Predicting decay in functional diversity in plant communities under future climates

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Abstract

Climate change is expected to cause widespread shifts in the composition of plant communities. However, the extent to which these changes will alter the composition and diversity of plant functional traits is less clear. Here, we assess how climate change may reshape the functional diversity of 32,996 plant communities by combining data on their traits and realised climatic niches with future climate projections. We find that under projected climates for the decade centred on 2070, up to a third of species will experience conditions outside of their current realised climatic niches. Yet, on average, the loss of these species from communities represented a potential decay in functional diversity of 9.5%, indicating some functional redundancy. We found weak evidence that patterns of species replacement, which are contingent on their realised climate niches, resulted in increased loss of functional diversity compared with random removal. Patterns of relative functional decay (i.e., projected loss of functional diversity relative to species loss under future climate) were distinct between woody and non-woody species, with woody species being at higher risk of declines in functional diversity, particularly in alpine and woodland communities. These results highlight the importance of integrating additional metrics, such as functional diversity and functional decay, into conservation and restoration planning to prioritise elements of biodiversity that help explain vulnerability to climate change. Using this dimension of biodiversity can help conservation practitioners identify areas vulnerable to climate change, inform management actions, and prioritise the planting of key growth forms to restore ecosystem function and identity.

Keywords

“biodiversity conservation”, “community ecology”, “ecological restoration”, “functional redundancy”, “hypervolume”, “plant traits”

Introduction

Climate change is forecast to continue having a widespread impact on the membership and function of plant communities globally (Franklin et al., 2016). Projecting the composition of future vegetation is challenging due to the complex interaction between heterogenous environmental change (Braga & Laurini, 2024) and species’ varied responses due to their growth forms and functional traits (Verslues et al., 2023). Plant traits underpin fundamental mechanisms of reproduction, dispersal, adaptation, and stress tolerance (Suding et al., 2008). By aggregating these attributes, we can capture information about these key functional properties of an ecosystem using a suite of functional diversity (FD) metrics (Díaz & Cabido, 2001). Predicting the extent and effects of declines in FD under future climates can proactively inform conservation and restoration practice and help safeguard plant biodiversity and its vital functions.

Plants have persisted through dramatic changes in climate for millennia, primarily through movement to track suitable conditions for growth and reproduction (Pardi & Smith, 2012) (Wang et al., 2022). Some plant species may also adapt in situ, especially where their dispersal capacity and population movement lags behind by the rate of climate change, else become locally or globally extinct (Corlett & Westcott, 2013a; Jackson & Overpeck, 2000). Given unprecedented rates of Holocene climate change (Marcott et al., 2013) many plant species are projected to experience conditions which outpace their capability to adapt or disperse and widespread extinctions are forecast (Bachman et al., 2024; Corlett & Westcott, 2013a). There is already extensive documentation of plant species shifting their ranges in response to anthropogenic climate change (Bradley et al., 2025; Rubenstein et al., 2023; Wiens, 2016).

Additionally, experimental evidence from the field attribute climate change as the driver of shifts in the abundance, identity and diversity of species in plant communities (Elmendorf et al., 2015; Reich et al., 2022). Although some species increase in abundance in response to new growing conditions, while others may persist in lower densities despite climate stress, these compositional changes are still able to underpin substantial changes to ecosystem function (Vellend et al., 2013). While species-specific predictions are valuable, anticipating the broader ecological consequences of climate change requires generalisable frameworks which can guide actionable conservation and restoration strategies at local scales. Focusing on the functional attributes of species, rather than their taxonomic identity, offers a flexible and generalisable predictive framework for forecasting vegetation change.

Climate related changes in the composition of plant communities will vary depending on the traits and tolerances of individual species (Kress & Krupnick, 2022; Pacifici et al., 2017). Functional traits capture the variety of ecological strategies that plants use to grow, reproduce, and survive in the environment (Li & Prentice, 2024). As such, many traits exhibit associations with climate (Dell et al., 2011; Moles et al., 2014). As growing conditions change, the rate at which species are affected may be mediated through the traits that define their adaptive capacity (Andrew et al., 2022; Kühn et al., 2021), or the traits which shape their ability to disperse and track suitable climates (Angert et al., 2011). Functional diversity (i.e., the values and distribution of traits) defines the amount of occupied niche space in a community, which is a predictor of ecosystem attributes like productivity and resilience (Legras et al., 2018). Incorporating FD metrics into forecasting climate change impacts on plant communities enables us to extend and generalise predictions of which *types* of species might be at risk, but also how their loss may risk the functioning and resilience of the ecosystems they inhabit.

While measures of species diversity have long been used as a tool for directing conservation planning (Fleishman et al., 2006), there is still a great potential to expand climate change adaptation strategies through the integration of traits and FD (Gallagher et al., 2021; Mammola et al., 2021; Pollock et al., 2017). Since FD offers a powerful tool to measure aspects of biodiversity that are sensitive to climate change (Heilmeier, 2019) and describes ecologically meaningful processes (Lavorel, 2013), forecasting its decay in response to a changing climate offers important context for adaptation planning. For instance, predicting spatial patterns of the loss of FD through potential species loss under future climate change (i.e. functional decay) allows conservation practitioners to pre-emptively prioritise and protect vulnerable areas and to devise and prescribe actions for functionally-informed ecological restoration. Updating conservation strategies with specific advice regarding climate change is becoming an increasingly important and common practice and is already being applied to assessments of species vulnerability (Alagador & Cerdeira, 2017), key biodiversity areas (Beier & Brost, 2010), and restoration (Simonson et al., 2021).

In this study, we assess how climate change may alter the FD of plant communities by combining data from floristics plots, species traits, and current and future climate data. We aim to quantify the climate change-induced decay in community level FD, as it relates to the potential loss of species and the unique trait space they contribute to their communities. Specifically, we ask: 1) How much FD may be lost from climate-change induced reductions in species richness (i.e. functional decay)? 2) How do patterns of climate-change induced declines in FD and species richness vary between growth forms (woody or non-woody)? And 3) What are the patterns of functional decay relative to species richness decay spatially, and between different vegetation types?

Methods

We use a large network of plant community data (floristic plots) from eastern Australia to estimate the decay of FD due to climate change, based on filtering out species based on the limits of their present realised climate niches relative to projected climate conditions.

Calculations of functional decay are derived using the proportion of overlapping trait space, based on multidimensional hypervolumes, between baseline and future-climate filtered species lists for each plot (Figure 1).

Floristic plot data

Floristic plot data were acquired from the Vegetation Condition Benchmarks Dataset (Somerville et al., 2019) across New South Wales (NSW), Australia, derived from the Systematic Flora Surveys module of the BioNet Atlas (NSW Office of Environment and Heritage, 2023). The dataset includes 32,996 floristic survey plots that list all vascular plant species within a 400m^2 survey area (Figure 1a). This dataset includes plots that represent all 16 broad vegetation classifications in the study area (NSW)(Department of Planning and Environment, 2017) as it is used to derive condition benchmarks for monitoring and compliance assessments by government agencies (Somerville et al., 2019). These floristic plots span baseline climate conditions between 4.7-21°C of mean annual temperature (MAT), and 205-2054 mm of mean annual precipitation (MAP) (Australian Bureau of Meteorology, 2019). Species names in the plot dataset were standardised against the Australian Plant Census (Australian National Herbarium, 2023) using the 'APCalign' package (Wenk et al., 2024a) to create a consistent taxonomic backbone prior to integrating additional data sources.

