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Amphibian phylogenetic diversity in the face of future climate change: not so good news for the chilean biodiversity hotspot

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Abstract

Climate change is projected to be the most extensive human-induced disturbance to occur on natural ecosystems, inducing changes in different biodiversity features including the evolutionary history of a region through the decline and loss of its phylogenetic diversity. Amphibians, given their ectothermic life cycle and critical conservation status, would potentially be exposed to extinction processes under conditions of climate change, with the corresponding loss of evolutionary history in regions of high biodiversity. This research addresses the effects of climate change on the evolutionary history of amphibians in the Chilean Biodiversity Hotspot, by estimating the PD (Phylogenetic diversity) and PE (Phylogenetic endemism) of 27 species. Using different RCP (RCP 4.5 and 8.5) and time frames (years 2050 and 2070), we create species distribution models (SDM) to evaluate the species range dynamics and the phylodiversity in the Hotspot. Also, given that Protected Areas (PA) are the main global strategy to ensure the conservation of species and their features, we evaluate the capacity of PA to conserve the evolutionary history in the Hotspot. Our results show a set of modeled species that will become extinct, or will experiment changes in their distributional ranges, inducing a clear decline of amphibian evolutionary history for the next 30 to 50 years, and a worrying low capacity of the PA to contain current and future PD and PE. Given the critical amphibian scenario, our results highlight the need for further research to improve the decision-making process in the hotspot area addressing the potential amphibian extinction risk, the lack of protection by the PA system, and the loss of evolutionary history as a key aspect of biodiversity.

Keywords Climate change · Species distribution models · Global change

Introduction

Climate change is projected to be the most extensive human-induced disturbance to occur on natural ecosystems (Chapin et al. 2000, Sala et al. 2000, Pereira et al. 2010, Beaumont et al. 2011, Li et al. 2018, Nolan et al. 2018, Weiskopf et al. 2020). The impact of climate change on biodiversity has been widespread and has involved several types of responses Parmesan 2006; Chown et al. 2010, Hoffman & Sgro 2018). In particular, several studies have shown shifts in phenologies and geographic ranges of a large number of taxa during the last 30 to 50 years (e.g. Walther et al. 2002, Parmesan and Yohe 2003, Root et al. 2003, Nussey et al. 2005, Pörtner and Knust 2007, Charmantier et al. 2008, Chen et al. 2009, Radchuk et al. 2019, Pecl et al. 2017). A large body of literature has been accumulating on forecasting future consequences of warming, particularly regarding changes in distributional ranges (e.g. Pearson and Dawson 2003, Thomas et al. 2004, Thuiller et al. 2005, Araujo & New 2007, Keith et al. 2008, Brook et al. 2009, Pereira et al. 2010, Sinervo et al. 2010, Beaumont et al. 2011, Dyderski et al. 2018, Nunez et al. 2019, Sirois-Delisle and Kerr 2018). In this context, a better understanding of how species respond to climate change is crucial to assess their potential vulnerability and avoid further biodiversity loss (Williams et al. 2008; Dawson et al. 2011; Moritz and Agudo 2013).

It is expected that climate change will induce changes in different biodiversity features, including the evolutionary history of a region through the decline and loss of its phylogenetic diversity (hereafter PD; Faith 1992, Young et al. 2016, Carvalho et al. 2017). The PD measures a fundamental dimension of biodiversity beyond species richness, namely the amount of evolutionary history of a particular system. Therefore, PD represents the accumulation of evolutionary adaptations and also the evolutionary potential in a community assemblage (Faith 1992, Forest 2007, Asmyhr et al. 2014). Through the identification of areas that represent young and old clades, or highly clustered and overdispersed communities, PD provides key information about the history of diversification that may have modeled contemporary assemblages of species (Fritz and Rahbek 2012; Tucker and Cadotte 2013; Rodrigues et al. 2005; Winter et al. 2013). The PD framework (i.e. includes several metrics *sensu* Tucker et al. 2017) englobes the idea of *"option value"*, which means that the preservation of PD maximizes the possibility of having the right features available in an uncertain future (Forest et al. 2007). Thus, major changes generated by the action of anthropogenic divers, such as climate change, could induce the loss of variants with potential to thrive under novel environmental conditions (Barker 2002), and therefore, alter species interactions that eventually might change the structure and functioning of ecosystems (Devictor et al. 2010, Gonzalez-Orozco et al. 2016). Similarly, for an adequate characterization of PD, it has been highlighted the importance of knowing the degree of the spatial restriction of phylogenetic branches in an area relative to the remaining areas (Rosauer et al. 2009; Rosauer and Jetz 2015). This approach defines what is known as Phylogenetic Endemism (hereafter PE), a useful metric to identify areas that hold relatively unique phylodiversity, and would support the identification of areas for PD conservation (Gonzalez -Orozco 2016, Pollock et al. 2017). One of the greatest impacts caused by climate change occurs on ectothermic vertebrates, given their dependence on environmental temperature to regulate their body temperature (Deutsch et al. 2008; Kearney and Porter 2009) and their reduced capacities to track their climatic niches compared to endotherms (Aragón et al. 2010). With half of the species threatened by extinction, amphibians are a key symbol of the biodiversity loss

