of pollination services and
603 genetic erosion in the agricultural frontier, while at the same time bolstering the dispersion and
604 population breeding between (and within) remaining habitat patches (Opdam and Wascher,
605 2004).

606

607 **5.4 Final remarks**

608 Several sources of uncertainty may influence the results we provide here. These include the
609 primary biodiversity data, the climate data and the climate envelope modeling (Braunisch et al.,
610 2013; Pearson et al., 2006; Ramirez-Villegas and Challinor, 2012). Although these uncertainties
611 are carried into the analysis, we argue that our results provide important insight on a globally
612 important biodiversity hotspot. Importantly, our results agree and partly complement with
613 previous regional and global studies (see Warren et al. 2013; Still et al., 1999; Thomas et al.,
614 2004; Feeley and Silman, 2010). Improvement to our modeling approach for future studies may
615 be warranted through achieving better spatial representativeness of both species and climate
616 observations, the use of abundance data (in addition to presence-only data), better constraining
617 species migration patterns, the inclusion of changes interannual variability and their effects on
618 species distributions, the use of higher resolution climate models that resolve local climatic

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619 change patterns in a more detailed manner, as well as a detailed assessment of relevant local
620 processes driving extinctions.
621
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634
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976 **Table 1** List of bioclimatic variables used in the modeling

ID	Variable name	Units
P1	Annual mean temperature	°C
P4	Temperature seasonality (standard deviation)	°C
P5	Maximum temperature of warmest month	°C
P6	Minimum temperature of coldest month	°C
P12	Annual precipitation	mm
P15	Precipitation seasonality (coefficient of variation)	%
P16	Precipitation of Wettest quarter	mm
P17	Precipitation of Driest quarter	mm
Io	Ombrothermic index	mm $°C^{-1}$
Iod2	Ombrothermic index of the driest 2-months of the driest quarter	mm $°C^{-1}$

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978 **Table 2** List of all and available GCMs and principal characteristics (resolutions)

Model	Country	Atmosphere**	Ocean**	A2*
BCCR-BCM2.0	Norway	T63, L31	1.5x0.5, L35	A
CCCMA-CGCM3.1 (T47)	Canada	T47 (3.75x3.75), L31	1.85x1.85, L29	
CCCMA-CGCM3.1 (T63)	Canada	T63 (2.8x2.8), L31	1.4x0.94, L29	
CNRM-CM3	France	T63 (2.8x2.8), L45	1.875x(0.5-2), L31	
CSIRO-Mk3.0	Australia	T63, L18	1.875x0.84, L31	A
CSIRO-Mk3.5	Australia	T63, L18	1.875x0.84, L31	A
GFDL-CM2.0	USA	2.5x2.0, L24	1.0x(1/3-1), L50	A
GFDL-CM2.1	USA	2.5x2.0, L24	1.0x(1/3-1), L50	A
GISS-AOM	USA	4x3, L12	4x3, L16	
GISS-MODEL-EH	USA	5x4, L20	5x4, L13	
GISS-MODEL-ER	USA	5x4, L20	5x4, L13	
IAP-FGOALS1.0-G	China	2.8x2.8, L26	1x1, L16	
INGV-ECHAM4	Italy	T42, L19	2x(0.5-2), L31	
INM-CM3.0	Russia	5x4, L21	2.5x2, L33	A
IPSL-CM4	France	2.5x3.75, L19	2x(1-2), L30	
MIROC3.2-HIRES	Japan	T106, L56	0.28x0.19, L47	
MIROC3.2-MEDRES	Japan	T42, L20	1.4x(0.5-1.4), L43	A
MIUB-ECHO-G	Germany/Korea	T30, L19	T42, L20	
MPI-ECHAM5	Germany	T63, L32	1x1, L41	
MRI-CGCM2.3.2A	Japan	T42, L30	2.5x(0.5-2.0)	
NCAR-CCSM3.0	USA	T85L26, 1.4x1.4	1x(0.27-1), L40	A
NCAR-PCM1	USA	T42 (2.8x2.8), L18	1x(0.27-1), L40	A
UKMO-HADCM3	UK	3.75x2.5, L19	1.25x1.25, L20	
UKMO-HADGEM1	UK	1.875x1.25, L38	1.25x1.25, L20	

979 *A: Monthly maximum and minimum temperature available **Horizontal (T) resolution indicates number of cells
980 in which the globe was divided. Vertical (L) resolution indicates the number of layers in which the atmosphere was
981 divided. When a model is developed with different latitudinal and longitudinal resolutions, the respective cellsizes
982 (LonxLat) in degrees are provided instead of a unique value.

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987 **Table 3** Change in distributional range for the Andean bird genera *Eriocnemis* and *Grallaria*.

