Title: Expected impacts of climate change on tree ferns distribution and diversity patterns in subtropical Atlantic Forest

Running title: Tree ferns distribution and diversity in future scenarios

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ABSTRACT

Tree ferns are common elements in subtropical Atlantic Forest, sometimes reaching more than half of dominance at forest sites. Climate change could impact the distribution and diversity of tree ferns, hence impacting the ecological processes provided by them. We investigate the impacts of future climate changes in the potential distribution of tree fern species, as well as in the α- and β-diversity. Our first hypothesis (H_1) is that warmer climate in the future will increase the distribution of Cyatheaceae species and decrease the distribution of Dicksoniaceae species. Due to a larger number of Cyatheaceae than Dicksoniaceae, the richness will tend to increase. However, the increase in richness might lead to homogenization, consequently decreasing β-diversity (H_2). To test these hypotheses, we used distribution models to predict the potential species distribution in the present and future. Then, we calculated the changes in potential distribution areas, α-, and β-diversity components between scenarios. Finally, we assessed the impact of these changes within Protected Areas. We found that Dicksoniaceae distribution tends to shrink in the future, while half of Cyatheaceae tend to lose distribution. Species richness tends to decrease in the future, as well as β-diversity. At least 43% of the sites tend to have their species richness reduced, while only 26% of sites tend to gain species. Our results suggest that species associated with cold environments will lose suitable areas and are more threatened. Richness tends to decrease mainly in sites with high precipitation seasonality. Furthermore, the tree ferns assemblage tends to homogenize.

Keywords: Beta-diversity, climate change, Cyatheaceae, Dicksoniaceae, species distribution modelling, tree ferns.

1. INTRODUCTION

Tree ferns are expressive elements in (sub)tropical forest formations (Tryon & Tryon, 1982), sometimes establishing monodominant forests (Gasper et al., 2011). For this reason, tree ferns act in the dynamics of the ecosystem and may affect the regeneration of wood species and nutrient cycling (Brock, Perry, Lee & Burns, 2016). In addition, they contribute to the process of ecological succession (Arens & Baracaldo, 1998), to biomass stock in tropical forests (Sarmiento, Pinillos, & Garay, 2005), and provide microhabitat for several epiphytic plants, many of them occurring exclusively on tree ferns caudices (Wagner, Mendieta-Leiva, & Zotz, 2015).

Tree ferns have suffered intense exploitation in the tropical forest due to ornamental use of their caudices (Eleutério & Pérez-Salicrup, 2006; Hoshizaki, & Moran, 2001), causing populational exhaustion of many species (Santiago, Mynssen, Maurenza, Penedo, & Sfair, 2013). Combined with this, the actual high forest fragmentation in the Atlantic Forest (Ribeiro, Metzger, Martensen, Ponzoni, & Hirota, 2009) plus climate change scenarios (IPCC, 2014; Lima, Ribeiro, Grelle, & Pinto, 2019) are other potential sources of threat to tree ferns. These threats might change the density and distribution of these species as well as the locations of the suitable area to grow and reproduce.

Moreover, despite tree ferns being an important group in forest structure, these plants are historically neglected in floristic and ecological studies in Brazil (Weigand & Lehnert, 2016). There are two main families of tree ferns in subtropical Atlantic Forest: Dicksoniaceae and Cyatheaceae. The former is represented by *Dicksonia sellowiana*, a species that inhabits high and cold environments (Gasper et al., 2011), and *Lophosoria*

quadripinnata, that grows in ravines, doing best on moist, well-drained soil and in full sun until 2000 m in eastern Brazil (Lehnert & Kessler, 2018). The later family (about 14 species) exhibits a preference for warm, humid and low seasonal climates (Bystriakova, Schneider, & Coomes, 2011) and may benefit from warmer climate regimes.

The Intergovernmental Panel on Climate Change (IPCC) predicts shifts in rainfall and temperature in the subtropical Atlantic Forest, and since water availability is correlated to fern species richness (Kessler, Kluge, Hemp, & Ohlemüller, 2011), rainfall regimes modifications could impact in ferns distribution. Therefore, a reduction of cold environments and increasing of warm and humid environments may impact *D. sellowiana*, an already endangered species, through the restriction of its occurrence area.

In this regard, our study sought to predict the impacts of future climate changes in α- and β-diversity of tree ferns in the subtropical Atlantic Forest, as well as to predict the impact in the potential distribution of each species. Our first hypothesis (H_1) is that species from both families will change their potential distribution areas. We expect Dicksoniaceae species will have their potential distribution area reduced (specially *Dicksonia sellowiana*) because of their association with cold habitats — and Cyatheaceae species will increase their potential distribution areas since they generally occur along hot and humid regions. Our second hypothesis (H_2) is that the changes in species distribution will affect α- and β-diversity. Since we expect the Cyatheaceae species will increase their distribution range, hence increasing the overlap in species areas, we expect an increase in α-diversity and a decrease in β-diversity, i.e., less variation in species composition among sites, leading to a homogenization of our study region.

2. MATERIAL AND METHODS

2.1 Study area

The study area is delimited by the subtropical Atlantic Forest, floristically distinct from the tropical Atlantic Forest (Eisenlohr & Oliveira-Filho, 2014). The subtropical Atlantic Forest occurs in southern Brazil as well as in parts of Argentina and Paraguay (C. Galindo-Leal & Câmara, 2005). The predominant climate type is Cfa (temperate humid with hot summer), with some areas fluctuating to Cfb (temperate humid with temperate summer). The relief ranges from sea level to altitudes near 1200 m, including peaks that reach almost 1900 m. Distinct forest types can be found, which includes Restinga, on the coastal areas; Rainforests, in low altitudes at the coastal region (< 800–900 m); Mixed Forests (*Araucaria* forest), generally in areas with altitudes over 800 m; and Semideciduous Forests in low areas with high seasonality in precipitation (Figure 1).

2.2 Data gathering

Among the species that occur in subtropical Atlantic Forest (following Brazilian Flora 2020 in construction 2019; Fernandes 1997; Lehnert & Weigand, 2016), we identified 1167 records and 15 species from the literature (Fernandes, 1997; Vibrans, Sevegnani, Lingner, Gasper, & Sabbagh, 2010) and herbaria (using SpeciesLink – http://splink.cria.org and GBIF Global Biodiversity Information Facility – https://www.gbif.org). Uncertain records were removed, as well as *Cyathea uleana* due to its low number of registers.

The data available for the study area was under the effect of sampling biases. This issue arose because, in the state of Santa Catarina, the Floristic and Forest

Inventory of Santa Catarina (IFFSC; Vibrans et al., 2010) had strongly sampled the state's territory — an effort that did not take place in other locations of the study area. To minimize sample biases, we performed spatial filtering on the occurrence data by defining a buffer of ≅ 30 km as a minimum distance between occurrence records in all study areas (Fourcade, Engler, Rödder, & Secondi, 2014).