crisis (González-del-Pliego et al. 2019). Global amphibian population declines arise as a consequence of the action of multiple interacting stressors, including anthropogenic land use change (Sala et al. 2000; Hoffmann et al. 2010; Newbold et al. 2015) climate change (Pounds 2001; Hof et al. 2011), water pollution (Beebee and Griffiths 2005), overexploitation and trade (Lips et al. 2005; Mendelson et al. 2006), increased UV radiation (Kiesecker et al. 2001; Blaustein et al. 2003), invasive species (Nunes et al. 2019) and emerging infectious diseases such as the chytrid fungus (Bacigalupe et al. 2017, 2019; Scheele et al. 2019). Although the overall potential impact of climate change on amphibian PD remains obscure (Loyola et al. 2014), the evidence suggests that decline, loss and phylogenetic homogenization should be expected (Menendez-Guerrero et al. 2020, Nowakowski et al. 2018). These changes in PD are mostly related to the rearrangement of the assembly of species and the spatial change of PD to high latitudes and elevations (Thuiller et al. 2011).

Under climate change conditions, Protected Areas (hereafter PAs) the classic global strategy for biodiversity conservation, would be unlikely to meet the conservation needs of multiple species Araujo et al. 2004, 2011; Possingham et al. 2006; Hannah et al. 2007, Kujala et al. 2013). Indeed, given the expected change in species assemblages, climate change might increase the spatial mismatch between species distributions and already established PAs with the consequent loss of diversity (Araujo et al. 2011). Thus, although PAs are an essential tool for preserving evolutionary history (Frishkoff et al. 2014), in the case of amphibians, it remains unclear how much their evolutionary history will be retained by PAs through the future dynamics of PD.

Here, we assess how projected climate change will affect amphibian PD and PE in the Chilean winter rainfall Valdivian forest biodiversity hotspot, one of the 35 key terrestrial areas in the globe for biodiversity conservation (Myers 2011, Marchese 2015), encompassing the area of highest amphibian richness in the country (Vidal and Díaz-Páez 2012). One of the most critical conditions of the Chilean biodiversity hotspot is its lack of protection, where approximately less than 10% of the region is under formal PA and the protection of the amphibian tree of life by PAs has been poorly addressed (Jofré and Mendez 2011). Thus, in this critical context, we incorporate the evaluation of the expected future spatial configuration of PD and PE and measuring how the Chilean Protected Area System (SNASPE) in the hotspot would have the conditions to sustain the potential future change in the PD configuration.

Methods

Study area

The Chilean biodiversity hotspot extends for ca. 3,000 km (25ºS – 47ºS) and includes several ecosystems, vegetational formations and climates (Myers 2011). At the same time, embraces almost 75% of Chile's gross domestic product and about 80% of the human population (Barbosa and Villagra 2015). We identified 40 species in the hotspot (IUCN 2020), representing approximately 2/3 of the total species in the country.

