University of Southampton

Faculty of Environmental and Life Sciences

School of Biological Sciences

The Potential Impacts of Climate Change on the Extent, Distribution and Functional Diversity of the Montane-Specialist Tree Community in Mesoamerica

by

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Abstract

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Tropical montane forests (TMFs) are ecosystems that, despite their small global area, have rich biodiversity, provide essential ecosystem services, and are of cultural significance. Their restricted distribution makes them highly vulnerable to anthropogenic pressures, including climate change. Their high ecological variability in remote areas with complex terrain makes their study challenging and costly, which may impede the design and implementation of conservation measures to preserve their biota and functions. In Chapter 2, I systematically review and critically assess the research on global TMFs under ongoing climate change, revealing some important methodological, geographical, and thematic biases: trees are the most studied group, and the Neotropics is the most studied region. Although these biases should be overcome, they also constitute an opportunity to direct efforts within well-studied areas that could serve as proxies for analogous tropical montane regions while local research is underway. One such area of opportunity is the montane-specialist tree community in Mesoamerica. The Neotropics are a hotspot of montane biota and of rare and range-restricted species, yet there is a dearth of information on the impacts on the montane-specialist tree subcommunity of its forests. Given the disproportionate role that rare and range-restricted species play in terms of richness, functional diversity, and support of vulnerable ecosystem functions, it is critical to forecast their potential responses to climate change. In Chapter 3, I use species distribution modelling to project the range and distribution of 272 Mesoamerican montane-specialist tree species under current and future climatic conditions by the end of the century, showing that this subcommunity will undergo significant losses of suitable habitat, but with some relicts remaining even under a severe climate change scenario. In Chapter 4, I incorporate the modelling results into a functional trait analysis to identify spatial patterns of functional diversity throughout the Mesoamerican region, their potential carbon stock capacity in aboveground biomass, and how these are predicted to shift in response to climate change. I found that despite the projected losses in suitable habitat, the functional diversity and aboveground biomass spatial patterns will persist, although they will be less conspicuous. These findings highlight the contribution of the montane-specialist trees' role in providing functional redundancy and stability to their communities and call for urgent action to complement existing conservation measures with active efforts such as seed banks and assisted migration programmes for montane-specialist trees globally. Overall, this thesis establishes a framework for future research and policy to protect montane-specialist trees in Mesoamerica, highlighting the need for integrated evidence, cross-disciplinary methods, and international collaboration to address climate change challenges.

Table of Contents

Abstract ... 5

Table of Contents ... 6

Table of Tables ... 10

Table of Figures ... 11

Research Thesis: Declaration of Authorship ... 14

Author contributions ... 15

Acknowledgements ... 16

Definitions and Abbreviations ... 17

Chapter 1. General Introduction ... 18

1.1 Overview of Tropical Montane Forests ... 18

1.1.1 Definition and Global Extent ... 18

1.1.2 General characteristics of TMFs and vulnerability ... 19

1.2 Introduction to climate change ... 21

1.2.1 Definition and magnitude ... 21

1.2.2 General impacts of climate change on biodiversity ... 21

1.2.3 Plants and forests in relation to climate change ... 22

1.3 Tropical Montane Forests in Face of Climate Change ... 23

1.3.1 General impacts on tropical montane flora ... 23

1.3.2 The significance of rare and montane-specialist species in Mesoamerica ... 25

1.4 Overview of the Thesis ... 26

1.4.1 General Aim and Objectives ... 26

1.4.2 Structure of the thesis ... 26

Chapter 2. Impacts of anthropogenic climate change on tropical montane forests: an appraisal of the evidence ... 29

2.1 Abstract 29

2.2 Introduction 30

2.3 Methods 32

2.3.1 Systematic Review 32

2.3.2 Evaluation of evidence quality 34

2.4 Results 38

2.4.1 Distribution of studies 38

2.4.2 Trends in the literature 41

2.4.3 Assessment of the evidence on the impacts of climate change in TMFs 41

2.4.4 Evidence-based synthesis of impacts of climate change on TMFs 44

2.5 Discussion 50

2.5.1 Methodological gaps and opportunities 51

2.5.2 Geographical and taxonomic gaps and opportunities 53

2.5.3 Thematic gaps and opportunities 53

2.6 Recent work 55

2.7 Conclusions 55

Chapter 3. Distributional Responses of Mesoamerican Tropical Montane Tree Communities under Climate Change 57

3.1 Abstract 57

3.2 Introduction 57

3.3 Methods 60

3.3.1 Study area and data sources 60

3.3.2 Species distribution modelling 62

3.3.3 Post-SDM Analysis 63

3.4 Results 65

3.4.1 Characterisation of the Mesoamerican TMF tree communities 65

3.4.2 Suitable area projection of the Mesoamerican montane-specialist tree community and predicted impacts of climate change 67

3.5 Discussion 70

3.5.1 The Mesoamerican montane-specialist tree community 71

3.5.2 Impacts of climate change on Mesoamerican montane-specialist trees 73

3.5.3 Conservation management under global change 75

3.6 Conclusions 77

Chapter 4. Functional Traits of the Montane-Specialist Tree Community of Mesoamerica in the Present and under a Changing Climate 79

4.1 Abstract 79

4.2 Introduction 79

4.3 Methods 83

4.3.1 Study area, data sources and missing data imputation 83

4.3.2 Analysis of spatial patterns of functional traits and their influence on species' distributional changes under climate change 84

4.4 Results 85

4.4.1 Current functional diversity patterns 85

4.4.1 PCA of the montane-specialist trees functional traits 89

4.4.2 Future functional diversity patterns 90

4.4.3 Functional traits' influence on projected habitat suitability loss under climate change 93

4.5 Discussion 94

4.5.1 Current distributional patterns of functional diversity 94

4.5.2 Phylogenetic divergence of functional diversity 95

4.5.3 Predicted effects on functional traits and aboveground biomass under climate change 97

4.5.4 Caveats and limitations 99

4.6 Conclusions 100

Chapter 5. General Discussion 101

5.1 Significance of biases in existing research on impacts of climate change on tropical montane forests 101

5.2 Significance of the predicted distributional changes of Mesoamerican montane-specialist trees in response to climate change 102

5.3 Significance of the predicted functional diversity shifts of Mesoamerican montane-specialist trees in response to climate change 105

5.4 Implications for the direction of future research 106

5.5 Conclusions 108

References 110

Appendices 137

Table of Supplementary Tables 137

Table of Supplementary Figures 139

Appendix A. Supplementary Information for Chapter 2 140

Appendix B. Supplementary Information for Chapter 3 157

Appendix C. Supplementary Information for Chapter 4 181

Table of Tables

Table 1.1 Logical Framework of the project "The potential impacts of Climate Change on the Extent, Distribution and Functional Diversity of the Montane-Specialist Tree Community in Mesoamerica". 28

Table 2.1 Classification of studies according to Level of Evidence adapted from Mupepele et al. (2016). Modelling studies included in LoE2a-c correspond roughly to the silver, bronze and deficient levels proposed by Araújo et al. (2019), since their *gold standard* is unattainable for TMFs studies in relation to climate change. 36

Table 4.1 Summary statistics of the community-weighted means of functional traits and aboveground biomass of the montane-specialist tree species throughout the study area on a pixel-by-pixel basis, under current (1981–2010 baseline) and future (2071–2100) climate conditions under RCP8.5. Change refers to the average difference per site (pixel) between current and future raster layers, so they do not correspond to a simple subtraction of the bulk current and future values. 86

Table of Figures

Figure 2.1 Systematic search protocol. The final count includes all relevant studies from 1994 to 2021.......... 32

Figure 2.2 Schematic representation of the adapted and combined evidence assessment used in this study. 35

Figure 2.3 Geographical distribution of retrieved studies by country or territory. The number of studies is indicated by the colour scale from light blue to purple. In orange: countries with TMF cover for which no relevant studies were found. In yellow: countries with contested TMF cover (see Karger et al., 2021; Los et al., 2019; Mulligan, 2010).......... 38

Figure 2.4 Number of studies published from 1994 to 2021 on the effects of climate change on tropical and subtropical montane forests for different (A) geographic regions, (B) effect types, (C) ecological levels, and (D) taxonomic groups. 39

Figure 2.5 Distribution of reviewed studies according to levels of evidence hierarchy. (A) Percentage of studies by level of evidence, with solid-coloured portions indicating the frequency of modelling studies within each level; and levels of evidence of reviewed studies by (B) research topic, (C) ecological level, and (D) taxonomic group. 43

Figure 3.1 The study area. The altitude categories are based on historically reported elevational limits of TMFs in Mesoamerica, typically located between 700 and 2,700 m asl, with rare occurrences as low as 700 m asl, and as high as 3000 m asl (Ressl and Morales, 2008). The distribution of tropical montane forests is shown in yellow, based on the combined estimates by Villaseñor (2010) and Los et al. (2021). Lake Cocibolca (in light blue) creates a latitudinal discontinuity for montane habitats, but Costa Rica and Panama are included nonetheless as part of the Mesoamerican bioregion.......... 60

Figure 3.2 Characterisation of Mesoamerican montane-specialist tree subcommunity (N = 272) by latitudinal (N = north, N – C = north-central, C = central, C – S = central-south, S = south, and W = widespread) and altitudinal (VH = very high, H = high, MH = mid-high, M = mid, ML = mid-low, L = low, and VL = very low; see Table S6) classes. Darker colour indicates projected suitable climatic conditions for higher number of montane-specialist tree species. 66

Figure 3.3 Projected climatically suitable habitat extent for the 272 montane-specialist trees by number of species under current (1981–2010 baseline) and future (2071–2100) conditions under a severe climate change scenario RCP8.5. 67

Figure 3.4 Number of montane-specialist tree species (richness) projected to have suitable habitat with current (A) and future (2071–2100) climatic conditions under RCP8.5 (B); change in climatic suitability from current to future conditions (C); and species turnover estimated with the Jaccard similarity index from current to future conditions (D). 69

Figure 3.5 Interaction of latitudinal and altitudinal classes of Mesoamerican MST species on projected range losses by the end of the century under RCP8.5. Boxes indicate the interquartile range (25^{th} and 75^{th} percentiles) and the whiskers extend to the 10^{th} and 90^{th} percentiles. 70

Figure 4.1 Heatmap of Pearson correlation coefficients of the elevational and latitudinal gradients and the CWMs of functional traits (H, DBH, WD, LA and LMA) and AGB_P of the species projected to have suitable area per pixel under current (1981–2010 baseline) and future (2071–2100) climate conditions (RCP8.5 scenario). Darker colours indicate stronger correlations. p-values < 0.0001 in all cases. 86

Figure 4.2 Community-weighted means of tree height (A and B) and leaf area (C and D) of the montane-specialist tree species projected to be present under current (A and C) and future (B and D) climatic conditions. 88

Figure 4.3 PCA of five functional traits of montane-specialist tree species by most frequent taxonomic groups. The directions and lengths of the loadings indicate that most of the variation in PC1 aligns with adult tree size-related variables (tree height and DBH), whereas PC2 is more aligned with leaf traits and biomass (leaf area, LMA and to a lesser extent WD). Height and DBH remain closely aligned when plotting other components. Most Conifers are outliers, occupying a distinct functional space along the PC1 separated from most Angiosperm families. Other outliers, belonging to different Angiosperm lineages (see Table S4), are show with numbers: (1) *Meliosma alba* [Clade Basal Eudicots, Proteales, Sabiaceae], (2) *Hypericum irazuense* [Clade Rosid I/Fabids, Malpighiales, Hypericaceae], (3) *Juglans pyriformis* [N-fixing Clade, Fagales, Juglandaceae], (4) *Ulmus Mexicana* [N-fixing Clade, Rosales, Ulmaceae], (5) *Chiranthodendron pentadactylon* [Clade

Rosid II/Malvids, Malvales, Malvaceae], (6) *Clethra suaveolens* [Clade Basal Asterids, Ericales, Clethraceae], (7) *Comarostaphylis arbutoides* and (8) *C. longifolia* [Clade Basal ASterids, Ericales, Ericaceae], and (9) *Buddleja nitida* [Clade Asterid I/Lamids, Lamiales, Scrophulariaceae]. 89

Figure 4.4 Community-weighted mean (CWM) of potential aboveground biomass (AGB_P) based on the species projected to have suitable conditions under current climate conditions (A), projected changes in CWM AGB_P under future with respect to current climatic conditions (B), functional dispersion of the 272 montane-specialist tree species under current conditions (C) and projected changes in functional dispersion under future with respect to future climatic conditions (D). 92

Figure 4.5 Relationship between adult tree height and log-transformed total losses projected suitable extent under future climate conditions relative to current conditions by taxonomic group and latitudinal class. 94

Research Thesis: Declaration of Authorship

Erik Omar Mata Guel

The Potential Impacts of Climate Change on the Extent, Distribution and Functional Diversity of the Montane-Specialist Tree Community in Mesoamerica.

I declare that this thesis and the work presented in it are my own and has been generated by me as the result of my own original research.

I confirm that:

1. This work was done wholly or mainly while in candidature for a research degree at this University;

2. Where any part of this thesis has previously been submitted for a degree or any other qualification at this University or any other institution, this has been clearly stated;

3. Where I have consulted the published work of others, this is always clearly attributed;

4. Where I have quoted from the work of others, the source is always given. With the exception of such quotations, this thesis is entirely my own work;

5. I have acknowledged all main sources of help;

6. Where the thesis is based on work done by myself jointly with others, I have made clear exactly what was done by others and what I have contributed myself;

7. Part of this work have been published as:

Mata-Guel, E.O., Soh, M.C., Butler, C.W., Morris, R.J., Razgour, O. and Peh, K.S.H., 2023. Impacts of anthropogenic climate change on tropical montane forests: an appraisal of the evidence. *Biological Reviews*, *98*(4), pp.1200-1224.

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Chapter 1: I compiled, synthesised and structured the information to write this chapter with guidance and feedback from my supervisors Dr Kelvin Peh, Dr Orly Razgour and Dr Rebecca J Morris.

Chapter 2: I conceptualised this chapter and study design with guidance of my supervisors Dr Kelvin Peh, Dr Orly Razgour and Dr Rebecca j. Morris. A pilot literature search was done by Connor W. Butler, and subsequently, I carried out the bulk of the literature review, its processing and data analysis. All listed co-authors contributed with the revisions that led to the publication.

Chapter 3: this chapter was conceptualised jointly by me and my supervisors Dr Kelvin Peh and Dr Orly Razgour. I carried out the data collection, data processing and data analysis, with major guidance from Dr Orly Razgour, who helped me with the development of the modelling protocol. I received help from Dr Roberto Novella with the automation process for large data analysis and from PhD candidate Penelope Fialas with beta-diversity calculations. The final version was revised by my three supervisors Dr Kelvin Peh, Dr Orly Razgour and Dr Rebecca j. Morris.

Chapter 4: I conceptualised this chapter jointly with my supervisor Dr Kelvin Peh, and received major support and feedback from him and my supervisors Dr Orly Razgour and Dr Rebecca J Morris. I carried out all the data collection, processing, analysis and writing up. I also received help from PhD candidate Penelope Fialas with the calculation of functional diversity indices and community-weighted means.

Chapter 5: I carried out the synthesis of the findings of the other chapters to elaborate upon their significance in the wider ecological context to write up this chapter, with feedback from my supervisors Dr Kelvin Peh, Dr Orly Razgour and Dr Rebecca J Morris.

I carried out the final revision and corrections, as instructed by my examiners Dr Patrick Doncaster and Dr Aida Cuni-Sánchez.

During the execution of this PhD, I contributed with revisions to another project independent from this thesis, which led to the following publication:

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Definitions and Abbreviations

AGB: Aboveground biomass

ANN: Artificial neural networks

BIEN: Botanical Information and Ecological Network; https://bien.nceas.ucsb.edu/bien/

CWM: Community-weighted mean

DBH: Diameter at breast height

FD: Functional diversity

FDis: Functional dispersion

GAM: Generalised additive models

GBIF: Global Biodiversity Information Facility; https://www.gbif.org/

GBM: Generalised boosted models

IUCN: International Union for Conservation of Nature; https://iucn.org/

LMA: Leaf mass per area

MaxEnt: Maximum entropy

PC: Principal component

PCA: Principal component analysis

SDM: Species Distribution Models

TMCF: Tropical montane cloud forest

TMF: Tropical montane forest

TRY: Plant Trait Database; https://www.try-db.org/TryWeb/Home.php

WD: Wood density

Chapter 1. General Introduction

1.1 Overview of Tropical Montane Forests

1.1.1 Definition and Global Extent

Tropical montane forests (TMF) are an ecosystem type that occurs within the tropical (0° to 23.4° N and S) and subtropical (23.4° to 30° N and S) latitudinal boundaries (Corlett, 2013). They are often constrained to narrow altitudinal belts enveloped by trade-wind orographic clouds that result in a typically uneven canopy persistently exposed to mist or fog, and with an extraordinary abundance and diversity of epiphytes (Comarazamy and González, 2011; Foster, 2001; Hamilton, 1995; Loope and Giambelluca, 1998; Richter, 2008; Still et al., 1999). It is due to these defining features that TMF if commonly used as a synonym of cloud forests, mist forests, mossy forests, among many more alternative names, especially in Latin America (Brown and Kappelle, 2001), where most studies have been conducted (Laurance et al., 2011; Soh et al., 2019).

It should thus be noted that the nomenclature of TMFs is ambiguous (Bruijnzeel, 2001; Shi and Zhu, 2009). A clear distinction of tropical montane cloud forests (TMCF) as a subcategory of TMFs has been proposed (Bruijnzeel et al., 2011a) and a classification of TMFs based on elevation, fog incidence and canopy structure has existed for over two decades (Bruijnzeel and Hamilton, 2000). However, Bruijnzeel et al. (2011b) posit that TMCFs constitute a small fraction (6.6%) of the total TMF global area, without clarifying what vegetation type or types constitute the remaining area. More recent reviews on the environmental determinants of these forests' traits and distribution use TMF, TMCF, montane forest and cloud forest interchangeably (e.g. Dalling et al., 2016; Fahey et al., 2016; Oliveira et al., 2014). For the sake of consistency, henceforth TMF will be the preferred term in this document, encompassing all the alternative names and subtypes encountered in previous publications.

In addition to the ambiguity in the definition of TMFs, efforts to accurately quantify their extent and map their distribution have resulted in wildly varying estimates. A quantitative "hydro-climatic" definition of TMFs or "cloud-affected forests," estimated 2.21 M km^2 of TMF globally (Mulligan, 2010), and a pantropical land cover classification based on topographic and satellite vegetation data estimated 2.12 M km^2 of TMF (Los et al., 2021). These studies are an order of magnitude greater than the previously estimated 380,000 km^2 of 'potential area' and 215,000 km^2 of 'actual' TMF area worldwide that were based on an elevation approach rather than a hydro-climatic one (Bruijnzeel et al., 2010; Oliveira et al., 2014). Similarly, a study using a combination of distribution modelling, cloud cover metrics and climatic variables estimated ~624, 200 km^2 of TMCF in 2001 with an average annual rate of loss of ~2.4%, i.e., ~612,000 km^2 left by 2018 (Karger et al., 2021). At a more regional scale, Helmer et al. (2019) used cloud immersion and frost frequency metrics to calculate 401,370 km^2 of TMCF in the Neotropics, which is consistent with global estimates of ~2 M km^2 (Los et al., 2021; Mulligan, 2010).

The comparatively smaller estimations are often the result of a very restrictive TMF definition, but many tropical mountains in fact support forests that do not fit into the narrow definition of 'cloud-affected' systems (Martin and Bellingham, 2016). For example, Mexico, which harbours the most intensively studied TMFs globally (Soh et al., 2019), has historically employed in its national inventories the highly restrictive "mountain mesophyll forest" category (INEGI, 2024; www.inegi.org.mx/temas/usosuelo/#mapas). This category covers 1% (~18,500 km^2) of the country, and only 0.4% if only primary vegetation is considered (Gual-Díaz and Rendón-Correa, 2014). However, employing a the broader "mountain humid forest" category (Villaseñor, 2010), the estimate increases to ~139,400 km^2 (7% of Mexico's territory) (Cruz-Cárdenas et al., 2012), with the aim of informing conservation efforts that would otherwise overlook areas with potential high diversity and endemism.

1.1.2 General characteristics of TMFs and vulnerability

TMFs typically occur at altitudes from 1,000 to 3,500 m asl (Scatena et al., 2010). With topography being a defining feature, the general characteristics of TMFs are as variable as the land on which they grow. The presence and characteristics of arboreal vegetation on tropical mountains responds to the physical terrain (i.e., size of the mountain, distance to the ocean, slope, aspect and ruggedness) and the associated environmental conditions, such as temperature, precipitation, solar irradiance, cloud cover, among others. The gradual change in environmental conditions along mountain slopes influences the type of vegetation that grows at different elevations (Bruijnzeel and Hamilton, 2000; Scatena et al., 2010). The combination of lapse rate (the rate at which atmospheric temperature decreases with an increase in altitude) with humidity (including precipitation and ground-level clouds) generally allows distinguishing three or four distinct TMFs subtypes, albeit with a high degree of variation between montane regions (Bruijnzeel et al., 2011a; Scatena et al., 2010):

- *Lower montane forest*: from ~1,000 (usually starting where average temperature drops below 18 °C) to ~1,500 m asl; multilayered canopy of 15–30 m with emergent individuals up to 37 m.
- *Upper montane forest*: from ~1,500–2,000 to ~2,800–3,000 m asl with mean temperatures ~14–17 °C; multilayered canopy of 1.5–18 m with rare emergent individuals up to 26 m.
- *Sub-alpine forest*: from ~2,800–3,000 (usually where mean temperature drops below 10 °C) to ~3,200 m asl, usually limited by the frostline; single or two-layered canopy of 1.5–9 m and very rare emergent individuals of up to 15 m.
- *Elfin forest*: on some montane regions, the high exposure to wind-driven fog and rain promotes the formation of a dense mass of gnarled, tangled stems, generally associated with upper altitudes, but without clear boundaries. The constant presence of ground-level clouds allows the establishment of mosses and vascular epiphytes, giving these assemblages an eerie appearance, for which they are commonly referred to as “elfin,” “fairy” or “mossy” forests.

One of the most common environmental characteristics of TMFs is the presence of frequent and persistent cloud immersion, which represents a constant input of water in addition to precipitation (Bruijnzeel et al., 2011a; Mulligan, 2010), often referred to as ‘horizontal precipitation.’ This continuous source of water combined with the multi-storeyed and complex structure of TMFs’ canopies translates into high microclimatic variability, and thus high levels of diversity and endemism (Oliveira et al., 2014), especially of bryophytes and vascular epiphytes, birds, amphibians and invertebrates (Berrios et al., 2022; Gotsch et al., 2015; Jankowski and Rabenold, 2007; Tobar-Suárez et al., 2022).

Montane forests had historically been shielded from direct human impact because they often occur in remote, difficult-to-access areas. Unfortunately, the same environmental conditions that shape these extraordinary biological communities in TMFs make them highly vulnerable to indirect anthropogenic pressure in the form of climate change. Their distribution on mountain slopes, naturally separated by lowland areas, results in small, discontinuous (or ‘archipelagic’) communities that are often isolated from each other. Aside from a constant influx of water, ground-level clouds have the effect of shielding the canopy from strong solar irradiance and maintaining vapour pressure deficit low, which has resulted in the selection over evolutionary time of species that have poor water use efficiency and low resistance to water stress (Esperón-Rodríguez and Barradas, 2014, 2015a). Thus, changing climatic patterns accelerated by human emissions of greenhouse gases can severely jeopardise the environmental conditions that support montane communities. As a result, climate change, in combination with accelerating rates of human encroachment (Bruijnzeel and Hamilton, 2000), has made these ecosystems some of the most endangered on Earth (Soh et al., 2019), but also of major interest for thorough research aimed at informing urgent conservation and restoration efforts (Christmann and Menor, 2021; Peh et al., 2011).

1.2 Introduction to climate change

1.2.1 Definition and magnitude

The United Nations Framework Convention on Climate Change (UNFCCC) in its Article 1 (Sands, 1992), defines *anthropogenic* climate change as "a change of climate which is attributed directly or indirectly to human activity that alters the composition of the global atmosphere and which is in addition to natural climate variability observed over comparable time periods." The main driver of anthropogenic climate change is human activities that emit large amounts of greenhouse gases to the atmosphere, increasing mean global surface temperature by as much as 1.1 °C from 2011 to 2020 above the 1850-1900 average (Lee et al., 2024). According to the most recent update from the Intergovernmental Panel on Climate Change, the observed human-induced warming during the last decade (2014-2023) was 1.19 [1.0 to 1.4] °C (Forster et al., 2024). In 2023, human-induced warming reached a record of 1.43 [1.32 to 1.53] °C, with and increase rate of 0.26 [0.2–0.4] °C per decade over 2014–2023 (Forster et al., 2024).

1.2.2 General impacts of climate change on biodiversity

Human-induced increases in global temperatures provoke rapid changes in the atmosphere, ocean, cryosphere and biosphere (Lee et al., 2024). Besides temperature, measures of climate change on a global scale including changes in precipitation patterns, snow cover, sea and river ice, glaciers, sea level, climate variability and extreme climatic events affect weather and climate extremes (Gitay et al., 2002). These measures in turn lead to widespread feedback loops, generally with negative impacts to natural systems and human populations (Lee et al., 2024).

In the case of biodiversity, modelling and empirical evidence indicate that climate change may induce significant widespread changes in global ecosystems in terms of their distribution, composition, physical and trophic structure, successional and community dynamics, functioning and productivity (Campbell et al., 2009). These impacts are evident at all levels of ecological organisation, from genes to populations, to whole biomes (McCarty, 2001; Pauls et al., 2013; Peñuelas et al., 2013), and are often aggravated by other human activities, such as habitat destruction, resource extraction, pollution and introduction of invasive species.

1.2.3 Plants and forests in relation to climate change

Plants are a relatively well-studied taxonomic group in relation to climate change. A review conducted in 2015 found that plants were the focus of ~42% of studies on species' responses to changing climate (Parmesan and Hanley, 2015). Plants respond to climate change through phenological shifts (e.g., germination, leaf emergence and senescence, flowering and fruiting times, and greening-up events), distributional changes (e.g., latitudinal and altitudinal range shifts, range contractions and expansions, local extirpations and colonisation of novel areas), plasticity, adaptation, evolution and biotic interactions (Parmesan and Hanley, 2015).

Forests cover ~42M km^2 (~30%) of the Earth's land surface, providing invaluable ecological economic, sociocultural, and aesthetic services to human populations, as well as having a key biological role as major hubs of biodiversity (Bonan, 2008; Sanquetta et al., 2011). In the context of climate change, forests are widely regarded as both major players in the efforts to combat its adverse effects by helping with mitigation and adaptation efforts, and as a source of major concern due to their potential negative responses to shifting climatic patterns (Keenan, 2016).