Species	IUCN 2010 category[1]	Endemic to Andes[2]	Elevation range (m)[3]	Range change (%)[3]			
				2020		2050	
				Full	Null	Full	Null
Eriocnemis alinae	**LC**	**-**	**2300-2800**	**-16.8**	**-23.3**	**-32.6**	**-37.0**
Eriocnemis cupreoventris	NT	-	1950-3000	149.4	-44.8	101.2	-68.6
Eriocnemis derbyi	**NT**	**-**	**2500-3600**	**-31.3**	**-45.3**	**18.0**	**-48.3**
Eriocnemis luciani	**LC**	**-**	**2800-3800**	**41.8**	**-13.6**	**-9.4**	**-30.3**
Eriocnemis mosquera	**LC**	**-**	**1200-3600**	**-17.8**	**-20.8**	**-34.0**	**-37.9**
Eriocnemis nigrivestis	CR	EC	1700-3500	261.4	-30.0	92.1	-65.0
Eriocnemis vestita	**LC**	**-**	**2800-3500**	**8.4**	**-29.7**	**-1.7**	**-52.0**
Grallaria alleni	VU B1a+b(i,ii,iii)	-	1800-2500	46.7	-31.5	3.1	-59.1
Grallaria erythroleuca	**LC**	**PE**	**2150-3000**	**38.8**	**-21.6**	**-12.5**	**-46.9**
Grallaria flavotincta	**LC**		**1300-2350**	**50.9**	**-16.7**	**-8.4**	**-47.6**
Grallaria gigantea	VU B1a+b(i,ii,iii)	-	1200-2600	> 500	-26.1	> 500	-54.0
Grallaria guatimalensis	LC	-	200-3000	10.0	-31.3	2.4	-50.8
Grallaria haplonota	**LC**	**-**	**700-2000**	**11.0**	**-55.1**	**-18.9**	**-82.7**
Grallaria hypoleuca	LC	-	1400-2300	170.3	-12.7	71.1	-63.0
Grallaria nuchalis	LC	-	1900-3150	73.1	-10.3	25.9	-36.4
Grallaria quitensis	**LC**	**-**	**2200-4500**	**-8.4**	**-38.2**	**-48.5**	**-66.6**
Grallaria ruficapilla	LC	-	1200-3600	28.6	-15.5	18.1	-35.0
Grallaria rufocinerea	VU B1a+b(i,ii,iii)	-	2200-3150	11.1	-31.1	60.7	-42.2
Grallaria rufula	LC	-	2300-3650	30.8	-25.9	10.4	-52.9
Grallaria squamigera	**LC**	**-**	**2000-3800**	**5.6**	**-26.0**	**-21.8**	**-50.7**
Grallaria watkinsi	**LC**	**-**	**600-1700**	**43.6**	**-20.8**	**33.7**	**-49.9**

988 [1] Status of the species according to the IUCN red list of threatened species: LC: least concern, NT: near-
989 threatened, VU: vulnerable, EN: endangered, CR: critically endangered. Additional criteria as in
990 http://www.iucnredlist.org/static/categories_criteria_3_1
991 [2] Country where endemic, if endemic to the Andes. EC: Ecuador, PE: Peru, BO: Bolivia
992 [3] Range change under different periods and for two dispersal scenarios. Full: unlimited dispersal, Null: no
993 dispersal
994 Species in **bold** depict range contractions (either by 2020 or 2050) regardless of migration assumptions.

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Table 4 Change in distributional range for the Andean plant genera *Gynoxis* and *Polylepis*.

Species	IUCN 2010 category	Endemic to Andes	Elevation range (m)	Range Change (%)[3]			
				2020		2050	
				Full	Null	Full	Null
Gynoxis acostae	LC	EC	2700-4300	> 500	-36.8	> 500	-84.0
Gynoxis asterotricha	n/a	-	3100-4100	> 500	-21.0	> 500	-65.5
Gynoxis baccharoides	VU D(ii)	-	3300-4200	233.3	-41.4	109.6	-69.2
Gynoxis buxifolia	**n/a**	**-**	**2500-4100**	**-12.8**	**-21.9**	**-52.1**	**-56.9**
Gynoxis caracensis	**LC**	**PE**	**2800-4335**	**-13.3**	**-69.0**	**-39.6**	**-81.3**
Gynoxis cuicochensis	NT	EC	2500-4050	90.9	-21.7	53.8	-39.3
Gynoxis fuliginosa	**n/a**	**-**	**2700-4150**	**-7.3**	**-26.7**	**-35.1**	**-52.6**
Gynoxis hallii	LC	EC	2500-4100	266.4	-17.7	198.6	-39.6
Gynoxis miniphylla	NT	EC	3100-4000	223.8	-36.6	44.6	-64.4
Gynoxis oleifolia	**LC**	**PE**	**3380-4900**	**-58.8**	**-81.6**	**-90.1**	**-94.5**
Gynoxis parvifolia	n/a	-	2900-4100	> 500	-22.1	> 500	-42.5
Gynoxis psilophylla	n/a	BO	2800-3900	> 500	-7.6	> 500	-14.6
Gynoxis reinaldii	n/a	-	2400-3300	165.2	-44.9	226.1	-64.5
Gynoxis sodiroi	VU B1ab(iii)	EC	2900-4286	55.5	-15.8	21.4	-37.6
Polylepis incana	**no**	**-**	**2450-3800**	**-39.1**	**-64.8**	**-55.8**	**-83.3**
Polylepis lanuginosa	VU B1abIII	EC	2600-3630	> 500	-26.1	> 500	-49.1
Polylepis pauta	**no**	**-**	**2700-4200**	**8.3**	**-59.7**	**-61.1**	**-87.5**
Polylepis reticulata	**VU A4c**	**EC**	**3200-4450**	**-28.9**	**-52.3**	**-31.3**	**-81.3**
Polylepis sericea	**no**	**-**	**2500-3900**	**-39.1**	**-63.6**	**-52.6**	**-83.8**
Polylepis besseri	no	-	2500-4100	12.8	-24.5	8.4	-32.4
Polylepis racemosa	no	-	2900-4500	23.8	-16.4	30.2	-31.5
Polylepis tomentella	no	-	2800-4700	71.9	-7.2	59.0	-16.2
Polylepis weberbaueri	**no**	**-**	**2700-4800**	**-38.0**	**-60.3**	**-46.7**	**-73.0**