Climate data were obtained for the present and future climate change scenarios from WorldClim v2.0 (Fick & Hijmans, 2017) and WorldClim v1.4 (Hijmans, Cameron, Parra, Jones, & Jarvis, 2005), respectively. Data for the future scenario refers to the year 2050 and was based on the climate projections made by IPCC. We used the mean of four Earth System Models (ESM) for future modeling: NorESM1-M (Bentsen et al., 2013), MIROC-ESM (Watanabe et al., 2011), HadGEM2-ES (Jones et al., 2011), and CNRM-CM5 (Voldoire et al., 2013). For these models, two future scenarios were selected: the most optimistic (RCP 2.6) and the most pessimistic (RCP 8.5), which were elaborated and named accordingly to the concentration of carbon dioxide in the atmosphere (W/m^2) between the pre-industrial period and the year 2100 (IPCC, 2014). In all tested ESMs we chose 19 environmental variables (bioclim) with 2.5' resolution. We used the Variance Inflation Factor (VIF) via the *vifstep* function on *usdm* R-package (Naimi, Hamm, Groen, Skidmore, & Toxopeus, 2014; R Core Team 2015) to remove collinearity. This function generates 5,000 random points across the climate layers and calculates the VIF for each variable using a step-by-step procedure. At each step, the layer with the highest VIF is removed from the set. The process is repeated until only variables with a VIF value below a defined threshold remain. We used a restrictive VIF value (VIF = 5) that reduced the 19 climate variables to six: Annual Mean Temperature (BIO01), Isothermality (BIO03), Temperature Annual Range (BIO07), Mean Temperature of Driest Quarter (BIO09), Precipitation of Wettest Month (BIO13) and Precipitation of Coldest Quarter (BIO19).

2.3 Species Distribution Models (SDMs)

We used SDMs to predict current and future distributions of each species. Our SDMs were formulated using the *sdm* R-package (Naimi & Araújo, 2016) through five different algorithms: MaxEnt (Phillips, Dudík, & Schapire, 2004), Boosted Regression Trees (BRT; Friedman, 2001), Generalized Linear Models (GLM; McCullagh & Nelder, 1989), Multivariate Adaptive Regression Spline (MARS; Friedman, 1991), and Support Vector Machine (SVM; Vapnik, 2013), all with default settings. Moreover, as proposed by Barbet-Massin, Jiguet, Albert, & Thuiller (2012), we applied 10 000 pseudo-absences for each species randomly created across the study area.

We evaluated the performance of the models generated by each algorithm through a cross-validation procedure with 10 repetitions, where data were randomly divided into training, amounting to 70% of the data, and testing, comprising the remaining 30%. Two statistical methods were responsible for evaluating the robustness of the models: Area Under the Curve (AUC) of the receiver operating characteristic curve (ROC; Fielding & Bell, 1997) — a threshold-independent metric — and True Skill Statistic (TSS; Allouche, Tsoar, & Kadmon, 2006) — a threshold-dependent metric. All generated models were above the previously established cutoff values (AUC > 0.70 and TSS > 0.40).

To avoid biases created by choosing a single statistical algorithm, as pointed out by several studies (e.g., Araújo & New, 2007; Grenouillet, Buisson, Casajus, & Lek,

2011), we built a single final model through an ensemble approach. In this sense, the models generated by statistical algorithms were projected using current and future climate, and the ensemble model for each scenario was considered as the mean weighted by the AUC value of the projections of each algorithm. Binary maps (presence-absence) were made based on the threshold that maximizes the sensitivity plus specificity of the models (maxSSS; Liu, Newell, & White, 2016), calculated with the R-package *SDMtools* (VanDerWal, Falconi, Januchowski, Shoo, & Storlie, 2014). The threshold was calculated for each species based on the final ensemble map of the current scenario. Likewise, the resulting threshold value for the current scenario was also used to build binary maps for future scenarios.

2.4 Species distribution and diversity patterns

To understand how climate change will affect species distribution and diversity patterns, the study area was divided into hexagonal cells of ≅ 200 km². Based on SDM predictions for tree ferns, we measured for each scenario the changes of suitable cells for all species and the α- and β-diversity indexes. We also evaluate the same metrics exclusively in grid cells that overlap Protected Areas (PAs) in the study region. We obtained the boundaries of PAs from the World Database on Protected Areas (UNEP-WCMC, 2019). Therefore, we can compare species distribution and diversity indexes between protected and non-protected areas for any specific scenario.

We measured α-diversity as the number of species present in each cell, i.e., the predicted species richness. We calculated the β-diversity as the total variance of the community (Total β-diversity, or BD_{TOTAL}; *sensu* Legendre & De Cáceres, 2013) and it was subsequently decomposed in Local Contributions to β-Diversity (LCBD). LCBD represents the ecological uniqueness of each hexagonal cell regarding species composition. We calculated BD_{TOTAL} based on the Jaccard dissimilarity coefficient (1–similarity). Then, the LCBD was determined based on the partition of BD_{TOTAL} between the cells (Legendre & De Cáceres, 2013). The significance of the LCBD values for each cell was obtained through 999 permutations, where the species are distributed randomly and independently along the grid and the LCBD values are calculated for each random distribution. Significant LCBD values were those with p-value < 0.05.

Finally, we followed Legendre (2014) by partitioning the BD_{TOTAL} in species replacement and richness difference components, using the replacement ($Repl_J$) and richness difference ($RichDiff_J$) indexes of the Podani family (*sensu* Podani & Schmera, 2011; Legendre, 2014). All β-diversity metrics were calculated with the *adespatial* R-package (Dray et al., 2019) using the functions *beta.div* and *beta.div.comp*. We used QGIS 3.8 (QGIS Development Team 2019) to generate the maps. Lastly, to test differences in α-diversity between current and future scenarios, we conducted the Wilcoxon Rank-Sum Test, a non-parametric unpaired test, for the current scenario × RCP 2.6 scenario and current scenario × RCP 8.5 scenario.

3. Results

All models had a good predictive power (AUC: mean = 0.825, sd = 0.05; TSS: mean = 0.623, sd = 0.14 — for individual distribution maps and statistical performance

of each SDM see the supporting information S1, and the interactive maps in https://avrodrigues.shinyapps.io/tferns/). Overall, the predicted impacts of climate change on species were similar in both RCP 2.6 and RCP 8.5 (Table 1), where Dicksoniaceae species tend to lose area of distribution and half of Cyatheaceae species tend to increase its area of occurrence — while the other half tends to decrease. Also, currently suitable areas inside PAs also suffer a significant reduction for many tree ferns.

3.1 α-diversity

The highest α-diversity values (current and future scenarios, Figure 2) were found in the Rainforest areas. In the west, where α-diversity is low in the current scenario, many sites tend to become unsuitable for all the species in the future. Furthermore, PAs harbor a higher average of α-diversity when compared to the subtropical Atlantic Forest. At the same time, PAs will lose fewer species than non-PAs and gain more species than non-PAs regions (Table 2).