Plant trait data

Trait data for all species occurring in floristic plots was sourced from the AusTraits database (Falster et al., 2021; Wenk et al., 2024b)(Figure 1bi). Each species was assigned a category of woodiness (i.e., woody/non-woody) from Wenk et al. (2024b), and the species means of all available values for six traits were used to calculate FD: leaf nitrogen per dry mass (mg/g), leaf area (mm^2), leaf mass per area (g/mm^2), maximum plant height (m), seed dry mass (mg), and wood density (mg/mm^3). These traits encompass variation in plant resource use (Parkhurst & Loucks, 1972; Wright et al., 2004), competition (Grime, 1988), and reproduction (Bekker et al., 1998), and have varied but well studied relationships with temperature and precipitation (Dwyer et al., 2014; Moles et al., 2009; Towers et al., 2024; Wright et al., 2001).

Where empirical values were not available for some species-trait combinations, we used gap-filling procedures to impute trait values, a common method in macroecological studies (Schrodt et al., 2015; Andrew et al., 2025). Imputation was performed using the R package ‘Rphylopars’ (Goolsby et al., 2017) which uses phylogenetic information in combination with data from known trait values to estimate missing values across a taxon-trait matrix. We created a phylogenetic tree for use in imputation using ‘V.PhyloMaker2’ (Jin & Qian, 2022) using taxonomic information from AusTraits (Falster et al., 2021). To improve the predictions of gap-filling for leaf and seed traits, two additional predictive leaf traits (leaf width and leaf length) and four seed traits (fruit length, fruit width, seed length, and seed width) were also provided to the imputation method.

Baseline and future climate data

Baseline climate data for the period 1971-2000 was derived from Australian Gridded Climate Data (AGCD; (Australian Bureau of Meteorology, 2019). MAT and MAP were used as variables for analysis since they are known to influence plant community composition (MacArthur, 1972; Moles et al., 2014). The AGCD uses 5km grid cells which were aggregated using bilinear re-gridding to match the spatial resolution of future climate projections. Future climate projections were sourced from NARCliM version 2.0 (Di Virgilio et al., 2025), an ensemble of dynamically downscaled climate models at 10km resolution across NSW. The raw NARCliM data was bias-corrected against the AGCD data for the period between 1971-2000 to improve compatibility between datasets. For this, raw NARCliM data was changed from a rotated pole grid to rectilinear using bilinear re-gridding and then bias removed using a quantile mapping approach. For analysis, MAT and MAP projections for the year 2075 (i.e., the decade centred on 2070-2080) were used for Representative Concentration Pathway (RCP) 4.5, representing projected climate across NSW in 50 years under a moderate emissions scenario (Thomson et al., 2011).

Species climate niches

Realised climate niches were defined for each species within the floristic plot data by combining with locations of occurrences in the Atlas of Living Australia (ALA; Belbin et al., 2021). ALA records were cleaned by excluding all but herbarium specimen records, records prior to 1970, duplicates, records with spatial issues, and records from external territories. ALA data was extracted and manipulated using the 'galah' package (Westgate et al., 2025), and taxa were aligned to APC names and species not native anywhere in Australia were removed using 'APCalign' (Wenk et al., 2024a). Spatial coordinates of the remaining 2,035,172 records were matched with the AGCD (Australian Bureau of Meteorology, 2019) to extract data on climate conditions (MAT, MAP) across the species range (i.e., 'climate niches') for each species. We followed previous calculations of climate niches which used the 2nd and 98th percentiles of records to represent upper and lower climatic tolerances (Gallagher et al., 2019; Esperon-Rodriguez et al., 2022)(Figure1bii).

Calculating functional decay

Functional decay is calculated from the reduction of FD, as a result of filtering out species according to future climate projections. The FD of each floristic plot was measured using multidimensional hypervolumes, a common method in ecology to quantify niche space and functional trait space of plant communities (Blonder, 2018). Hypervolumes were constructed for each of the 32,996 floristics plots using original species lists and the six traits mentioned above. To simulate the effect of species loss from communities as a function of their realised climate niche limits under climate change, a second set of hypervolumes were constructed using a subset of species from each plot, filtering out species if the projected MAT and MAP of a plot location in 50 years under RCP 4.5 is expected to be outside of a species' current climate niche limits. For each pair of baseline and climate-filtered hypervolumes calculated for each floristic plot, the proportion of overlapping trait space was calculated to define the relative loss in functional trait space due to climate filtering (Figure 1c). In plots where changes in climate are projected to be minor, and/or where the species present have wide climate niches, fewer species are filtered, and the proportion of hypervolume overlap relative to the baseline hypervolume will be higher, meaning low functional decay. In cases where places are forecast to undergo major changes in climate that will exceed the present species' climate niches, more species will be filtered. If these filtered species represent unique or extreme parts of the community's trait space, then functional decay will be high, as defined by the much-reduced overlap between the baseline and climate-filtered hypervolumes.

To test the effect of climate change induced losses in FD relative to losses in species richness, the above protocol was replicated by randomly removing species from hypervolumes instead of selectively by climate niche. At each floristic plot, the same number of species were removed as defined in the structured climate niche removal, producing a hypervolume for each floristic plot with the same number of species as the climate reduced hypervolumes, that could be overlapped with the original hypervolumes. This procedure provided a form of null expectation of the effect of species loss for comparison.

All calculations of hypervolume overlaps were performed using the *dynRB* package (Junker et al., 2016). Since hypervolumes can't be calculated with fewer observations (species) than dimensions (traits), we performed a PCA on the six plant traits to reduce dimensionality to the first three principal components which captured 73% of variation. This method had little effect when compared with hypervolumes constructed with all six traits, but allowed more plots included (n = 32,996), and improved accuracy of overlaps from geometric scaling inherent to hypervolume approaches at higher dimensionality (Mammola, 2019) (S1). To calculate relative functional decay (i.e. functional decay relative to the proportion of species richness lost from climate filtering), the proportion of FD decline from climate filtering was divided by the proportion of species richness lost from the same filtering. Resulting values close to one represent where the proportion of species being removed from a plot translates to a similar proportion of reduction in the functional hypervolume. Values are bound at zero, where there hypervolumes do not decay from species removal due to either no species being vulnerable to climate changes in their present location or functional redundancy with the species being dropped. High values occur where the species being lost contribute disproportionately to FD, for example, a relative FD loss of two would mean that the proportion of species being lost results in twice as much functional decay.

Statistical Analyses

To examine the relationship between reductions in species richness and hypervolume overlap between baseline and climate-subset species lists, we fit generalized additive models (GAMs) using the *mgcv* package (Wood, 2017). The analysis included both growth form (woody or non-woody) and scenario (structured loss via climate margins or random loss) as explanatory factors. We compared two models: one with a single smooth term for loss of hypervolume overlap shared across scenarios and growth forms, and another that included separate smooths for each scenario and growth form combination, allowing the shape of the decline to differ between groups. Model improvement was evaluated using an F-test, comparing the two models to determine whether structured and random species removals exhibited significantly different trajectories of functional decay. All models were fitted using restricted maximum likelihood. To quantify overall magnitude differences between scenarios, we estimated marginal means from the full model using the emmeans package (Lenth & Piaskowski, 2025), providing average differences in overlap between structured and random removals for each growth form. We compared two models: one with a single smooth term for species loss and a parametric effect of growth form (woody vs non-woody), and another that included separate smooth terms for each growth form to allow the shape of the relationship to differ between groups. All data manipulation, visualisation, analysis, and mapping were done using R in the integrated development environment R Studio (R Core Team, 2025). Data and code used in this analysis can be accessed online here: https://figshare.com/s/d071cf38009f34856090.

Results

Future climate projections for floristics plots indicate, on average, 2.37°C (σ = 0.79) of warming in MAT and a decrease in MAP by 234.8 mm (σ = 143.9) by 2075 (Figure 2). The median percentage of species removed from plots due to local climates moving beyond species present day climate niches was 35.5%, although this varied greatly across the plots examined (interquartile range [IQR] = 37.2%; S2). Functional decay associated with this species loss was relatively lower, with the plot median being a 9.5% reduction (IQR = 11.7%) in hypervolume after species removal. If only woody species are considered, slightly higher proportion of species were removed (median = 36.5%; IQR = 38.9%) but resulted in much higher reductions in functional trait space (median = 17.1%; IQR =28.2%). In contrast, non-woody species experienced the same loss in species (median = 37.9%; IQR =39.5%) but retained higher overlaps in hypervolume space (median = 11.9%; IQR =20.3%).