[1] Status of the species according to the IUCN red list of threatened species: LC: least concern, NT: near-threatened, VU: vulnerable, EN: endangered, CR: critically endangered. Additional criteria as in http://www.iucnredlist.org/static/categories_criteria_3_1

[2] Country where endemic, if endemic to the Andes. EC: Ecuador, PE: Peru, BO: Bolivia

[3] Range change under different periods and for two dispersal scenarios. Full: unlimited dispersal, Null: no dispersal

Species in **bold** depict range contractions (either by 2020 or 2050) regardless of migration assumptions.

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1007 **FIGURE CAPTIONS**
1008

1009 **Figure 1** Study area. A. Elevation (in meters) across the tropical Andes countries overlaid with locations
1010 of weather stations in WorldClim; B. Number of modelling occurrences in 0.5 degree cells and key sites
1011 with high projected impacts (mentioned throughout the text).

1012 **Figure 2** Evaluation of Maxent models. Distribution of the Area under the ROC Curve (AUC) for A. All
1013 vascular plants; B. All birds. Training AUC values are plotted for training (grey bars) and test (black bars)
1014 sets. AUC values of individual species are averages of 10 cross-validated runs with 10-20% of the input
1015 points drawn randomly.

1016 **Figure 3** Modeled current species richness for A. Vascular plants and B. birds in the Tropical Andes as
1017 derived by the sum of binned species distributions models. Values are counts of species occurring in a 25
1018 km^2 pixel.

1019 **Figure 4** Spatial patterns of changes in species richness for birds and vascular plants under both migration
1020 scenarios and time periods. Values are percentage change in species richness from the present-day value
1021 shown in Figure 3.

1022 **Figure 5** Species turnover for birds and vascular plants, for both periods. Community turnover can only
1023 be calculated for scenarios that somehow assume migration as this calculation requires that species can
1024 move to more suitable environments whenever possible. Values are percentages of change in community
1025 turnover as calculated by Eq. 1 (see Sect. 3.4 for details).

1026 **Figure 6** Climate change impacts on individual species. Change in range size for birds (white bars) and
1027 vascular plants (grey bars) for A. Unlimited dispersal and B. No dispersal, for the SRES-A2 emission
1028 scenario and both periods (2020s and 2050s) (outliers have been removed from the plot for easier
1029 visualization). Box plots were constructed with n=1,456 and n=9,062 for birds and vascular plants,
1030 respectively.