3.2 β-diversity

The BD_{TOTAL} is predicted to decrease with climate change (Table 3**)**, indicating community homogenization along the study area. The LCBD values (Figure 3) ranged from 3.2×10^{-4} to 5.5×10^{-4} in the current scenario, 2.7×10^{-4} to 5.8×10^{-4} in the RCP 2.6 scenario and 2.5×10^{-4} to 6.0×10^{-4} in the RCP 8.5 scenario, indicating an increase in LCBD at some locations in the future and a decrease in others. Considering only PAs, community homogenization is like the rest of the study area (Table 3).

4. DISCUSSION

4.1 Changes in species distribution

Our first hypothesis was that the species potential distribution would change in future climate scenarios — where Dicksoniaceae species would have its distribution reduced and Cyatheaceae species would have its distribution increased. We partially confirmed our expectation since Dicksoniaceae species indeed had their potential distribution reduced in both future scenarios, but only half of Cyatheaceae species had their potential distribution increased. *Cyathea phalerata*, a common species in southern Brazil, had its potential area reduced in both scenarios, and *Alsophila setosa* — one of the most common tree ferns in forest communities in subtropical Atlantic Forest (Lingner et al., 2015) — increased its area of occurrence in scenario RCP 2.6 but decreased in RCP 8.5. *Cyathea feeana*, recently segregated from *C. corcovadensis* (Lehnert & Weigand, 2013), had the most reduced potential occurrence area. All Cyatheaceae species that were predicted to lose potential area in the future by our model showed climatic suitability for the coldest regions of the study area at the current scenario (Figures S1-S14 and interactive maps in https://avrodrigues.shinyapps.io/tferns/).

A global analysis of Cyatheaceae distribution patterns showed a clear preference of the group for hotter and wetter locations with low seasonality, although some species show a capacity to occupy relatively cold regions, rarely where minimum temperatures

drop below freezing (Bystriakova et al., 2011). Half of the Cyatheaceae species analyzed here were able to occupy the coldest areas of the current Mixed Forest. However, few Cyatheaceae species were able to occupy areas of Semideciduous Forest, where there is higher seasonality in precipitation than areas of Rainforest and eastern Mixed Forest (Oliveira-Filho, Budke, Jarenkow, Eisenlohr, & Neves, 2015; Oliveira-Filho & Fontes, 2000). The effects of high temperatures, reducing the cold intensity (Wilson, Walters, Mayle, Lingner, & Vibrans, 2019), set up favorable conditions for some of Cyatheaceae, since they expanded to it.

Dicksonia sellowiana, a species able to withstand colder environments (Gasper et al., 2011), was restricted to higher and colder locations, wherein the Mixed Forest predominates. In the same way, *L. quadripinnata* seems able to inhabit the colder regions of the Mixed Forest. In addition, the warmer areas of the Rainforest also showed high climatic suitability for this species. In fact, *L. quadripinnata* has been recorded in both temperate (Ricci, 1996) and tropical (Bernabe et al., 1999) Rainforests. In this sense, precipitation rather than temperature appears to be the main determinant of *L. quadripinnata* occurrence.

4.2 Changes in diversity patterns

The second hypothesis was that the different responses of Dicksoniaceae and Cyatheaceae species should cause significant changes in diversity patterns. Our expectation was that species richness (α-diversity) would increase, following the increase in overlapping distributions of Cyateaceae species, which in turn would lead to a decrease of β-diversity. Our expectation about α-diversity was not confirmed since we found evidence for less species richness per cell in the future scenarios. Nevertheless, we found evidence for future homogenization of cells in the subtropical Atlantic Forest, since both BD_{TOTAL} and LCBD values tend to decrease in future scenarios.

Our estimates indicate that at least 40% of the sites (cells) will lose richness in the subtropical Atlantic Forest. This is due to a reduction in the area of distributions for Cyatheaceae species, against our expectations. This loss can affect the conservation of other biological groups that depend on tree ferns caudices, such as epiphytes — some of these growths exclusively in tree ferns (Mehltreter, 2008). These changes may be associated with variations in precipitation since water availability seems to be an important species richness predictor for ferns, as pointed out by several authors (Aldasoro, Cabezas, & Aedo, 2004; Gasper, Eisenlohr, & Salino, 2015; Kessler et al., 2011). The current low α-diversity of tree ferns observed in the midwest and far west, as well as the reduction of α-diversity in the future scenarios, may be caused by the rainfall regime and high climatic seasonality (Bystriakova et al., 2011; Cabré, Solman, & Núñez, 2016).

In the study region, the BD_{TOTAL} is more influenced by $RichDiff_J$ among the sites than by $Repl_J$ (Figure 3 and Table 3). Together, these results indicate that local extinctions are important drivers of homogenization in species composition of tree ferns. The high influence of $RichDiff_J$ in β-diversity could be explained by the environmental filtering process (Atmar & Patterson, 1993). This process is evidenced by the loss of species in the west region, with high precipitation seasonality, and by the loss of more suitable areas by species from a colder climate. Working with ferns communities in the northern Atlantic Forest, da Costa, Arnan, de Paiva Farias, & Barros (2019) found a higher contribution of species replacement than of species differences in ferns on a regional and local scale in tropical Atlantic Forest. The authors listed high

dispersal ability, niche preferences, and the heterogeneous structure of the Atlantic forest as the three main factors that contribute to species replacement. Two differences between the studies could explain the different results: i) our study used local assemblage in coarse scale, provided by the stack of SDMs, which is not able to capture differences in microhabitats as in da Costa et al. (2019), and ii) we focused on the tree ferns niches rather than all ferns species in the subtropical Atlantic Forest.

4.3 Implications for biodiversity conservation agenda

Our results provide relevant insights to the conservation of tree ferns by predicting which species will lose larger proportions of adequate habitat in the future (Table 1). We note that this phenomenon will mainly affect species that occupy higher areas such as *A. capensis*, *C. feeana*, *C. villosa*, *S. gardneri*, *L. quadripinnata*, and *D. sellowiana*. Although extinctions did not occur in our study area, historically endangered species such as *D. sellowiana* had their potential distribution area greatly diminished (–42%). Also, the climate changes can alter the threatened status from endangered (Santiago et al. 2013) to critically endangered (considering IUCN 2012 criteria of population size reduction: A1cB1b). *C. feeana* and *S. gardneri* (currently present in 424 and 386 cells, respectively), both endemic to eastern Brazil, seem to be the most threatened species in the future considering the reduction in cell numbers (to 208 in RCP 2.6 and 95 in RCP 8.5 for *C. feeana;* and to 217 in RCP 2.6 and 134 cells in RCP 8.5 for *S. gardneri*). *Cyathea feeana* occurs from 500 to 1750 m and was recently segregated from *C. corcovadensis* (Lehnert & Weigand, 2013), which may bias our results since not all herbaria (despite our best efforts) review these records. *Sphaeropteris gardneri* occurs between 450–600 m and few individuals were found by the IFFSC project (Gasper et al., 2018).