Forests are a major player in the global carbon cycle, representing ~45% of terrestrial carbon, and ~50% of terrestrial net primary production (Bonan, 2008). Through photosynthesis, forests sequester carbon from the atmosphere, and lock it in their living biomass; they can also store carbon in their soils through leaf litter, woody debris, and roots. Some of that carbon is returned to the atmosphere as CO_2 through respiration and degradation of organic matter. Although the estimates of carbon stores in global forests vary between ~650 and

~1,100 metric gigatonnes and their rates of carbon sequestration vary across world regions, forests are widely seen as net carbon sinks and thus play a major role in climate change mitigation (Brack, 2019).

However, climate change threatens forests by increasing the intensity and frequency of natural disturbances, including fires, droughts, storms, snow, and ice, as well as subjecting them to physiological stress and altered biotic interactions that can make them vulnerable to pests and diseases. There are major concerns that physiological stress, heightened mortality, and increased decomposition rates due to climate change – often compounded with other human impacts such as deforestation – global forests can transform into net carbon emitters (Tubiello et al., 2021). It is thus crucial to expand our understanding of the potential impacts of climate change on forests across all world regions and scales to help with the design and implementation of effective policies aimed at preserving forests, their biodiversity, and the invaluable services they provide.

1.3 Tropical Montane Forests in Face of Climate Change

As temperature rises around the globe, rainfall patterns and overall water regimes shift unevenly at both spatial and temporal scales (Murugan et al., 2009). That heterogeneity is influenced by topography. There is abundant evidence that mountainous regions tend to warm up faster than surrounding lowland areas and enhance the effects of increasing temperatures on upper communities in the long term (Aiba and Kitayama, 2002; Karmalkar et al., 2008; Loarie et al., 2009; Torres et al., 2008; Williams et al., 2018). Raising levels of cloud formation and decreased or steady water regimes (i.e. rainfall, humidity and fog incidence) in mountainous regions have indeed been observed in association with higher mean annual temperatures in mountainous regions over the past decades (e.g. Crausbay and Hotchkiss, 2010; Murugan et al., 2009; Sperling et al., 2004).

1.3.1 General impacts on tropical montane flora

For more than two decades, it has been recognised that the particular climatic and topographic features of TMFs (i.e., their steep environmental gradients and their naturally discontinuous distribution on mountainous regions) make them especially susceptible to environmental change (Loope and Giambelluca, 1998). During that same decade, it was suggested that they could serve as early warning systems, as some of their biotic components were expected to respond quickly to shifting environmental patterns (Hamilton, 1995).

Thus, the assumption that montane communities will migrate following their suitable climatic thresholds is the basis for most predictive ecological distribution models, although it is acknowledged that plants may also be able to acclimate or adapt to new climatic conditions (Heilmeier, 2019), A pioneering ecological niche modelling study (Still et al., 1999) projected that as cloud base levels rise due to global warming, whole ecological communities would be forced to migrate upslope in order to persist.

Since then, it has been observed that most montane communities migrate at a much slower rate than that of ongoing climate change (Bergamin et al., 2024; Feeley et al., 2011; Lutz et al., 2013). Moreover, unlike their temperate counterparts, a recent review found no evidence that tropical communities are successfully migrating latitudinally (Colwell and Feeley, 2024). This increases the risk of running out of available space in the mountaintops (Freeman et al., 2018; Jankowski et al., 2010; but see Elsen and Tingley, 2015).

Even where physical space is not a limiting factor, temperature rises and seasonal shifts in water regime can severely impact upper plant communities (Hiltner et al., 2016; Ortega et al., 2024) by making them more likely to be outcompeted by invasive lowland species (Loope and Giambelluca, 1998; Mamantov et al., 2021; Oliveira et al., 2014). The gradual upward displacement of species may thus lead to 'thermophilisation' at mid and high elevations, i.e., a replacement of a community by another one adapted to warmer regimes (Duque et al., 2015), and biotic attrition at lower elevations (Colwell et al., 2008).

Since plants can only migrate at a population level, individuals already established will experience continuous stressful ambient conditions. Trees, despite having access to ground water, are subjected to hydric stress as vapour pressure deficit increases due to a combination of higher air temperature and constant or decreased humidity (Esperón-Rodríguez and Barradas, 2014, 2015b). That stress could have possibly led to the widespread browning trend noticeable since the mid-1990s over large areas (Krishnaswamy et al., 2014).

The situation might be more severe for epiphytic communities that cannot access ground moisture. Despite showing some plasticity, field and laboratory experiments with bryophytes have found increased respiration and mortality rates under warmer, dryer air conditions

(Bader et al., 2013; Metcalfe and Ahlstrand, 2019; Wagner et al., 2014). Similarly, studies on vascular epiphytes have recorded decreases in growth rates, leaf production and reproductive success, as well as increased mortality (Crain and Tremblay, 2017; Nadkarni and Solano, 2002; Song et al., 2012; Zotz et al., 2010).

1.3.2 The significance of rare and montane-specialist species in Mesoamerica

Globally, species richness increases from the poles towards the equator. The proportion of rare species follows the same latitudinal pattern, but with precipitation acting as a major promoter of rarity at regional and local scales (Hordijk et al., 2024). Hence, even in boreal regions, rare species are important contributors to local richness (Heegaard et al., 2013). Although there are several approaches to measure rarity, including low local abundance, restricted distributional range and habitat breadth (Leitão et al., 2016), montane regions are known to have a high proportion of rare species (Mouillot et al., 2013).

The Neotropics host the highest number of plant species on Earth and also have a higher proportion of montane communities than other world regions (Laurance et al., 2011). It has also been noted that Neotropical montane regions concentrate most collections of rare species, including Central America (Zizka et al., 2018). A study in Panama found that the proportion of rare species peaked at the highest elevations (~2,000 m asl), which compounded with decreasing species richness with elevation, meant that the highest number of rare species was concentrated between 750 and 1,000 m asl (Tokarz and Condit, 2021). This pattern was especially noticeable in the western part of the country bordering Costa Rica. Therefore, it is reasonable to expect a similar pattern in the rest of Mesoamerica.

It has been recognised that rare species play a disproportionate role in expanding the functional and environmental niche space of their community (Mi et al., 2021). By presenting some of the most original (i.e., less redundant) combinations of traits, rare species often support the most vulnerable functions of their ecosystems (Mouillot et al., 2013), especially within species that have large-sized individuals (Kearsley et al., 2019). A study along an altitudinal gradient from 0 to 2,000 m asl in Mexico found that the high functional diversity and redundancy of montane communities made them resilient to disturbances, especially at mid elevations (Monge-González et al., 2021). In contrast, a simulation study found that extirpating rare species resulted in a disproportionately larger impact on levels of functional richness, specialisation and originality than if species disappeared randomly (Leitão et al., 2016).

Three main points can be derived from this knowledge: 1) Mesoamerica is a global hotspot of montane flora, 2) montane regions harbour a high proportion of rare species, and 3) rare species contribute disproportionately to their communities' diversity, functional niche and resilience in face of disturbance. However, despite having some of the most intensely studied tropical montane forests (Soh et al., 2019), to my knowledge, no studies focusing specifically on the montane-specialist tree community of Mesoamerica have been conducted. With the threat of on-going climate change, it is thus crucial to characterise the montane-specialist tree community of Mesoamerica and predict the potential impacts that changing climatic regimes could have on their extent, distribution, community composition and functional diversity.

1.4 Overview of the Thesis

1.4.1 General Aim and Objectives

This project's aim was to produce a comprehensive picture of the state of knowledge of tropical montane forests globally with utility for future action and improved protection of these ecosystems. Subsequently, it focused on the Mesoamerican region as a case study, seeking to create a species-level characterisation of its montane-specialist tree community along latitudinal and altitudinal gradients. This characterisation served as the foundation to project the potential impacts of climate change on their extent, distribution, and community composition by the end of the century. Furthermore, to get a more in-depth understanding of the significance of those impacts, I described the spatial patterns of functional diversity of the Mesoamerican montane-specialist community and test the potential effects of climate change and its significance in terms of carbon storage capacity in aboveground biomass.

1.4.2 Structure of the thesis

The structure of the research chapters with their aims, objectives and main methodological approaches are presented in the Logical Framework (Table **1.1**). A brief description of each chapter is presented below.

Chapter 2 consists of systematic review complemented with an evidence-based assessment that sought to compile and rank studies of the effects of climate change on TMFs based on the quality of their evidence. With this information, I built a consistent and comprehensive narrative of the effects of climate change on global TMFs prioritising the strongest available evidence.

In Chapter 3, I compiled tree species from national inventories of countries in Mesoamerica characterised the montane-specialist community based on those with ≥75% (upper 3 quartiles) of their occurrence records inside tropical montane areas. I then used an ensemble Species Distribution Modelling approach to estimate their current extent and distribution and forecast the potential impacts in terms of species richness and community composition of a severe climate change scenario (RCP8.5) by the end of the century.

In Chapter 4, I build upon the distribution models of Chapter 3 to describe the Mesoamerican montane-specialist tree community in terms of its functional diversity throughout space and phylogeny. Then, I predict the potential impacts of climate change on the montane-specialist tree community's functional diversity by the end of the century, and how it might affect its carbon storage capacity in terms of aboveground biomass.

Finally, Chapter 5 integrates the general findings of the whole thesis, discusses its relevance and contribution in the context of wider global science, and outlines the possible direction of further research.

Table 1.1 Logical Framework of the project "The potential impacts of Climate Change on the Extent, Distribution and Functional Diversity of the Montane-Specialist Tree Community in Mesoamerica".

Chapters	**Aims**	**Objectives**	**Main analytical method**
Chapter 2. Impacts of anthropogenic climate change on tropical montane forests: an appraisal of the evidence	Identify and critically assess the available knowledge of the impacts of climate change on global TMFs.	1. Identify the general thematic and methodological trends in research on the impacts of climate change on TMFs. 2. Determine the implications of those trends in terms of our understanding and application of the current knowledge. 3. Rank the existing knowledge in terms of the quality of the evidence it provides. 4. Synthesise the general forecasts for TMFs globally based on the most reliable evidence.	Systematic search of primary research. Ranking of evidence quality.
Chapter 3. Distributional Responses of the Montane-Specialist Tree Community in Mesoamerica under Climate Change	Determine the current and future extent of the montane-specialist tree community in Mesoamerica under current and future climatic conditions.	1. Characterise the MST community of Mesoamerica beyond political boundaries (including establishing whether its southern limit extends to the Panama-Colombia border, or it is truncated by Lake Cocibolca). 2. Estimate the spatial extent and distribution of Mesoamerican MSTs. 3. Assess the potential impacts of projected climate change on Mesoamerican MSTs in terms of extent, distribution, and species richness by the end of the century.	Species Distribution Modelling under current and future climatic conditions. Post-SDM analysis, including estimations of richness, turnover and relation to latitudinal and elevational gradients.
Chapter 4. Functional Traits of the Montane-Specialist Tree Community of Mesoamerica in the Present and under a Changing Climate	Understand the components and spatial patterns of the functional diversity of montane-specialist tree communities throughout Mesoamerica and investigate how it will be impacted by climate change.	1. Assess the functional trait diversity of the montane-specialist tree community in Mesoamerica in terms of PCA axes and spatial patterns. 2. Project how functional trait diversity of those tree communities will change under climate change by the end of the century. 3. Evaluate how functional diversity translates into potential aboveground biomass of the montane-specialist tree community in Mesoamerica and how it might change under climate change by the end of the century.	Functional traits analysis, including community-weighted means, PCA of functional diversity, and functional dispersion.
Chapter 5. Synthesis and Conclusions			

Chapter 2. Impacts of anthropogenic climate change on tropical montane forests: an appraisal of the evidence

2.1 Abstract

In spite of their small global area and restricted distributions, tropical montane forests (TMFs) are biodiversity hotspots and important ecosystem services providers, but are also highly vulnerable to climate change. To protect and preserve these ecosystems better, it is crucial to inform the design and implementation of conservation policies with the best available scientific evidence, and to identify knowledge gaps and future research needs. We conducted a systematic review and an appraisal of evidence quality to assess the impacts of climate change on TMFs. We identified several skews and shortcomings. Experimental study designs with controls and long-term (≥10 years) data sets provide the most reliable evidence, but were rare and gave an incomplete understanding of climate change impacts on TMFs. Most studies were based on predictive modelling approaches, short-term (<10 years) and cross-sectional study designs. Although these methods provide moderate to circumstantial evidence, they can advance our understanding on climate change effects. Current evidence suggests that increasing temperatures and rising cloud levels have caused distributional shifts (mainly upslope) of montane biota, leading to alterations in biodiversity and ecological functions. Neotropical TMFs were the best studied, thus the knowledge derived there can serve as a proxy for climate change responses in under-studied regions elsewhere. Most studies focused on vascular plants, birds, amphibians and insects, with other taxonomic groups poorly represented. Most ecological studies were conducted at species or community levels, with a marked paucity of genetic studies, limiting understanding of the adaptive capacity of TMF biota. We thus highlight the long-term need to widen the methodological, thematic and geographical scope of studies on TMFs under climate change to address these uncertainties. In the short term, however, in-depth research in well-studied regions and advances in computer modelling approaches offer the most reliable sources of information for expeditious conservation action for these threatened forests.

Key words: biodiversity, cloud forests, conservation, ecological levels, ecosystem functions, evidence quality, global warming, research rigour, systematic review, tropical mountains.

2.2 Introduction

Tropical montane forests (TMFs) are typically an evergreen ecosystem constrained to a narrow altitudinal belt, with an uneven canopy layer frequently enveloped by orographic clouds (Foster, 2001; Hamilton, 1995; Loope and Giambelluca, 1998; Richter, 2008; Still et al., 1999). These forests often harbour high abundance and diversity of epiphytes (Collin, 2001; Foster, 2001; Loope and Giambelluca, 1998). TMFs are thus commonly referred to as cloud forests, mist forests or mossy forests, as well as numerous names in other languages, especially in Latin America (Brown & Kappelle, 2001) where most studies of this ecosystem have been conducted (Laurance et al., 2011; Soh et al., 2019).

Unlike other vegetation types defined by their taxonomic affiliation (e.g. coniferous forest, oak forest, etc.) or their phenological structure (e.g. deciduous forest, xerophilous shrubland, etc.), TMFs are characterised by the intersection of an atmospheric phenomenon (fog incidence) and a topographic feature (mountain slopes). Such restrictive features highlight the potential vulnerability of TMFs in the face of climate change (Hamilton, 1995), but also make impacts difficult to isolate from these ecosystems' intrinsic climatic variability (Vuille et al., 2003). In this study, we focus on *anthropogenic* climate change, as defined by the United Nations Framework Convention on Climate Change (UNFCCC) in its Article 1 (Sands, 1992): a change of climate which is attributed directly or indirectly to human activity that alters the composition of the global atmosphere and which is in addition to natural climate variability observed over comparable time periods.

There is abundant evidence that mountainous regions tend to warm up faster than their surrounding lowland areas and thus amplify the effects of increasing temperatures on upland plant communities in the long term (Aiba & Kitayama, 2002; Karmalkar, Bradley & Diaz, 2008; Loarie *et al*., 2009; Nyongesa *et al*., 2019; Ohmura, 2012; Torres, González & Comarazamy, *et al*., 2008; Williams *et al*., 2018; but see Pepin *et al*., 2022). Rising altitude of cloud formation, and decreased water availability have been observed in response to higher mean annual air temperatures in tropical mountainous regions over recent decades (e.g. Crausbay & Hotchkiss, 2010; Murugan *et al*., 2009; Sperling, Washington & Whittaker, 2004; but see Cuervo-Robayo *et al*., 2020). Even if average annual changes in precipitation are small, marked seasonal shifts in water regime could severely impact upper montane communities (Hiltner et al., 2016) that already live near their physiological limits (e.g. Catenazzi, Lehr &

Vredenburg, 2014; Crausbay *et al*., 2014; Muñoz *et al*., 2016) and with little space to expand their ranges (the "escalator to extinction"; Freeman *et al*., 2021, p. 1,706). These combined pressures could make them more likely to be outcompeted by climate-driven upwardly migrating species from nearby lowland areas (Loope and Giambelluca, 1998; Oliveira et al., 2014; Sukumar et al., 1995).

Despite recent advances in our understanding of the impacts of ongoing anthropogenic climate change on TMFs, concrete empirical evidence (i.e. compelling proof of climate change-driven impacts occurring in real time) remains scarce, partly because such assessments require long-term data that are difficult to collect and analyse (Wauchope et al., 2021). For instance, it has been observed that plant communities can shift their distributions, but these movements lag behind the velocity of climate change (Corlett and Westcott, 2013; Feeley et al., 2011; Sáenz-Romero et al., 2016). Therefore, key ecosystems like TMFs help advance our knowledge of climate-driven distributional change because the microclimatic variability and island-like distribution of mountainous biomes not only results in high levels of endemism, but also high extinction vulnerability (Freeman et al., 2018), making them natural laboratories for climate change (Silveira et al., 2019; Tito et al., 2020).

Numerous literature reviews on TMF research have been conducted, focusing on various aspects, including environmental determinants (Fahey et al., 2016; Oliveira et al., 2014), ecosystem functions (Dalling et al., 2016), ecosystem services (Buytaert et al., 2011), habitat degradation (Soh et al., 2019), conservation (Peh et al., 2011) and restoration strategies (Christmann and Menor, 2021), specific geographical regions (Rosas Rangel et al., 2019; Sáenz-Romero et al., 2020a; Tovar et al., 2022) or taxonomic groups (Gotsch, Nadkarni & Amici, 2016; He, He & Hyvönen, 2016), and use of remote sensing technologies (Altarez et al., 2022). Our study goes beyond a systematic review on the effects of climate change on TMFs by critically evaluating the quality of the available evidence. We applied a replicable evidence-appraisal methodology to synthesise the current knowledge of the impacts of climate change on TMFs, with a focus on the published studies yielding the strongest evidence. This will serve as a guide to identify knowledge gaps and inform future research on this highly threatened biome.

Specifically, our research questions are: (1) what are the general thematic and methodological trends in the published studies investigating the impacts of climate change on TMFs? (2) What are the implications of those trends in terms of our understanding and application of the current knowledge? (3) How reliable is the literature in terms of the quality of the evidence it provides? (4) What are the general forecasts for TMFs globally based on the most reliable evidence and the direction of future research?

2.3 Methods

2.3.1 Systematic Review

We carried out a literature search on the effects of climate change on TMFs following a standard systematic review protocol (PRISMA-P; Shamseer *et al*., 2015). The search terms were based on the most common names of TMFs and terms associated with long-term changes in the climate. The final search terms used were "(trop* monta* forest* OR cloud forest* OR trop* high* elevation* forest* OR trop* mid* elevation* forest* OR trop* mount* forest*) AND (climat* chang* OR glob* warm* OR temperature ris* OR clima* vari*).

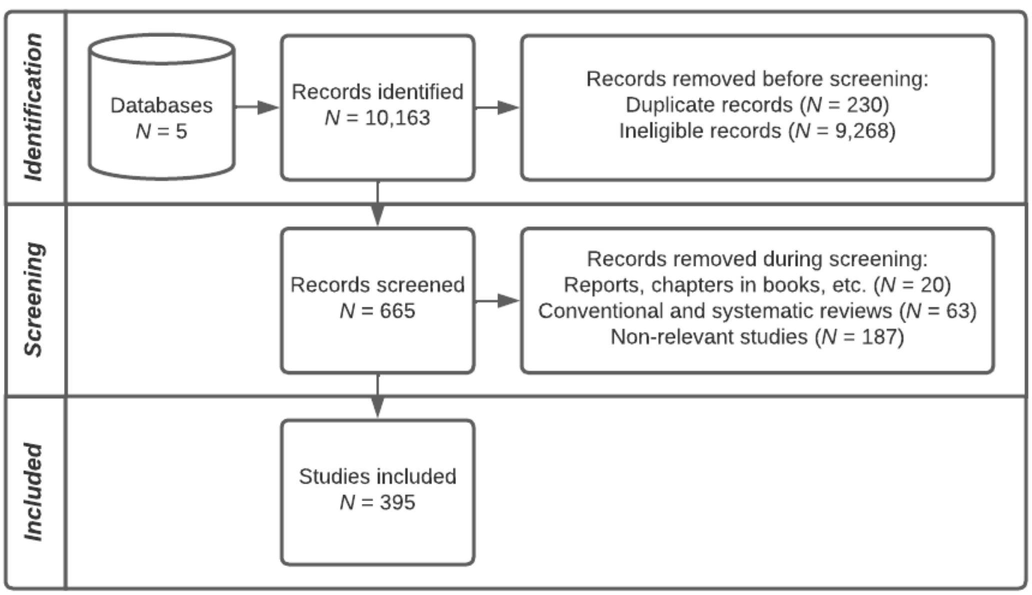

Figure 2.1 Systematic search protocol. The final count includes all relevant studies from 1994 to 2021.

We conducted the literature search in July 2022 using four academic databases (*Web of Science*, *Scopus*, *ScienceDirect* and *Google Scholar*) and the *Google* search engine for the period 1994–2021. Relevant studies published during 2022 were omitted from the statistical analysis but are included in the discussion. The search string was run choosing the 'all fields' option if available, hits were sorted alphabetically, and relevant publications – i.e. studies that explicitly reported observed or projected effects of anthropogenic climate change on TMFs (or any component thereof) — were chosen based on title and abstract. Duplicated records were removed, and studies that mentioned climate change only as a potential future threat but did not investigate its effects on TMFs were excluded. Likewise, studies that described diversity patterns along elevational gradients but did not link them to environmental factors to allow inferences of potential future shifts were also discarded (Figure **2.1**). For *Google* searches, only the first 100 hits were reviewed to ensure that no relevant studies had been missed. A search in Spanish was conducted in the *Redalyc* database (www.redalyc.org), which did not yield any relevant studies, however, publications in Spanish and French listed under an English title, and thus captured by the search string, were retained. Since we aimed to rank the quality of the available scientific evidence, only peer-reviewed primary research articles were retained (see Section 2.3.2). However, we checked reviews for 'snowballing' purposes (i.e. pursuing references of references; see Greenhalgh & Peacock, 2005). We excluded book chapters, reports and other grey literature. Palaeoclimatic and palaeoecological studies were excluded because our focus was on current anthropogenic climate change only.

We extracted from individual studies information on study region, country, methodology, study duration, publication date, ecological level, taxonomic group, research topic, as well as measures of climate change and their observed effects in TMFs. We considered patterns of changing climate (rising temperature and increased or reduced precipitation) as binary variables (i.e. whether these measures were reported or tested in the study or not), and the observed impacts as categorical variables: occurrence of extreme events [strong El Niño Southern Oscillation (ENSO), cyclones, fires, frost, landslides], species' distributional changes [habitat losses (range contractions or fragmentations), and upslope or downslope shifts], impacts on biodiversity (increased vulnerability, population decline, extinctions, reduced genetic diversity, invasion of neighbouring communities and lowland biotic attrition), and other local-scale effects (alterations to carbon and other nutrient fluxes and stocks, soil functions, and phenological/physiological patterns).

To elucidate general trends in TMF research related to climate change, we investigated if studies were skewed towards a particular geographical region, taxonomic group, ecological level, or research topic; and if there were any associations between these parameters. We also investigated if studies reporting or testing an impact on TMFs tend to look at a particular measure of climate change or taxonomic group in their research. Only categories represented by at least 10 studies were analysed using chi-squared tests to avoid inferring spurious associations. These associations should be interpreted with caution because they are based solely on what individual studies reported (observations or historic records) or tested (experimentally or through modelling). Statistical tests were carried out in R 4.1.1, using *ggplot2* v.3.3.5 (Wickham, 2011) and *corrplot* v.090 (Wei et al., 2017) packages.

2.3.2 Evaluation of evidence quality

The evidence assessment tool devised by Mupepele *et al*. (2016) is designed to rank the quality of evidence provided by individual reviews and studies based primarily on their study designs. Studies that are poorly conducted (e.g. unclear research questions, inadequate sample sizes, lack of controls, etc.) are downgraded in the evidence hierarchy. The discrete categories of evidence quality and assessment criteria of the evidence assessment tool were specifically designed for conservation studies, but can be adapted for other fields of research. Our evidence assessment was an adaptation of Mupepele *et al*. (2016). We removed the 'review' (systematic and conventional) category, ranked highest by Mupepele *et al*. (2016), as its inclusion would result in double-counting of studies (i.e. studies captured by our search string are likely to occur in other reviews; Figure **2.2**). In addition, we carried out a methodological appraisal of each study. Studies with flaws and biases identified in terms of data collection and analyses or that employed outdated methodologies (mainly applicable to modelling studies) were downgraded to a lower level in the evidence hierarchy. Studies that combined multiple methodological approaches were classified according to their highest level in the evidence hierarchy [e.g. a study with both *ex-situ* experiments and modelling would be ranked as Level of Evidence 1b (LoE1b); Figure **2.2**, Table **2.1**].

To define the hierarchical LoEs, we considered the following features: use of references and controls, execution of the study in the field (i.e. *in-situ*), long-term collection of data, high regularity of survey, and corroboration of findings derived from models with empirical observations (Table 1). Experimental study designs with a control carried out in the field provide very strong evidence (LoE1a). Experiments conducted *ex situ* yield only strong evidence (LoE1b) because artificial environments introduce potential biases to the study (Tito et al., 2020). Observational studies spanning a minimum of 10 years with observations at regular intervals (weekly, monthly, etc.) also provide strong evidence by accounting for the long-term nature of climate patterns (Bruijnzeel, 2004; Chapman et al., 2018), and are included in LoE1b. We prioritised the highest-ranked study design in studies that had employed both an *ex-situ* experiment and a modelling approach and therefore assigned these to LoE1b. None of the studies captured by our search string had conducted both field experiments and modelling.