Results of the GAMs reveal evidence that structured losses of species based on climate niches reduces FD faster than random species loss (Figure 2). Structured removal of species resulted in an additional 1.56% loss in hypervolume overlap compared to random species removal (Table 1). When comparing growth forms, woody communities exhibited the greatest declines in FD overlap, showing a 10.2% lower overlap than declines when all species are used, compared with the non-woody subset which was 4.2% lower. Finally, significant positive interaction terms suggest that subsets of specific growth forms slightly moderate the negative impact of structured loss. Smooth terms from the GAM further demonstrated strong nonlinear relationships between proportional species loss and hypervolume overlap, with distinct decay trajectories across growth forms. Woody and non-woody subsets showed distinct patterns of functional decay ($p < 0.001$). Together, these results provide evidence that climate-induced structural filtering accelerates functional decay, and that growth form can modulate the shapes the trajectory of these reductions.

Across NSW, most areas are projected to experience slower declines in FD than in species richness (Figure 4). Median relative FD loss was 0.34 across all plots, meaning that for every 1% loss in species richness, FD is expected to decline by roughly one-third as much (S2). When separated by growth form, woody species exhibited a higher median relative loss (0.54) than non-woody species (0.37), meaning that woody species contribute more to plant community's functional trait space and/or are at greater risk of changing climates. Woody FD appears at greatest risk of loss, relative to species loss along the Great Dividing Range and western slopes of NSW, and in the southwestern arid zone.

A similar pattern of slower functional decay relative to species richness was also observed across vegetation types (Table 2). All vegetation types had values below 1, with the highest being alpine vegetation (0.509), suggesting that the rate of functional decay will be just over half that of the projected decay of species richness. In contrast, wet sclerophyll forests showed the lowest relative loss (0.304 for grassy and 0.311 for shrubby types), indicative of higher functional redundancy. When considering growth forms separately, woody species in wooded communities (woodlands and dry and wet forests) had greater relative functional decay than non-woody species in the same communities. The opposite pattern was observed in vegetation types not dominated by woody vegetation (grasslands and chenopod shrublands), where non-woody species exhibited greater relative functional decay.

Discussion

Predicting the risk of FD decline in vegetation communities offers a potential guide for prioritising conservation under climate change. While functional redundancy remains a key mechanism to slow the decline in FD of plant communities, we show that calculating functional decay can reveal areas of vulnerability, when patterns of functional decay are decoupled regarding growth form or vary spatially. By quantifying where and which plant types are at risk of rapid FD decline, our framework allows conservation practitioners more resolution to prescribe interventions and identify areas of high climate risk. For example, these insights can inform proactive restoration strategies, such as the pre-emptive planting of slower maturing woody species in anticipation of future stress on these species, ensuring that the structural and functional membership of these ecosystems is maintained as climate-sensitive species decline in abundance (Di Sacco et al., 2021).

When designing strategies that account for functional decay, practitioners need to be strategic with prioritising species with unique trait combinations that are not predicted to become climate displaced in the future, but it will also be important to simultaneously protect for high functional redundancy. Due to the inherent uncertainty in climate models and predictions of species persistence, it is risky to manage for communities with only the minimum trait diversity preserved. Instead, restoring for redundancy acts as a critical insurance policy to maximise FD despite possible additional species loss. Measuring and maintaining areas with high functional redundancy will also be of vital importance for providing a buffer against rapid declines of FD and collapse of ecosystem processes as climate change affects community membership (Biggs et al., 2020).

Functional redundancy is a critical biological insurance policy that protects ecosystem stability by filling ecological niches with species that perform similar roles (Lawton & Brown, 1994). When species are at risk of being lost due to changing growing conditions, high redundancy allows the community to maintain its overall functionality because remaining species fulfill similar ecological niches that fill the gap of missing species (Fonseca & Ganade, 2001). This mechanism explains the observed and predicted decoupling of species richness loss and FD loss seen in this study as well as other Australian (Gallagher et al., 2013) and European (Thuiller et al., 2006) plant communities under future climate scenarios. However, the protection offered by redundancy is finite and depends heavily on the distribution of traits within the community (Reich et al., 2012). While functional redundancy can be greater with more species in a community, the extent and potential loss of FD, as measured using the hypervolume approach, is defined by the extreme trait values (Blonder, 2018). This means that safeguarding FD from changes in community composition should aim to retain the species with extreme trait values that contribute disproportionately to functional trait space and whose loss would result in the greatest changes in the functional properties of the community. By combining broad functional redundancy with the targeted protection of these extreme trait values, conservation managers can hedge against predictive uncertainty and ensure vital ecological niches remain filled.

Measuring the risk of FD loss from climate change provides a more mechanistic understanding of ecosystem vulnerability and offers specific insights that can better shape conservation and restoration priorities. Our finding that woody communities are more vulnerable to functional decay is particularly concerning given the life histories and adaptive capacities of these species. Since these woody species generally have longer generation times and slower growth rates (Petit & Hampe, 2006), their capacity for *in situ* adaptation as well as dispersal speed is lower than species that are faster to grow and reproduce (Corlett & Westcott, 2013). Consequently, there is a heightened likelihood of local extinctions when the rate of climatic shift is expected to exceed the current distributions of woody species. With climate risk of plant communities being biased toward a particular suite of species that contribute a distinct and important part of trait space to FD, conservation interventions may be needed to prevent decline (Nicotra et al., 2015).

One method for stemming the decline in FD would be to prioritise species with unique trait combinations that are vulnerable to decline in restoration efforts using climate-adjusted provenancing. This strategy involves strategically selecting genetic material for restoration from regions with growing conditions more similar to conditions projected in the future, as opposed to local provenancing which prioritises source material that is locally adapted to current growing conditions (Prober et al., 2015). This method bypasses some of the limitations of natural dispersal and adaptation, expediating the resilience of communities to novel climates (Di Sacco et al., 2021). Strategic selection of species and provenancing, can also reinforce resilience to disturbances that may be exacerbated by climate change by restoring traits that are robust to specific threats such as fire resprouting capacity or drought tolerance (Laughlin et al., 2017). Further, proactively planting species that contribute disproportionately to community FD that are also at the greatest risk of decline from changing growing conditions could help safeguard FD under climate change. In the case of woody species, which represent a suite of species at greater risk of climate induced reductions of FD, pre-emptive planting may buy time for slow to adapt species, while also slowing down FD decline. However, implementing these proactive strategies at scale faces significant practical hurdles, primarily, the availability and supply of seeds from diverse, climate-ready provenances remains a critical bottleneck for restoration practitioners (Andres et al., 2024). Addressing these supply-chain issues will be another essential challenge for successfully safeguarding the FD of ecosystems into the future.

Due to the uncertainty, and variability in how areas will experience climate changes, it is likely that the effect on plant persistence and growing conditions will also vary greatly. This means that while local extinctions are already widespread and predicted to continue (Wiens, 2016), many species will first experience declines in abundance, fitness, and recruitment as conditions shift away from a species' physiological optimum (Condit et al., 1996; Huang et al., 2025). Since the hypervolume approach utilised in this study is not weighted by abundance, the implications of these "declining but persisting" species may be overstated in terms of absolute trait space loss. However, even if species persist in lower densities, their reduced abundance will drive their functional contributions to ecosystem processes (Winfree et al., 2015), suggesting that our results still correctly identify the species most likely to contribute to functional decay.