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Figure 1

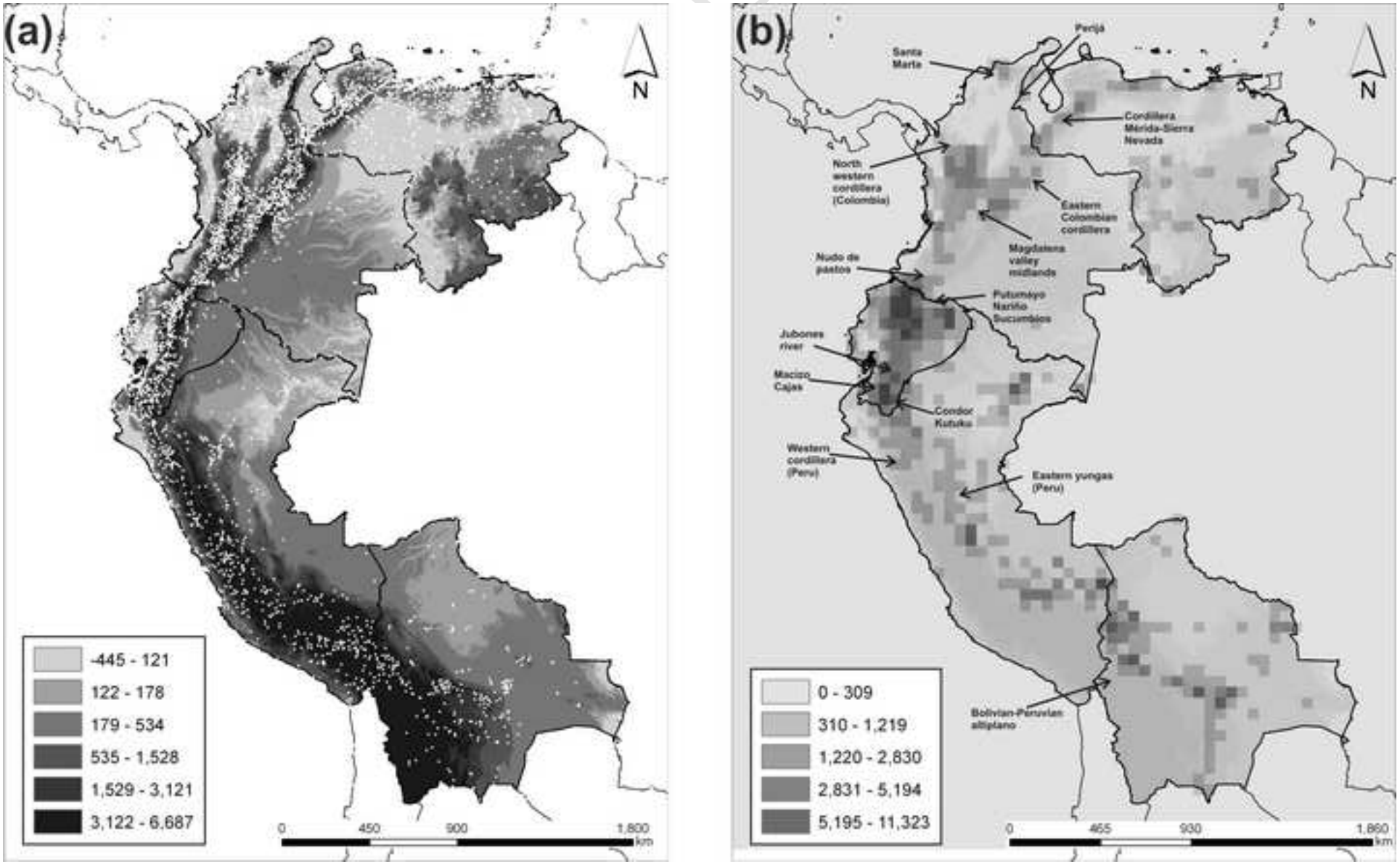

(a)
N
-445 - 121
122 - 178
179 - 534
535 - 1,528
1,529 - 3,121
3,122 - 6,687
0
450
900
1,800
km
(b)
N
Perijá
Santa Marta
Cordillera Mérida-Sierra Nevada
North western cordillera (Colombia)
Eastern Colombian cordillera
Magdalena valley midlands
Nudo de pastos
Putumayo Nariño Sucumbíos
Jubones river
Macizo Cajas
Condor Kutuku
Western cordillera (Peru)
Eastern yungas (Peru)
Bolivian-Peruvian altiplano
0 - 309
310 - 1,219
1,220 - 2,830
2,831 - 5,194
5,195 - 11,323
0
465
930
1,860
km