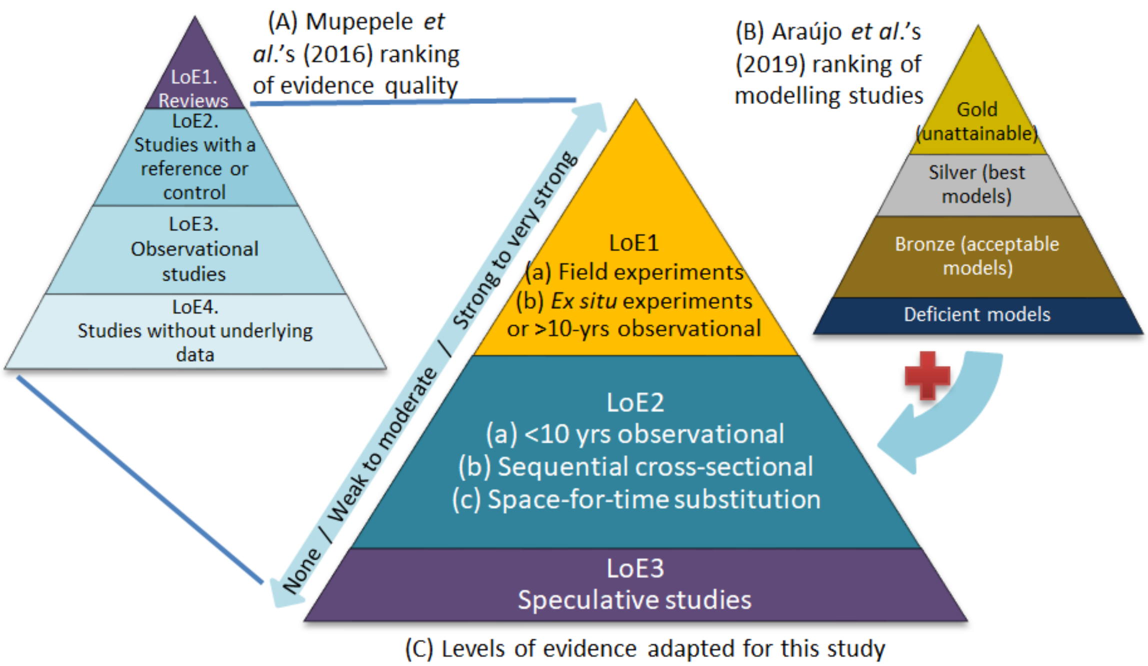

Figure 2.2 Schematic representation of the adapted and combined evidence assessment used in this study.

LoE2 comprises short (<10 years) longitudinal and all cross-sectional observational studies, as well as the bulk of studies that relied on computer-based simulations (hereafter 'modelling'), i.e. projections of future climate conditions and especially ecological niche models, which include species distributions, life zones and relative abundance projections. Non-modelling studies are subdivided into <10-year long longitudinal studies (LoE2a, moderate evidence), sequential cross-sectional studies, i.e. resurveys (LoE2b, inconclusive evidence), and all other cross-sectional studies (LoE2c, circumstantial evidence), including comparative surveys (between different sites) and along altitudinal or latitudinal gradients (space-for-time substitution approach).

Table 2.1 Classification of studies according to Level of Evidence adapted from Mupepele et al. (2016). Modelling studies included in LoE2a-c correspond roughly to the silver, bronze and deficient levels proposed by Araújo et al. (2019), since their *gold standard* is unattainable for TMFs studies in relation to climate change.

Level of Evidence (LoE)		Main study design	Explanation
*** Reviews are not included in this classification.**			
1. Very strong to strong evidence of climate change effects.	**1a**	Field experiments.	Experimental studies that had a controlled design and were carried out in the field provide the strongest and most direct evidence.
	1b	Ex situ experiments OR time-series studies (≥10 years of regular —yearly, monthly, etc.— observations).	Long-term datasets provide strong evidence. Experiments carried out in artificial conditions may introduce uncertainty. (Studies that employed both an *ex situ* experiment and a modelling approach belong to this level.)
2. Moderate, inconclusive and circumstantial evidence of climate change effects, including observational and modelling studies.	**2a**	Time-series studies (<10 years of regular —yearly, monthly, etc. — observations) OR modelling studies verified with empirical evidence.	Short-term datasets provide evidence of changes taking place over time, but they may not be long enough to confidently attribute observed changes to shifting climatic patterns. Similarly, models that have been corroborated with field observations provide moderate evidence of changes that might occur in the future (comparable to the *silver standard* in Araújo et al., 2019).
	2b	Sequential cross-sectional studies OR modelling studies not verified by empirical evidence.	Cross-sectional studies that provide two or more snapshots of a system but lack regular/continuous data (resurveys), can inform of changes in a system. However, they do not allow an adequate analysis of the environmental influence on the observed changes. Similarly, models that have not been verified with field observations provide inconclusive evidence of the effects of climate change (comparable to the *bronze standard* in Araújo et al., 2019).
	2c	Cross-sectional and comparative surveys, space-for-time substitution approaches OR non-forecasting modelling studies.	Descriptive studies of conditions at a certain moment provide circumstantial evidence of environmental determinants of observed patterns. These include the use of spatial comparisons as a surrogate for temporal data, i.e., comparative surveys (e.g., temperate vs. tropical sites) or altitudinal/ latitudinal gradients as proxies for changes over time (space-for-time substitution). Modelling studies making inferences about future changes even though they are not designed to make forecasts or predictions are included in this level (comparable to the *deficient category* in Araújo et al., 2019).
3. Studies without underlying data (no evidence).	**3**	Speculative studies OR data-deficient modelling-based inferences.	Experts' educated guesses of expected future changes, usually based on past observed changes in analogous systems, are not considered as evidence. Modelling studies based on the knowledge from other systems are also included in this level.

Given the requirement for long-term data in climate change research and the urgency to implement well-informed policies to preserve ecosystems from anthropogenic threats, modelling is widely used to forecast outcomes both in the near and distant future. Our scoring of evidence strength for modelling studies (LoE2a–c and LoE3) was based on the model's function and whether their conclusions were validated by field observations, and corresponds loosely to the silver, bronze and deficient standards for models proposed by Araújo *et al*. (2019) (Table 1, Figure **2.2**). In principle, modelling approaches using ideal data and next-generation tools that are still under development (the 'aspirational' gold standard) is unattainable for predictive climate change studies and is therefore not considered in our evidence quality scale. The silver standard corresponds to best modelling practices, i.e. a combination of the best-available data and tools to account for or quantify bias and uncertainty (LoE2a). The bronze standard implies limited acceptable practices that allow inferring implications from their results (LoE2b,c). Lastly, the deficient category corresponds to insufficiently robust data or modelling practices (LoE3), and this level also comprises all speculative studies, i.e., not supported by empirical data. These include expert opinions and speculations based on knowledge of other regions or ecosystem types.

2.4 Results

2.4.1 Distribution of studies

Our initial search string retrieved 10,163 studies. After successive elimination stages, the final data set included 395 studies (see Appendix A. Supplementary Information for Chapter 2, Table S1 for full list), published between 1994 and 2021 (Figure **2.1**). The number of publications increased from less than ten annually before 2006 to an average of 30 from 2013 onwards, potentially reflecting an increased interest in TMFs.

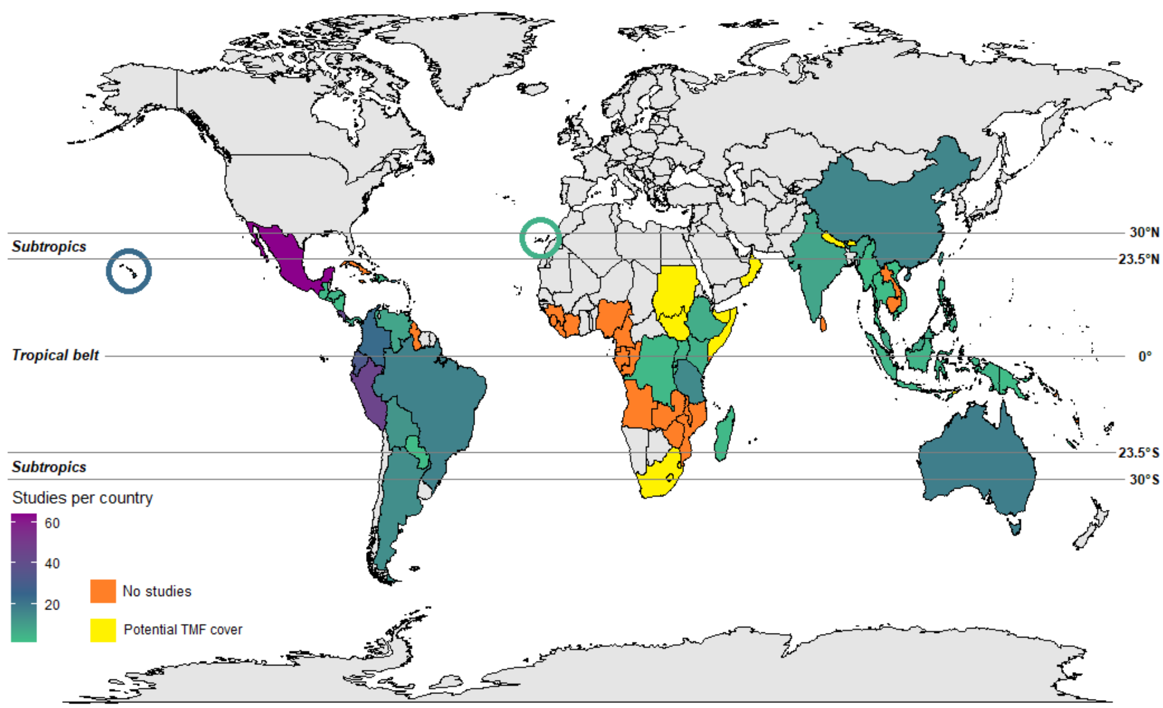

Figure 2.3 Geographical distribution of retrieved studies by country or territory. The number of studies is indicated by the colour scale from light blue to purple. In orange: countries with TMF cover for which no relevant studies were found. In yellow: countries with contested TMF cover (see Karger et al., 2021; Los et al., 2019; Mulligan, 2010).

Inconsistent definitions of TMFs, ambiguous latitudinal limits and different mapping methods have resulted in estimates of their global distribution and extent that vary by as much as an order of magnitude. However, an estimate of ~2M km² globally (Los et al., 2019; Mulligan, 2010) has been reported relatively consistently, and is compatible with other recent regional estimates (e.g. ~401,300 km² TMF cover in the Neotropics; Helmer *et al.*, 2019). Around 75 nations are thought to have some TMF cover, however, there is no consensus regarding many countries such as Bhutan, East Timor, Eritrea, Eswatini, Lesotho, Nepal, Oman, Somalia, South

Africa, South Sudan and Sudan (Figure **2.3**), and some authors count subnational territories separately, e.g. New Caledonia, Tahiti and La Réunion Island (see Karger *et al*., 2021; Mulligan, 2010).

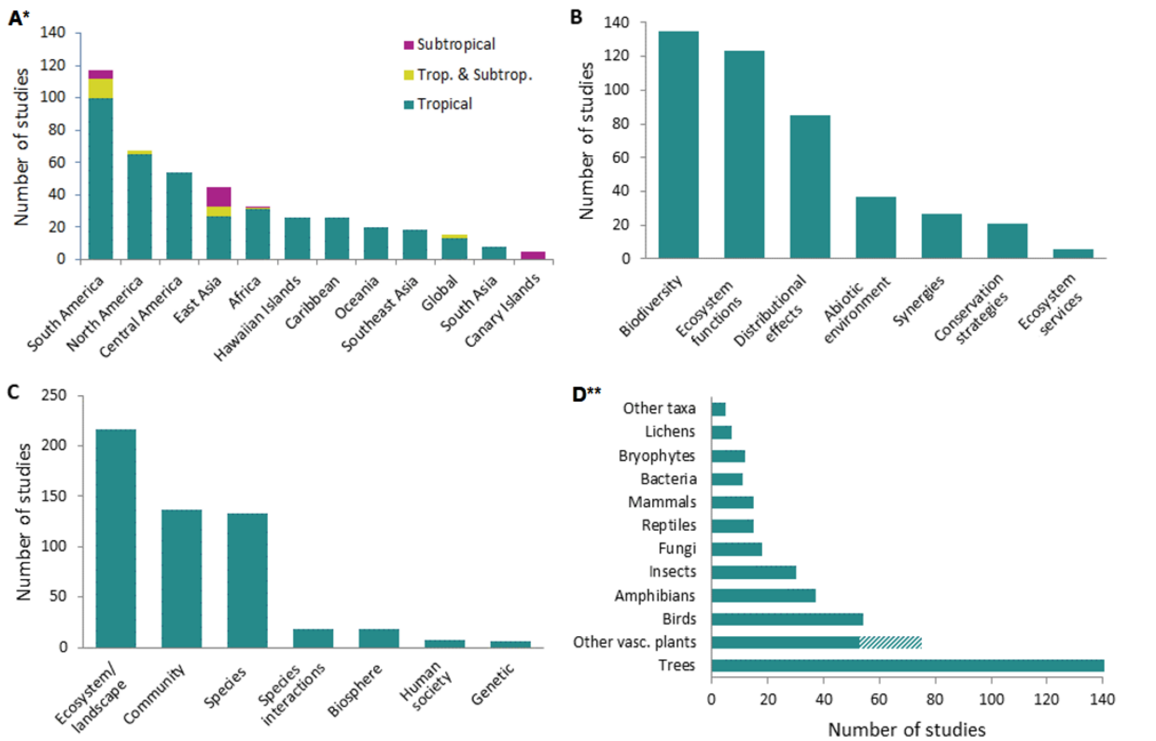

Figure 2.4 Number of studies published from 1994 to 2021 on the effects of climate change on tropical and subtropical montane forests for different (A) geographic regions, (B) effect types, (C) ecological levels, and (D) taxonomic groups.
* Oceania comprises Australia, Papua New Guinea and other islands of the South Pacific; the Hawaiian Islands are shown separately.
** The striped portion of the "other vascular plants" group corresponds to epiphytes.

Of the 395 reviewed studies, 66.8% (264) were carried out in the Neotropics. South America had the highest number of studies (117, 29.6%), followed by North America (Mexico) with 67 (17.0%) and Central America with 54 (13.7%). Only 26 studies (6.6%) focused on the Caribbean (Figure **2.3** and Figure **2.4**A). Elsewhere, we found 71 (18.0%) studies in total for Asia; and there were 33 (8.4%) studies in Africa, 20 (5.1%) in Oceania (Australia, Papua New Guinea, and Melanesian islands), and 31 (7.8%) in other oceanic regions (Hawaii and Canary Islands). Relatively few studies examined TMFs in relation to climate change at a global scale (15, 3.8%; Figure **2.4**A). The vast majority of studies took place in tropical regions (0° to 23.5° N and S), whereas exclusively subtropical (23.5° to 30° N and S; *sensu* Corlett, 2013) studies were only those conducted in the Canary Islands (Figure **2.4**A). Some studies included tropical–subtropical overlapping regions, mainly in Taiwan, Argentina and Brazil.

We found studies conducted in 32 countries and six subnational territories (Figure **2.3**). The most studied TMFs were those in the Neotropics, namely Mexico (64 studies, 16.2%), Peru (45, 11.4%), Costa Rica (42, 10.6%), Ecuador (32, 8.1%), Colombia (23, 5.8%), Brazil (17, 4.3%), Argentina (14, 3.5%) and Bolivia (12, 3.0%). In other regions, only Australia (18, 4.6%), Tanzania (15, 3.8%) and Taiwan (12, 3.0%) stood out. Well-represented subnational territories were the Hawaiian Islands (24, 6.1%) and Puerto Rico (20, 5.1%). We found less than 10 studies each for other countries and territories. Countries with potential areas of TMF (Karger et al., 2021; Mulligan, 2010), for which we did not find any relevant studies include Angola, Cambodia, Cameroon, Comoros, Cuba, Equatorial Guinea, Gabon, Guinea, Guyana, Ivory Coast, Jamaica, Laos, Liberia, Malawi, Mauritius, Mozambique, Myanmar, Nigeria, Republic of the Congo, Sierra Leone, São Tomé and Príncipe, Saint Kitts and Nevis, Saint Vincent and the Grenadines, Solomon Islands, Sri Lanka, Vanuatu, Zambia and Zimbabwe.

Concerning the impacts of climate change on TMF components (Figure **2.4**B), most studies examined the effects on biodiversity (135, 34.2%), ecosystem functions (123, 31.1%) and distributional effects (85, 21.5%). Abiotic environment, synergies with other drivers of change and conservation strategies were the focus of 37 (9.4%), 27 (6.8%) and 21 (5.3%) studies, respectively, and despite the recognition of TMFs as important provider of ecosystem services, these were the focus of only six (1.5%) studies. Research effort at different ecological levels was also noticeably imbalanced (Figure **2.4**C). Most studies focused on the ecosystem or landscape (216, 54.7%), community (137, 34.7%) and species (132, 33.4%) levels. Considerably fewer studies investigated species interactions and biosphere (18, 4.6% each), human societal (7, 1.8%) and genetic (6, 1.5%) levels.

Taxonomically, vascular plants were the most researched taxonomic group (Figure **2.4**D), with trees alone considered in 141 (35.7%) studies, followed by other vascular plants (75, 19.0%), of which 22 (5.6%) studies corresponded to epiphytes. The next best studied taxonomic groups were birds (54, 13.7%), amphibians (37, 9.4%), insects (30, 7.6%), fungi (18, 4.6%), mammals (15, 3.8%), reptiles (15, 3.8%), bryophytes (12, 3.0%) and bacteria (11, 2.8%), while other groups were the focus of less than ten studies each.

2.4.2 Trends in the literature

We found that research from certain regions tended to focus on a particular taxonomic group (chi-squared = 129.77, df = 81, $p < 0.001$). Research efforts in North America (i.e. Mexico) were skewed towards trees; in Oceania (mainly in the Wet Tropics Bioregion, northeastern Australia) were skewed towards insects; and in Central America (mainly in Costa Rica and Panama) towards both bryophytes and fungi. Conversely, studies on trees were scarce in Oceania, with only one study from Papua New Guinea (Venter et al., 2017).

Some research topics were associated with certain geographic regions (chi-squared = 104.25, df = 45, $p < 0.001$). For example, studies in the Caribbean tended to focus on the effects of changing abiotic conditions, such as changes in fog immersion, air temperature and streamflow. We also found an association between research topics and taxonomic groups (chi-squared = 268.83, df = 45, $p < 0.001$). This was mainly driven by studies on ecosystem functions whose focus was on trees and bacteria. Tree studies focused on large-scale processes such as primary productivity, carbon sequestration and distributional shifts, whereas bacteria studies mainly examined soil functions, such as nutrient cycling and decomposition. There was also a dearth of diversity (e.g. abundances and community composition) studies on trees, compared to other taxa, such as birds and insects.

We did not find evidence that studies reporting or testing climate change impacts tend to use a particular measure of climate change or consider a particular taxonomic group in their research (i.e. no association between focus on climate change measures and on taxonomic groups). Studies reporting habitat losses (range contractions or fragmentations) focused more frequently on mammals (chi-squared = 41.07, df = 10, $p < 0.001$), and were also more likely to report biodiversity losses in terms of abundances, species richness or species turnover, and extinctions (chi-squared = 10.32, df = 4, $p = 0.035$).

2.4.3 Assessment of the evidence on the impacts of climate change in TMFs

We found 30 (7.6%) field experimental studies (Figure **2.5**A), i.e. providing a 'very strong' LoE (LoE1a). They reflected the general geographical trend described in Section 2.4.1, with five studies from Mexico, although four of these were conducted by one research team (García-Hernández et al., 2019; García-Hernández and Toledo-Aceves, 2020; Toledo-Aceves et al., 2019; Toledo-Aceves and del-Val, 2021a). Four LoE1a studies were from Peru, followed by three each from China, Costa Rica and Hawaii, two each from Puerto Rico, Taiwan and Tanzania, and one study for Colombia, Ecuador, India, Panama, the Philippines and Rwanda. The scarcity of field experimental studies could be due to the relative inaccessibility of tropical montane regions, often in poorly connected rural areas of low-income countries, which makes field experiments impractical and costly in the absence of well-established research groups. By contrast, strong evidence (LoE1b; *ex-situ* experiments and long-term data sets), was found in 94 studies (23.8%). Around 13% of these studies (12, 3% of total) included the use of modelling methods. Studies that scored as moderate evidence (LoE2a) accounted for 58 studies (14.7%), including mainly <10 year data sets and a small contribution of modelling studies (6, 1% of total) that supported their forecasts with field observations. A third of the reviewed studies provided inconclusive evidence (LoE2b; 132 studies). The reliance of climate change studies on modelling methods was evident: modelling studies accounted for 81.8% (108, 27.3% of total) of studies in this level. The remaining studies in LoE2b were non-longitudinal observational resurveys that provided two or more snapshots suggesting temporal changes but did not allow identification of trends. Studies yielding circumstantial evidence (LoE2c) were common (73, 18.5%), mostly consisting of cross-sectional observational studies, and a small contribution from modelling approaches (11, 2.8% of total). Finally, speculative studies (LoE3a) were the least frequent (8, 2.0%).

Modelling methods were used by 144 (36.5%) studies in total. Climatic envelope models [mainly species distribution models (SDMs), as well as projections of species' population decline, extirpation or extinction under climate change] comprised over two thirds (102, 70.8%) of these; 20 (13.8%) were future climate projections (mainly estimations of future temperature and precipitation regimes), and the remainder (22, 15.3%) were a diverse array of computer-based simulations of changes in biomass, evapotranspiration, albedo, erosion, water runoff, etc. From 1994 to 2005, we found 14 studies that employed modelling methods, with no more than three studies annually, and zero modelling studies for some years, including 2005. Of these, only two were SDMs (Miles et al., 2004; Williams et al., 2003). From 2006 onwards, the use of modelling methods increased to an average of seven studies per year until 2018, and over 10 per year after 2019. Of the 130 modelling studies retrieved after 2006, 96 (73.8%) were SDMs. This trend coincides with the development of the modelling algorithm Maximum Entropy (MaxEnt; Elith *et al*., 2006), which was employed by 58.5% of

SDM studies, either on its own or in combination with other algorithms [e.g. Random Forests, Genetic Algorithm for Rule-Set Production (GARP), etc.].

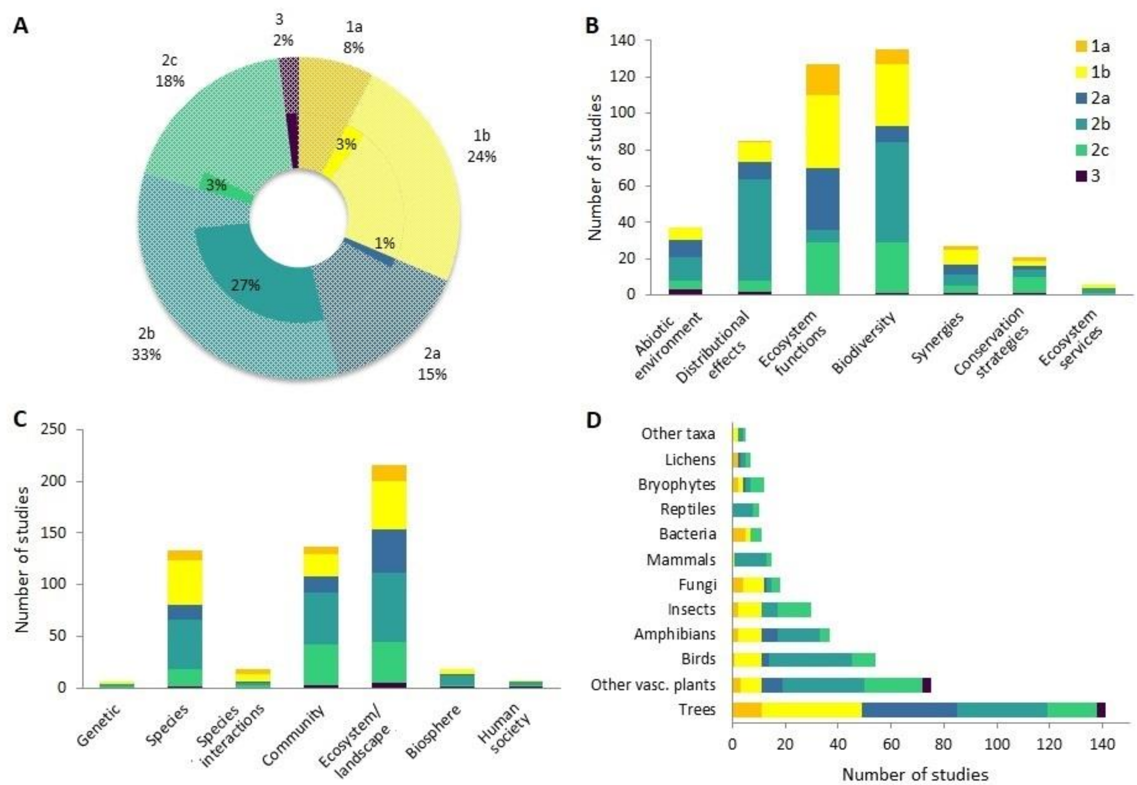

Figure 2.5 Distribution of reviewed studies according to levels of evidence hierarchy. (A) Percentage of studies by level of evidence, with solid-coloured portions indicating the frequency of modelling studies within each level; and levels of evidence of reviewed studies by (B) research topic, (C) ecological level, and (D) taxonomic group.

An association between the LoE strength and research topics (chi-squared = 116.11, df = 20, $p < 0.001$; Figure **2.5**B) revealed some methodological trends. This association partly was an artefact of the ranking of most modelling studies in LoE2b (i.e. distributional shifts as a main research topic was strongly associated with LoE2b). However, the association persisted after removing distributional studies (chi-squared = 68.45, df = 16, $p < 0.001$) because studies on biodiversity also relied more on LoE2b approaches that yield inconclusive evidence (modelling and resurveys). Conversely, studies on ecosystem functions contained the most study designs providing strong evidence (LoE1b, experiments and long-term dendrochronological records).

Considering only the studies at species, community, and ecosystem or landscape levels (i.e. ecological levels with >100 studies each; Figure **2.5**C), we found an association of these studies with certain LoE strength (chi-squared = 23.44, df = 10, p = 0.009). Whilst provision of inconclusive evidence (LoE2b) was most frequent in studies at these three ecological levels, the species and ecosystem or landscape levels also included a non-negligible proportion of studies yielding strong evidence (LoE1b). For the species level, these corresponded mainly to *ex-situ* experiments (e.g. thermal and drought tolerance experiments). For the ecosystem or landscape level, these were based on long-term data sets.

Regarding taxonomic groups, trees and other vascular plants, birds, amphibians, and insects were the only taxa represented in more than 20 studies each (Figure **2.5**D). By retaining only these groups and grouping together the remaining taxa, we found that studies of some taxonomic groups were associated with certain LoE strength (chi-squared = 79.49, df = 20, p < 0.001). Studies of trees commonly provided evidence ranging from strong (LoE1b) to inconclusive (LoE2b), whereas studies of birds were associated more often with inconclusive evidence (LoE2b, modelling studies) and those of insects more with circumstantial evidence (LoE2c, comparative cross-sectional surveys). Very strong evidence (LoE1a) was provided most often in studies of other taxa (i.e. fungi, mammals, bacteria, reptiles, bryophytes, lichens, and others), driven by the use of field transplant experiments in studies of fungi, bryophytes, lichens, and bacterial (soil) communities.