The future trajectory of FD of plant communities will also be shaped by the colonisation of new species. Modelling species gain from climate induced community turnover adds additional levels of complexity because it requires accounting for varying dispersal capacities (Garcia et al., 2014) and the overcoming of biotic resistance from the resident community (Urban et al., 2012). Due to the lack of data on dispersal capacity for species at large scales as well as the highly stochastic and dynamic nature of species-specific interactions, modelling novel species under future scenarios can introduce considerable uncertainty. While there is already some work to predict climate induced range shifts in plant species (Bradley et al., 2024; Tomiolo & Ward, 2018), future research that integrates both climate filtering and dispersal-constrained colonisation models would provide a more holistic view of community reassembly. This approach has the potential to identify where the introduction of novel trait space might compensate for the decline or accelerate the change in functional membership of the original community.

Predicting potential losses in FD provides a critical macroecological tool for prioritising the selection of new protected areas. Many mechanisms exist to expand protected area networks, including the rapid growth of privately protected areas, offering diverse opportunities to safeguard biodiversity and improve the representativeness of conservation reserves (Alves-Pinto et al., 2021; Bingham et al., 2021). By integrating functional decay predictions into selection processes for expanding conservation areas, practitioners have more information to align investments to conservation priorities. Areas predicted to be highly resilient to climate induced functional decay, may be desirable for conservation as refugia for FD. These places would require minimal intervention and act as stable reservoirs of ecosystem processes and services and provide vital insurance populations that can support broader landscape stability (Keppel et al., 2012). Alternatively, areas identified as high risk of rapid FD loss may be eligible of significant proactive intervention to rebuild trait space and maximise redundancy. Identifying areas of high biodiversity and implementing protection measures is in line with conservation goals globally (Maxwell et al., 2020). FD represents a dimension of biodiversity that is being increasingly recognised as a key goal for safeguarding ecosystems and their vital functions (Pollock et al., 2020; Tobias et al., 2025). Whether used to identify resilient strongholds or vulnerable biodiversity, the addition of FD metrics offers additional resolution to inform conservation planning.

As climate change continues to reshape global vegetation, broadening our focus to include FD decline is essential for predicting the future of ecosystem stability. While functional redundancy offers a vital buffer against rapid loss of FD, our findings reveal patterns of vulnerability in certain growth forms and habitats. Integrating metrics like functional decay into conservation and restoration planning provides a more mechanistic lens to identify where ecosystem stability is most at risk and how this may be remedied. By prioritising the protection of FD and implementing proactive strategies, practitioners can safeguard the essential ecological processes that sustain biodiversity in a rapidly changing climate.

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Declaration of generative AI technologies in the manuscript preparation process

During the preparation of this work the authors used Gemini 3.1 Pro for editing text and code. After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the content of the published article.

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Figures

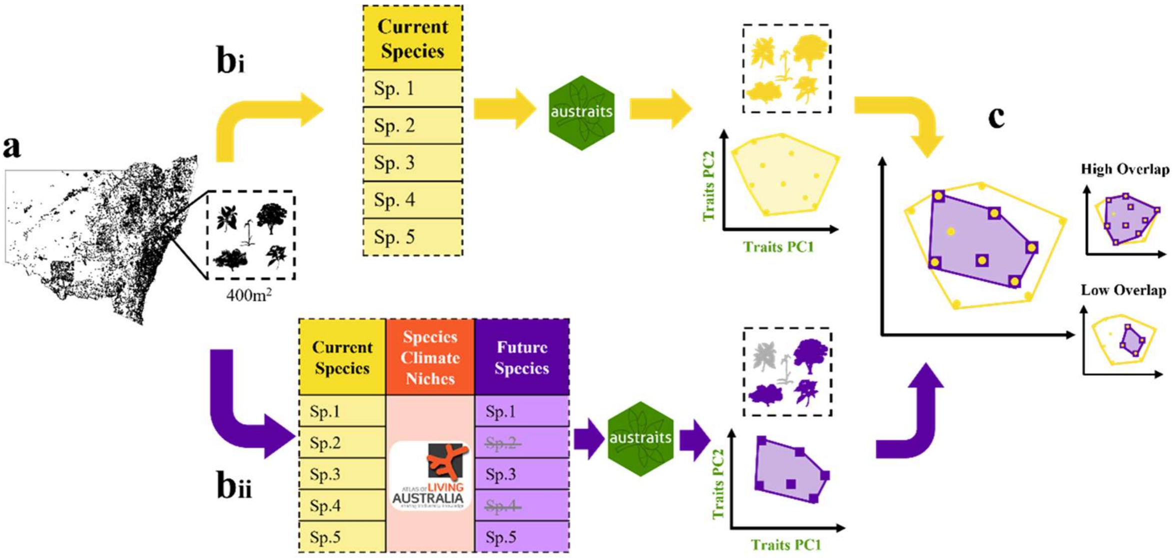

Figure 1. (a) Species lists from 32,996 floristic survey plots were (bi) joined with gap-filled trait information from AusTraits (plant height, leaf area, leaf mass per area, leaf nitrogen per dry mass, seed mass, and stem density; Falster et al., 2021) to calculate hypervolumes for each survey plot. (bii) Future hypervolumes were created by defining climate niches for each species based on their current extents of occurrence from the Atlas of living Australia (Belbin et al., 2021) and filtered out of plot level lists if a species is not expected to remain within its current climate niche in 2075 under a climate change scenario of RCP 4.5. (c) The proportion of overlap of current and future hypervolumes was measured for each floristics plot using all species, and again separately for woody, and non-woody species.