Figure 2

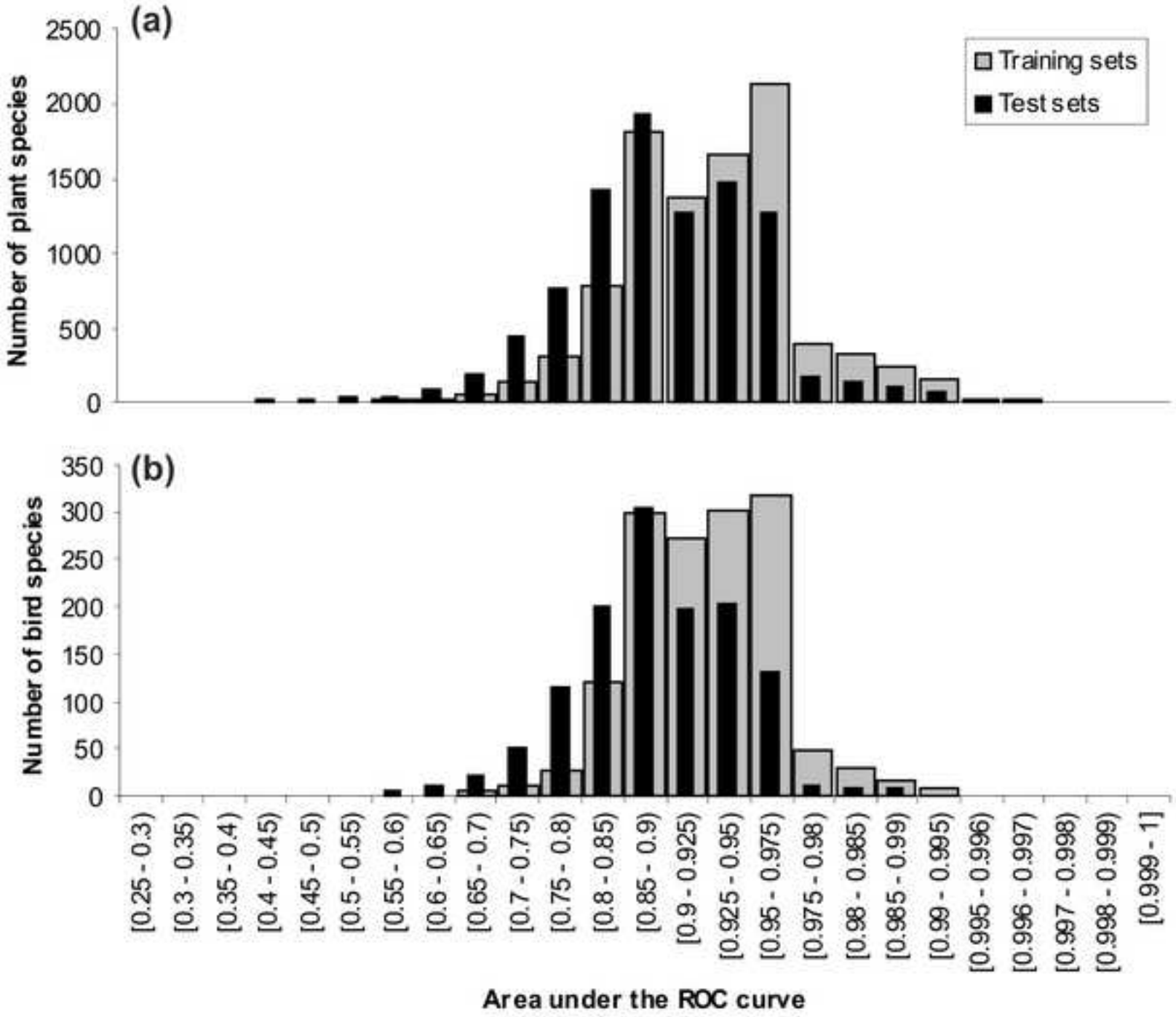
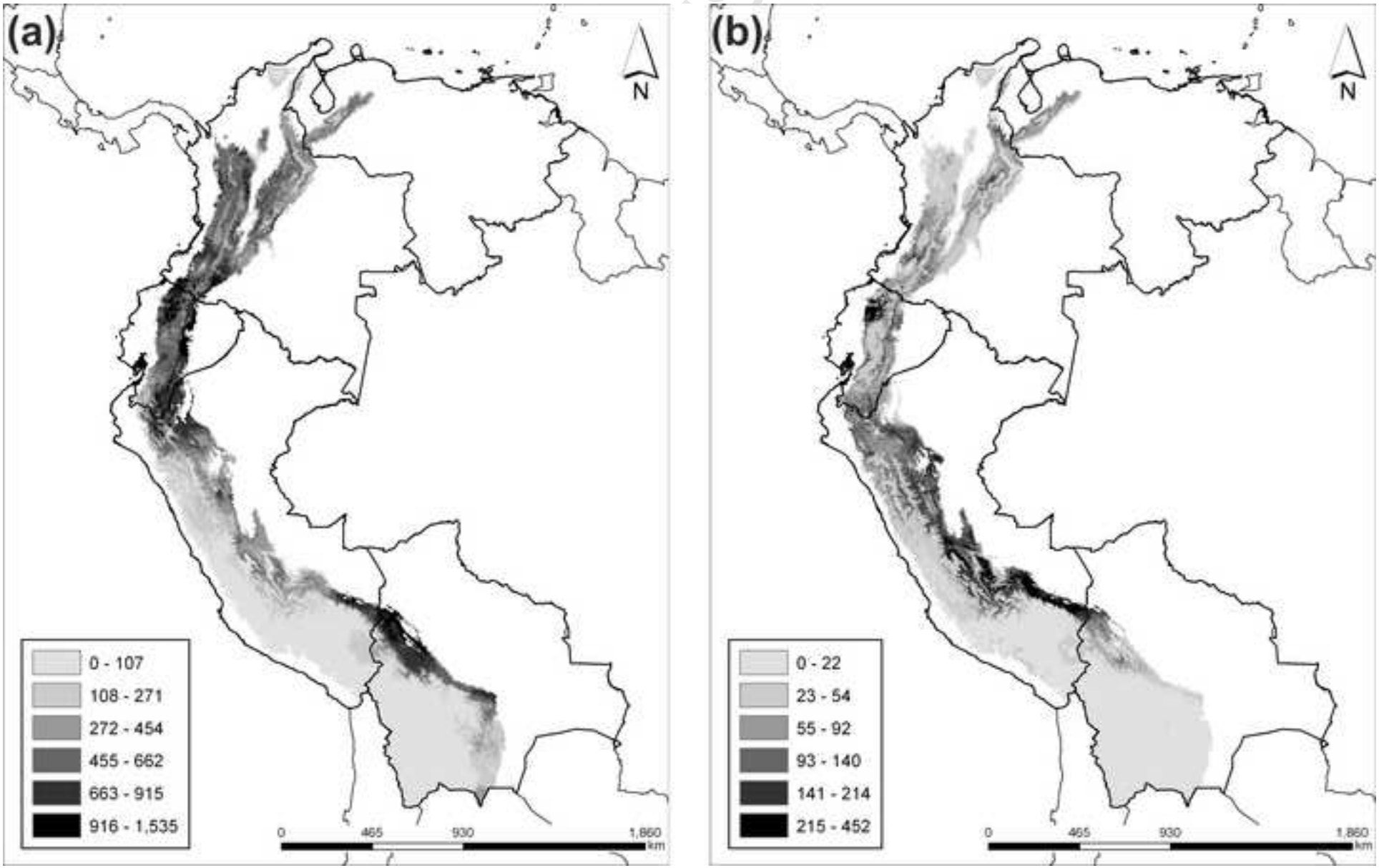