Occurrence records and climate data

Species records were obtained for 27 of the 40 species currently distributed in the hotspot. Occurrences were obtained from an extensive literature search carried out in ISI Web of Knowledge and Google Scholar and through the collaboration with amphibian experts. To reduce the effect of biased species occurrence, we utilized the SpThin Package in R (Aiello-Lammens et al. 2015), selecting 1 km as a minimal distance between records. Nineteen bioclimatic layers obtained from the WorldClim database (Hijmans et al. 2004) were used as predictors in the statistical modelling. All layers were cropped to our study area and resampled to a 1-km resolution in ArcGIS 10.3. Considering the wide climatic extension of the *hotspot*, and based on the known distribution of the species, we selected specific bioclimatic variables, which represent ecological conditions suitable for the species. We evaluated the collinearity between those retained bioclimatic variables with a Pearson correlation, selecting variables with relations smaller than 0.7. In this way, none of the modeled species used all of the 19 bioclimatic variables. Two future climate change scenarios were selected: Representative Concentration Pathways (RCP) 4.5, an optimistic scenario where emissions peak around 2040, and RCP 8.5, a pessimistic scenario of high emissions. Three future global circulation models (GCM) were used to obtain consensus of the climatic predictions for 2050 and 2070: MIROC (Model for Interdisciplinary Research on Climate, Japan), HadGEM2-ES (Met Office Hadley Centre, UK) and CCSM (NCAR-UCAR, USA). The SDM was carried out using Maxent (Phillips & Dudik, 2008). Maxent is a machine-learning algorithm that minimizes the relative entropy of the probability densities calculated from the presence records versus those calculated from randomly sampling the study region or background. Background is the representation of a potentially accessible area or one likely to be explored by the species (Peterson et al. 2011), and is used to contrast the information of presence points, allowing training of the model (Merow et al. 2016). Models were evaluated using the area under the curve (AUC) of the receiver operating characteristic (ROC). The AUC measures the probability of correctly classifying the background presence points, with values ranging from 0.5 (explained by chance) to 1 (perfect discrimination between points of presence and background; Phillips et al. 2006, Riquelme et al. 2018).

Our selected models considered an AUC value of 0.7 as a measure of model accuracy (Fielding and Bell 1997). For every model, a 10-fold cross-validation scheme was used to validate the modelling results. Using this cross-validation scheme, the dataset was divided into 10 subsets. Then, the model was fit using nine of the subsets, and the remaining subset (independent) was used to test (validate) the fit. This procedure was repeated 10 times, and the AUC and jackknife values reported represent the average of the 10 tests. The probabilistic SDM were converted into a binary scale (absence=0, presence=1) utilizing the threshold "Maximum Training Sensitivity plus Specificity", which has proven to generally produce more accurate results than other thresholds, reducing errors of omission and commission in the model classification (Liu et al. 2005, Jimenez-Valverde & Lobo 2007, Fajardo et al. 2014, Guisan et al. 2017). The resulting presence/absence models were transformed to raster and subsequently stacked to calculate phylogenetic diversity metrics.

Spatial analysis and PD metrics

For the resulting SDM of each species for each time frame and RCP, we calculated its PD and PE using the most recent published amphibian phylogenetic tree (Jetz and Pyron 2018) and the "*Picante*" package version 1.7 (Kembel et al. 2010) implemented in R (R Core Team 2018). The PD was calculated as the sum of the length of the branches (Lc) in the phylogenetic tree on the spanning path linking a set of taxa to the root of the tree, as a proportion of the total length of the tree. For calculation purposes, we defined C as the set of branches in the minimum spanning path joining the taxa to the root of the tree, and c is a branch (a single segment between two nodes) in the spanning path C (Eq. 1).

(1)

$$PD = \sum_{c \in C} Lc$$

For PE, we utilized Eq. 2, which corresponds to a relative measure of endemism with the contributions of each PD unit (defined as Lc) through areas where it occurs (defined as Rc), aiming to identify spatially restricted concentrations of PD (Cadotte & Davies 2010).