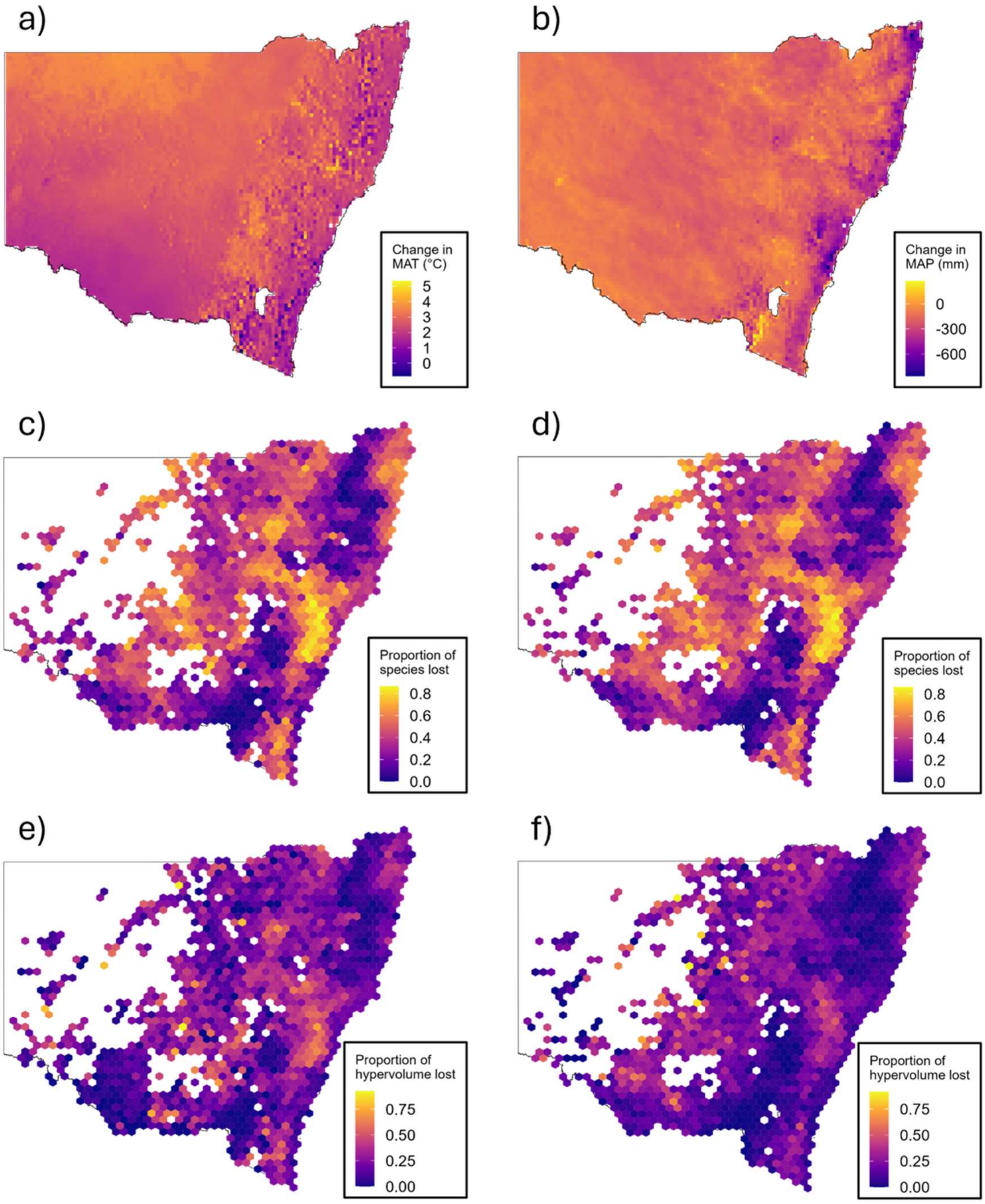

Figure 2. Maps of change in New South Wales between current and 2075 climate under RCP4.5. Maps (a) and (b) respectively show forecasted change in mean annual temperature at sea level and change in mean annual precipitation based on NARCliM 2.0 (Di Virgilio et al., 2025). Map (c) shows the proportion of woody species, and (d) non-woody species that have been removed from each floristics plot based on species climate niches relative to each plot's future climate. Maps (e) and (f) show the proportions of woody and non-woody community hypervolume lost (functional decay) from the same species removal. Maps C-F display values for each floristics plot aggregated into hexbins and coloured by mean values.

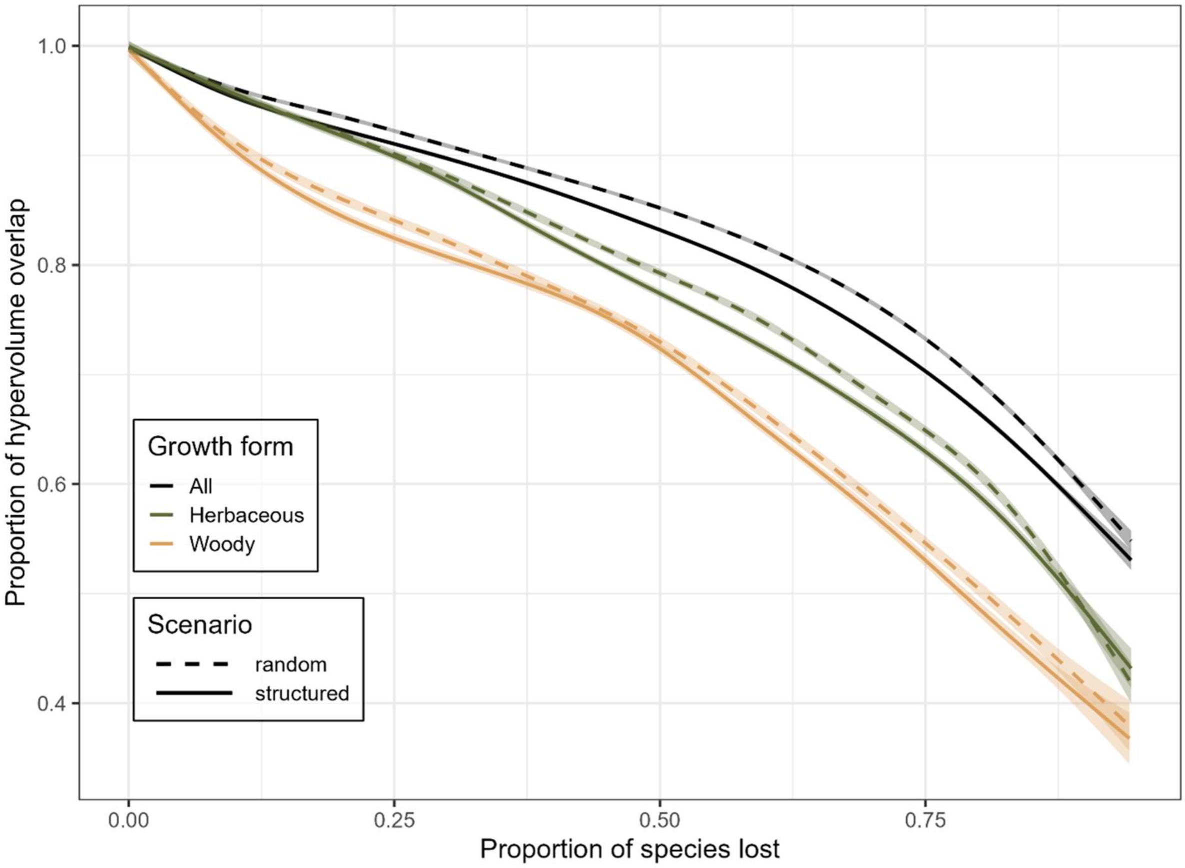

Figure 3. Decline in plot level functional diversity from hypervolume overlaps resulting from species removal based on 2073 RCP4.5 climate projections for plot locations relative to species' current climate niches. Curves represent smoothed trends from generalised additive models (GAMs) for 32,996 plots for all species (black), only woody species (brown), and only non-woody species (green). Models were fit for reductions in hypervolumes via structured removal of species based on their climate niches (solid), and separately via random species loss (dashed). Ribbons represent 95% confidence intervals.

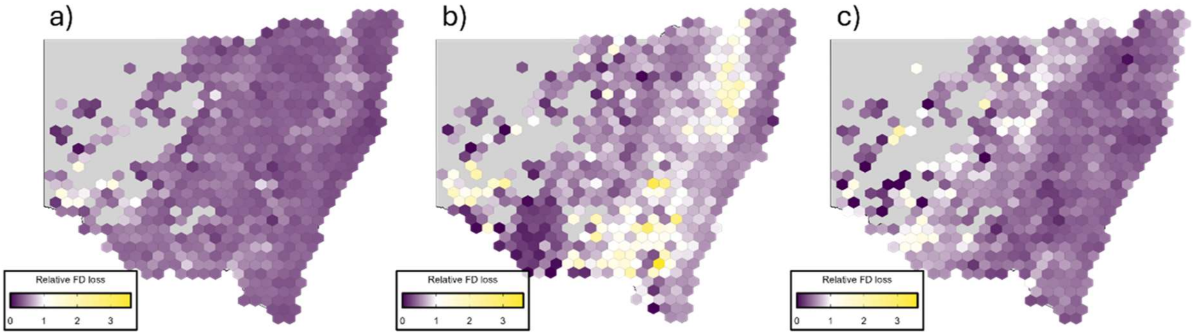

Figure 2. Maps of proportion of functional diversity decline relative to proportion of species decline after climate filtering. Values of one represent areas where the proportion of species removed due to projected climate conditions in 2075 translates to the same proportion of functional decay in the original hypervolume. Values close to zero represent areas with high functional redundancy that are protected from rapid functional decay when species are lost. Values above one are areas where the rate of functional decay outpaces the rate of species decay, i.e. a value of two means that for each proportion of species removed from climate filtering, two times as much functional trait space is lost. Maps are produced for the relative decay of all species (a), only woody species (b), and only non-woody species (c). Relative functional diversity loss values for all 32,996 plots have been aggregated using the mean values of each hexbin.

Tables

Table 1. Summary of parametric terms for the Generalized Additive Model (GAM) of hypervolume overlap after structured and random removal of species from functional trait space. Estimates represent the linear offsets for decline scenarios (random/structured by climate niche) and growth forms (all species, woody only, and non-woody only) relative to baseline (random/all species).