2.4.4 Evidence-based synthesis of impacts of climate change on TMFs

Ranking the published studies according to LoE strength allows us to outline better the current state of knowledge on the present or expected future impacts of climate change on TMFs. In this section, we summarise the findings on climate change impacts on the abiotic environment, biodiversity and ecosystem functions of TMFs, focusing primarily on the studies ranked highest in our evidence hierarchy in each case.

a) Changes in atmospheric conditions

None of the published studies on atmospheric conditions employed field experiments (i.e. no LoE1a studies). Albeit scarce (7, 1.8%), evidence based on long-term studies (≥10-year observations; LoE1b) suggested that atmospheric changes in TMFs were a local or regional effect, rather than a global trend. A study analysing 100 years of meteorological data in

Mexico showed that the subtropical (Nearctic) northern mountainous regions have experienced more consistent increments of both atmospheric temperature and precipitation since 1970 than the (Neo)tropical southern mountainous areas (Cuervo-Robayo et al., 2020). Elsewhere, significant reductions in precipitation have been reported over recent decades (e.g. the Indian Western Ghats region; Murugan *et al*., 2009). However, a decrease in rainfall does not necessarily affect the capacity of TMFs to intercept water from the atmosphere (e.g. La Hispaniola; Comarazamy *et al*., 2015). TMFs in Puerto Rico had a higher likelihood of fog immersion during the dry periods of the year compared to the rainy season (Van Beusekom, González & Scholl, 2017). In addition, over 40 years of observations in Puerto Rico revealed no significant change of cloud base levels in the mountains (Miller et al., 2018). Therefore, these data suggest that water availability within TMFs may not be adversely affected by reductions in precipitation.

Climatic trends can be confounded by large-scale factors such as topography (Aiba & Kitayama, 2002; Van Beusekom, González & Rivera, 2015; but see Loarie *et al*., 2009) and global atmospheric patterns, including the ENSO cycle (Crausbay et al., 2014), as well as by regional-scale phenomena like cyclones and fires. There is evidence of a feedback mechanism between defoliation caused by strong hurricanes and rising cloud levels (Scholl et al., 2021), and accelerated thermophilisation (i.e. intrusion of lowland species adapted to warmer climates into cooler communities; Duque, Stevenson & Feeley, 2015; Fadrique *et al*., 2018) of montane forests in the Caribbean region (Tanner et al., 2022). Increased fire incidence linked to dryer atmospheric conditions could facilitate the expansion of fire-resistant species (Grau and Veblen, 2000), for instance, serotinous pine trees (Climent et al., 2004; Rodríguez-Trejo and Fulé, 2003). In some regions (e.g. Mexico), however, fire incidence has declined over recent decades because of human management, which could negatively affect the regeneration of fire-adapted arboreal species (Cerano-Paredes et al., 2021; Yocom and Fulé, 2012).

b) General decline of TMFs

Most models (LoE2b) predict substantial losses of TMF area and biodiversity in the coming decades. Mexico is projected to suffer important reductions in TMF area, population declines and even extinctions by the end of the century, either due directly to climate change or in combination with other drivers of change, notably land-use change (e.g. Correa Ayram *et al*., 2017; Golicher *et al*., 2008; Ponce-Reyes *et al*., 2017, 2013). Similar outcomes have been forecast for other montane regions, including Costa Rica (Colwell et al., 2008), the tropical Andes (Godoy-Bürki, 2016; Ledo et al., 2009; Tejedor-Garavito et al., 2015), the 'campos rupestres' (Bitencourt et al., 2016) and the Atlantic forest biomes in Brazil (Castro et al., 2020), the Neotropical realm as a whole (Helmer et al., 2019), the Western Ghats region in India (Chakraborty et al., 2013), China's *Abies* forests (Liao et al., 2020), Myanmar's natural protected areas (Nwe et al., 2020), and the Australian Wet Tropics (Costion et al., 2015).

c) Distributional effects on flora

Atmospheric variables play a key role in shaping the distributions of species and whole communities, thus shifts in species' ranges are one of the expected consequences of climate change. The presence of clear boundaries between forests and other vegetation types (treelines) is primarily defined by temperature, precipitation and fog incidence, and has been confirmed in various tropical montane regions, including the Cordillera Central in the Dominican Republic (Martin et al., 2007; Martin and Fahey, 2014), the Afromontane forests in Ethiopia (Schmitt et al., 2013), and in Hawaii (Crausbay and Hotchkiss, 2010). Upslope displacements of treelines have been observed over periods of 10 years or more (LoE1b) in Mexico (Jiménez-García et al., 2021), protected areas in the tropical Andes (Lutz et al., 2013), Taiwan (Greenwood et al., 2014), Hawaii (Koide et al., 2017) and Mount Kilimanjaro (Shugart et al., 2001), as well as changes in community composition that reflect upslope migrations of lowland plant species in Costa Rica (Feeley et al., 2013).

Yet, many species' responses to spatial changes in temperature and precipitation may not be occurring at a sufficiently fast pace to keep up with the rate of climate change (Feeley *et al*., 2013, 2011; Lutz *et al*., 2013; but see Lu *et al*., 2020). An analysis on avian seed dispersal in the Peruvian Andes concluded that several long-distance dispersal events would be necessary for the treeline to keep up with warming rates (Nowak et al., 2022). And even if that dispersal occurs, experimental studies have found that species-specific thermal and drought tolerances might influence seedling recruitment rates at higher altitudes (Esperón-Rodríguez and Barradas, 2014; Fadrique et al., 2018; Rehm and Feeley, 2016), potentially stymieing treeline expansion. However, an analysis of historic data of Taiwanese montane trees showed dissimilar responses at both intra- and interspecific levels; species already adapted to higher elevations moved upslope at higher rates, but these responses varied among life stages (O'Sullivan et al., 2021).

The main bulk of evidence of distributional changes comes from forecasting modelling studies (LoE2b), which overwhelmingly project range contractions, population declines, local extinctions or a combination of these in TMF tree communities (e.g. John *et al*., 2020; Neto dos Santos, Silva & Higuchi, 2020; Rojas Briceño *et al*., 2020) and herbaceous plants (Setyawan et al., 2020). These predicted range contractions are partially explained by the topography of mountains themselves because as species migrate upwards, the available area decreases. However, upward area reduction does not happen monotonically in over half of the world's mountainous regions (Elsen and Tingley, 2015), and the influence of topography is complex, in some cases potentially leading to horizontal rather than vertical displacements (Lippok et al., 2014).

Several studies have looked at the potential limiting factors for treelines to track new climatically suitable areas. These factors include frost (Joshi et al., 2020; Rehm et al., 2021; Rehm and Feeley, 2015), hydraulic stress (Song et al., 2016b), fruit or seed production (Chapman et al., 2018), seed dispersal (Hillyer and Silman, 2010; Nowak et al., 2022; Rehm and Feeley, 2013), germination rates (Center et al., 2016), and even the absence of nurse plants (Soto-Correa et al., 2013). By contrast, a field transplant experiment carried out in the Peruvian Andes concluded that soil was not a limiting factor for the establishment of trees at higher elevations (Tito et al., 2021). It has been suggested that plants that have evolved in nutrient-poor soils might be 'pre-adapted' to cope with other environmental stressors (Whitman et al., 2021). Thus, intra- and interspecific differential migration rates, in combination with other processes such as higher mortality of cold-resistant species and intrusion of lowland species into montane areas (e.g. de Gasper *et al*., 2021), may lead to the formation of new communities adapted to warmer regimes (Wright et al., 2009), i.e. thermophilisation (Duque et al., 2015; Fadrique et al., 2018).

While experimental studies to test responses of tree species to climate change rarely went beyond seed dispersal and seedling establishment, another defining component of the TMF flora, epiphytes, has been researched more often through experimental manipulations (LoE1a and LoE1b). Field transplant experiments to different elevations to simulate changing climatic regimes on bryophytes (Nadkarni and Solano, 2002; Song et al., 2012; Wagner et al., 2014) and ferns (Hsu et al., 2014) consistently found slower rates of growth and leaf production, and higher mortality, even if some species or individuals displayed some plasticity. Similar results were obtained from *ex-situ* experiments with both bryophytes and vascular epiphytes (Gotsch et al., 2015; Zotz et al., 2010). Epiphytes' reliance on different water sources seems to be linked to their taxonomic affiliation; Liu *et al*. (2021) found that in a Chinese subtropical montane forest, bryophytes and ferns obtained water both from humus and fog, whereas lichens and seed plants relied almost exclusively on fog. Although significant tolerance to desiccation (Bader et al., 2013) and temperature rise (up to an average of 3 °C) has been observed for some epiphytic species (Müller et al., 2017), their ability to track new climatically suitable areas was not experimentally tested.

d) Effects on fauna

Birds were the best-studied taxonomic group after all vascular plants (Figure **2.4**D). There was some empirical evidence (LoE1a and LoE1b) for the effects of climate change on avian species, such as recorded cases of upslope migrations in Honduras (Neate-Clegg et al., 2018), the tropical Andes (Forero-Medina et al., 2011; Freeman et al., 2018; Hayes et al., 2018; Hermes et al., 2018), Tanzania (Neate-Clegg *et al*., 2021*b*) and New Guinea (Freeman, 2016). Elevational shifts have also been reported for moth assemblages on Mount Kinabalu, Malaysia (Chen et al., 2009), and bats in Costa Rica (LaVal, 2004). However, these responses might be species specific (Anderson et al., 2013) and not all species can successfully expand their ranges (Campos-Cerqueira and Aide, 2017; Neate-Clegg et al., 2020). Even if elevational shifts do occur, they may result in intense competition for space and resources near mountaintops, triggering aggressive behaviours (Jankowski et al., 2010), or leading to higher morbidity (Freed and Cann, 2013) and mortality rates (Shiao et al., 2020).

Modelling studies (LoE2b) outline similar trends, overwhelmingly predicting range contractions, population declines, local extinctions or a combination of these in TMF birds (Colyn et al., 2020), mammals (Ramírez-Bautista *et al*., 2020; Raman *et al*., 2020*b*,*a*), amphibians (Cordier et al., 2020; Cruz-Elizalde et al., 2020) and scorpions (de Araujo-Lira et al., 2020). However, the influence of climate change on upslope migrations could be challenging to distinguish from that of land-use change (see Jacob *et al*., 2015*b*,*a*).

The prospect for ectotherms is a matter of much concern. Thermal tolerance experiments (LoE1b) conducted with amphibians (González-del-Pliego et al., 2020; Rueda-Solano et al., 2016; von May et al., 2017), reptiles (Muñoz *et al*., 2016; Piantoni, Navas & Ibargüengoytía, 2016; Strangas *et al*., 2019; but see Tao *et al*., 2021) and insects (e.g. Montejo-Kovacevich *et al*., 2020; Polato *et al*., 2018; Shah *et al*., 2017*b*) showed that tropical montane species are more susceptible to changes in temperature regimes relative to their lowland or temperate counterparts. However, some level of adaptive capacity was observed in some frog (von May et al., 2017) and ant (Nowrouzi et al., 2018) species, and in some cases, upslope migrations might be favoured by a relief from cold stress at higher elevations (Muñoz et al., 2022). Many tropical montane amphibians have also been severely impacted by the compounding effect of pathogens (e.g. Anchukaitis & Evans, 2010; Catenazzi *et al*., 2014, 2010; Neely *et al*., 2020), with long-term studies confirming substantial population declines and even extinctions in recent decades (e.g. Barker & Ríos-Franceschi, 2015; Campos-Cerqueira & Aide, 2017; Pounds, Fogden & Campbell, 1999). Amphibian microhabitats can offer some buffering from impacts of climate change, especially for arboreal species (Scheffers *et al*., 2013*a*,*b*, 2014), yet the overall expected outcome is loss of amphibian diversity at a global scale. Assemblages are projected to become much more homogenous due to selective extinctions of specialists (Menéndez-Guerrero et al., 2020) and the intrusion of invasive lowland generalists (Rödder, 2009).

Empirical evidence showing effects of climate change on other major animal taxa, such as mammals, reptiles and most invertebrates, is too limited to identify any clear patterns. For instance, a resurvey of tropical montane ants in Costa Rica concluded that over a decade, the community became less diverse, with upland areas becoming more similar to lowland ones (Warne *et al*., 2020) − suggesting that thermophilisation of communities is not limited to flora −, however, this finding was promptly contested (Klimes *et al*., 2021).

e) Effects on ecosystem functions

Ecosystem functions are commonly studied through experimental manipulation in the field (LoE1a) or laboratory (LoE1b). Multiple soil transplant and litter decomposition experiments have found that increasing temperatures, altered water status or both can change decomposition rates and have a negative influence on the capacity of TMF soils to retain organic matter, potentially turning the systems into carbon emitters (e.g. Becker & Kuzyakov, 2018; Looby & Treseder, 2018; Nottingham *et al.*, 2019*b*, 2016). Higher temperatures also make nutrients more readily available, with potential cascading effects on vegetation and other soil properties (Dantas de Paula et al., 2021). Some studies, however, have reached opposite conclusions or found no clear relationship between temperature or hydric regime and soil properties (e.g. He *et al.*, 2010; Scowcroft, Turner & Vitousek, 2000). Such equivocal conclusions could be due to the heterogeneity and localised nature of soil properties.

Montane trees are also expected to experience thermal and hydraulic stress under climate change. This may lead to increased respiration and reduced growth rates that translate into diminished carbon sequestration (e.g. Esperón-Rodríguez & Barradas, 2015*b*; Feeley *et al.*, 2020; Gutiérrez-García & Ricker, 2019). This vulnerability is partly due to the reliance of TMF plant communities on atmospheric water and the shielding effect of fog from direct solar radiation; both climatic factors at present attenuate vapour pressure deficit, to which TMF species are particularly responsive (e.g. Correa-Díaz *et al.*, 2020; Gotsch *et al.*, 2014*b*,*a*; Rodríguez-Ramírez *et al.*, 2020). These climatic factors also influence the growth rates of TMF trees ; but see Camarero *et al.*, 2021), seed production (Pau et al., 2020; Rodríguez-Ramírez et al., 2019), establishment (Chirino et al., 2017; Toledo-Aceves et al., 2019), and even plant–herbivore interactions (Bendix et al., 2021; Toledo-Aceves and del-Val, 2021a). The generalised persistent stress caused by climate change could explain high tree mortality events at a global scale (Allen et al., 2010). It could also be linked to a browning trend observed in pantropical forests since the mid-1990s (Krishnaswamy et al., 2014), or the contrasting greening trend during the past 20 years in central Mexico (Correa-Díaz et al., 2021). Such large-scale vegetation changes can further boost rising temperatures by altering the forests' albedo effect (Doughty et al., 2018) and render plant communities more susceptible to droughts.

2.5 Discussion

This review of the literature on impacts of climate change on TMFs shows that: (1) the rates of climate change are generally intensified rather than attenuated by elevation, making montane communities more susceptible to their effects; (2) tropical montane communities might be able to respond by shifting their distributions primarily upslope, but (3) not all species seem able to shift their distributions and the factors preventing them from tracking or establishing in new climatically suitable areas are unknown or not well understood; (4) even if montane species are able to track suitable conditions fast enough, they risk running out of physical space; (5) the impacts of climate change on genetic diversity and species interactions within tropical montane ecosystems remain largely unknown; and (6) the loss of biodiversity and functions of TMFs could result in the loss of valuable ecosystem services for human populations living close to tropical montane regions, with repercussions at a broader scale.

Additionally, this study revealed some significant knowledge gaps in several aspects (i.e. methodological approaches, geographical and taxonomic skews, and research topics) that need to be addressed, but also shows that there are some areas of opportunity, either expanding on the available knowledge, or by employing methodologies and data sources that have not been properly explored.

2.5.1 Methodological gaps and opportunities

The nature of climate change as a global and long-term phenomenon limits our ability to produce in the short term abundant empirical evidence of its effects in real time on particular ecosystem types or any of their functions and biotic components. Instead, our assessment of evidence strength showed that there has been greater reliance on study designs that yield moderate, inconclusive and circumstantial evidence. Given that conventional study designs that yield strong evidence tend to be time-consuming, effort-intensive and costly in remote mountainous regions, a practical strategy is to accumulate independent lines of moderate, inconclusive or circumstantial, yet coherent evidence that build up the same narrative. Thus, greater effort is needed to reconcile contradicting findings across different study sites and spatiotemporal scales, as well as attempts to disentangle the synergistic influences of multiple environmental factors on the diversity and functions of TMFs. Albeit likely geographically biased and difficult to interpret, long-term data sets are becoming increasingly abundant and accessible (Wauchope et al., 2021).

Additionally, environmental gradients along mountain slopes make TMFs ideal locations to conduct field manipulation experiments (e.g. transplant experiments along temperature gradients). Field transplant experiments are recognised as a powerful tool capable of yielding robust evidence by replicating complex projected environmental conditions more accurately than laboratory trials (Nooten and Andrew, 2016; Silveira et al., 2019; Tito et al., 2020). These types of experiments are useful to inform assisted migration programmes (e.g. Castellanos-

Acuña, Lindig-Cisneros & Sáenz-Romero, 2015; Sáenz-Romero *et al*., 2020*b*), changes in ecological networks (e.g. Maunsell *et al*., 2015), adaptation to urbanisation (e.g. Martin *et al*., 2021), among others. Alternative methods to produce strong evidence in the short term include analysis of historical remote sensing data (e.g. aerial photography and satellite imagery to detect changes in ecosystem boundaries and canopy spectral changes over time), as well as 'natural' long-term records (e.g. dendrochronological studies; Rodríguez-Ramírez *et al*., 2022).

Even though our review focuses exclusively on anthropogenic climate change, the potential value of palaeoclimatic and palaeoecological studies cannot be disregarded. In principle, studies of Quaternary-time and deep-time face similar challenges as long-term future-modelling studies, namely: "the impossibility of distinguishing between true and false" (Biondi, 2014, p. 1), and their findings should be interpreted cautiously. For instance, Fitzpatrick *et al*. (2018) projected that by 2090 (i.e. seven decades), climates in North America will have shifted by as much as they did during the past 13,000 years. Hence, it seems unreasonable to expect that species will be able to replicate in just a few decades past migrations spanning millennia. Conversely, a few decades might simply be too little time to detect distributional changes, especially for long-living, slow-growing organisms such as trees than can live for centuries. In fact, a common criticism of ecological niche modelling approaches is their underlying assumption that present distributions reflect the whole set of conditions in which a species can persist (Feeley & Silman, 2010*a*; Sax, Early & Bellemare, 2013), which is not necessarily true and needs to be accounted for. Some studies warn that relying exclusively on 'realised distributions' as input for predictive distributional studies could overly restrict potential future suitable habitats and overestimate risks of extinction and extirpation (Sax et al., 2013; Veloz et al., 2012). However, we argue that given the current rate of anthropogenic climate change, it is preferable to avoid overly optimistic assumptions that may lead to inaction, especially for montane ecosystems globally. Moreover, sets of good practices have been suggested to improve the accuracy of palaeoecological reconstructions (e.g. Nogués-Bravo, 2009) and a combination of short-term ecological studies with long-term palaeoecological evidence can help us to understand the impacts of climate change better (Lamentowicz et al., 2016). These research strategies would help resolve conflicting lines of evidence to enable rapid preventive and adaptive responses to climate change impacts on TMFs.

2.5.2 Geographical and taxonomic gaps and opportunities

Studies have been heavily concentrated in the Mesoamerican and Andean regions, both part of the Neotropical biogeographic realm. The fragmented evidence from other world regions suggests that other tropical montane regions may share similar climate change-induced impacts, albeit with some degree of local variation. Research efforts should be refocused on understudied regions to find out if there are any major discrepancies among them. Nonetheless, a few intensively researched areas, such as TMFs in Mexico, Costa Rica and Peru – representative of subtropical, mid-latitude tropical and nigh-equatorial TMFs, respectively – can be considered suitable proxies for environmental management while local studies elsewhere are in progress.

Most research on TMFs focuses on only a few taxa, yet these ecosystems are considered hotspots of biodiversity, much of which remains undescribed. For example, a recent survey of rove beetles (Staphylinidae), one of the largest families of organisms in the world, along an elevational gradient in Honduras found that they reached peak diversity precisely in the highly vulnerable TMF altitudinal belt (Dolson et al., 2021). Such lack of knowledge on biotic components of TMFs obscures our understanding of their ecological networks, ecosystem functions, and the magnitude of potential losses if cascading extinctions occur. Fortunately, surveys of soil, understorey and canopy biota can be carried out relatively quickly and are less costly and effort intensive than long-term monitoring or manipulative study designs. Additionally, knowledge biases towards charismatic taxa can be exploited to set up 'umbrella species' conservation schemes.

2.5.3 Thematic gaps and opportunities

Many studies support the notion that climate change will result in physiological pressures and distributional shifts of tropical montane communities, but assumptions of general climate-driven range shifts should be avoided (Rubenstein et al., 2020). Empirical evidence also shows that climatic conditions can impede the effective establishment of tree communities (Joshi et al., 2020; Rehm and Feeley, 2013; Song et al., 2016b) and likely other components of tropical montane biota. In fact, species' ability to persist or migrate is influenced by their interactions with other ecosystem components, both biotic and abiotic (Jankowski et al., 2010; Joshi et al., 2020; Quiroga et al., 2018; Ramirez-Villegas et al., 2014), but few studies have looked at ecological networks in TMFs (Benning et al., 2002; Hillyer and Silman, 2010; Jankowski et al., 2010; Ornelas et al., 2018; Tito et al., 2021). Improving our understanding of these interactions would help improve the accuracy of forecasts both in terms of distributional responses and potential future assemblages. Although field studies are needed to elucidate how tropical montane networks respond to climate change and other disturbances, existing databases can be used to construct and conduct robust analyses on ecological networks (e.g. de Almeida & Mikich, 2018; Fricke & Svenning, 2020), and project their responses under climate change scenarios.

More concerning is the paucity of studies at the genetic level in TMF research in relation to climate change, which has been previously acknowledged (Pauls et al., 2013). For example, tropical tree populations are experiencing genetic bottlenecks following intense disturbance events, but tropical montane regions are understudied (Pautasso, 2009). For mountainous regions, this might be crucial because microevolutionary processes operate differently within a population along an elevational gradient, i.e. the leading edge, the central population and the rear edge (Kremer et al., 2014). Sudden disturbance-induced migrations may lead to decreased phenotypic variability, further jeopardising the plasticity and ability of trees both to reach and to establish populations in new areas (Pertoldi et al., 2007). Also, highly variable conditions may not result in an adaptive response because selection processes are multi-directional and the existing genetic variation in a population might be insufficient to generate the genotypic combinations required for it to persist under new environmental conditions (Alfaro et al., 2014). As the most conspicuous biotic component of forests, declines of trees could trigger negative cascading effects (Bawa and Dayanandan, 1998; Nagel et al., 2019). Thus, the genetic status of trees is a factor that ought to be taken into account for assisted migration programmes (Alfaro et al., 2014), and all these considerations are equally valid for other taxa. Unlike other knowledge gaps, the scarcity of studies at the genetic level is difficult to overcome though indirect and remote methods, however, the increasing accessibility and affordability of sequencing methods should facilitate extensive genetic surveys of tropical montane populations in the short term.

2.6 Recent work

There have been recent developments since our systematic review was carried out. For example, Rico *et al.* (2023) reported that the genetic health of a tropical montane tree species in Mexico is threatened by human activity and climate change.

Research on treeline expansion has also shown that dry conditions in the boundary between TMF and *páramo* in the Venezuelan Andes support a seedling bank, but with slow growth (Ramírez et al., 2022); and some species may need treatment, such as scarification, to increase germination rates (Liyanage et al., 2022).

Thus, understanding the genetic adaptations and environmental influence of seeds and seedlings is crucial for conservation efforts. However, long-term monitoring is also necessary as some abiotic effects, such as soil properties, may not be noticeable for years (Martínez-Ramos et al., 2022).

Lastly, new knowledge of diversity patterns and spatial partitioning in various regions (Berrios et al., 2022; Morton et al., 2022) emphasise the importance of traditional biodiversity surveys in TMFs, especially in understudied regions.

2.7 Conclusions

(1) We highlight the long-term need to widen the methodological, thematic, taxonomic and geographical scope of studies on TMFs under climate change. In the short term, however, the accumulation of moderate to circumstantial evidence constitutes the most accessible and reliable tool to address uncertainties and gaps in current knowledge. As such, in-depth research in well-studied regions, use of alternative data sources (remote sensing and 'natural' long-term records), palaeoecological supporting evidence and advances and refinements of forecasting modelling techniques offer the most reliable and immediate sources of information for expeditious conservation action for these threatened forests.

(2) Natural variability within TMF regions represents a challenge for the generalisation of the findings of individual studies, but it simultaneously represents an exceptional opportunity to generate high-quality evidence of the impacts of climate change for both tropical montane species and lowland species. Environmental gradients along mountain slopes have been identified as natural laboratories, where field manipulation experiments (e.g. field transplant

experiments, rain exclusion experiments, etc.) can be conducted in the short term to simulate complex projected environmental conditions with greater accuracy than can be achieved *ex situ*.

(3) We highlight the importance of modelling approaches in TMF research and encourage further refinement and development of these methods. To enhance their effectiveness, novel, more robust forecasting algorithms should be developed to account for uncertainties and sampling biases. Additionally, incorporating ecological information, such as species dispersal limitations, biotic interactions, and analyses of ecological networks, can make commonly used modelling approaches more informative. Optimizing these approaches with more ecological information is crucial for the success of conservation strategies, such as the design of protected areas that consider future suitable habitats for whole biotic communities, and minimizing losses of biodiversity and ecosystem functions.

(4) Despite the undeniable importance of trees in forests, the responses of other taxonomic groups to climate change should not be overlooked, as intraspecific interactions could prove decisive for the success of conservation measures. Similarly, the impacts of climate change at the genetic level remain largely unknown and the loss of genetic diversity can threaten the long-term viability of TMF populations.

(5) We urge scientists to conduct similar evidence quality assessments in their respective fields. Experts in each area of research should critically ponder what study designs and data sources yield the most robust body of evidence and take them into consideration when carrying out reviews and planning future research.