(2)

$$PE = \sum_{\{c \in C\}} \frac{Lc}{Rc}$$

For each species and climate change scenario, range shifts and directions were calculated using the centroid of the core distributional area, using the '*gCentroid*' function in the R package '*rgeos*' (v. 0.3–26; Bivand et al. 2014). We measured the shift as the linear distance and direction between the species centroids considering the present distribution and the predicted distribution in each time period and RCP scenario. The analysis of species range distribution (expansion and/or contraction) was performed in ArcGIS 10.3 (ESRI 2015). We evaluated the differences in ranges through the stacked distribution of each time period. Finally, with the extension "*spatial analyst*", for each emission scenario we evaluated whether there was a gain (PD>0), maintenance (PD=0) or loss (PD<0) of amphibian PD between time periods (current time – 2050 and between 2050–2070). Finally, we used the layers of the Chilean Protected Area System (Congress National Library 2019) to evaluate the amount of amphibian PD retained by pixel in each time period and RCP scenarios and then express it as percentage of PD in protected area across time.
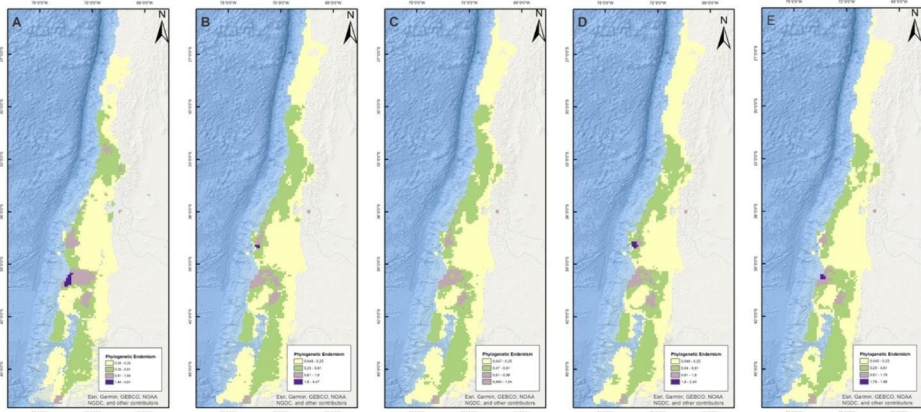
Fig. 1 Phylogenetic endemism from current to 2050 and 2070 for amphibians from the Chilean biodiversity hotspot under climate change scenarios RCP4.5 and 8.5. A: current time; B year 2050 RCP 4,5; C: - year 2070 RCP 4,5 D: year 2050 RCP 8,5 and E: year 2070 RCP 8,5

Results

Species distributional changes

Overall, the 19 bioclimatic variables used can be grouped into variables corresponding to "temperature" (Bio1 to Bio11) and "precipitation" (Bio12 to Bio19; Hijmans et al. 2005). Our results show that temperature variables were the most significant predictors for 19 species, while precipitation variables were important in the remaining 8 species (see Supplementary Online Material, SOM, for details).

Irrespectively of the temporal horizon and greenhouse gas emissions scenario, all species will change the range of their current distribution. In scenario RCP 4.5, at the end of the evaluated period, 6 species will increase, in average, their distributional range, whilst 19 are expected to decrease it in different magnitudes (Table 1, SOM). Under the more climatic adverse scenario (RCP 8.5), at the end of the evaluated period, 15 species are expected to increase their ranges, whilst 8 are expected to shrink their distribution (Table 2, SOM). The cases of decrease in distribution include 2 species that are predicted to disappear by 2070 under scenario RCP 4.5 (*Alsodes norae, Insuetophrynus acarpicus*) and 4 species for scenario RCP 8.5 (*Alsodes barrioi, A. valdiviensis*, *A. norae, Insuetophrynus acarpicus*). In general, regardless of the year and RCP scenarios, more than 70% of the species shifted their ranges towards southern directions (SE-S-SW), while around 20% are predicted to shift their ranges in a N-NW direction (Fig. 1, SOM).

Changes in phylogenetic diversity

Our results show an increment of PD in those new areas where expansions occur (i.e. number of pixels) occupied by amphibians species in each scenario and a decrease on the total PD in the Chilean biodiversity hotspot (Fig. 2 SOM). This is mainly associated to the predicted rearrangement of amphibians assemblages and the expected extinction of species under climate change scenarios. For both climate change scenarios (RCP 4.5 and RCP 8.5)

Table 1 PD configuration for comparative time periods and each climate change scenario (RCP 4.5 and 8.5) compared value of PD pixel by pixel for every climatic condition)