(a)
Number of plant species
2500
2000
1500
1000
500
0
Training sets
Test sets
(b)
Number of bird species
350
300
250
200
150
100
50
0
[0.25 - 0.3)
[0.3 - 0.35)
[0.35 - 0.4)
[0.4 - 0.45)
[0.45 - 0.5)
[0.5 - 0.55)
[0.55 - 0.6)
[0.6 - 0.65)
[0.65 - 0.7)
[0.7 - 0.75)
[0.75 - 0.8)
[0.8 - 0.85)
[0.85 - 0.9)
[0.9 - 0.925)
[0.925 - 0.95)
[0.95 - 0.975)
[0.975 - 0.98)
[0.98 - 0.985)
[0.985 - 0.99)
[0.99 - 0.995)
[0.995 - 0.996)
[0.996 - 0.997)
[0.997 - 0.998)
[0.998 - 0.999)
[0.999 - 1]
Area under the ROC curve

Figure 3


(a)
N
0 - 107
108 - 271
272 - 454
455 - 662
663 - 915
916 - 1,535
0
465
930
1,860
km
(b)
N
0 - 22
23 - 54
55 - 92
93 - 140
141 - 214
215 - 452
0
465
930
1,860
km

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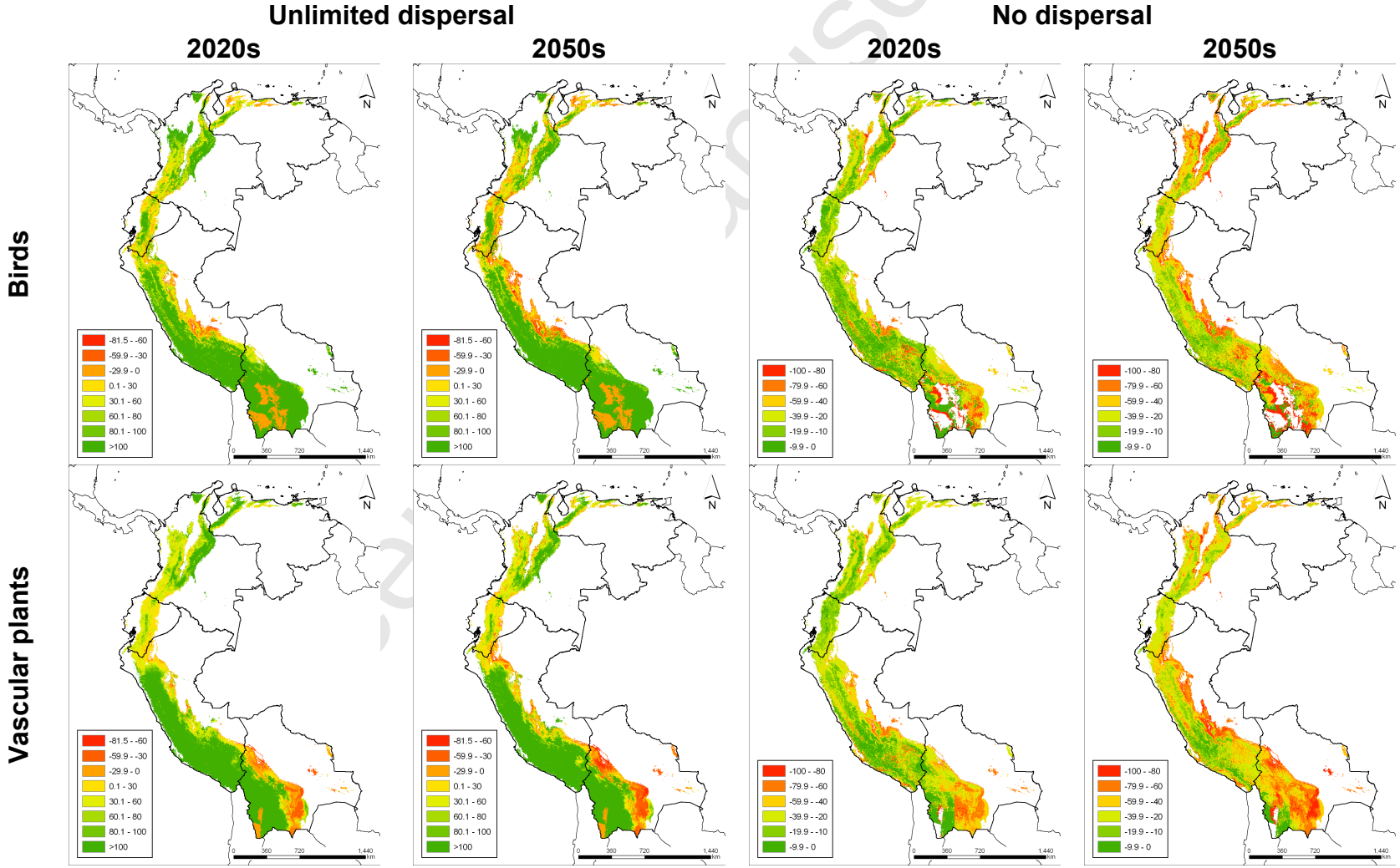