More than half of the studied species are predicted to lose suitable areas, even inside PAs (Table 2). So, in order to safeguard these species, new PAs will be needed as they play an essential role in protecting species *in situ* (Chape, Harrison, Spalding, & Lysenko, 2005). However, these new PAs may too have reduced long-term effectiveness in species conservation since they are also going to be impacted by climate change. Then, to optimize cost-benefit analysis of implementing new PAs, lawmakers and specialists should always consider species conservation now and in the future (Araújo, Alagador, Cabeza, Nogués-Bravo, & Thuiller, 2011). Also, LCBD values should be interpreted with caution when looking at alternative indexes for conservation use since they are not always positively correlated to species richness (Legendre & De Cáceres, 2013; Silva & Hernández, 2014).

Unfortunately, some PAs in higher altitudes, such as Parque Nacional de São Joaquim — where *D. sellowiana*, as well as other threatened species (such as *Araucaria angustifolia*; Wilson et al., 2019) may find suitable areas in the future — are threatened to be downsized (see the workgroup created to study the protected area boundaries; ICMBio 2019).

At last, we expected that some tree fern species could expand their distributions as a result of climate change. However, we must not extrapolate, without deep analysis, this expansion to other groups. It is well known that animals (Radchuk et al., 2019) and plants (Corlett & Westcott, 2013) will not be able to keep up, in general, with the changes affecting the planet.

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Table 1. Comparisons between potential areas occupied by the tree ferns in all scenarios (current, RCP 2.6, and RCP 8.5). The percentages are based on the subtropical area of the Atlantic Forest (≅ 731.788 km²) and Protected Areas inserted into the study region. Negative values (**bold**) indicate a reduction of the potential area, while positive values indicate an increase in the potential area. The "Current" column indicates the predicted number of suitable cells at the present.

Protected Areas			**Species (record number)**	**Subtropical Atlantic Forest**		
Current	**RCP 2.6**	**RCP 8.5**		**Current**	**RCP 2.6**	**RCP 8.5**
			CYATHEACEAE			
340	**-27.06**	**-43.82**	*Alsophila capensis* (20)	815	**-40.61**	**-55.71**
477	**-9.43**	**-24.32**	*Alsophila setosa* (144)	1239	4.68	**-17.68**
219	41.10	12.33	*Alsopahila sternbergii* (19)	297	86.87	21.89
481	**-9.15**	**-10.81**	*Cyathea atrovirens* (204)	1084	4.89	3.97
349	**-0.57**	2.87	*Cyathea corcovadensis* (142)	615	7.15	17.40
385	1.56	17.92	*Cyathea delgadii* (95)	849	14.02	39.58
188	**-40.43**	**-69.68**	*Cyathea feeana* (29)	424	**-50.94**	**-77.59**
301	3.32	**-2.33**	*Cyathea hirsuta* (29)	470	28.30	20.21
248	24.60	41.53	*Cyathea leucofolis* (20)	312	85.58	145.19
371	**-6.47**	**-9.16**	*Cyathea phalerata* (205)	777	**-5.15**	**-6.18**
-	-	-	*Cyathea uleana*[1] (6)	-	-	-
354	**-25.99**	**-49.72**	*Cyathea vilosa* (21)	801	**-41.32**	**-68.16**
205	**-35.61**	**-57.56**	*Sphaeropteris gardneri* (21)	386	**-43.78**	**-65.28**
			DICKSONIACEAE			
304	**-22.70**	**-39.80**	*Dicksonia sellowiana* (146)	923	**-25.79**	**-42.15**
336	12.50	0	*Lophosoria quadripinnata* (66)	1255	**-22.23**	**-29.48**

[1]Species not considered in the analysis due to the low number of records.

Table 2. Mean species per cell and changes in α-diversity in the future scenarios (RCP 2.6, and RCP 8.5) for the whole study area and PAs only. "Gain", "equal" and "loss" indicates the proportion of cells that will respectively gain, keep the same number, or lose species in each future scenario.

Protected Areas			**α-diversity**	**Subtropical Atlantic Forest**		
Current	**RCP 2.6**	**RCP 8.5**		**Current**	**RCP 2.6**	**RCP 8.5**
3.6	3.4	3.1	**Mean per cell**[2]	5.53	5.15	4.68
-	26.56	21.83	**Gain (%)**	-	23.52	18.06
-	30.32	25.97	**Equal (%)**	-	28.00	20.36
-	43,12	52.20	**Loss (%)**	-	48.48	61.58

[2]Mean differences between current and future scenarios were supported by the Wilcoxon Rank-Sum Test (p-value < 0.001).

Table 3. BD_{TOTAL} and its components (*RichDiff*$_J$ and turnover) between the current and future climate scenarios for the whole study area and PAs only.

Protected Areas			**β-diversity**	**Subtropical Atlantic Forest**		
Current	**RCP 2.6**	**RCP 8.5**		**Current**	**RCP 2.6**	**RCP 8.5**
0.316	0.303	0.3	**BD_{TOTAL}**	0.320	0.305	0.294
83.3	91.2	91.3	***RichDiff*$_J$ (%)**	80.8	88.3	90.4
16.7	8.8	8.7	**Turnover (%)**	19.2	11.7	9.6

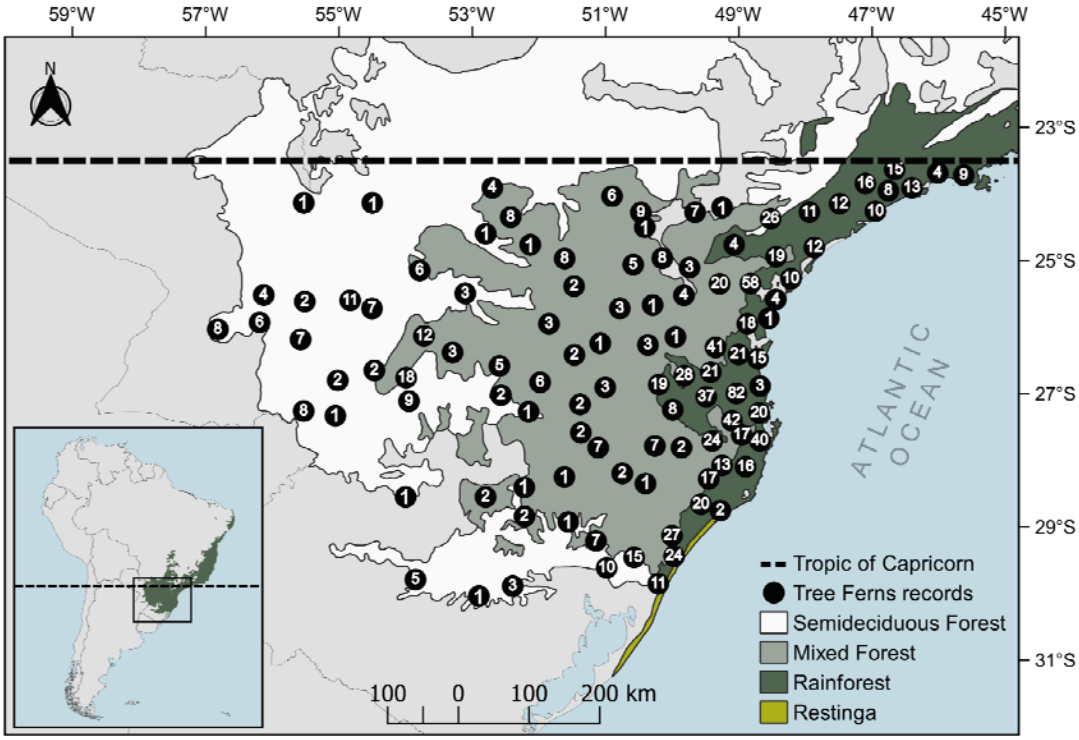


Figure 1. Study area and forest types of subtropical Atlantic Forest. The occurrence data (black dots) was grouped into ≅ 30km² clusters to simplify the visualization. The respective numbers on the dots represent the number of records found in each cluster.