Parameters	**Estimate**	**St. error**	**t-statistic**	**p-value**
Intercept	0.875	0.0007	1300.284	<0.001
Structured decline	-0.016	0.001	-16.424	<0.001
Non-woody only	-0.041	0.001	-42.964	<0.001
Woody only	-0.102	0.001	-104.023	<0.001
Structured:non-woody interaction	0.005	0.0014	4.003	<0.001
Structured:woody interaction	0.006	0.0014	4.097	<0.001

Table 2. Median and interquartile ranges (IQR) of the relative functional decay (proportion of functional diversity decline relative to proportion of species decline after climate filtering) for vegetation formations across New South Wales. Vegetation classification from (Department of Planning and Environment, 2017). Relative functional diversity loss values for each formation are presented for all species, only woody species, and non-woody species.

Vegetation Formation	**median (all species)**	**IQR (all species)**	**median (woody)**	**IQR (woody)**	**median (non-woody)**	**IQR (non-woody)**	**n plots**
Alpine complex	0.509	0.411	2.184	2.914	0.281	0.346	21
Arid shrublands (Acacia)	0.357	0.254	0.460	0.388	0.558	0.519	237
Arid shrublands (Chenopod)	0.377	0.301	0.055	0.411	0.557	0.648	427
Dry sclerophyll forests (Grassy)	0.328	0.204	0.646	0.562	0.290	0.292	4379
Dry sclerophyll forests (Shrubby)	0.341	0.200	0.560	0.436	0.358	0.365	7356
Forested wetlands	0.359	0.225	0.379	0.760	0.441	0.389	1760
Freshwater wetlands	0.372	0.259	0.506	0.852	0.422	0.402	1289
Grasslands	0.405	0.248	0.030	0.847	0.475	0.349	1235
Grassy woodlands	0.365	0.233	0.751	0.847	0.326	0.314	5629
Heathlands	0.308	0.175	0.461	0.428	0.378	0.473	1046
Rainforests	0.321	0.200	0.436	0.361	0.409	0.445	1471
Saline wetlands	0.416	0.242	0.103	0.541	0.585	0.540	214
Semi-arid woodlands (Grassy)	0.377	0.268	0.423	0.546	0.569	0.574	1246
Semi-arid woodlands (Shrubby)	0.406	0.243	0.558	0.524	0.500	0.456	1376
Wet sclerophyll forests (Grassy)	0.304	0.213	0.481	0.464	0.297	0.317	2297
Wet sclerophyll forests (Shrubby)	0.311	0.204	0.480	0.398	0.352	0.331	2335

Supplementary material for: *Predicting decay in functional diversity in plant communities under future climates*

S1 Comparison of results from decline in hypervolume overlap from species removal between raw traits (leaf area, leaf mass per area, leaf nitrogen content, plant height, seed mass, and stem density) and the first three principal components of a PCA of these traits.

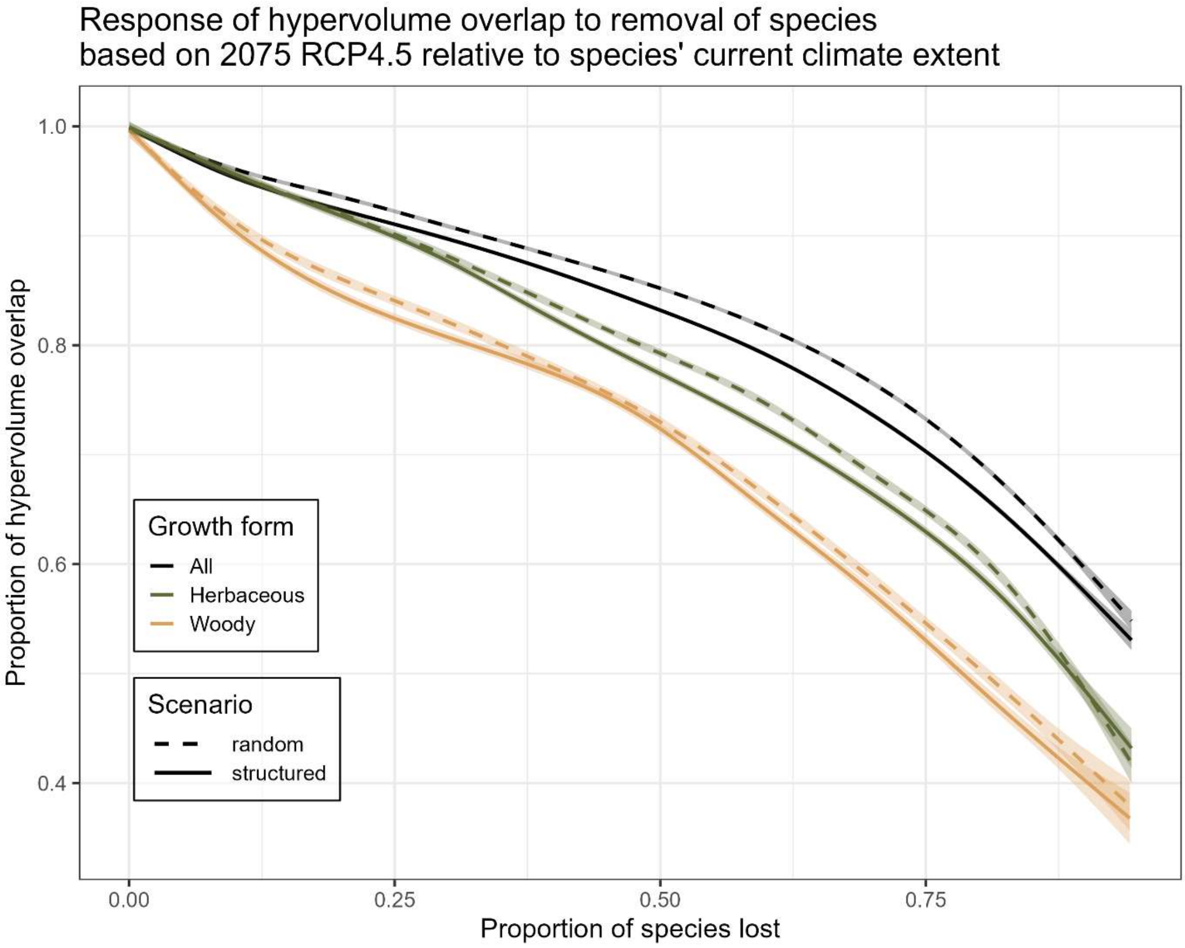

Figure 1. Figure 3 from main paper. Decline in plot level functional diversity from hypervolume overlaps resulting from species removal using first three principal components of six functional traits.