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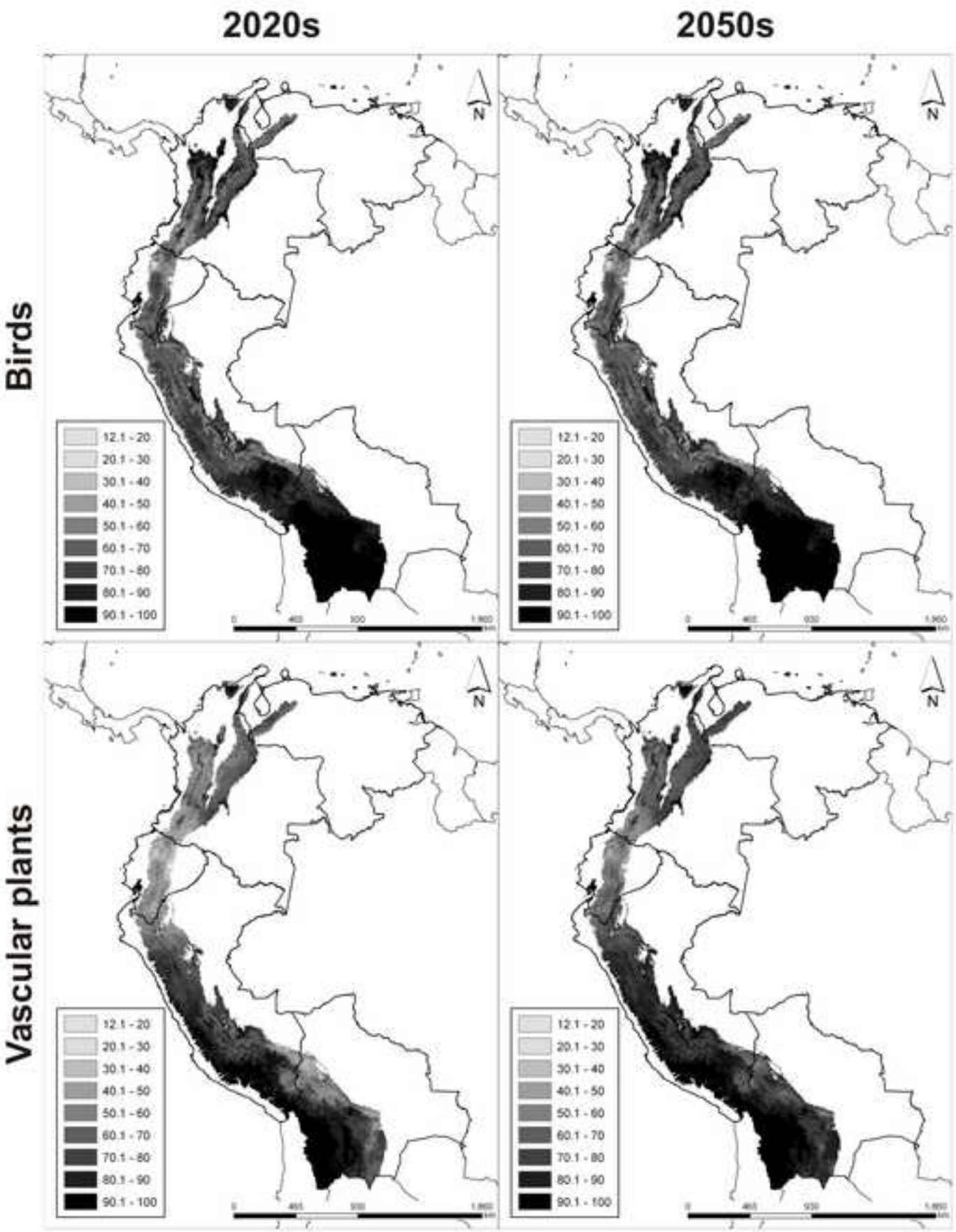
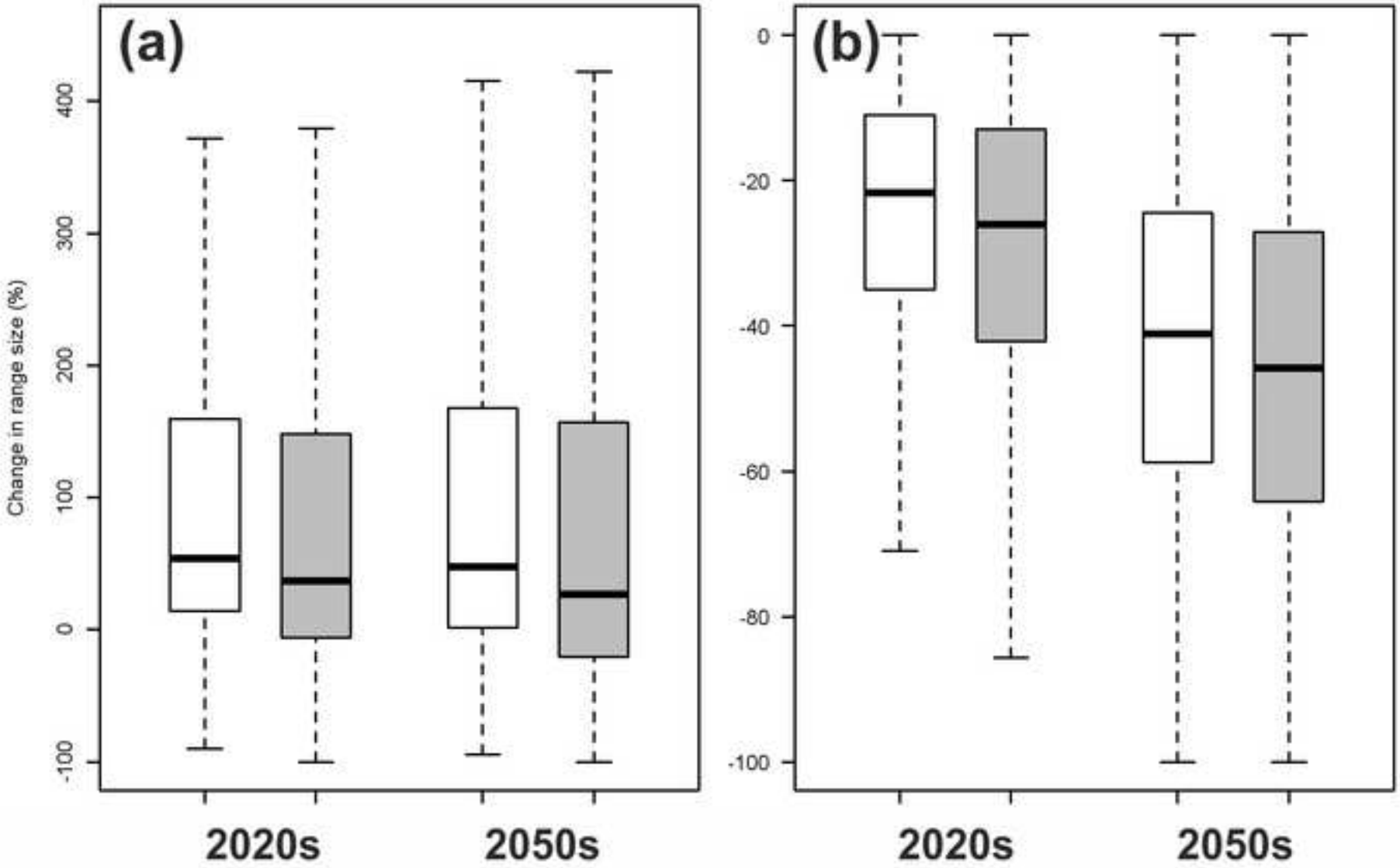


Figure 6


(a)
(b)
Change in range size (%)
400
300
200
100
0
-100
-20
-40
-60
-80
2020s
2050s
2020s
2050s