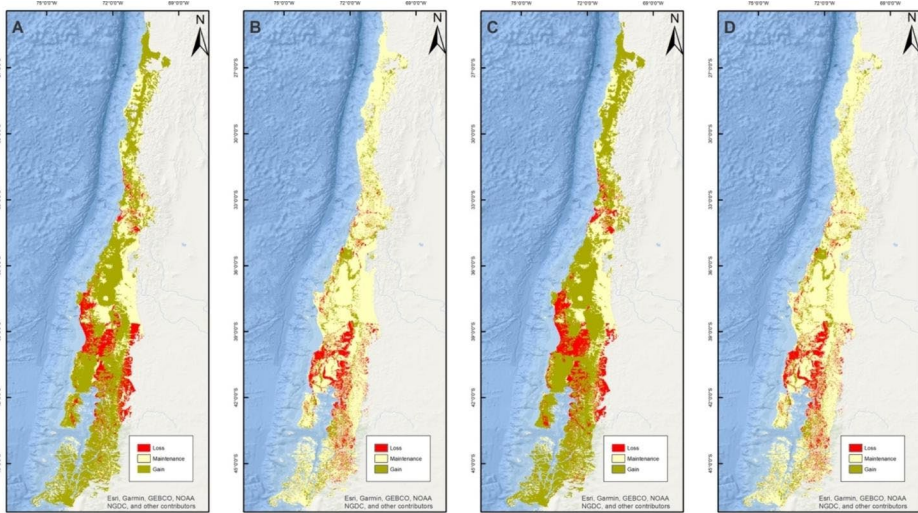| Time period	Proportion of PD gain	Proportion of PD maintenance	Proportion of PD loss
Current – year 2050 RCP 4.5	61,3%	26,91%	11,72%
Year 2050–2070 RCP 4.5	12,95%	76,4%	10,6%
Current – year 2050 RCP 8.5	63,73%	23,38%	12,87%
Year 2050–2070 RCP 8.5	24,15%	63,33%	12,5%

Fig. 2 Phylogenetic diversity gain, maintenance and loss in each cell for all climatic scenarios. A: current time - year 2050 RCP4,5; B: year 2050- year 2070 RCP 4,5; C: current time - year 2050 RCP 8,5; D: year 2050 - year 2070 RCP 8,5. The calculations were made pixel by pixel,

we compared changes in PD at a pixel by pixel scale, to understand how changes in the distribution of amphibians would influence changes in PD (Table 1). Although we have detected a progressive decrease of PD into the future, the pixel dynamics in the hotspot show that initially almost the 60% of the available pixels increase their PD levels between the current time and the year 2050 for both climate change scenarios (Fig. 2; Table 1). For the next evaluated time period (2050 to 2070), the spatial configuration of PD is dominated by a high number of cells that maintain PD values trough time (Fig. 2; Table 1). Regarding amphibian PE, the results show a sustained decline for each climate change scenario. In the RCP 4.5 scenario the maximum PE values tend to remain until the year 2050 to abruptly decline in the year 2070. While, for the RCP 8.5, the decline begins in the year 2050, to continue declining towards the year 2070, although in a less pronounced way (Fig. 1).

Protected Areas and amphibian phylodiversity

Given the rearrangement of species distribution and the range expansion of a set of species (Tables 1 and 2, SOM), a consistent increment of PD in the area occupied inside PAs is observed, regardless of the climate projection scenario. These occupied new areas also rep-

resent new PD values inside PA. When we evaluated the characteristic of the PD contained in the PAs, we found that for each evaluated period, it is possible to find pixels that contain the maximum PD values (Fig. 3). In order to assess if high PD values represent in turn a measure of the effectiveness of PAs in maintaining phylogenetic diversity, we evaluated these PD values comparing the performance of the highest quartile in each time period and climate change scenario. Our results show a decrease in the top values of PD contained in the PAs for future scenarios, and most relevant, the amount of the highest PD values contained by PAs just represent a small fraction of the highest PD value compared to the entire biodiversity hotspot (see Table 4, SOM). Our results also show that, in the case of PE, the current system of PAs harbor, on average, low levels of PE compared to the maximum possible, estimated for each period of time (Table 5, SOM). In addition, as result of the future action of climate change, PAs would not be able to sustain high PE areas, independent of the RCP.