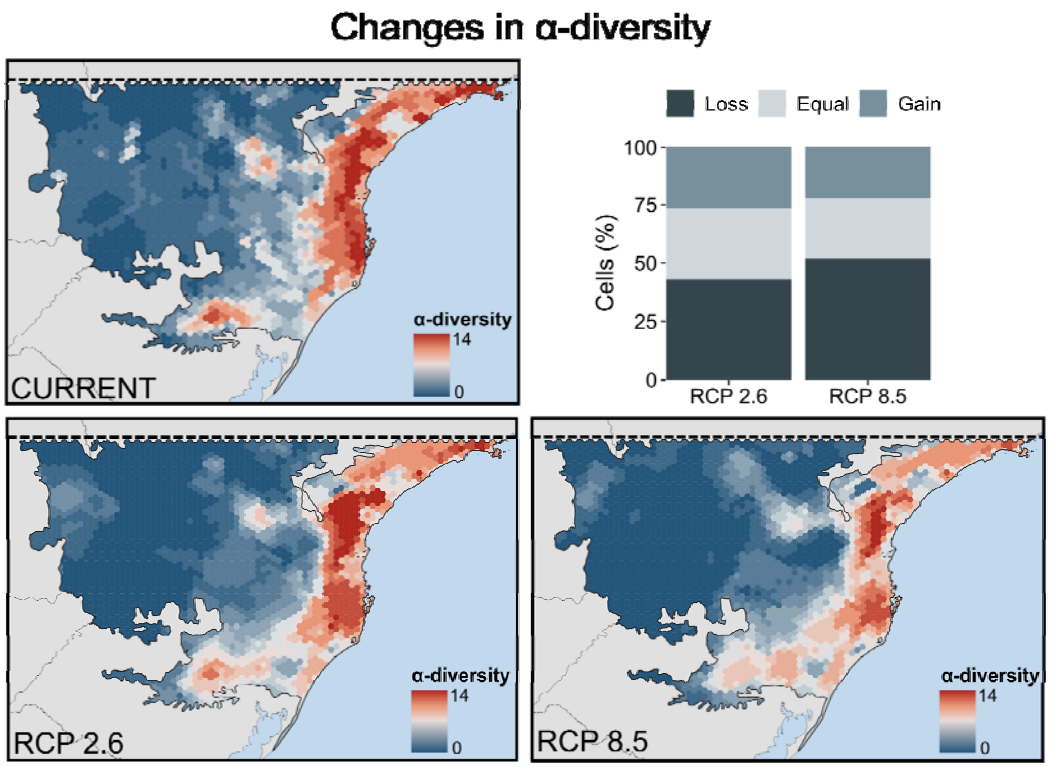


Figure 2. α-diversity of tree ferns in the subtropical Atlantic Forest based on current and future (RCP 2.6 and RCP 8.5) climate scenarios. The bar chart demonstrates the predicted proportion of cells that will lose, keep and gain α-diversity in the two future scenarios.