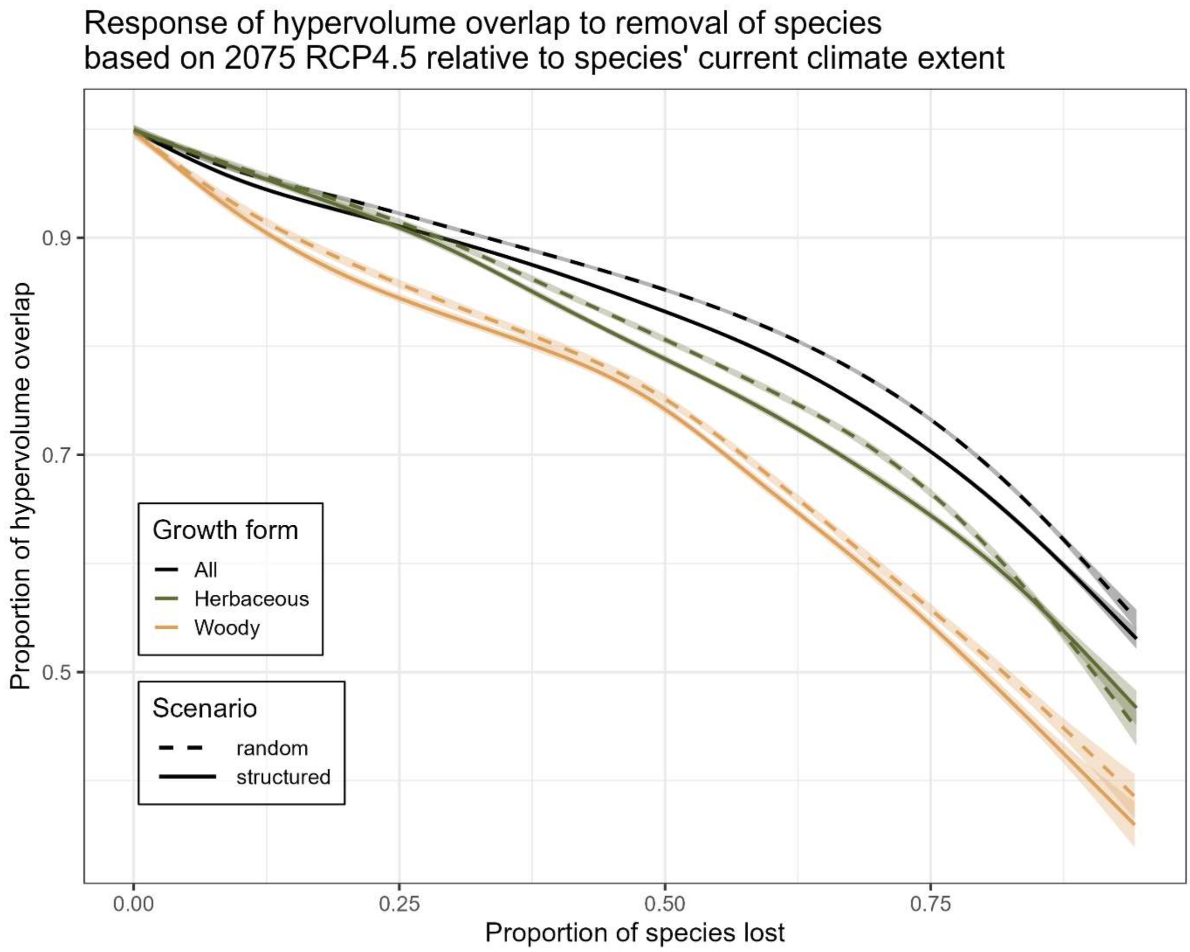

Figure 2. Decline in plot level functional diversity from hypervolume overlaps resulting from species removal using all six log transformed trait values without reduction via PCA. This method excludes vegetation surveys with fewer than seven species present since the number of observations for calculating a hypervolume must be greater than the number of dimensions.

S2 Histograms of floristic plot network's projected climate change, change-induced number of species removed, and subsequent functional diversity decline.

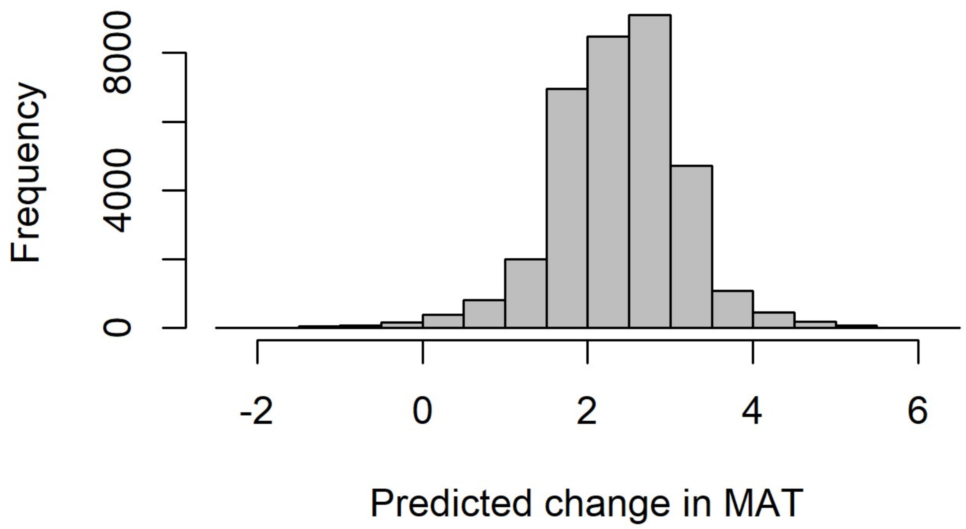

Figure 3. Projected change in mean annual temperature for each plot according to NARCliM 2.0 (Di Virgilio et al., 2025).

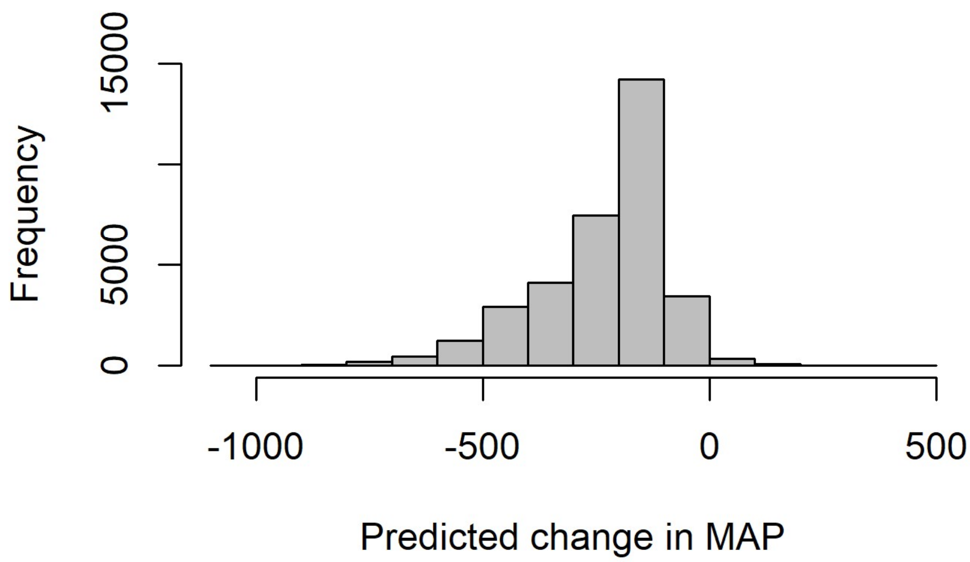

Figure 4. Projected change in mean annual precipitation for each plot according to NARCliM 2.0 (Di Virgilio et al., 2025).

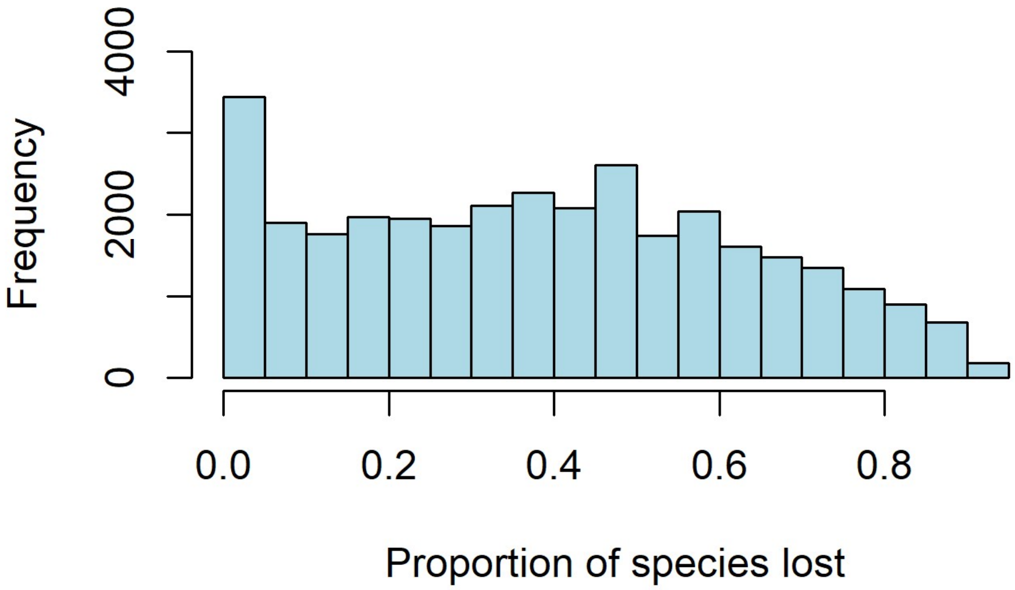

Figure 5. Proportion of species removed at each floristics plot after filtering by species climate niche relative to projected climate at that plot location.