Discussion

Climate change will induce major changes in species distributions (Thomas et al. 2004, Hijmans & Graham 2006) and will promote several changes in the ecology and evolution of species (Thuiller et al. 2011). In this context, a better understanding of how species might respond to this threat is crucial for assessing their vulnerability and guiding efforts to avoid potentially severe biodiversity loss (Williams et al. 2008; Dawson et al. 2011; Moritz and Agudo 2013). The results of our SDM show that: (i) some species will become extinct, some will contract and some will expand their distributional ranges, where most shifts in ranges will occur towards the south, (ii) there is a clear decline of amphibian evolutionary history in the hotspot for the next 30 to 50 years under different climate change scenarios, and (iii) Chilean PAs show a low capacity to contain current and future phylogenetic diversity.

Changes in Amphibian distributions

Our results show that species could increase or decrease their distributional range under climate change scenarios (Tables 1 and 2, SOM), and most of them show a tendency to migrate to southern directions in the hotspot (Fig. 1 SOM). Species with wider distribution ranges will be less affected by future warming, whereas species with smaller distributional ranges are projected to face extinction (Purvis et al. 2000; Thomas et al. 2004; Gaston and Fuller 2009; Urban 2015). In particular, our results indicate that there is a group of 12 species for which the tendency to decrease their distribution ranges is permanent, regardless of the climate change scenario (Tables 1 and 2, SOM). This group of species could be considered extremely vulnerable to climate change, considering that 5 of them are already categorized as at risk of extinction (EN) by the IUCN Red List and the remaining 7 species are currently categorized as Least Concern (LC, IUCN 2019) or Near Threatened (NT., *A. nodosus*). Thus, it is expected that the reduction in their distributional ranges could increase their risk of extinction in the future (Thuiller 2004). Conversely, a group of 6 modelled species shows a constant increment in their distributional ranges regardless of the climate change scenario, with 5 of them having some conservation risk status (Table 6 SOM). Overall, our results show an increment in the amphibian extinction risk in the region (mostly year 2050, with the
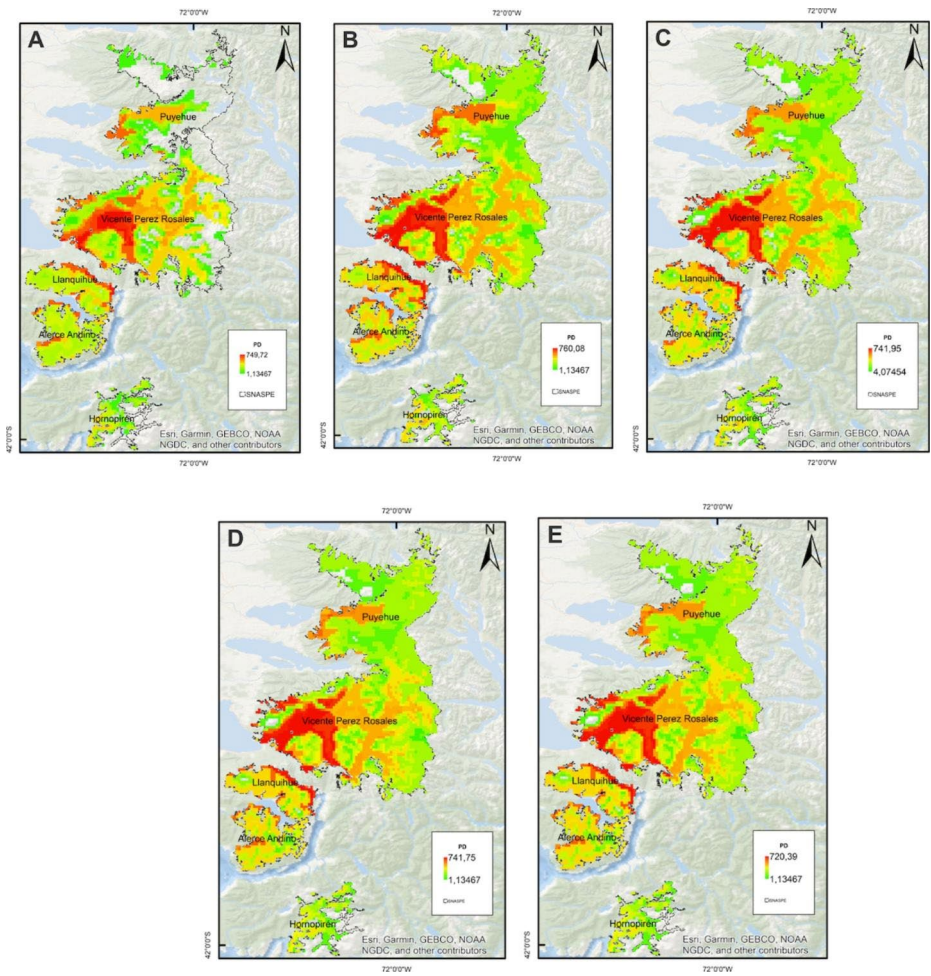
Fig. 3 PD dynamic inside PA, were (A) current time; (B) year 2050 RCP 4.5; (C) year 2070 RCP 4.5; (D) year 2050 RCP 8.5 and (E) year 2070 RCP 8.5. We utilized the PAs "Puyehue", "Vicente Perez Rosales", "Llanquihue" and "Hornopiren" as references. In the figure, through time, new areas of the PAs increase their PD levels