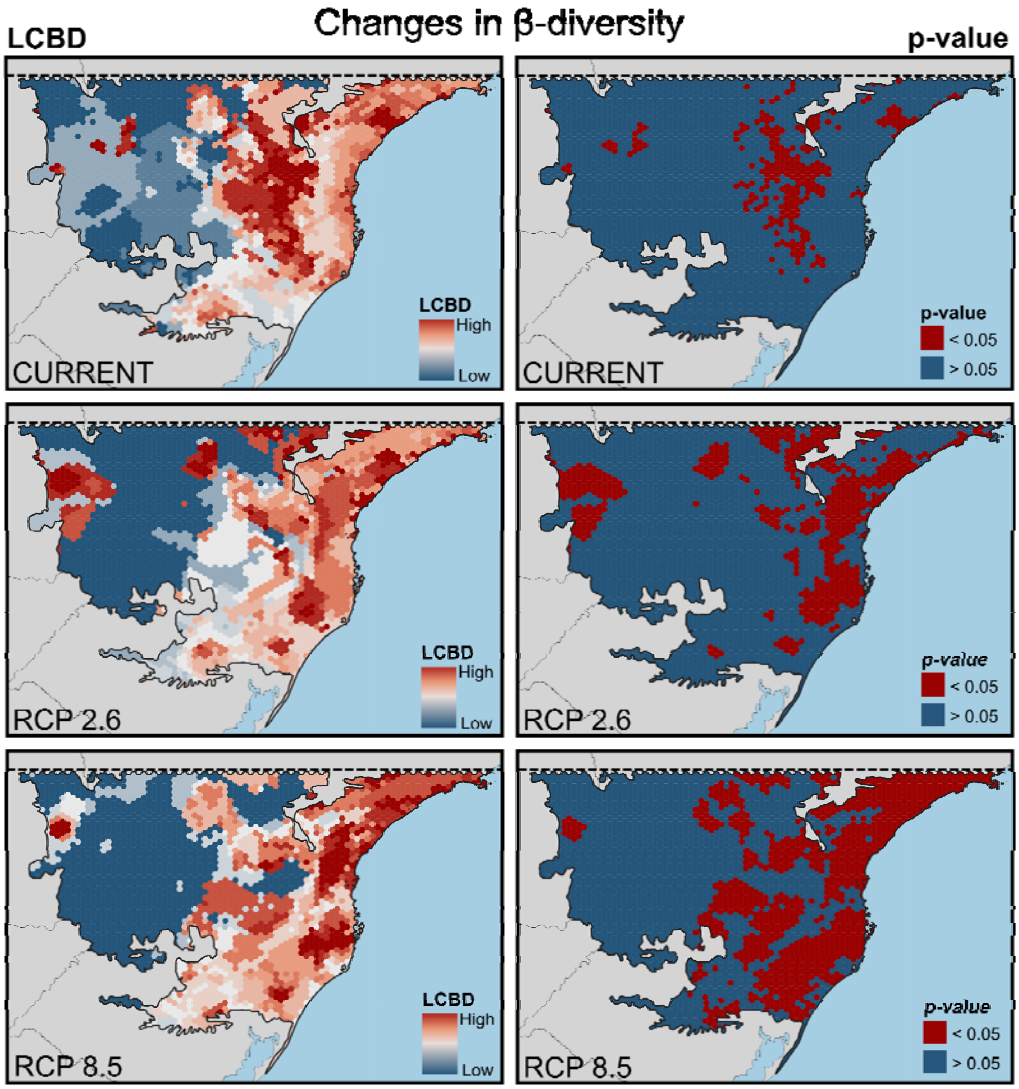
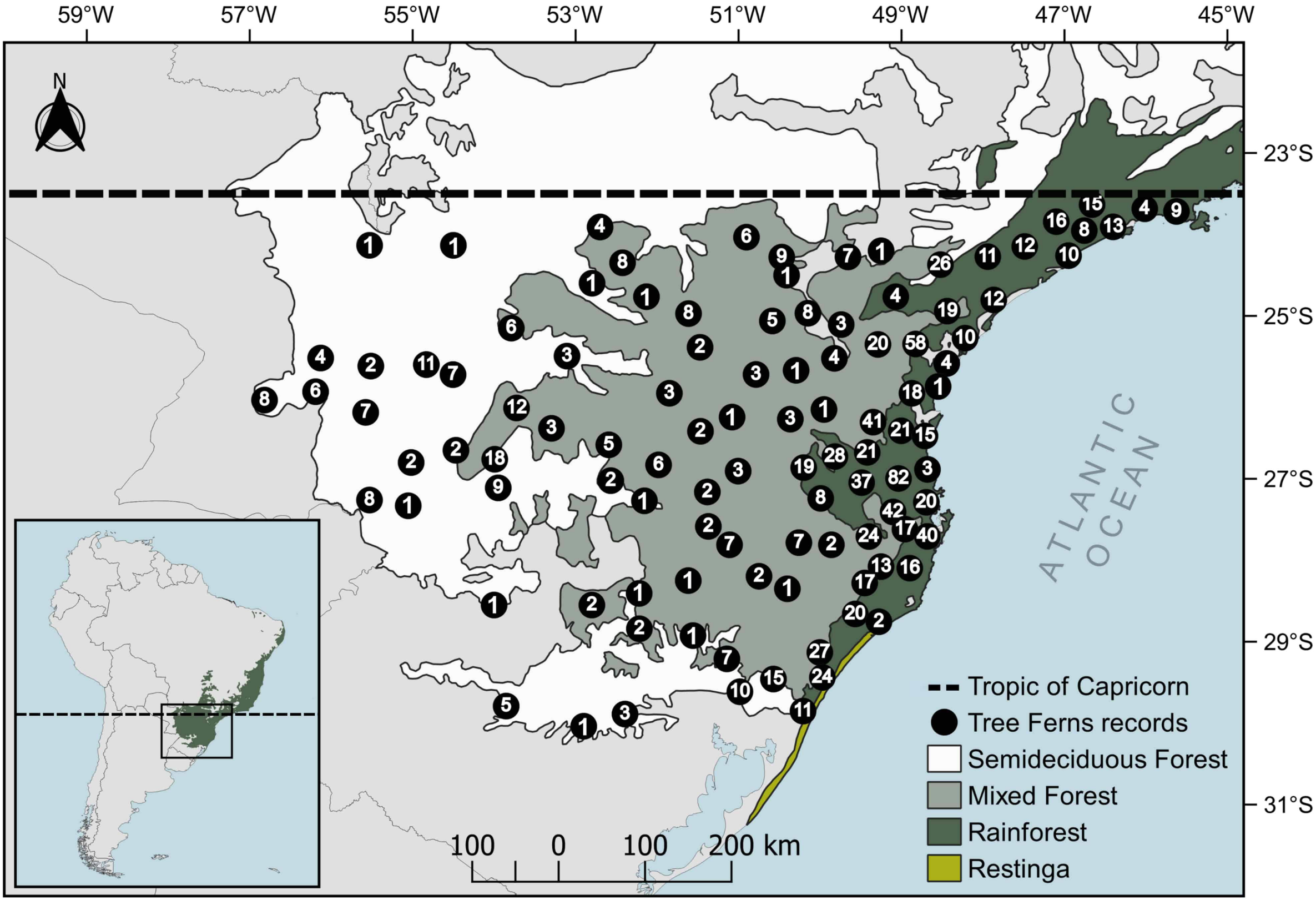


Figure 3. Current and future (RCP 2.6 and RCP 8.5) distribution of LCDB (left column) and LCBD p-values (right column) of tree ferns in the subtropical Atlantic Forest. Cells with p-value < 0.05 show significantly higher than average LCBD values.