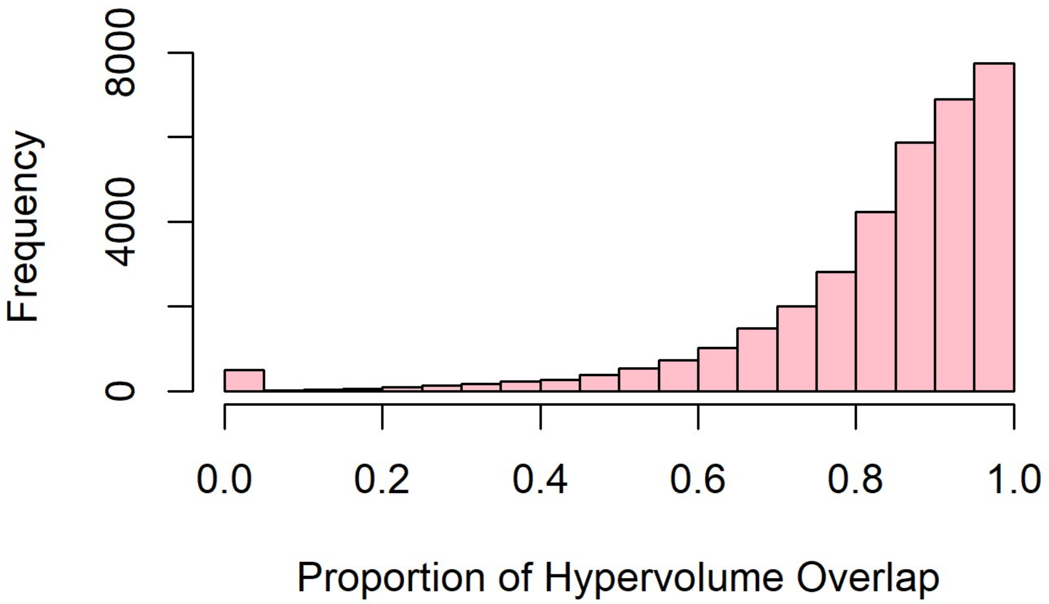

Figure 6. Proportion of overlap between the new climate-filtered trait space hypervolume, and original hypervolume for each floristics plot using all species. Values of 1 represent plots that retain full

overlap and therefore lose no functional diversity after removal of species that will no longer be within their current climate niches. Values of 0 represent plots which will have all species removed from the climate filtering process and therefore lose all functional diversity.

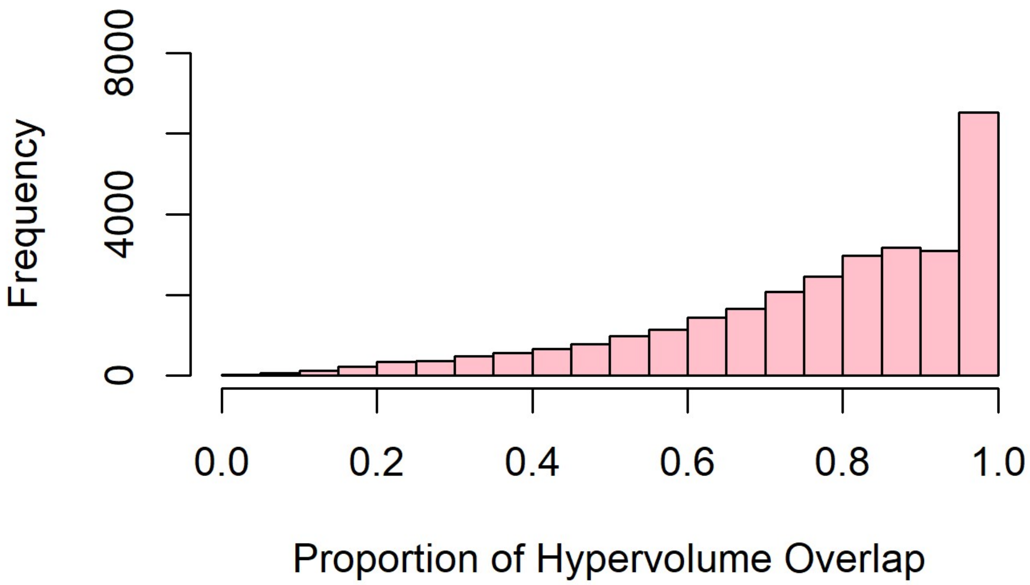

Figure 7. Proportion of overlap between the new climate-filtered trait space hypervolume, and original hypervolume for each floristics plot using woody species only. Values of 1 represent plots that retain full overlap and therefore lose no functional diversity after removal of species that will no longer be within their current climate niches. Values of 0 represent plots which will have all species removed from the climate filtering process and therefore lose all functional diversity.

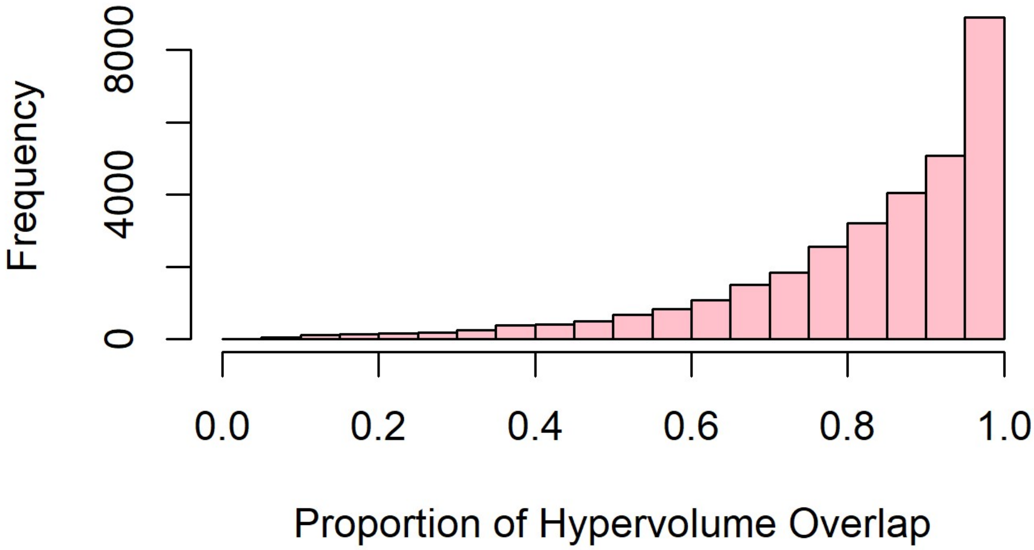

Figure 8. Proportion of overlap between the new climate-filtered trait space hypervolume, and original hypervolume for each floristics plot using non-woody species only. Values of 1 represent plots that retain full overlap and therefore lose no functional diversity after removal of species that will no longer be within their current climate niches. Values of 0 represent plots which will have all species removed from the climate filtering process and therefore lose all functional diversity.