exception of A. barrioi). For the scenario RCP 4,5 there are 2 extinct species, whilst the scenario RCP 8.5 involves a greater number of projected extinct species (4 species, see SOM).

Species distribution models has been utilized as a tool to evaluate the impacts of climate change and their effect in species extinction risk (Malcolm et al. 2006, Lee and Jetz 2008, Warren et al. 2013, Foden et al. 2013). However, these kinds of projections have some limitations that are important to keep in consideration. First, there is still a lack of consistent global estimates of species extinctions attributable to climate change, and although wide percentages of extinctions have been linked to the action of this driver, there is not yet a clear proximal explanation Cahill et al. 2013; Urban 2015, but see Sinervo et al. 2010). Indeed, one of the main gaps of the use of correlative models and their projections based in climatic envelopes is their failure to account for important processes that influence extinc-

tion outcomes, such as interactions between species and habitat shifts, landscape structure, species demography and dispersal capacities (Akcakaya et al. 2006, Thuiller et al. 2008, Keith et al. 2008, Urban 2015). Moreover, our approximation does not consider important biological mechanisms, such as species interactions, plasticity, evolution, landscape dispersal barriers, and intraspecific trait variation (Buckley et al. 2010; Sinervo et al. 2010; Huey et al. 2012; Ruiz-Aravena et al. 2014). Finally, our approach does not consider the synergistic effect between climate change with other anthropic drivers, which influence the accelerated loss of biodiversity in areas under pressure for multiple global change drivers, as the current and future scenario of the Chilean hotspot (Northrup et al. 2019).

Changes in spatial configuration of PD and PE

Our results show that, the expected rearrangement of amphibian distributions (which implies variations both in the area and in the direction of geographic extension) together with changes in species richness, will influence the spatial configuration of PD and PE in the Chilean biodiversity hotspot (Figs. 2 and 1). The spatial reorganization of distribution ranges and changes in the number of species influence a reduction in the maximum values of amphibian PD and PE across time periods and RCP scenarios (Table 3 SOM). This decline in PD and PE may be associated with the extinction of species that our models have projected, considering its contribution of evolutionary history in the reference phylogenetic tree for the species in the area. For example, *I. acarpicus* is predicted to be extinct by 2070 in both climate change scenarios, while this particular species has, within our set of modeled species, the highest values of accumulated evolutionary history (Jetz and Pyron 2018). However, despite this general decline in accumulated PD through time frames and RCPs, we find specific areas of the hotspot that gain PD along time periods, as a result of species distribution rearrangements and overlapping, particularly as a consequence of those species that expand their range. It is also important to highlight, that these areas of PD gains correspond to new areas which did not previously have amphibians presences (see Figs. 2 and 1).

Additionally, our results show how the future configuration of PD changes in space and time. Whilst at present PD is highly concentrated in the south-central zone of the hotspot, in the future high PD values will tend to be distributed along the south and south-west directions (Fig. 1, SOM). Additionally, given these changes in PD, the PE will tend to decrease in terms of values and geographical extension in the hotspot, in both climate change scenarios and time periods (Fig. 1). As PE is a measure of rarity, the decline in endemism levels means that those areas that hold restricted range species and concentrate high proportion of PD relative to their range, will tend to disappear in the future due to climate change.