Chapter 3. Distributional Responses of Mesoamerican Tropical Montane Tree Communities under Climate Change

3.1 Abstract

Tropical Montane Forests are an ecosystem type with high ecological and cultural significance for their biodiversity and ecosystem services but that is threatened by climate change. In Mesoamerica, Tropical Montane Forests extend from the central mountain ranges parallel to the east and west coasts of Mexico, southwards into Central America, with a wide latitudinal discontinuity created by Lake Cocibolca in Nicaragua, and a smaller one at the Isthmus of Tehuantepec. I provide a latitudinal and altitudinal characterisation of the montane-specialist tree community from Mexico to Panama, based on the occurrence records of 272 species, that will serve as a baseline for conservation efforts of this biome. Furthermore, I employ ensemble Species Distribution Models to predict climatically suitable areas for this montane-specialist community under current (1981–2010 baseline) and future conditions by the end of the century (2071–2100) under a severe emissions scenario (RCP8.5). Areas with suitable climatic conditions for a high number (≥50) of montane-specialist species currently occupy ~136,000 km^2 out of ~1,236M km^2 of aggregated suitable areas for the 272 species. Climatically suitable areas could be reduced by >60% by the end of the century due to severe climate change. Species turnover will also be affected, being more pronounced at low (<1,000 m asl) and mid elevations (<1,500 m asl), indicating species migrating upslope, and the highest communities remaining largely unchanged. The potential impacts on montane-specialist tree species were mainly driven by latitude, with greater losses in the northern part of the study area, with a compounding effect of increasing elevation. The severity of suitable habitat loss will be proportionately greater for already restricted species, which highlights the patchy nature of montane communities and the likely inefficacy of passive conservation policies, such as protected areas. I thus call for those policies to be reinforced and supplemented with active conservation efforts such as seed banks and assisted migration programmes.

3.2 Introduction

Tropical montane forests (TMF) are highly vulnerable to environmental changes because their extent and distribution are constrained by topographic boundaries (Salinas et al., 2021).

In addition to threats common to most terrestrial ecosystems, such as agricultural expansion and urban encroachment, TMF communities face unique stressors for mountainous areas, such as fog cover reductions and subsequent increases of solar radiation and vapour pressure deficit (Christmann and Menor, 2021). Not even remote mountainous areas free from direct human disturbance are spared, as there is little evidence that undisturbed vegetation provides additional mitigation to global warming (Trew et al., 2024). Within the tropics, Latin America, and especially Mesoamerica, have been identified as highly vulnerable regions to environmental change (Pacifici et al., 2015; Trew et al., 2024).

Mesoamerica, extending from the northern mountains of Mexico through Central America, is a bio-cultural region that hosts remarkably high biological diversity, the product of its recent geological history (Cody et al., 2010). TMFs are found in Mexico and Central America, and cover ~56,000 km^2 out of ~401,300 km^2 in the Neotropics (Helmer et al., 2019). Despite their smaller size compared to South American TMFs, Mesoamerican montane regions are critical for their unique biodiversity and as a convergent zone between Nearctic and Neotropical biotas (Corral-Rosas and Morrone, 2016). This region is also of great cultural and agroecological importance because it is inhabited by ethno-linguistically diverse indigenous groups and is centre of diversity and domestication of several world staple crops (Boege et al., 2008).

Mesoamerica is projected to be severely impacted by climate change by the end of the century (Williams et al., 2018), leading to 60% or more habitat losses (Altamirano-León et al., 2022; Ortega et al., 2024; Ramírez-Amezcua et al., 2016) that could threaten key natural protected areas (Esperón-Rodríguez et al., 2019). Furthermore, large climatically suitable areas for TMF species do not fall within current protected area schemes (Jiménez-García and Peterson, 2019; Rojas-Soto et al., 2012; Sierra-Morales et al., 2021). Understanding how Mesoamerican TMFs will respond to climatic stressors is complex due to variations in species sensitivity (Esperón-Rodríguez and Barradas, 2014) and varying quality of the research evidence (Mata-Guel et al., 2023).

The high heterogeneity of montane regions in Mesoamerica has led to inconsistent classifications of its vegetation. For example, the Mexican government conflates TMFs with the highly restricted 'mountain mesophyll forest' (INEGI, 2024; https://www.inegi.org.mx/temas/usosuelo/#mapas), although mountainous regions in

Mesoamerica harbour a wider variety of assemblages, including pine, oak, mixed-broadleaf, seasonally-dry, among other forest types (Gual-Díaz and Rendón-Correa, 2014). Additionally, in its southern part, it is not clear whether Mesoamerican TMFs form one continuous community or are disrupted by Lake Cocibolca in Nicaragua (Morrone, 2020).

Mapping TMFs' extent and distribution has been equally challenging. Species Distribution Models (SDM) are an essential tool for projecting current areas of TMFs and predicting possible future changes in response to environmental changes (Araújo and Peterson, 2012; Elith and Leathwick, 2009; Pacifici et al., 2015). However, varying methodological choices, data availability, assumptions of species' representativeness of the whole biome, among other biases, have resulted in inconsistent estimates of Mesoamerican TMFs extent and distribution (Alfonso-Corrado et al., 2017; Gómez-Pineda et al., 2020; López-Arce et al., 2019; Rojas-Soto et al., 2012).

To address these challenges, I propose a novel and easily replicable and expandable methodology to assess the extent and distribution of TMFs under current and future climatic conditions, using the montane-specialist tree subset of Mesoamerican TMFs as a case study. By characterising the montane-specialist tree community without *a priori* assumptions of their representativeness of TMFs, this approach allows to forecast changes and make inferences related to community composition in addition to distributional shifts and allows to easily expand the approach to incorporate more community elements and taxa.

Thus, my study aims to contribute to a wider understanding of Mesoamerican montane-specialist trees and their responses to global change by estimating their current and future extent and distribution in Mesoamerica using ensemble SDMs with straightforward reproducibility in other communities and world regions. The specific objectives are to: (1) characterise the montane-specialist tree community beyond political boundaries with Mesoamerica as a case study; which includes determining whether its southern limit extends to the Panama-Colombia border or is cut off by Lake Cocibolca; (2) estimate the spatial extent and distribution of montane-specialist trees in Mesoamerica; and (3) assess the potential impacts of projected climate change on that montane-specialist tree community in terms of extent, distribution, and species richness by the end of the century.

3.3 Methods

3.3.1 Study area and data sources

I focused on the Mesoamerican region, covering the portion of Mexico below 25° N and Central America, and combined the tropical montane region estimates by Villaseñor (2010) and Los et al. (2021), allowing some buffer area north and south into the USA and Colombia, respectively (Figure **3.1**) to capture potential unoccupied climatically suitable areas.

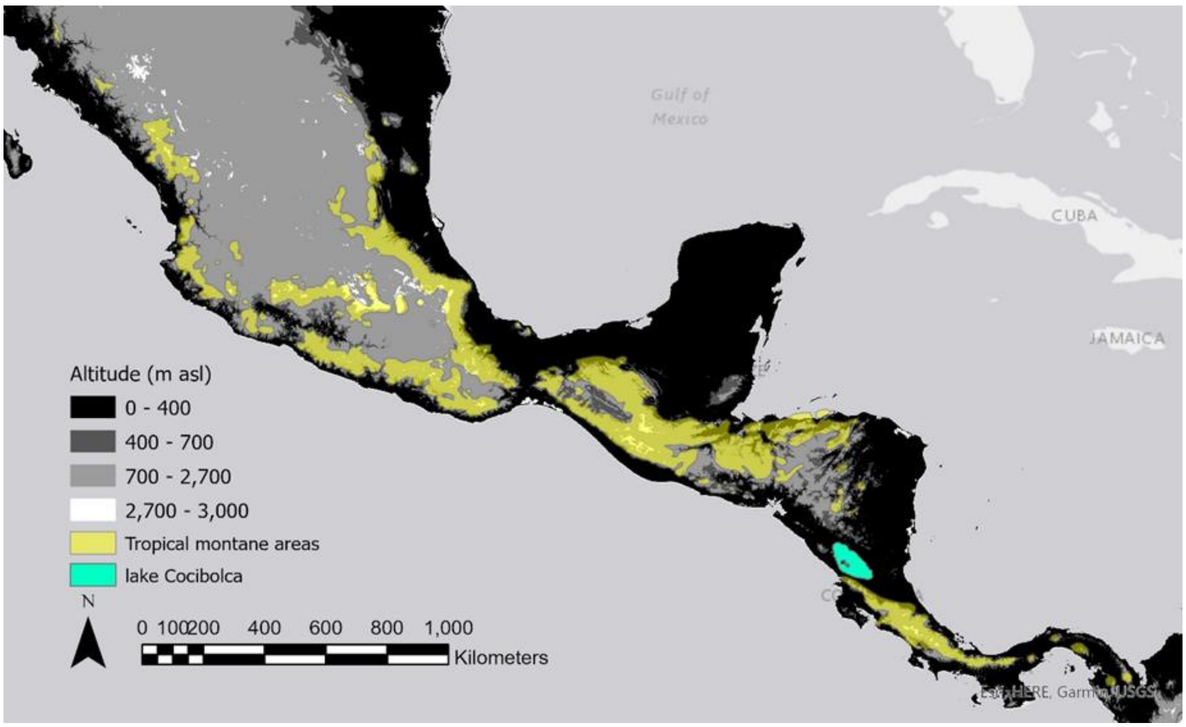

Figure 3.1 The study area. The altitude categories are based on historically reported elevational limits of TMFs in Mesoamerica, typically located between 700 and 2,700 m asl, with rare occurrences as low as 700 m asl, and as high as 3000 m asl (Ressl and Morales, 2008). The distribution of tropical montane forests is shown in yellow, based on the combined estimates by Villaseñor (2010) and Los et al. (2021). Lake Cocibolca (in light blue) creates a latitudinal discontinuity for montane habitats, but Costa Rica and Panama are included nonetheless as part of the Mesoamerican bioregion.

Occurrence records. I compiled a harmonised list of tree species from national catalogues and inventories from Mexico and Central America (*n* = 5,307 species) (Table S2), excluding Belize, whose territory falls entirely outside of out tropical montane polygons (Figure **3.1**). I downloaded the species (*n* = 4,342) occurrence records from GBIF (www.gbif.org; 21st June 2022, https://doi.org/10.15468/dl.dsbdrh) with filters for accepted names and exact matches, and intertropical latitudinal limits of 30°N and 30°S (*rgbif* R package; Chamberlain et al., 2017). To keep montane-specialist species only, I filtered the occurrence records in

ArcMap 10.8 by the tropical montane polygons in Figure **3.1**, retaining species with ≥75% (upper quartile) of records inside the polygons. I thinned occurrence points to 1 km^2 using the *spThin* package (Aiello-Lammens et al., 2019), and discarded species with <20 usable occurrence records (Table S3). Finally, I verified the growth forms of the remaining species from digitised herbarium specimens available in GBIF to keep only tree species, although some can also occur as shrubs. The final dataset included 272 montane-specialist tree species and 15,555 occurrence records (Tables Table **S4** and Table **S5**).

To test whether there is one continuous montane-specialist tree community along the whole latitudinal range or if Lake Cocibolca imposes a barrier, I used the Bray-Curtis dissimilarity index above and below the 12° N parallel (approximate mid-section of the lake). I also tested the second widest topographic discontinuity (terrain <400 m asl) in the Isthmus of Tehuantepec (~16.5° N; Figure **3.1**), even though it occurs on a longitudinal axis and, to my knowledge, does not represent a significant dispersal barrier.

Environmental layers. I downloaded baseline (1981–2010) climatic variables from CHELSA v 2.1 (http://chelsa-climate.org/bioclim/, Karger et al., 2017), since it has been reported to perform slightly better than other sources in mountainous regions (Bobrowski et al., 2021; Noce et al., 2020). I initially retrieved 19 bioclimatic layers plus frost change frequency and net primary productivity, which could have biological significance for TMFs. I downloaded WorldClim's digital elevation model (www.worldclim.org/data/worldclim21.html, Fick and Hijmans, 2017) and used it to derive aspect, slope, and ruggedness in ArcMap 10.8. I obtained soil types from the FAO's Harmonized World Soil Database v 1.2 (Fischer et al., 2008). All environmental layers had a 30-sec resolution and WGS84 projection.

Variable selection. I clipped all layers and occurrences to the study area and used the Spearman test for raster objects using the *Correlation* function in *ENMTools* (Warren et al., 2019) to check for collinearity. For variable selection, I then ran pilot tests using MaxEnt (Phillips et al., 2006) with combinations of non-correlated environmental variables on the 20 species with most records, the 20 with fewest records, and 20 random species to ensure good representation of the community. After removing low-contributing and highly correlated variables (>|0.75|), the retained explanatory variables were slope, ruggedness, soil type, temperature seasonality (BIO4), mean temperature of the wettest quarter (BIO8), mean temperature of the driest quarter (BIO9), precipitation of the driest month (BIO14), precipitation of the wettest quarter (BIO16) and precipitation of the coldest quarter (BIO19).

Future climate change scenarios. To forecast the effects of climate change on the distributions of the tropical montane trees, I employed the RCP8.5 scenario (rising radiative pathway leading to 8.5 W m^{-2} [~1,370 ppm CO_2 eq] and 3.7±1.1°C temperature rise by 2100; Leggett, 2021) by the end of the century (2071–2100). I chose the long-term severe pathway because observed emissions and existing global fossil fuel infrastructures already surpass emission thresholds of moderate pathways (Dhakal et al., 2022), and tropical montane trees have been reported to migrate more slowly than observed rates of climate change (Alexander et al., 2018; Corlett, 2015; Corlett and Westcott, 2013). I downloaded the projections for the six selected bioclimatic variables from CHELSA under the and the upgraded versions of three General Circulation Models recommended by the Mexican government (https://www.gob.mx/inecc/acciones-y-programas/escenarios-de-cambio-climatico-80126) or previously used for the Mesoamerican region (e.g., Esperón-Rodríguez et al., 2019; Helmer et al., 2019; Ramírez-Bautista et al., 2020; Williams et al., 2018): GFDL-ESM4 (Dunne et al., 2020), MPI-ESM-HR (Müller et al., 2018) and UK-ESM1 (Sellar et al., 2019).

3.3.2 Species distribution modelling

I generated SDMs under baseline (henceforth 'current') and future conditions for the 272 tropical montane tree species, using an ensemble of four complementary algorithms (GAM, ANN, GBM and MaxEnt; Guisan et al., 2017) in the *biomod2* package (Thuiller et al., 2016). I kept the spatial resolution at 30-arc seconds because coarser resolutions could miss important details in narrow TMF areas.

Due to the size of the study area and the number of species, I used the default parameters of the algorithms, whilst relying on the high number of species to dilute potential biases of any individual tree species. Therefore, the robustness of this models is based on the number of species used to model the Mesoamerican tropical montane tree community.

I parallelised the process on the Iridis5 High Performance Computing Facility of the University of Southampton, using the *foreach* (Weston, 2019) and *doParallel* (Calaway et al., 2015) packages (scripts available in SI). I divided the species into those with <50 occurrences (*n* =

168) and ≥50 occurrences (n = 106), and then into 11 batches of ~30 species. Each batch took ~20 processing hours.

For each species, I created 1,000 random background (pseudoabsences) points. To validate models, I used 5 cross-validation replications for species with <50 records and 10 for species with ≥50 records. I combined the model outputs of the repetitions (5 or 10) and four algorithms and three GCMs for every species (*biomod2* 'PA+run' assembly rule), using a TSS > 0.5 (true skill statistic, Allouche et al., 2006) threshold for model inclusion. I overlapped the final ensemble present and future maps to analyse shifts in climatic suitability between present and future conditions. I generated categorical maps for each species with four possible values: −2 predicted to be lost, −1 predicted to remain occupied, 0 predicted to remain unoccupied, and 1 predicted to be gained (Thuiller et al., 2023).

3.3.3 Post-SDM Analysis

I imported each species' ensemble map under current and future conditions to R v4.1.1, using the *raster* package (Hijmans, 2020), and stacked them to generate current and future species richness maps and calculate predicted changes in richness between the two time periods. I converted the current and future richness maps to a projected coordinate system to estimate their extent in km^2. To estimate turnover per pixel (i.e., each pixel as a site), we generated binary presence matrices, with species arranged in columns and pixels in rows. I used the *betapart* package (Baselga et al., 2018) to estimate the Jaccard dissimilarity index for each pixel, excluding those with incomplete data, i.e., pixels not containing any species in one of the two time periods to compare. I converted the turnover matrix into a map using the *raster* package.

To test whether projected distributions were associated with phylogenetic affiliation, I ran correlation tests between projected extent under current and future conditions and taxonomic groups at the family and order levels. Since Conifers and Angiosperms tend to form distinct communities that are often listed as separate assemblages in national inventories (e.g., pine forests distinct from oak forests, broadleaf forests, etc.), I tested differences between both clades per species (i.e., not aggregated) in terms of projected current and future extent, as well as percent range change, using the non-parametric Kruskal-Wallis test.

To characterise the Mesoamerican TMF tree communities throughout the study area and assess the changes in their suitable range sizes, I classified species based on their projected current latitudinal and altitudinal ranges. For the latitudinal range, I used 17.5° N (approximate northern limit of the Isthmus of Tehuantepec, where mountainous regions extend longitudinally rather than latitudinally) and 12.5° N (north of lake Cocibolca) as dividers for 'northern' (N), 'central' (C), and 'southern' (S) species. I labelled species spanning two contiguous areas 'northern–central' (N – C) or 'central–southern' (C – S), and 'widespread' (W) those with even broader ranges. The boundaries between altitudinal subtypes of TMFs (i.e., lower montane, upper montane and alpine) vary from 1,200–1,500 m asl (lower boundary) to 2,000–3,000 m asl (upper boundary), and are influenced by the height of the mountains and distance from the sea (Scatena et al., 2010). Thus, for altitudinal range, I based the classes on interquartile ranges into 'very low' (VL), 'low' (L), 'mid-low' (ML), 'middle' (M), 'mid-high' (MH), 'high' (H), and 'very high' (VH) (detailed criteria in Table S6). I used these classes to generate a species-level characterisation of the montane tree community in Mesoamerica.

To test an interactive effect of latitude and altitude on forecast climatic suitability changes, I simplified the classes above to ensure every class had enough species to compare and eliminate overlapping classes. Latitude classes were thus reclassified as 'northern' ('N' ∪ 'N – C'), 'centred' ('C' ∪ 'W') and 'southern' ('C – S' ∪ 'S'), and altitudinal classes as 'low' ('VL' ∪ 'L'), 'mid' ('ML' ∪ 'M' ∪ 'MH') and 'high' ('H' ∪ 'VH'). I ran GLMs with these broad classes on percentage loss and log-transformed absolute loss in suitable range size, using the Gamma family with a log function for positive continuous data and to account for overdispersion of the data. Finally, I ran a linear regression of percent losses in response to log-transformed current projected size of suitable areas to test whether more restricted species will be more severely impacted than widespread ones. I report range sizes in km^2 but used current and future projections in their original 30 arc-seconds projections (i.e., pixels) for statistical tests to avoid errors caused by rounding while reprojecting to a 1 km^2 grid. All stats were done in R v.4.1.1 and the scripts are available in SI.

3.4 Results

3.4.1 Characterisation of the Mesoamerican TMF tree communities

Costa Rica had the highest number of TMF tree species (192) and almost as many occurrence records (6,180) as Mexico (155; 6,523) despite its considerably smaller size (Table S5). The distance between the southernmost TMF regions in Nicaragua and the northernmost TMF areas in Costa Rica is approximately 200 km. The Bray-Curtis index (BC = 0.799) indicated that there are two distinct montane-specialist tree communities separated by Lake Cocibolca. This is also evident in the number of occurrence records exclusive to Costa Rica and Panama, with 89 species of my dataset recorded in one or both countries and only four species shared between Costa Rica and Nicaragua. In contrast, the Bray-Curtis index seems to indicate another latitudinal discontinuity at the Isthmus of Tehuantepec (BC = 0.690), however, the subcommunity south of the isthmus harboured 264 out of the 272 species, whereas north of the isthmus there were 151 species, only 8 of which were exclusive (Table S7). Therefore, the dissimilarity index in this case showed that the subcommunity south of the isthmus was nearly twice as diverse as the northern one.

Based on the projected suitable habitats of 272 species under current conditions, I characterised the Mesoamerican montane-specialist tree community along altitudinal and latitudinal gradients (Figure **3.2**). I grouped the projected suitable areas for the species by elevation and latitude, with a separate category for widespread species. Most species (darker classes in Figure **3.2**) concentrated at low and mid elevations in the central and southern parts of the study area, indicating higher richness at lower latitudes. Few species fell neatly into the middle categories (mid elevation and central part of the range) and tended to skew instead into lower or upper elevations and northern or southern areas. There were suitable conditions spanning the whole latitudinal range for 51 species. Note that in Mexico, the northern montane areas are divided into two ranges along its east and west coasts, separated by the central highlands. Therefore, species categorised as northern might not necessarily co-occur longitudinally.

Figure 3.2 Characterisation of Mesoamerican montane-specialist tree subcommunity (N = 272) by latitudinal (N = north, N – C = north-central, C = central, C – S = central-south, S = south, and W = widespread) and altitudinal (VH = very high, H = high, MH = mid-high, M = mid, ML = mid-low, L = low, and VL = very low; see Table S6) classes. Darker colour indicates projected suitable climatic conditions for higher number of montane-specialist tree species.

3.4.2 Suitable area projection of the Mesoamerican montane-specialist tree community and predicted impacts of climate change

The aggregated extent of suitable conditions for Mesoamerican montane-specialist tree species was projected to decrease under future conditions from ~1,236M km^2 suitable for the 272 species included in the analysis, to ~970,000 km^2, equating to a loss of 21.5% (Figure **3.3** and Table S8). The extent of losses of climatically suitable areas would likely be greater because many areas predicted to become suitable for a few montane-specialist tree species are in low-elevation coastal regions and the Yucatán Peninsula (Figure **3.4**A and B), which do not qualify as TMFs. Future projections forecast an increase in low-diversity areas, with areas predicted to have suitable conditions for 1–5 tree species increasing from 548,061 km^2 currently to 648,145 km^2 (+18.3%) by the end of the century (Figure **3.3**). This represents a projected expansion of a handful of species into lower elevations and a wider latitudinal range, rather than of the whole or even partial tree community. In contrast, areas with higher diversity (predicted to be suitable for >5 tree species) show consistent declines in suitable range (Figure **3.3**; Figure **3.4**A and B).

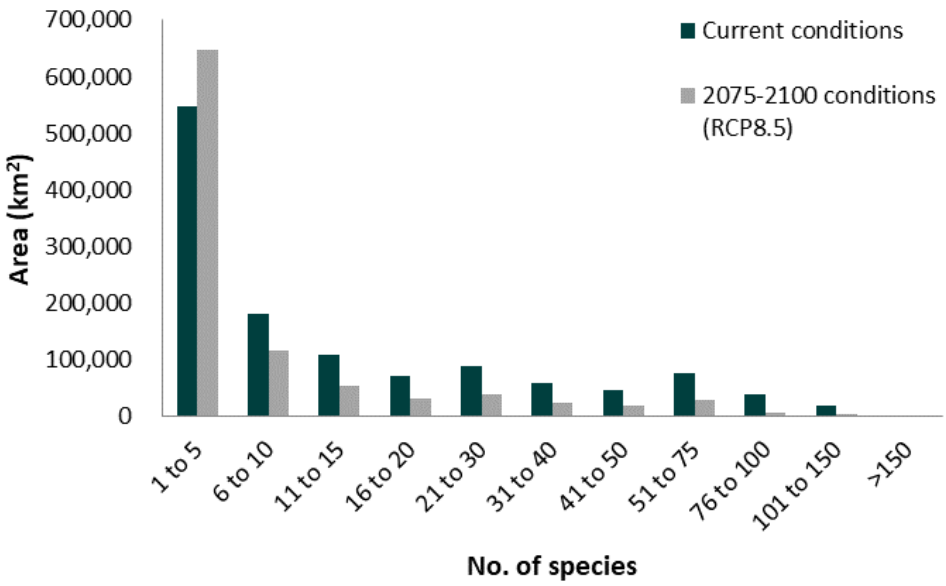

Figure 3.3 Projected climatically suitable habitat extent for the 272 montane-specialist trees by number of species under current (1981–2010 baseline) and future (2071–2100) conditions under a severe climate change scenario RCP8.5.

With few exceptions, montane-specialist tree species richness is predicted to decline under future conditions in almost all the TMF area in Mesoamerica (Figure **3.4**C), being more severe at higher montane elevations, with some high-altitude areas projected to become unsuitable for >50 species. In contrast, most areas predicted to gain in suitability for some species are lowland regions where there is no TMF, but that could allow for the establishment of a handful of species. In these cases, the expected gains in climatic suitability are for <5 species. Some small TMF areas that could see an increase in suitability for montane-specialist species are in the southernmost parts of the Mesoamerican bioregion, mainly Costa Rica and Panama, and even into the Andean region of South America, as shown by the southern buffer area in Figure **3.4**A.

The composition of the montane-specialist community was also predicted to change. The beta diversity analysis (Jaccard turnover index; Figure **3.4**D) showed that for most low- and mid-elevation TMF areas, the degree of dissimilarity between current and future conditions was higher than at the highest elevations, indicating that species currently at lower altitudes would experience proportionally larger (likely upslope) distributional shifts, changing the species assemblages along the way. The montane-specialist tree community compositions at the highest elevations were projected to remain relatively unchanged.

I did not find significant associations between taxonomic groups at the family and order levels with current distributional extent nor projected changes. I found that under current conditions and on a species-by-species basis (i.e., not aggregated), Conifers had on average larger areas with suitable climatic conditions than Angiosperms (Kruskal-Wallis chi-squared = 5.1306, d.f. = 1, p = 0.0235). However, I did not find any difference in habitat suitability between these groups under future conditions (chi-squared = 2.6709, d.f. = 1, p = 0.1022). There were no differences in the predicted percent losses in range suitability between groups (p = 0.8473).

I found moderate evidence that both elevation and latitude could jointly influence the magnitude of losses in climatically suitable areas (Figure **3.5**). 'Southern' (i.e., <12.2 °N) montane-specialist tree species were projected to experience the smallest losses, especially at higher elevations. Losses were greatest among species with predominantly suitable areas at mid latitudes ('centred', i.e., 12.5 °N to 17.5°N) and intermediate among 'northern' species (>17.5 °N). There was a strong interaction between the linear components of both latitude and altitude (t = -3.94, p = <0.001) on log-transformed absolute losses; however, I found no effect of other combinations of the linear and quadratic components of latitude and altitude. Both the linear (t = -5.60, p = <0.0001) and the quadratic (t = -6.13, p = <0.0001) components of latitude had a strong effect on predicted range losses, while only the quadratic component of altitude had a moderate decelerating effect on range losses (t = -3.01, p = <0.002), except for the 'northern-low' species. This suggests that latitude will have a more pronounced effect on range shifts than elevation, as shown with the elevation classes 'following' an arched pattern along the latitudinal axis of Figure **3.5**.