59°W
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ATLANTIC OCEAN
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Tree Ferns records
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Mixed Forest
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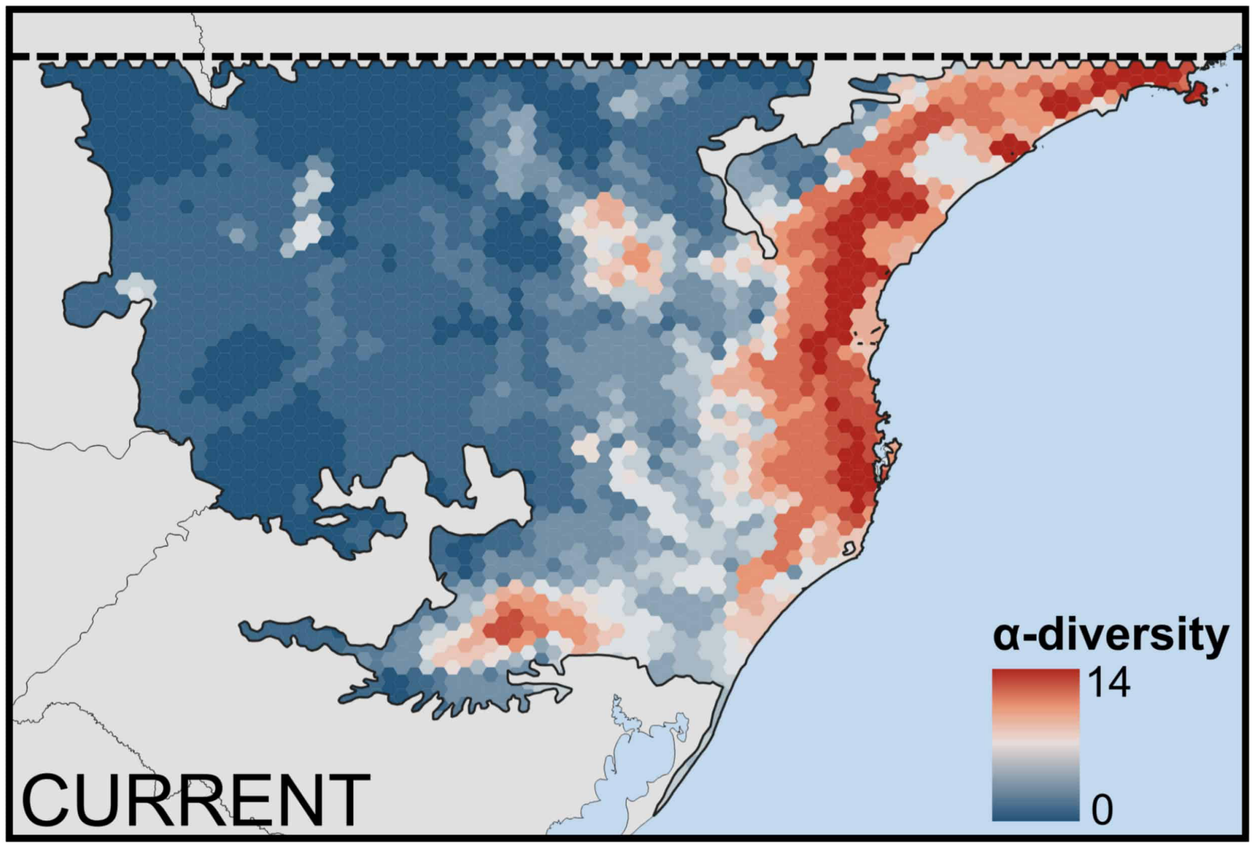
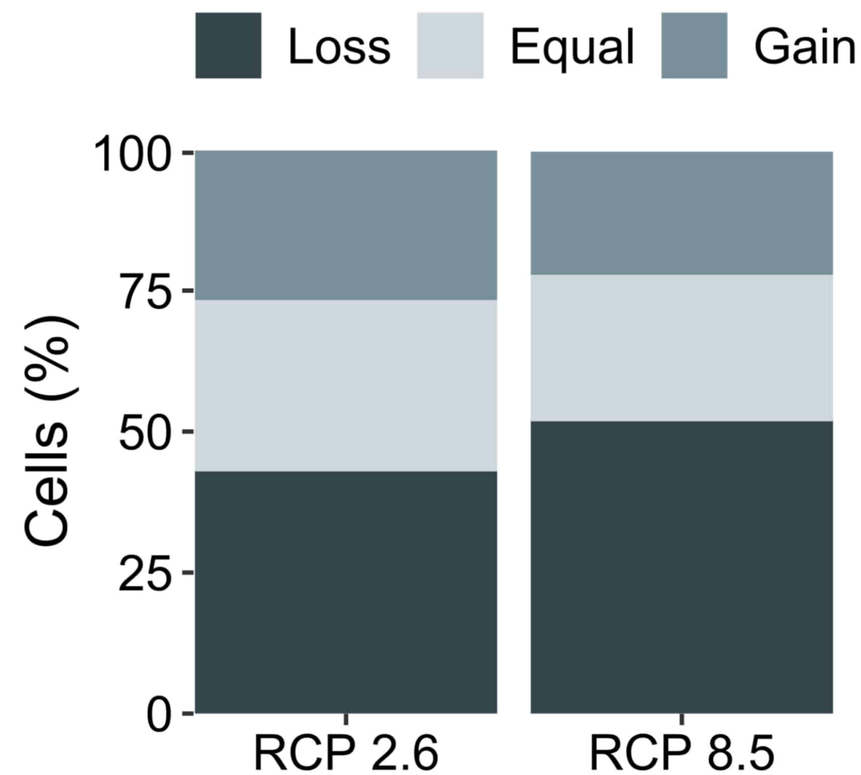
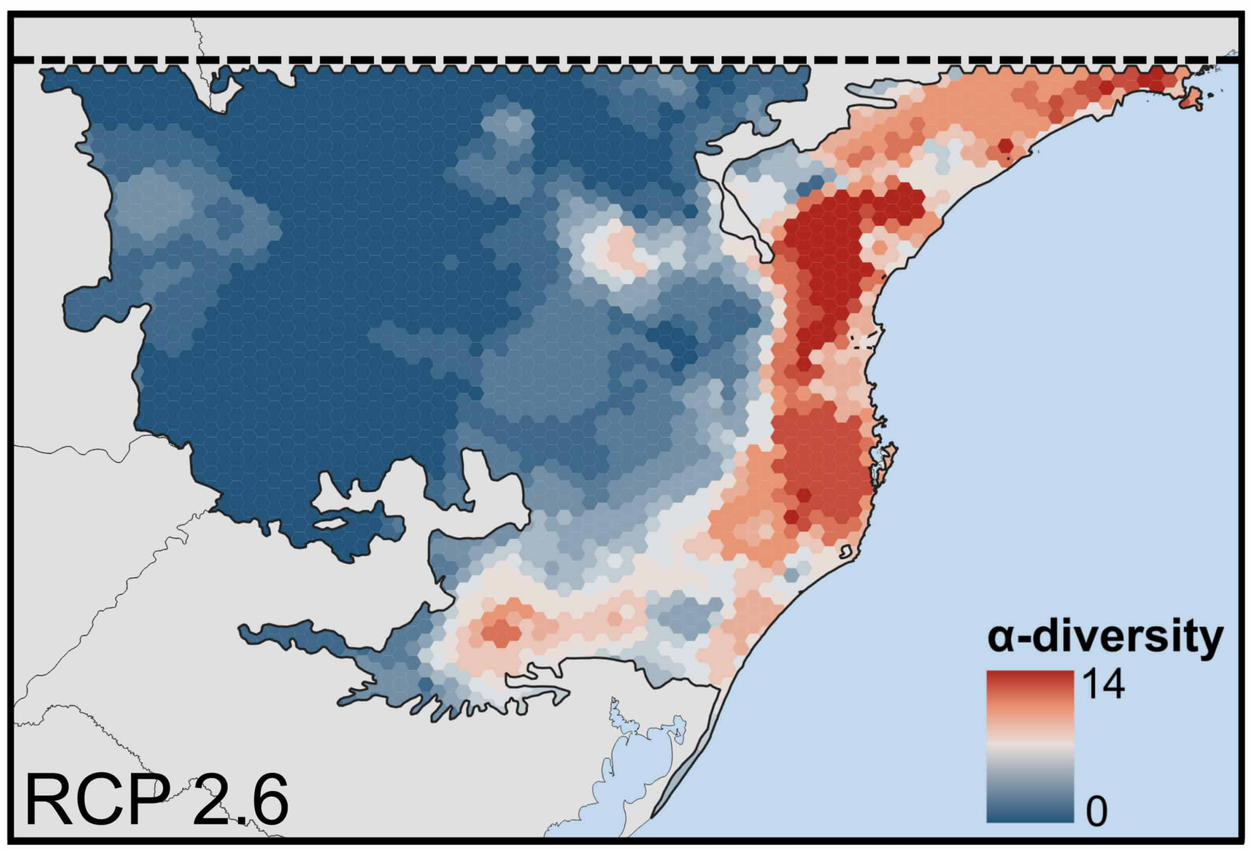
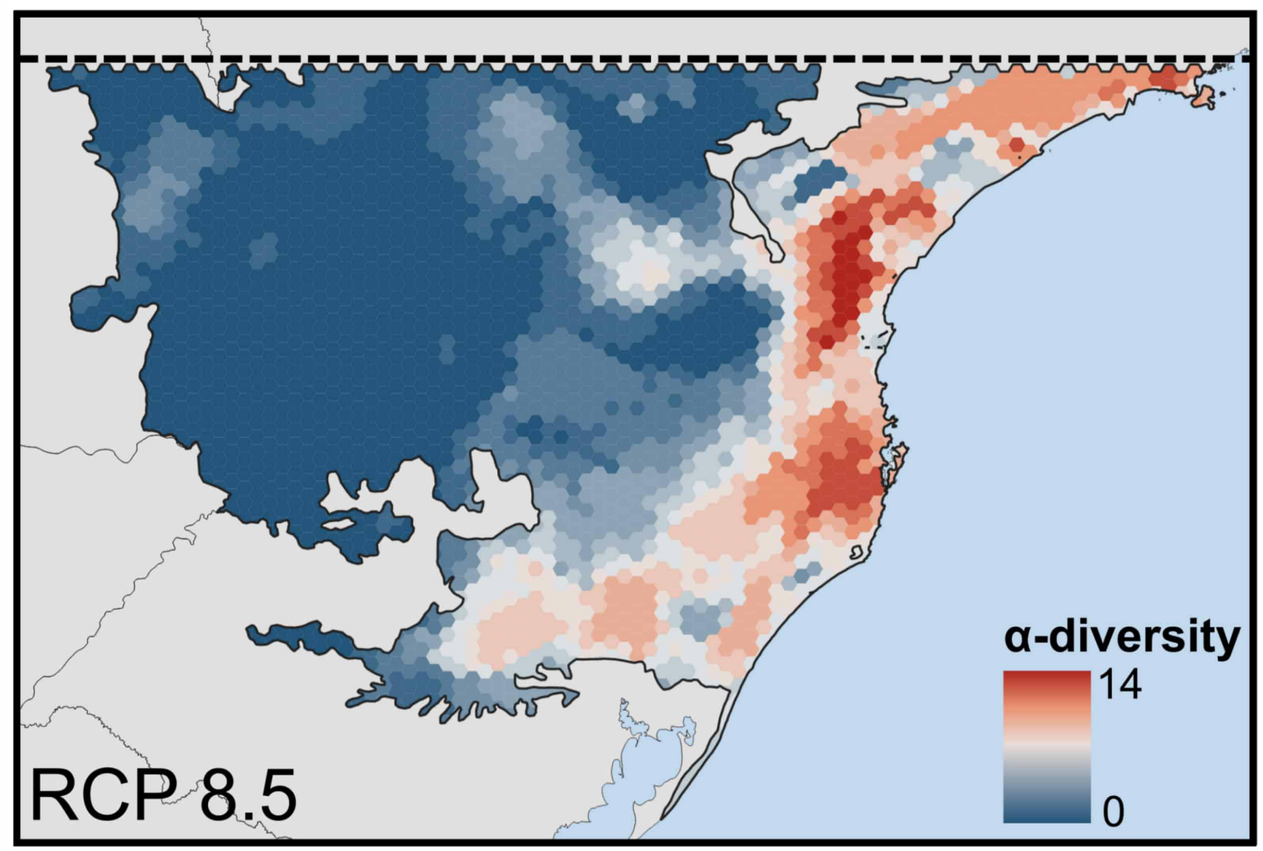
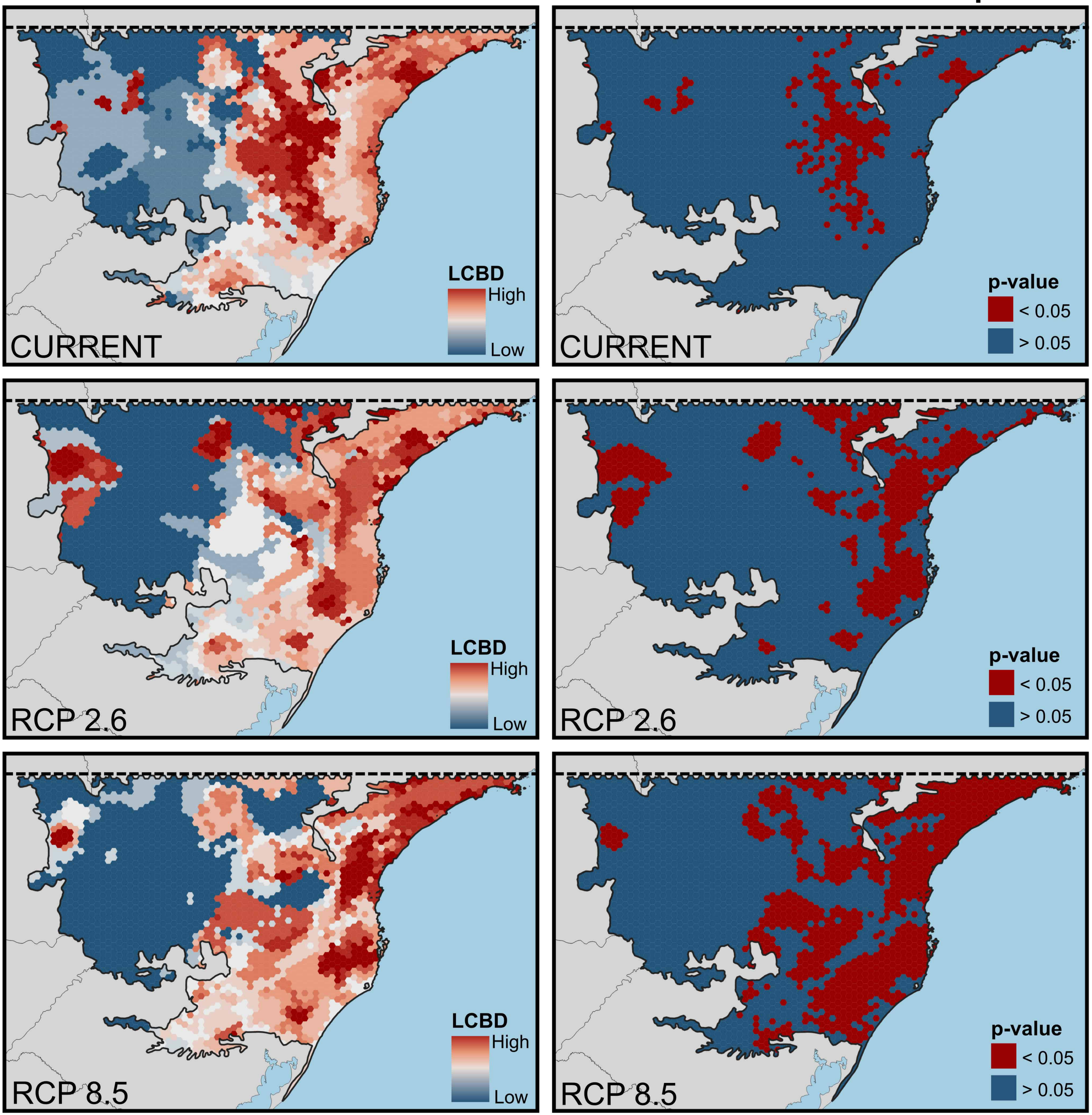
Changes in α-diversity









Changes in β-diversity