The evolutionary diversity of a system has an intrinsic conservation value (Mace et al. 2003; Winter et al. 2013; Frishkoff et al. 2014) and their loss inducted by anthropogenic climate change will impact the evolutionary history and future options for humanity in a region Faith 1992; Mace et al. 2003; Forest et al. 2007, Emerson & Gillespie 2008, Owen et al. 2019). The expected decline in PD highlights the critical importance of the concept of "option value" in the hotspot, considering that the loss of PD could jeopardize the possibility of having the right feature at hand in an uncertain future (Forest et al. 2007). The expected future change in PD would be critical, regarding the varied and significant roles of amphibians in ecosystems, from soil bioturbation and nutrient cycling to pest control and ecosystem engineering (Hocking and Babbitt 2014). Several studies suggest that the loss of

amphibians from stream ecosystems can alter primary production, algal community structure, food chains (from aquatic insects up to riparian predators), and reduce energy transfer among diverse ecosystems (Whiles et al. 2006; Hocking and Babbitt 2014; Meredith et al. 2016; Campos et al. 2017).

Phylogenetic diversity metrics and PAs

In terms of the current and future protection of PD, our results are a call for concern. Although there is a sustained increase in the area that species will occupy inside PAs as a result of the rearrangement in future distributions (Fig. 3), this effectively represents new areas with PD inside PAs, but conclusions about the effectiveness of them in maintaining adequate levels of PD should be considered with caution. These new values do not represent the highest values possible to obtain and in fact, through the comparison of the highest PD values in each time period and climate change scenario, we found that those areas that contained the highest PD values inside PAs, only represent a proportion of less than 5% of the total high PD pixels possible to find in the entire hotspot area (see Table 4, SOM). Unfortunately and certainly more worryingly, PE follows a similar pattern, with maximum values declining across time inside PAs, which implies the loss of unique genetic diversity (Rosauer and Jetz 2015, Gonzalez-Orozco et al. 2016). Thus, our results highlight that the greatest concentration of amphibian evolutionary history is currently located outside the Chilean formal PAs and, unless new PAs are created, this situation will continue across time (see Tables 4 and 5 SOM). This constitutes an adverse scenario, considering that the hotspot has already been exposed to intense land-use change during the last 50 years (Armesto et al. 2010). Also, in the face of climate change, this would exacerbate the possibility of loss of large amounts of both phylogenetic as well as functional traits (Redding and Mooers 2015, Collen et al. 2011, Mazel et al. 2018, Gumbs et al. 2018).

Our results represent a critical and much needed contribution in amphibian conservation, considering the Phylogenetic Diversity as a key aspect of biodiversity and the dynamics of species distributions. Particularly for decision-makers, the focus should be put in the expected species in potential extinction risk, species that will possible change to higher risk categories given their range contraction and the high amount of amphibian PD and PE without PAs protection for the present and future time independent of the climate change scenario.

Perspectives and limitations

Our results show the vulnerability of amphibians to climate change in a biodiversity hotspot and the loss of their evolutionary history in the face of climate warming. Although our predictions are based on Worldclim 1.0 bioclimatic data, this is an area of knowledge that is constantly improving, as is the appearance of new high-resolution climate databases (e.g. Chelsa, Worldclim 2.1), which certainly allow new research possibilities for species distribution models, climate change, and conservation. For example, there is evidence that SDMs based on Chelsa outperformed Worldclim -based models (Bobrowski et al. 2021). In addition, Worldclim 2.1 version incorporates the framework of Shared Socio-Economic Pathways (SSPs) derived within the Coupled Model Intercomparison Project Phase 6 (CMIP6) (Cerasoli et al. 2022), which is useful for increasingly updated climate projections.

Although our models were run at 30-arcsecond resolution (1×1 km at the equator), the perspective of using new climate databases under this fine-scale, or even finer resolution (100×100 m) (Poggio et al. 2018), would result in more accurate predictions and consequently, better conservation decisions. This is particularly true for those species whose predicted distributional ranges are so small that might put them to the limit of survival, or those whose predicted ranges results in their extinction, and whose loss of evolutionary history will be irreversible.

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