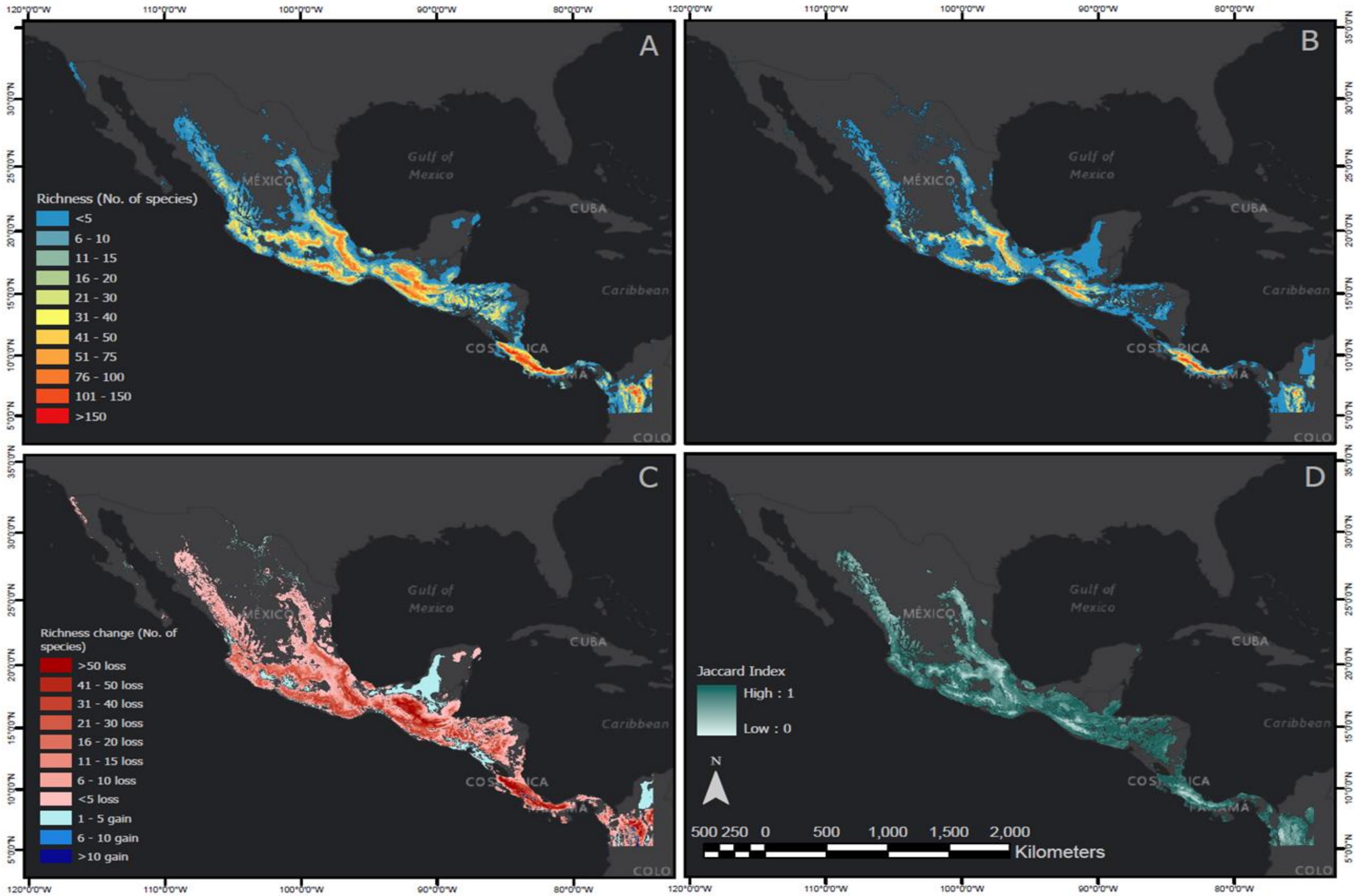

Figure 3.4 Number of montane-specialist tree species (richness) projected to have suitable habitat with current (A) and future (2071–2100) climatic conditions under RCP8.5 (B); change in climatic suitability from current to future conditions (C); and species turnover estimated with the Jaccard similarity index from current to future conditions (D).

The percent loss in predicted climatically suitable areas was negatively influenced by the projected range of the species. I found moderate evidence that the magnitude of projected percent losses of suitable area decreased with projected extent under current conditions (α =136.2, p <0.001; β = -5.85, p < 0.001; $F_{1,265}$ = 81.35, p < 0.001; R^2 = 0.23; Shapiro-Wilk's W = 0.992, p = 0.173; Fig. S1). This means that species with currently restricted distribution are more likely to experience the greatest proportional suitable habitat losses under future climate conditions, although the high variability in my models suggests that other factors could drive changes in range suitability in relation to the size of current suitable areas.

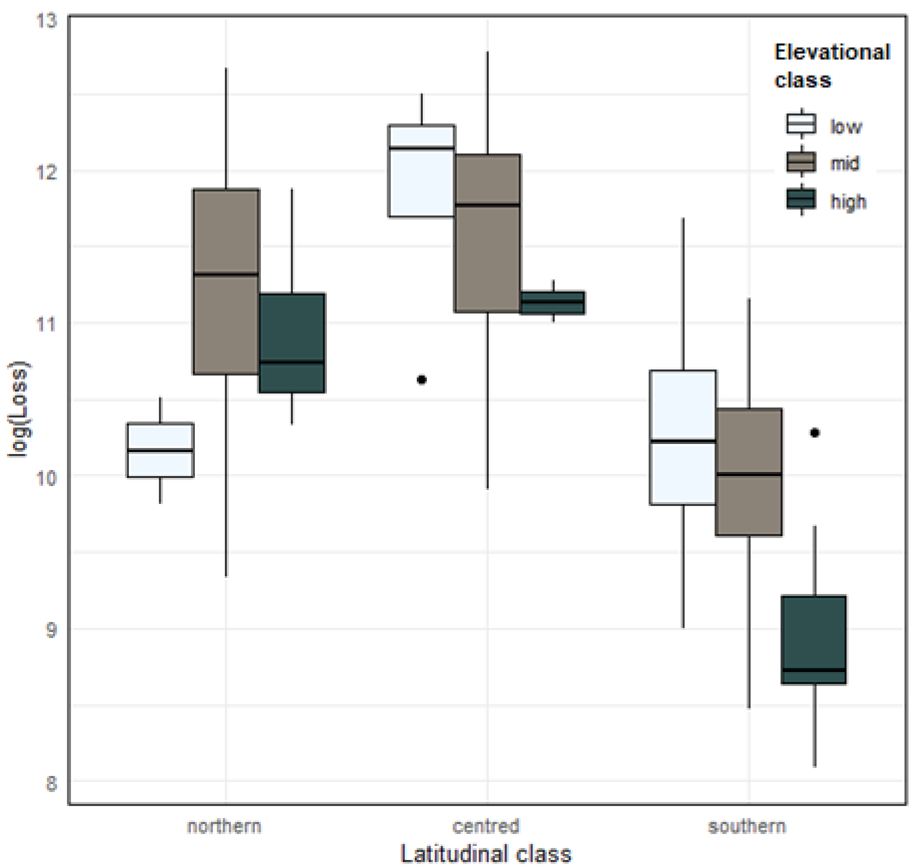

Figure 3.5 Interaction of latitudinal and altitudinal classes of Mesoamerican MST species on projected range losses by the end of the century under RCP8.5. Boxes indicate the interquartile range (25th and 75th percentiles) and the whiskers extend to the 10th and 90th percentiles.

3.5 Discussion

I conducted an assessment of the distribution of montane-specialist tree species in Mesoamerica based on climatic suitability with a transparent, replicable, and transferable methodology for other biological communities, world regions and ecosystems. The results indicate that Lake Cocibolca splits the Mesoamerican montane-specialist tree community into two distinct subcommunities, while species richness significantly decreases north of the Isthmus of Tehuantepec. Additionally, I characterised the montane-specialist tree community at the species level across latitudinal and altitudinal gradients in Mesoamerican montane areas, which is lacking from national inventories and could serve as a baseline for further research and conservation policy of these key ecosystems. Finally, my models predicted that climate change will lead to loss of suitable area and high species turnover at low and mid elevations of Mesoamerican montane-specialist tree species by the end of the century.

3.5.1 The Mesoamerican montane-specialist tree community

My estimate of the current extent of Mesoamerican TMFs of ~1.236M km^2 based on the combined suitable areas of 272 montane-specialist tree species is higher than previous estimates. These range between ~627K km^2 (Karger et al., 2021) and ~2M km^2 (Los et al., 2019; Mulligan, 2010) globally for the most restrictive and broadest definitions, respectively. The discrepancies with previous estimates arise from aggregating the predicted suitable areas of the 272 species, as some of those areas are suitable for only a handful of species and cannot confidently represent a TMF community. By increasing the threshold of species considered as indicative of TMF presence – i.e., the number of montane-specialist species with projected suitable climatic conditions per site (pixel)–, my estimates align more closely with previous studies. For example, areas suitable for ≥15 montane-specialist tree species amount to 400,021 km^2, and for ≥50 species to 136,199 km^2. These estimates, although based on a reduced number of montane-specialist tree species, are more consistent with regional estimates for the biome, e.g., ~401,300 km^2 of TMF cover for the entire Neotropics (Helmer et al., 2019) and ~139,400 km^2 for Mexico (Cruz-Cárdenas et al., 2012). The remaining differences between estimates are due to varying definitions of TMFs and the use of different methodologies (Mata-Guel et al., 2023), as my study does not restrict TMFs to areas frequently covered by ground-level clouds (i.e., cloud forests).

The high dissimilarity between montane-specialist tree communities north and south of 12° N suggest that Lake Cocibolca acts as a topographic obstacle for dispersal, marking the southern boundary of the montane-specialist component of Mesoamerican TMFs. In contrast, studies that focus on conservation policies, such as the Mesoamerican Biological Corridor (a chain of natural protected areas covering mainly lowland tropical forests), extend the Mesoamerican bioregion to Costa Rica and Panama (e.g., Ankersen, 2005; Barquet, 2015; Dettman, 2006). Additionally, the stark reduction in species richness north of the Isthmus of Tehuantepec may suggest another barrier for dispersal. However, a phylogeographic study on 16 populations of *Podocarpus matudae* (an endemic tree of Mesoamerican cloud forests) found that the greatest population divergence for this species could not be attributed to the formation of the Isthmus. Alternatively, the lower species richness north of the Isthmus may reflect the general global trend of lower species richness linked to lower productivity at higher latitudes (Gillman et al., 2015).

Regardless of the degree to which topographic discontinuities hinder dispersal, evidence of unambiguous latitudinal shifts in response to climate change within the tropics remains scant (Colwell and Feeley, 2024). Thus, although the lake may not constitute a dispersal barrier for lowland species and more motile animal taxa, my study provides evidence that the main portion of the Mesoamerican montane-specialist tree community ends at the mountainous regions of northern Nicaragua. Rising temperatures that push tree species upslope would further isolate already fragmented populations (Colwell and Feeley, 2024), and thus passive policies such as chains of protected areas, would need to be complemented with active conservation efforts specific for montane biota.

To my knowledge, this is the first study that provides a species-based characterisation of the Mesoamerican montane-specialist tree community, quantitatively separating montane species into altitudinal and latitudinal classes. The overlap between classes could be indicative of a gradual change in species composition along the altitudinal and latitudinal axes of the study area, except for the discontinuities at Lake Cocibolca and the Isthmus of Tehuantepec. However, since these altitudinal and latitudinal classes are based on model predictions, this characterisation should be corroborated with field surveys to confirm presence and co-occurrence of the listed species. This is particularly important at the northern end of the study area due to its funnel shape, as I do not make a longitudinal classification, and the Sierra Madre Occidental and Sierra Madre Oriental mountain ranges, extending along the Pacific and Atlantic coasts of Mexico, are separated by the Mexican central highlands. An additional consideration is that the species list is not exhaustive. Many tree species were excluded from the analysis because they did not meet the montane-specialist threshold (≥75% of records inside the TMF polygons), or due having insufficient usable records (<20) to generate SDMs.

Despite increasing availability of global occurrence data, important sampling biases persist even between contiguous regions, such as in Nicaragua compared to Costa Rica and Guatemala compared to Mexico (Ramírez-Barahona et al., 2023).

3.5.2 Impacts of climate change on Mesoamerican montane-specialist trees

The predicted percent reductions of suitable range are consistent with previous studies that have predicted losses of 50% or more of TMFs in the coming decades due to climate change alone or in combination with land use change in Mesoamerica (Ortega et al., 2024), Mexico (Ponce-Reyes et al., 2013, 2012; Rojas-Soto et al., 2012) and at local scales (Altamirano-León et al., 2022; Correa Ayram et al., 2017; Estrada-Contreras et al., 2015). Globally, TMFs have declined at a rate of 0.14% per year between 2001 and 2018, with Guatemala being one of the most severely affected countries (Karger et al., 2021). For the Mesoamerican bioregion, my results suggest that the loss rate could accelerate to as much as 0.24% per year under a high emissions scenario, resulting in a 21.5% loss of suitable areas for the montane-specialist tree community by the end of the century.

This estimate could be worsened by direct anthropogenic impacts, such as land use change, illegal logging, and forest fires. Climate change is pushing agriculture to higher elevations, especially crops typically grown in tropical montane regions, such as coffee (Ovalle-Rivera et al., 2015). When fire is used to open areas for cultivation, recovery is slow due to altered microenvironments, even if seed dispersal is not directly impacted (Lippok et al., 2013). A study on African montane forests found that deforestation enhances air temperature and cloud base height increases beyond what would be expected by climate change alone and even offsetting the buffering effect of the elevational gradient (Abera et al., 2024). Large infrastructure projects, such as the "Corredor Interoceánico del Istmo de Tehuantepec," consisting of a railway communicating Mexico's Atlantic and Pacific ports (Thierry-Aguilera et al., 2021), could further disrupt connectivity between montane regions. Fragmentation of TMFs increases the edge effect, which favours the proliferation of pioneer plant species (Lippok et al., 2014), while also decreasing albedo, thus enhancing warming at high elevations (Zeng et al., 2021). Hence, climate change in combination with other direct anthropogenic pressures can generate feedback loops that could push TMFs to a tipping point.

A modest, albeit clear, trend I found is that proportional losses of suitable areas will be more severe for species with restricted range, which has been identified as a factor that increases vulnerability (Pacifici et al., 2015). This is expected due to the already small surface potentially occupied by these species, and because the orographic discontinuity of the suitable areas along mountain ranges might impede effective dispersal and genetic flows. None of the 272 species in my study are categorised by the IUCN (International Union for Conservation of Nature) as critically endangered (CR); only five are endangered (EN), 18 are vulnerable (VU) and 33 are near threatened (NT), two are data deficient (DD) and 12 are not even registered in the IUCN website (NA) (Table S4). With the exception of *Magnolia schiedeana* (see below), all vulnerable species risk losing 60–90% of their climatically suitable habitat by the end of the century. Moreover, the estimated losses of suitable areas by latitudinal and altitudinal classes under the high emissions scenario were predominantly >60%, except for the northern, low-elevation class. Losses were also higher for species with narrow latitudinal ranges (central and southern), and for those at the highest elevations (>2,000 m asl).

Similar trends in latitude and elevation as determining factors of distributional patterns have been reported for other taxa. Epiphytes, a defining feature of TMFs, show increased richness at mid- and high elevations (Krömer et al., 2013; Pouteau et al., 2016), and high levels of vulnerability to climate changes (Hsu et al., 2012; Reyes-Chávez et al., 2021). A study of climate change-driven community disassembly (defined as loss of co-occurrences) along altitudinal gradients comparing ectotherms and endotherms found positive relationships between elevational range and latitude for all included taxa (Sheldon et al., 2011). Moreover, tropical communities are consistently more sensitive than temperate ones to disassembly with increasing temperature (Sheldon et al., 2011), possibly because temperate communities are more successful in tracking suitable climatic conditions latitudinally (Colwell and Feeley, 2024). Greater projected losses at higher elevations could result from the boosting effect of altitude on rising temperatures (Loarie et al., 2009; Williams et al., 2018), combined with highland communities living closer to their physiological limits (Crausbay et al., 2014; Esperón-Rodríguez and Barradas, 2015; but see Mamantov et al., 2021) and limitations to dispersion (Chapman et al., 2016; Nowak et al., 2022) and establishment (e.g., Joshi et al., 2020; Song et al., 2016b, 2016a). These losses are particularly concerning because Neotropical TMFs have a high number of taxa, including birds, mammals, amphibians, and tree ferns, that are strictly associated with them (Karger et al., 2021), i.e., higher proportion of montane-specialist taxa.

Montane oaks (*Quercus* spp.) are of special concern. A study of 12 endangered and range-restricted Mexican cloud forest trees projected severe habitat losses by 2050 under mid and high emissions scenarios, with the genus *Quercus* (*Q. germana, Q. sartorii* and *Q. xalapensis*) displaying the highest sensitivity (Jiménez-García and Peterson, 2019). Similarly, my models predicted that two oaks would experience the second largest losses in total potential suitable area (*Q. salicifolia*, –282,762 km^2, –74.4%) and percent of potential suitable area (*Q. crispipillis*, –18,173 km^2, –94.7%), whereas *Q. costaricensis* is categorised as vulnerable by the IUCN (Table S4). Conversely, my models predicted that *M. schiedeana* and *Lacmellea zamorae* will experience gains in suitable range of 185.9% and 27.4%, respectively. For *M. schiedeana*, most of the gains were projected to occur along lowland coastal areas, including the Yucatán peninsula. Previous studies of this species have also reported moderate habitat losses and some expansions to lower elevations (Altamirano-León et al., 2022; Vásquez-Morales et al., 2014). However, such large suitable range gains seem implausible, especially when other limiting factors, such as compromised genetic connectivity are considered (Rico et al., 2023). The projected changes for *L. zamorae* are likely an artefact of scale because it had the second smallest potential suitable range under current conditions (3,977 km^2), which means that small changes in predicted suitable area translate into large percent changes.

3.5.3 Conservation management under global change

TMFs are important contributors to global biodiversity and key ecosystem services, such as atmospheric water interception, landslide prevention, and carbon storage (Tognetti et al., 2010); thus, the projected losses are concerning. However, this study shows that even under the severe climate change scenario, there are large areas that could be placed under protection or restoration schemes (e.g., along the Sierra Madre Oriental, the range along the Chiapas-Guatemalan Pacific coast and Costa Rica-western Panama). Due to TMF communities' low ability to track new climatically suitable areas (e.g., Bergamin et al., 2024; Colwell and Feeley, 2024), protected areas schemes might need to be accompanied by active conservation strategies, such as seed banks and assisted migration. Although these strategies have not been widely tested, a transplant experiment of 12 cloud forest tree species (30 seedlings per species) to higher elevations in southern Mexico recorded >90% survival rates after two years, with canopy and herbaceous cover acting as the main promoters of establishment, and no effect from humidity (García-Hernández et al., 2019). Another transplant experiment of six TMF tree species (3 to 5 individuals per species) also found high establishment rates (>75%) at higher elevations, and detected moderate herbivory in all of them after four years, indicating that some trophic relationships can form or persist post-assisted migration (Toledo-Aceves and del-Val, 2021b).

Additionally, raising both public and academic awareness on the importance of montane flora and the threats it faces may be crucial for conservation efforts, especially for the most vulnerable species (Fišer et al., 2021). An analysis of publication trends on endangered plant species in Brazil found that their inclusion in its national Red List resulted in increased focus on those species, especially ones that are of no commercial interest (Andrade and Freitas, 2021). In that sense, the fact that none of the species in my study are categorised as endangered nor critically endangered by the IUCN might be counterproductive, and even deceptive if they have not been assessed adequately or recently, if at all (2 species are data deficient and 12 are absent [NA] from the IUCN's database; Table S4). For the species facing severe extinction risk, passive protection schemes are unlikely to be successful and may require complementary strategies, such as *ex situ* conservation, as well as population management to promote recovery within their historical range, and reintroduction in target areas (Heywood, 2019). These actions could be focused on species already identified as endangered and which are known to be indicative of TMF, such as the Conifers *Abies guatemalensis, A. hickelii* and *Podocarpus matudae* , and broadleaved Angiosperms like *Juglans pyriformis*, *Magnolia schiedeana, Persea schiedeana* and *Quercus costaricensis*. To increase success rate of those actions, it should also be a priority to deepen our knowledge of those species at the genetic, physiological and ecological levels.

Thus, a species-level characterisation of the biome throughout its latitudinal and altitudinal ranges is a crucial first step for designing effective conservation strategies for TMFs. Monitoring and taking measures to preserve the co-occurrence of the species that constitute the Mesoamerican TMF tree community would minimise the risk of community disassembly and loss of ecosystem functions and services. TMFs in Latin America have been the most intensively studied globally (Mata-Guel et al., 2023), especially with regard to ecological restoration after disturbances (Christmann and Menor, 2021), but have also been identified as the region with the fastest climatic shifts (Trew et al., 2024). Cloud-base height/fog cover was not included in this study to avoid overly restricting the montane-specialist community to the narrow definition of tropical montane cloud forest (See Chapter 1.1.1). However, my approach is complementary to a study on cloud immersion declines of montane cloud forests and *páramos* throughout the Neotropics, which projected 60–90% cloud cover declines even under moderate scenarios, and also identified key areas for protection (Helmer et al., 2019). Overall, data availability was the main limitation of this study, leading to the exclusion of a large number of montane-specialist species (Table S3) and the omission of potentially relevant environmental and ecological factors, e.g., lack of data on co-occurrence and interactions with other taxa (herbivores, pollinators, seed dispersers, etc.), especially for rare, range-restricted taxa.

Nonetheless, this study introduces a clear methodology that estimates habitat suitability using species occurrence records instead of non-specific remote sensing data for the montane-specialist component of TMF communities, and projects their risk under climate change. This is particularly relevant for other parts of the world where baseline information on TMF biological communities is lacking, such as central Africa and southeast Asia. Although my methodology is intended to be easily transferable, its application in other regions or ecosystem types, especially those with significantly different dynamics compared to tropical montane regions, would require *ad hoc* variable selection and calibration (Qiao et al., 2019). Several good-practices guides have been published to help researchers increase the accuracy and usefulness of species distribution models (Araújo et al., 2019; Soley-Guardia et al., 2024). Still, with increasing availability of large occurrence databases for species across the globe (Ivanova and Shashkov, 2021), and global climatic models, my approach remains valid for other ecosystem types and taxa, especially if it is complemented with field surveys and experiments.

3.6 Conclusions

In the Mesoamerican bioregion, this study projects 400,021 km^2 of potential suitable area for >15 species, and 136,199 km^2 for >50 species out of 272 montane-specialist tree species, which is consistent with estimates based on alternative approaches, such as remote sensing and cloud-base level measurements. The southern limit of the Mesoamerican bioregion may extend to the Panama–Colombia border for lowland species, but Lake Cocibolca in southern

Nicaragua imposes a latitudinal discontinuity around 12° N for montane-specialist tree communities, indicating limited dispersal across lowland regions. Also, the species richness of the montane-specialist tree community halves north of the Isthmus of Tehuantepec, in accordance with global patterns of decreasing richness at higher latitudes. I provide a species-specific characterisation of the Mesoamerican montane-specialist tree community across latitude and altitude, based on predictions of range suitability. Although it necessitates further refinement with field surveys, it should serve as a foundational framework for policy design to monitor and preserve species co-occurrences under climate change.

The power of my estimates comes from the aggregated forecasts for the species included in the models, focusing on the overall habitat suitability of Mesoamerican TMF tree communities, rather than on highly specific parameters for individual species, and the broad extent of the study throughout the Mesoamerican bioregion, ignoring political boundaries. I provide a broad definition of the biome based on habitat suitability of predominantly montane tree species, rather than on topographic features and ground-level clouds that provide little information about species composition. This methodology can thus be easily adapted to other regions, ecosystems, and taxa, providing baseline data to inform the design of conservation policies in poorly studied regions.

Out of the 272 modelled species, 270 were predicted to experience substantial losses of climatically suitable area, in most cases >50%, and with increasing severity at higher elevations, at lower latitudes and for range-restricted species. However, I showed that even under the more severe climate change scenario, some portions of the biome are predicted to remain suitable for a high number of species. This calls for immediate and more effective inclusions of those areas under transnational protection schemes to shield them from other anthropogenic disturbances and ensure connectivity, complemented with active conservation measures to preserve this biologically and culturally crucial biome that extends over seven sovereign countries.

Chapter 4. Functional Traits of the Montane-Specialist Tree Community of Mesoamerica in the Present and under a Changing Climate

4.1 Abstract

Global forests are key ecosystems in the face of climate change. Assessing their functional diversity can aid in predicting how plant communities will respond to shifting environmental conditions and in estimating their potential contribution to mitigating climate change. This study describes the functional diversity of 272 montane-specialist tree species in Mesoamerica based on five functional traits – tree height, diameter at breast height (DBH), wood density (WD), leaf area and leaf mass per area (LMA) – and explores their latitudinal and altitudinal patterns and their phylogenetic influences. It also estimates the carbon storage potential in aboveground biomass (AGB) for the 272 montane-specialist species and possible responses under future climate scenarios. The montane-specialist tree community's functional diversity is adequately described along two axes, corresponding to (1) plant size (tree height and DBH) and (2) leaf and nutrient economics. Tree height and leaf area were correlated positively and negatively, respectively, with latitude. Tree height had the greatest influence on projected potential AGB, whereas I found no influence from WD and DBH. The observed spatial patterns in tree height and leaf area might be associated with the distribution of Conifers that generally reach larger sizes and have smaller leaves, but whose functional space forms a continuum with that of oaks (*Quercus* spp.), whereas other clades form a clearly separate functional cluster. Predicted distributional changes under climate change resulted in the attenuation of the observed latitudinal patterns, but these did not translate into similar losses of potential AGB nor functional diversity. The predicted stability of the functional diversity and spatial patterns of the montane-specialist tree community in Mesoamerica may allow the persistence of these valuable forests and their ecosystem services if adequate conservation measures are implemented to safeguard their remaining suitable habitats.

4.2 Introduction

In light of the impending predicted impacts of climate change, understanding how organisms are able to cope with and adapt to shifting environmental conditions is crucial to preserve biodiversity and ecosystem services. Those mechanisms are largely dictated by the spatial distribution of organisms' functional traits (Violle et al., 2014). Functional traits are the morphological, physiological or phenological characteristics of an organism that determine its fitness in and interactions with other elements of its environment (Violle et al., 2007). The value, range and relative abundance of functional traits in a community or ecosystem constitute its functional diversity (Apaza-Quevedo et al., 2015).

Functional trait analysis is a crucial ecological tool to understand the mechanisms that make organisms resilient or vulnerable to environmental factors (Sodhi et al., 2008). A wide variety of functional trait sets and metrics have been proposed to describe functional diversity, such as functional richness, divergence and evenness (Laliberté and Legendre, 2010), trait probability density (Carmona et al., 2016), hypervolumes (Blonder, 2018), among others. These approaches seek to use functional diversity as a link between environmental factors and plant communities' responses to them (Adler et al., 2013; Li et al., 2020).

To simplify and describe the complexity of functional diversity, a common approach is hypervolumes, which are conceptually similar to an environmental niche in that they represent biological diversity as an n-dimensional hyperspace (Blonder, 2018; Blonder et al., 2018). For functional diversity, each axis represents the possible range of values for each trait. However, there is no consensus on the exact number and identity of those traits. For example, the "global spectrum of plant form and function" (Díaz et al., 2016) posits that most of the variability determinant for plants' survival and reproduction is captured by six aboveground traits distributed along two main axes: (1) plant size (adult plant height, specific stem density and seed mass), and (2) leaf economics spectrum (leaf area, leaf mass per area and leaf nitrogen content per mass unit). Nevertheless, other functional traits such as root traits – often ignored due to the technical and financial difficulties associated with their measurement – can provide key complementary information (Carmona et al., 2021).

Besides the complex nature of functional diversity, a challenge in functional traits analysis is data availability. The development of open access databases, such as TRY (Kattge et al., 2020) and BIEN (Maitner et al., 2018), has recently made global quantitative studies possible. However, with the exception of plant height and wood density, there is a stark mismatch between the most influential traits and those most widely available in global databases (Kühn et al., 2021). There is also an overrepresentation of studies focusing on leaf traits relative to traits that promote plants' positive responses to environmental changes, such as water use efficiency, resprouting ability, growth rate and clonality (Kühn et al., 2021). Some additional shortcomings of functional trait analysis are the difficulty in detecting and explaining 'holes' in hypervolumes, i.e., unrealised combinations of traits that hint at non-considered ecological or evolutionary processes (Blonder, 2016), and the prevalent neglect of intraspecific trait variation, in favour of average values at species level (Heilmeier, 2019).

Despite the limitations and challenges of functional trait analysis, some consistent trends have been identified in global functional diversity and in relation to plants' resilience or vulnerability in face of environmental change. Firstly, it has been shown that taxonomic diversity is not directly linked to functional diversity (Hernández-Vargas et al., 2019). Functional traits have varying degrees of inheritability reflected in clustering at the family level (Ahrens et al., 2020), which allows for the use of phylogeny as a proxy for missing data imputation (Flores and Coomes, 2011; Joswig et al., 2023). However, at small scales, functional diversity tends to show higher dispersion than phylogeny (Swenson and Enquist, 2009). This is likely an effect of competition, resulting in niche partitioning of the habitat (Adler et al., 2013) along multiple axes that enable coexistence (Kraft et al., 2015), albeit to a degree. Somewhat counterintuitively, interspecific variation can increase competition (Blonder, 2018) in the absence of temporal variation or due to dearth of resources (Adler et al., 2013), which may lead to functional convergence (Hernández-Vargas et al., 2019). Thus, diverse ecosystems maintain some degree of functional redundancy that makes the loss of some components without an immediate disruption of ecological functions possible (Aguirre-Gutiérrez et al., 2022; Carmona et al., 2016).

Secondly, functional diversity follows some identifiable geographical patterns, which are sometimes non-causally correlated to phylogenetic ones. Globally, functional diversity changes along latitudinal and altitudinal gradients, with the former having a stronger influence (Wieczynski et al., 2019). However, the mechanisms that drive these spatial patterns are highly disputed and are likely case-specific. Some studies have identified temperature as the single main driver of functional diversity (Moles et al., 2014) or in combination with other factors like nutrient availability (Apaza-Quevedo et al., 2015) or disturbances (van der Sande et al., 2023). Other studies have found water availability variables to be much more important than temperature (e.g., Álvarez-Dávila et al., 2017; Li et al., 2020). The actual mechanisms are likely more nuanced, with both energy and water-related variables playing compounding, and sometimes offsetting roles, depending on the site and the temporal scale. For instance, Wieczynski et al. (2019) found that while hydric pressure promotes functional diversity, high temperature variability is a limiting factor; in contrast, Wang and Ali (2021) concluded that high temperature and low aridity are positively correlated with functional diversity. Although the combination of temperature and water availability might explain up to 90% of local environmental variability (Vilanova et al., 2018), other factors such as nutrient availability, soil conditions, spatial and temporal heterogeneity and the presence of natural enemies can also shape a community's functional diversity (Adler et al., 2013; Apaza-Quevedo et al., 2015; Báez and Homeier, 2018). Furthermore, no two functional traits respond to the same set of environmental variables, leading to varying rates of trait divergence and evolution (Ahrens et al., 2020).

A third trend is the role that rare and range-restricted species have on functional diversity. It has been reported that despite their low abundance and narrow habitat breadths, rare species contribute disproportionately to their communities' functional diversity by occupying unique – i.e. non-redundant – functional spaces, which in turn support vulnerable ecosystem functions (Leitão et al., 2016; Mi et al., 2021; Mouillot et al., 2013). In the Neotropics, montane communities are larger than those of other tropical regions due to the presence of large and continuous mountain ranges (Laurance et al., 2011), and are thus a hotspot of rare species (Zizka et al., 2018). As climate change progresses pushing montane species to higher elevations, they are projected to run out of suitable areas faster than lowland and temperate communities (Freeman et al., 2021). Well-established mature communities (e.g., tropical montane forests) have developed over evolutionary time high phylogenetic and functional diversity that allow coexisting species to exploit many specialised microhabitats (Sodhi et al., 2008). They also sustain key ecosystem services, such as water interception (Ah-Peng et al., 2017; Holwerda et al., 2010) and carbon sequestration in standing biomass and soils (Augusto and Boča, 2022). In fact, there is evidence that carbon stocks in Neotropical montane regions have been underestimated (Álvarez-Dávila et al., 2017), highlighting the pertinence of characterising the functional diversity of its montane-specialist community along its geographical range and how it may respond to anthropogenic climate change.

In this chapter, I build upon the Species Distribution Models of Chapter 3 to characterise the montane-specialist subset of tree communities in Mesoamerica in terms of aboveground functional diversity. Firstly, I assess how the functional traits of the Mesoamerican montane-specialist trees are distributed throughout the latitudinal and elevational gradients of the study area and if their variation can be described along general PCA axes, such as tree size and leaf economics. Secondly, I explore how the functional diversity will be altered by climate change based on the projected suitable areas under current and future climatic conditions by the end of the century and if the projected changes can be explained by the montane-specialist trees' functional traits. Finally, I test the montane-specialist tree subcommunity's potential carbon storage capacity in the form of aboveground biomass under future compared to current climatic conditions. Even though this analysis could benefit significantly from abundance data, mainly to improve the accuracy of estimates locally, this study can still provide valuable insights. Examining the diversity, range and spatial distributions of functional traits within these montane subcommunity can shed light on their ecological roles, functional redundancy and potential resilience to environmental changes.

4.3 Methods

4.3.1 Study area, data sources and missing data imputation

Species Distribution Models. This study covered the montane-specialist community of the Mesoamerican montane forest region, as defined in Chapter 3.3.1. The community extends from the portion of Mexico below ~25° N throughout Central America to the Panama-Colombia border, and is based on the combined polygons of the tropical montane forest (TMF) area from Villaseñor (2010) and Los et al. (2021). The species list was built from the harmonised national inventories from Mexico and all Central American countries except Belize (whose territory lies entirely outside of my TMF polygons). I downloaded their occurrence records from GBIF (www.gbif.org; 21st June 2022, https://doi.org/10.15468/dl.dsbdrh) and used the combined TMF polygon to retain montane specialists, i.e., species with ≥75% of their occurrence records inside the TMF areas. I thinned the records to a 1 km^2 grid and discarded species with <20 occurrences, resulting in 272 montane-specialist species (Table S4). As static variables, I derived aspect, slope and ruggedness from WorldClim's digital elevation model (www.worldclim.org/data/worldclim21.html, Fick and Hijmans, 2017) and obtained soil type from the FAO's Harmonized World Soil Database v 1.2 (Fischer et al., 2008). I downloaded all bioclimatic variables from CHELSA v2.1 (http://chelsa-climate.org/bioclim/, Karger et al., 2017) under current (1981–2010 baseline) and future conditions by the end of the century (2071–2100) with the severe climate change scenario RCP8.5. After testing for collinearity, I retained the following six: temperature seasonality (BIO4), mean temperature of the wettest quarter (BIO8), mean temperature of the driest quarter (BIO9), precipitation of the driest month (BIO14), precipitation of the wettest quarter (BIO16) and precipitation of the coldest quarter (BIO19). I generated the Species Distribution Models with the *biomod2* R package (Thuiller et al., 2016) using the ten environmental variables and four complementary algorithms (GAM, ANN, GBM and MaxEnt; Guisan et al., 2017). See Chapter 3.3.2 for full details.

Functional Traits Analysis. For the 272 tree species, I obtained four out of the six key functional traits according to the "global spectrum of plant form and function" (Díaz et al., 2016): adult plant height, stem specific density (henceforth wood density, WD), leaf area (LA), leaf mass per area (LMA), because these were the only traits available for most (60.2%) of the genera. I included an additional trait – diameter at breast height (DBH) – that allowed estimating standing biomass, using the equation $AGB = 0.0673 \cdot (WD \cdot DBH^2 \cdot H)^{0.976}$ (best-fit pantropical model; Chave et al., 2014; Cuni-Sanchez et al., 2021). I used the averages of the tree size variables (H and DBH), so the standing biomass represents potential mean AGB, henceforth AGB_P. I downloaded the functional traits from the TRY (Kattge et al., 2020) and BIEN v4.2

databases (Maitner et al., 2018). To build the most complete pre-imputation functional traits dataset possible (Table S9), I prioritised values at the species level, otherwise I used the average at the genus level if available, or calculated the average from other available species of the same genus, since functional traits tend to be conservative within lineages (Flores and Coomes, 2011; Joswig et al., 2023). I complemented average height and DBH with records from scanned herbaria specimens available in GBIF (www.gbif.org; consulted March-April 2024), and WD with the Global Wood Density Database (Zanne, 2009) and Ordóñez Díaz et al. (2015). When specific leaf area ($mm^2 mg^{-1}$) was reported, I calculated the inverse to obtain LMA. I harmonised all the variables to standard units (tree height in m, DBH in cm, WD in g cm^3, LA in mm^2, and LMA in mg mm^{-2}). For missing data, I used the data imputation process described in (Carmona et al., 2024) with the *V.PhyloMaker* package (Jin and Qian, 2022), which uses phylogenetic relationships as proxy to fill in missing values (Table S10).

4.3.2 Analysis of spatial patterns of functional traits and their influence on species' distributional changes under climate change

To examine the spatial patterns of the functional traits across the study area, I added the imputed functional traits data to species richness matrices derived from the distribution models under current (1981-2010 baseline) and future (2071-2100, RCP8.5) climatic conditions generated in Chapter 3.3.3. These matrices contain predicted climatic suitability data for each of the 272 tree species per pixel at 30 arc-seconds resolution. I calculated the community-weighted mean (CWM) for each functional trait and AGB_P per pixel based on the frequency of species present in all pixels (Lavorel et al., 2008). Additionally, I calculated functional dispersion (FDis) using the four available traits (tree height, wood density, leaf area and LMA) out six listed by Díaz et al. (2016). FDis measures the spread of species traits relative to the centroid of their multidimensional trait space, reflecting the breadth of ecological strategies within the community (Laliberté and Legendre, 2010). I chose FDis because, unlike other functional diversity metrics, it emphasises trait spread, and it is not dependant on species abundance, although it can incorporate abundance data when available (Laliberté and Legendre, 2010). All CWM and FDis calculations were performed using the *FD* R package (Laliberté et al., 2009). I converted the resulting CWM and FDis matrices back into georeferenced raster layers using the *raster* R package (Hijmans, 2020) and estimated the changes in each trait by the end of the century by subtracting each current raster layer from its corresponding future one.

To determine whether each of the functional traits was associated with elevation and latitude, as has been shown globally (Wieczynski et al., 2019), I transformed the rasters into *SpatRaster* objects and loaded an elevation layer using the *terra* package (Hijmans et al., 2022). To account for spatial autocorrelation, I used the *spdep* package (Bivand et al., 2017) to estimate each functional trait layer's Moran's I coefficient (Table S11). I extracted the trait value and ran Pearson correlation tests between each trait and the associated elevation (m asl) and latitude per pixel. Since most montane-specialist tree species were projected to lose a large part of their suitable range (>60%), resulting in high species turnover at mid-elevations throughout the whole range (see Chapter 3.4.2), I expected these shifts to translate into significant changes in the spatial distribution of each functional trait by the end of the century.

To investigate whether the functional traits of the tree species have an influence on the predicted suitable areas under current and future climatic conditions, I ran linear regression models using each of the five functional traits for the species as explanatory variables for the predicted estimates of climatically suitable areas (in km^2) under current and future conditions, and on projected total losses and percent losses. To explore how well the variation of the tree species is captured by the set of five functional traits, I ran a PCA using the *ade4* package (Dray and Siberchicot, 2017) and estimated the proportion of variation explained by each axis and their level of significance using Monte-Carlo simulations (Díaz et al., 2016). To identify variation patterns in my data, I plotted the PCA by taxonomic group, latitudinal and altitudinal classes (see Chapter 3.3.3). To handle collinearity between explanatory variables, I ran pairwise Person correlation tests, using the rcorr() function of the *Hmisc* R package (Harrell Jr, 2019). To standardise the data (reduce the influence of extreme values and varying degrees of magnitude between variables), I log10-transformed and scaled the dataset (Carmona et al., 2024; Díaz et al., 2016) and ran a bidirectional stepwise regression using the stepAIC() function of the *MASS* R package (Ripley et al., 2013) to identify the best performing linear model for each response variable, based on the Akaike Information Criterion. I thus reduced the number of explanatory variables to include only those with a significant effect on range sizes and changes. I tested the models' assumptions with the Shapiro-Wilk and Breusch-Pagan tests. All calculations were done in R v4.4.0.

4.4 Results

4.4.1 Current functional diversity patterns

Using the distribution models of 272 Mesoamerican montane-specialist tree species developed in Chapter 3, I calculated the community-weighted mean of each functional trait (tree height, DBH, WD, leaf area and LMA) and AGB_P, based on species projected to have suitable areas under current and future climatic conditions (RCP8.5 scenario).

Among the traits, WD (0.54 ± 0.03 g cm^{-3}) and LMA (0.08 ± 0.01 mg mm^{-2}) exhibited the least variation, whereas leaf area (5,514.9 ± 2,035.6 mm^2) varied by several orders of magnitude (Table **4.1**), likely due to differences between scale- or needle-like leaves of Conifers and broadleaved Angiosperms. The mean sizes of the species were 15.28 (±4.28) m in height and 30.98 (±14.88) cm in DBH under current conditions (Table **4.1**). The varying height-to-DBH ratios translated into highly variable AGB_P (1,784.0 ±1,978.7 kg).

Table 4.1 Summary statistics of the community-weighted means of functional traits and aboveground biomass of the montane-specialist tree species throughout the study area on a pixel-by-pixel basis, under current (1981–2010 baseline) and future (2071–2100) climate conditions under RCP8.5. Change refers to the average difference per site (pixel) between current and future raster layers, so they do not correspond to a simple subtraction of the bulk current and future values.

Trait	Time frame	mean	sd	min	1st q	median	3rd q	max
H (m)	current	15.28	4.28	3.33	11.94	15.30	17.72	40.00
	future	16.02	4.36	3.83	12.72	15.88	18.75	38.17
	change	*0.13*	*3.79*	*-23.25*	*-1.97*	*0.20*	*1.83*	*24.75*
DBH (cm)	current	30.98	14.88	1.33	20.83	28.72	37.81	153.33
	future	33.94	16.29	1.90	22.28	31.37	40.61	140.00
	change	*1.50*	*14.00*	*-79.50*	*-5.06*	*0.66*	*6.91*	*107.65*
WD (g cm^{-3})	current	0.54	0.03	0.40	0.53	0.54	0.55	0.76
	future	0.53	0.03	0.40	0.52	0.53	0.55	0.70
	change	*-0.01*	*0.03*	*-0.19*	*-0.02*	*0.00*	*0.01*	*0.18*
LA (mm^2)	current	5,514.9	2,035.6	673.8	4,234.1	5,351.2	6,606.2	21,551.3
	future	5,724.7	2,310.1	789.1	4,212.4	5,444.5	7,025.5	21,551.3
	change	*69.5*	*1,394.1*	*-9,542.2*	*-601.2*	*31.1*	*681.7*	*11,070.9*
LMA (mg mm^{-2})	current	0.08	0.01	0.03	0.08	0.08	0.09	0.19
	future	0.09	0.01	0.03	0.08	0.08	0.09	0.18
	change	*0.00*	*0.01*	*-0.11*	*-0.01*	*0.00*	*0.01*	*0.10*
AGB_P (kg)	current	1,784.0	1,978.7	0.2	397.6	1,207.6	2,401.4	23,191.4
	future	1,868.2	1,836.2	1.2	421.7	1,334.8	2,666.3	15,420.9
	change	*-80.9*	*1,843.2*	*-18,181.3*	*-656.1*	*-13.4*	*506.5*	*12,668.5*

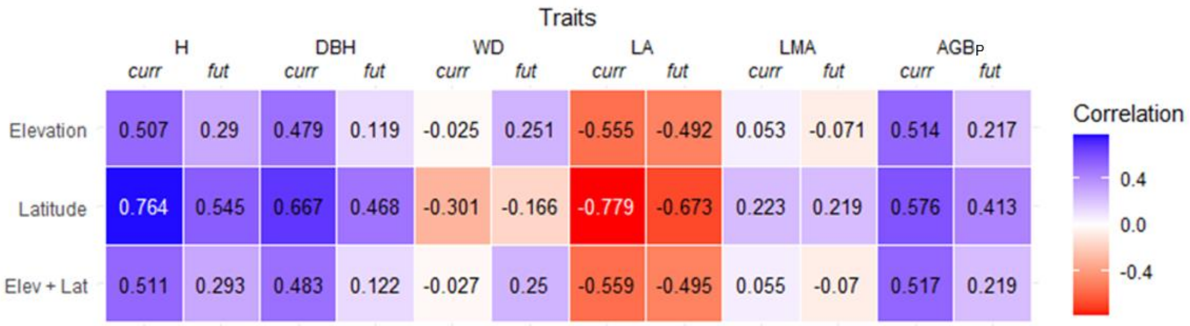

Figure 4.1 Heatmap of Pearson correlation coefficients of the elevational and latitudinal gradients and the CWMs of functional traits (H, DBH, WD, LA and LMA) and AGB_P of the species projected to have suitable area per pixel under current (1981–2010 baseline) and future (2071–2100) climate conditions (RCP8.5 scenario). Darker colours indicate stronger correlations. p-values < 0.0001 in all cases.

Out of the five traits and potential AGB_P, tree height and leaf area had the strongest correlations with elevation and especially latitude (Figure **4.1**). Suitable areas for taller tree species tended to concentrate in higher latitudes (r = 0.76, p < 0.0001; Figure **4.2**A) and elevations (r = 0.51, p <

0.0001), whereas areas suitable for species with large leaves were more common at lower latitudes (r = -0.78, p < 0.0001; Figure **4.2**C) and lower elevation (r = -0.56, p < 0.0001), both under current conditions. Taller tree species were projected to have more suitable conditions on the inland-facing edges of the Sierra Madre Oriental and Sierra Madre Occidental ranges, flanked by slightly shorter species at similar latitudes and a shortening height pattern further into Central America. Leaf area also had a clear north-south gradient, with big-leaved species being more common at lower latitudes, especially into southern Central America and into Colombia (southern buffer area), and a pattern on the northern part of the range of larger leaves on the ocean-facing slopes and smaller leaves on the slopes facing inland.

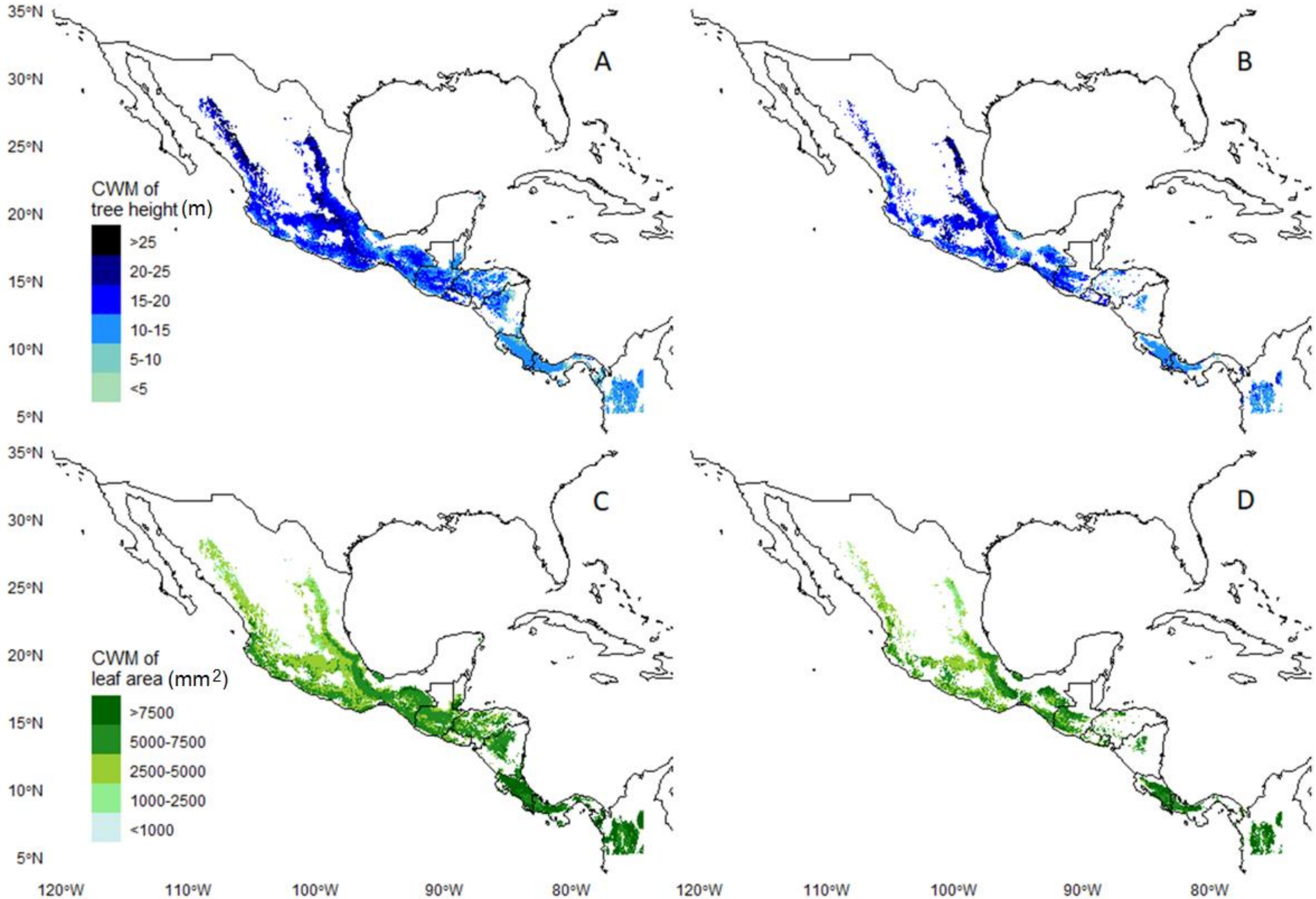

Figure 4.2 Community-weighted means of tree height (A and B) and leaf area (C and D) of the montane-specialist tree species projected to be present under current (A and C) and future (B and D) climatic conditions.

4.4.1 PCA of the montane-specialist trees functional traits

Out of the five functional traits, only height and DBH were moderately correlated ($r = 0.632$, $p < 0.0001$), which suggests varying ratios of height to stem width of tropical montane-specialist trees. The association between the two adult tree size-related traits was also apparent in the PCA, with tree height and DBH varying along the PC1 axis (Figure **4.3**). The variation of the functional traits was captured by five components, with plant size-related traits (PC1) and leaf and biomass traits (PC2) explaining an accumulated 62.63% of the total variation (Table S12).

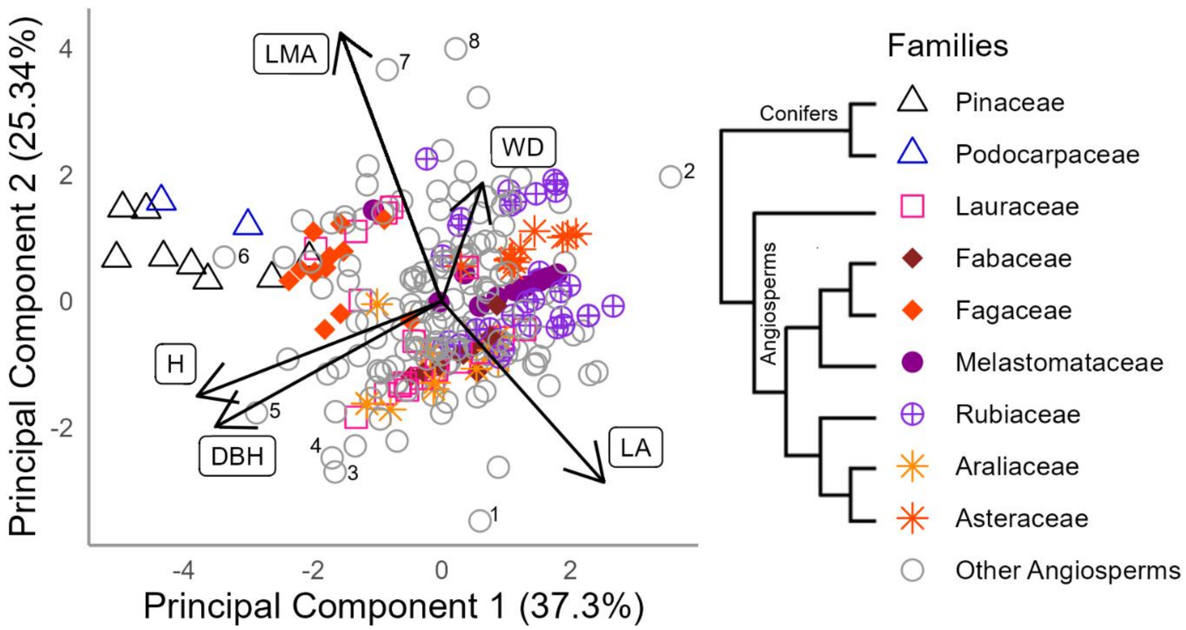

Figure 4.3 PCA of five functional traits of montane-specialist tree species by most frequent taxonomic groups. The directions and lengths of the loadings indicate that most of the variation in PC1 aligns with adult tree size-related variables (tree height and DBH), whereas PC2 is more aligned with leaf traits and biomass (leaf area, LMA and to a lesser extent WD). Height and DBH remain closely aligned when plotting other components. Most Conifers are outliers, occupying a distinct functional space along the PC1 separated from most Angiosperm families. Other outliers, belonging to various Angiosperm lineages (see Table S4), are show with numbers:
(1) *Meliosma alba* [Clade Basal Eudicots, Proteales, Sabiaceae],
(2) *Hypericum irazuense* [Clade Rosid I/Fabids, Malpighiales, Hypericaceae],
(3) *Juglans pyriformis* [N-fixing Clade, Fagales, Juglandaceae],
(4) *Ulmus Mexicana* [N-fixing Clade, Rosales, Ulmaceae],
(5) *Chiranthodendron pentadactylon* [Clade Rosid II/Malvids, Malvales, Malvaceae],
(6) *Clethra suaveolens* [Clade Basal Asterids, Ericales, Clethraceae],
(7) *Comarostaphylis arbutoides* and (8) *C. longifolia* [Clade Basal Asterids, Ericales, Ericaceae], and
(9) *Buddleja nitida* [Clade Asterid I/Lamids, Lamiales, Scrophulariaceae].

The PCA also revealed some important taxonomic trends (Figure **4.3**). Most Conifers were outliers and formed a clearly distinct group separated from the rest of the community along PC1, but they also seemed to form a continuous functional space with Fagaceae (*Quercus* spp.). Most species belonging to the remaining most abundant families formed another cluster separated from the Conifers-Fagaceae continuum along PC2. Only Lauraceae straddled these two distinct clusters, with some species intermixed with Fagaceae, and the rest with the bulk of other frequent Angiosperms. Additionally, the most closely related families do not form continuous clusters. Such is the case of Fabaceae and Fagaceae within the N-fixing clade (Rosid I *sensu* APW v14, 2017; Stevens, P.F. (2001 onwards) Angiosperm Phylogeny Website, www.mobot.org/MOBOT/research/APweb/), and of Araliaceae and Asteraceae within the Asterid II clade (*IDEM*), but that form distinct clusters separated from each other. The remaining families (Other Angiosperms) had fewer than 5 representatives each to confidently describe any patterns. Aside from Conifers, nine other species were outliers in the PCA, which did not belong to closely related Angiosperm lineages.

4.4.2 Future functional diversity patterns

Under the RCP8.5 scenario, the community-weighted means of all traits remained relatively stable by the end of the century (Table **4.1**). Except for wood density, most functional traits were projected to maintain their current distributional patterns throughout the study area, although correlations with latitude and elevation were noticeably less strong (Figure **4.1**). Changes in species composition per site (pixel) might also lead to a slight decrease in the potential community average biomass under future climatic conditions (-80.9 kg difference in potential community-weighted mean AGB_P of species with predicted suitable conditions across all sites; Table **4.1**).

The trends of higher community-weighted mean tree height at higher latitudes ($r = 0.55$, $p < 0.0001$; Figure **4.2**B) and elevations ($r = 0.29$, $p < 0.0001$), and of greater leaf area at lower latitudes ($r = -0.67$, $p < 0.001$; Figure **4.2**D) and elevations ($r = -0.49$, $p < 0.0001$,) were attenuated, although they were still present. The functional traits maps under future conditions also reflect the predicted size reduction of climatically suitable areas due to climate change. Out of the remaining traits, only DBH (Fig. S2) showed a similar albeit less strong trend to tree height; there were no clear geographical patterns for WD (Fig. S3) and LMA (Fig. S4).

The latitudinal pattern of tree height (and correlated DBH) resulted in an estimated community-weighted mean AGB_P with the same latitudinal pattern. Since the community-weighted tree height was inversely correlated with latitude, the average AGB_P also decreased southwards, where there is projected climatically suitable habitat for a higher proportion of short-statured tree species (Figure **4.4**A), some of which can occur as shrubs. When plotting the projected differences in potential community-weighted mean ABG_P between current and future conditions, there is a patchwork of areas projected to experience small potential gains and, to a lesser degree, losses in potential AGB_P throughout the study area (Figure **4.4**B). Note that the spatial AGB_P distribution is an estimate of the potential carbon stored by montane-specialist species under current or future climatic conditions in each site (pixel) within the study area. It should be taken as an indicator of where suitable conditions for montane-specialist species with high carbon storage capacity occur, rather than the actual amount of carbon in these ecosystems.

Functional dispersion (FDis) did not show a clear latitudinal pattern, but it seemed higher at higher elevations (centres of polygons) and decreased near the edges (Figure **4.4**C), coinciding with areas projected to have high species richness (see Chapter 3). When subtracting projected future FDis from current FDis, it seemed to remain largely stable, except for some small gains at the highest elevations and small losses at low elevations (Figure **4.4**D).

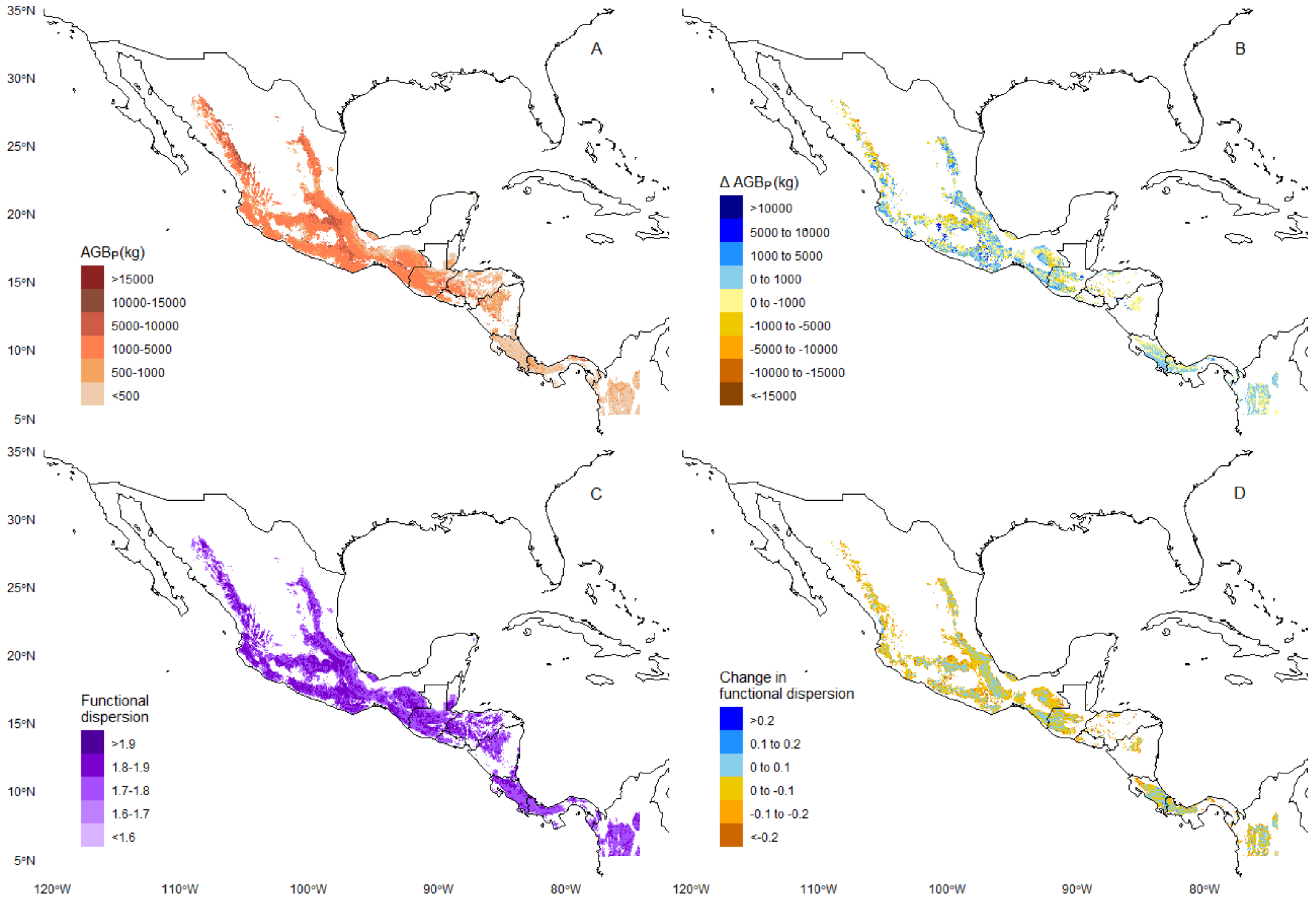

Figure 4.4 Community-weighted mean (CWM) of potential aboveground biomass (AGB_P) based on the species projected to have suitable conditions under current climate conditions (A), projected changes in CWM AGB_P under future with respect to current climatic conditions (B), functional dispersion of the 272 montane-specialist tree species under current conditions (C) and projected changes in functional dispersion under future with respect to future climatic conditions (D).

4.4.3 Functional traits' influence on projected habitat suitability loss under climate change

The stepwise regression showed that the combination of tree height and leaf area were the best predictors for both the current and future size of suitable ranges for the ensemble of species (best model in each case; **current** lm = 4.96 + 0.45H – 0.19LA; $F_{2, 269}$ = 21.92; $p <$ 0.0001; R^2 = 0.13; Shapiro-Wilk W = 0.99, p = 0.225; Breusch-Pagan BP = 2.08, p = 0.353;; **future** lm = 4.17 + 0.52H – 0.13LA; $F_{2, 269}$ = 13.9, p < 0.0001; R^2 = 0.09; W = 0.99, p = 0.321; BP = 3.26, p = 0.195). For both time frames, tree height and leaf area were the only variables with a significant effect on the size of the suitable area. Similarly, there was a small influence of height and leaf area on the total losses of predicted suitable areas under future with respect to present climatic conditions (Gamma glm = 1.62 + 0.08H – 0.05LA; W = 0.99, p = 0.063; BP = 0.16, p = 0.923). The taller species, which were predominantly distributed in the central and northern-central areas throughout the study region, and other widespread species were predicted to experience the largest losses in suitable area (Figure **4.5**), in line with the total predicted area losses being associated with current range size (see Chapter 3.4.2). However, neither tree height, nor any other functional traits, on their own or in combination, were correlated with predicted percent range losses under future conditions.

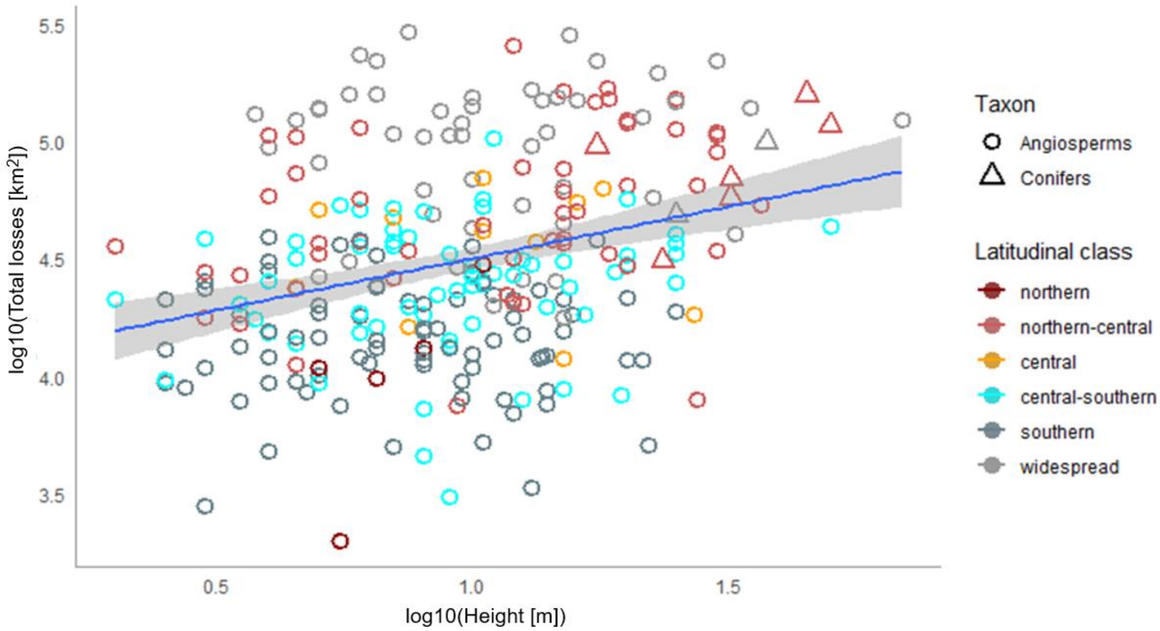

Figure 4.5 Relationship between adult tree height and log-transformed total losses projected suitable extent under future climate conditions relative to current conditions by taxonomic group and latitudinal class.

4.5 Discussion

I analysed five aboveground functional traits (tree height, DBH, WD, leaf area and LMA) of 272 montane-specialist tree species in Mesoamerica. The main drivers of functional diversity were tree height and leaf area. I also investigated how these traits translate into carbon storage capacity in the form of potential aboveground biomass (AGB_P) and estimated the functional dispersion of the montane-specialist community. AGB_P followed a similar pattern to tree height, with higher values at higher latitudes, whereas functional dispersion was more positively related to elevation. Under future conditions, both potential AGB_P and functional dispersion were projected to remain largely stable throughout the whole study area, but with marked local variation.

4.5.1 Current distributional patterns of functional diversity

The five functional traits included in this analysis revealed some significant trends. Within the range where tropical montane specialists are projected to occur currently, adult plant size tends to be larger, and leaf area tends to be smaller at higher latitudes. The greater variability in DBH compared to height is expected for montane communities, which either comprise several strata or transition from tall to short vegetation types within short distances along mountain slopes (Scatena et al., 2010). This variability is reflected in varying height–DBH ratios. The wide standard error in DBH could also be attributable to a few very large species, including the Mexican elm (*Ulmus mexicana*), Mexican white pine (*Pinus ayacahuite*) and the cloud forest oak *Quercus benthamii*.

The absence of clear geographical patterns for the other traits aligns with previous studies, which have shown that height and leaf area are more responsive to environmental factors than, for instance, wood density and seed mass (van der Sande et al., 2023). Wood density is more commonly linked to successional stages and their growth strategies than directly to environmental drivers. Early successional species with a high resource acquisition strategy typically have low wood density, while late successional species have higher wood density (Castillo-Figueroa et al., 2023). Similarly, leaf mass per area indicates a resource acquisition strategy (Díaz et al., 2016), so the lack of a clear spatial pattern across a large, rugged area with microhabitat variation could be expected.

Similar observations for other traits such as deciduousness have been made, showing a stronger association with growth strategy rather than environmental factors. For example, a study of two subcommunities (deciduous and evergreen) in a karst forest in southern China found that deciduous species are linked to a high resource acquisition strategy, whereas evergreen species favour resource conservation by investing in long-lasting, high-quality tissues (Wang et al., 2023). Regarding functional traits that influence water use, high wood density is associated with greater drought tolerance, but this relationship is context-dependent (Ahrens et al., 2020). Stomatal control, a key functional trait for hydraulic failure prevention, does not always correlate with the water gradient either (Sterck et al., 2011).

The community-weighted mean AGB_P of the montane-specialist tree community followed a similar latitudinal gradient to tree height under current climatic conditions. Wood density, one of the least variable traits, had a lesser influence on AGB_P. These findings provide additional evidence that tree size (i.e. height) and AGB are strongly correlated (Wang and Ali, 2021) whereas AGB has weak or no relation with wood density (Álvarez-Dávila et al., 2017).

4.5.2 Phylogenetic divergence of functional diversity

The distribution of the analysed functional traits along two main PCA axes – with PC1 associated with plant size (height and DBH), and PC2 related to leaf economics and nutrient acquisition strategy (wood density, leaf area, and leaf mass per area) – aligned with the "global spectrum of plant form and function" framework (Carmona et al., 2021; Díaz et al., 2016). The PCA also revealed some phylogenetic patterns. Firstly, the functional space of Conifers was clearly separated from the rest of the species along PC1 (tree size), indicating that Conifers tend to reach larger sizes than Angiosperms. This contributed to the larger community-weighted mean sizes in the less diverse northern parts of the study area. Secondly, the proximity of the functional trait spaces of Conifers and Fagaceae (*Quercus* spp.) suggests a functional gradient due to their similar roles as keystone species in montane regions, where they often form distinct pine- and oak-dominated communities with transitional mixed pine-oak forests between them (Alfaro et al., 2014). Thirdly, species within the same family tended to cluster in close functional spaces, reflecting a trend of trait conservatism within lineages at lower taxonomic levels. However, at higher taxonomic levels, closely related families tended to avoid each other's functional spaces. This pattern could be due to competitive pressures within lineages leading to niche partitioning, while environmental constraints push unrelated clades into narrower functional trait spaces.

These patterns match those observed in an Amazon-to-Andes altitudinal gradient, where closely related lineages were found to remain within similar elevations, whereas higher clades preferred different altitudes (Griffiths et al., 2021). Asterids were more dominant at higher elevations, Rosids at lower elevations and Podocarpaceae (Conifers) at mid elevations, whereas Magnoliids were split between mid and low elevations. Families within the Asterids and Rosids clades did not have such clearly split functional spaces, except for Fagaceae (Rosids) by forming a continuum with Conifers. Lauraceae straddled both functional space clusters, which could be related to the broader altitudinal range of the Magnoliid clade described by Griffiths et al. (2021). Additionally, other species with trait combinations that situate them on the edges of the functional trait space (outliers) belong to families spread across the whole Angiosperm cladogram, which might indicate non-redundant niches that, if lost, could result in a net reduction of the of the community's functional diversity. Among these, there are other large-statured Angiosperm species, such as *Chiranthodendron pentadactylon, Juglans pyriformis* and *Ulmus mexicana*.

While phylogenetic affiliation is generally a good proxy for ecological similarity, enabling missing data imputation (Jin and Qian, 2022; Joswig et al., 2023), co-occurring species face pressures that promote both functional differentiation (e.g., competition) and convergence (e.g., limiting factors) (Swenson and Enquist, 2009). Given that this database relied on phylogeny to fill data gaps, the observed family-clustering pattern could partly result from the data imputation method. Nevertheless, the greater differentiation at higher taxonomic levels (between families) compared to lower levels (within families) may reflect evolutionary timescales, as some functional trait changes in response to environmental factors have been detected over periods of 10,000 years (van der Sande et al., 2023). This suggests that families have had more evolutionary time to develop distinct functional spaces than the genera within each family.

4.5.3 Predicted effects on functional traits and aboveground biomass under climate change

The latitudinal trends of tree height and, to a lesser extent, that of leaf area were less apparent under projected future climatic conditions. The higher proportion of big-leaved species predicted to remain constant was especially marked in the central part of the study area, in the Mexican states of Oaxaca and Chiapas flanking the Isthmus of Tehuantepec, and southern Central America, especially in Costa Rica, which also correspond to the areas with the highest species richness (see Chapter 3.4.2). Thus, even if proportionally similar richness losses occurred throughout the whole latitudinal range, the relative proportion of tall, small-leaved species would remain more noticeable in the northern parts of the study area.

The attenuations of latitudinal trends of functional traits are likely due to a reduction in the size of climatically suitable areas for the ensemble of tree species, rather than the northward migration of shorter, big-leaved species from more southern areas. A recent review of distributional changes in response to climate change did not find evidence of topical communities successfully migrating latitudinally, as opposed to their temperate counterparts (Colwell and Feeley, 2024). Additionally, a study in the Brazilian Atlantic Forest biome found that half of the surveyed montane communities have not migrated in response to changing climate (Bergamin et al., 2024). Of the remaining communities, those at higher altitudes tended to migrate upward, whereas lowland communities migrated downward.

As the main driver of functional diversity of the montane-specialist tree community, tree height seemed to have an inverse relationship with the magnitude of suitable area losses, but that did not translate into a positive association between tree height and percent losses of suitable habitat. Likewise, tree height drove the latitudinal pattern of AGB_P under current conditions, but that did not translate into a similar pattern of projected losses in AGB_P under future climatic conditions. It has been noted that several auxiliary factors can have a greater influence than commonly expected on AGB, such as dominant tree species, herbaceous cover, canopy closure, among others (Su et al., 2024).

Functional dispersion did not have a clear latitudinal pattern either, although it seemed to increase at higher elevations. Under future climatic conditions, functional dispersion was projected to remain largely stable, but with slight increases at higher elevations and slight declines at lower elevations. This could be further indication of upland species predicted to migrate upslope, and of lowland species to migrate downslope (Bergamin et al., 2024). Alternatively, the loss of functional diversity at lower elevations could be an indication of lowland attrition, in which there are no extant communities adapted to warmer regimes to replace those migrating upslope (Colwell et al., 2008).

Taken together, my AGB_P and functional dispersion findings could be attributed to some rare, but resilient and influential large tree species. A study in tropical Africa found that rare tree species with large individuals display the highest functional specialisation – thus the lowest functional redundancy – and can sometimes be 'hyperdominant' in terms of carbon storage (Kearsley et al., 2019). If the latitudinal pattern of tree hight – and AGB_P – is preserved under future climatic conditions, it could be a reflection of the persistence of large-sized species at the northern parts of the range, and simultaneously, the relative stability of functional dispersion due to the preservation of rare species with the most specialised functional niches (Mi et al., 2021). However, conservation and recovery measures should be taken to preserve population of these very large species, as three of them (*Abies guatemalensis*, *A. hickelii* and *Juglans pyriformis*) are considered as endangered (EN) and one (*Podocarpus matudae*) as vulnerable (VU) by the IUCN (Table S4).

It has been reported that functional traits are not always clearly correlated to environmental variables (Heilmeier, 2019). For instance, a study conducted on a dominant canopy species in Western Australia found significant correlations between functional traits and their responses to climate change, but with low explanatory power, which was attributed to high intra-functional trait variation and differential evolutionary rates (Ahrens et al., 2020).

4.5.4 Caveats and limitations

This study found some clear trends that were consistent with previous research. However, there are some limitations that call for a careful interpretation of the results: (1) they are not based on field measurements, (2) they are not weighted by abundance or basal area data, and (3) the traits included in the models were limited by data availability. Furthermore, potential carbon storage capacity (AGB_P) omits abundant species that did not meet the threshold to be considered montane specialists.

Thus, the low explanatory power of my models is likely due to the limited number of functional traits available for this set of montane-specialist species. For example, only two out of the 23 species listed as vulnerable or endangered by the IUCN had complete data for all the traits included in this study, whereas the rest only had data available at genus level or had to be imputed based on their phylogeny. Otherwise, the inclusion of water-use efficiency or root traits, among others, would likely improve the accuracy of current habitat suitability estimates and resilience to changing climatic patterns (Kühn et al., 2021). Additionally, the restrictive species selection process meant omitting the functional diversity context found in other growth forms, such as herbs, shrubs. Potentially, if other biomes or growth forms were included, more obvious trends would become detectable. Moreover, this study relies on species' average values of each functional trait, whereas other significant trends might arise by including intraspecific variation to make comparisons between locations and lineages (Heilmeier, 2019).

Although beyond the scope of this study, the importance of dominant species cannot be overlooked. A study of pantropical lowland forests found that only 2.2–2.3% of 'hyperdominant' species make up half of all trees in Amazonia, Africa, and Southeast Asia (Cooper et al., 2024). Despite the absence of abundance data in this study, my dataset includes 19 tree genera featured in the list of 'hyperdominant' species (Cooper et al., 2024), 12 of which are found in the Neotropics (Amazonia): *Amaioua*, *Coccoloba*, *Croton*, *Eugenia*, *Guatteria*, *Inga*, *Lacmellea*, *Miconia*, *Ocotea*, *Pouteria*, *Sloanea*, and *Zygia*. Oaks only had representatives in southeast Asia, whereas pines were not included because the study did not cover montane regions.

Nonetheless, Mesoamerica has been reported to have a higher proportion than the global average of elevational specialist species (Laurance et al., 2011) and its mountainous regions are diversity hotspots for both pines (Farjon, 1996) and oaks (Valencia-A., 2004), Thus, it is possible that some species in this study's dataset could be abundant, if not 'hyperdominant,' and may have a significant influence on the functional traits of the montane-specialist tree community. Therefore, it seems likely that the conclusions of this study would hold up if corroborated by the addition of abundance data and field observations.

4.6 Conclusions

The combination of five aboveground functional traits produced a reasonable description of the functional diversity of the Mesoamerican montane-specialist tree community. Under current climatic conditions, tree height and leaf area were correlated positively and negatively, respectively, with latitude. However, a clear unidirectional relationship between each functional trait and the projected changes in suitable area under climate change was not accomplished.

This study showed that functional trait analysis without abundance data can still provide valuable insights into the functional traits present in an ecosystem. However, it should be emphasised that that this study analysed the functional traits that are less likely to be influenced by abundance, which can nevertheless help in understanding the general drivers of functional diversity within the community.

Similarly, the estimate of aboveground biomass can be taken as indication of the areas with the greatest potential carbon sequestration capacity, rather than accurate estimates of current carbon stocks. Calculating actual standing biomass would require complementary abundance, basal area or population structure to both generate precise carbon stocks estimates and pinpoint at a fine scale key conservation areas for carbon sequestration, in addition to those for biological diversity (de Albuquerque et al., 2015).

Finally, the inclusion of more functional traits, intraspecific variation and abundance data could help establish clearer relationships between functional diversity and environmental factors, and potentially identify traits or combinations of traits that can make tree species more resilient or vulnerable to the projected effects of climate change in the Mesoamerican biodiversity hotspot.

Chapter 5. General Discussion

5.1 Significance of biases in existing research on impacts of climate change on tropical montane forests

The systematic search of the literature on the impacts of climate change on global tropical montane forests (TMF) conducted in Chapter 2 revealed important methodological, geographical and thematic trends. The most evident one is the disparity between world regions, with the Neotropics concentrating by far the majority of studies. Not only do vast regions with known presence of TMFs, such as Southeast Asia and Africa, have been less studied, but there were more than 20 countries for which not a single study meeting the inclusion criteria was found. Such a dearth of knowledge represents a serious obstacle in our ability to link ecological science with environmental policy during the present time of accelerated climatic change.

The observed geographic research bias might even call into question some basic notions that seem well established. For example, it is generally agreed that the Neotropics have the largest areas of TMF globally (Los et al., 2021; Mulligan, 2010), although conflicting definitions and detection methods lead to inconsistencies (e.g., Scatena et al. (2010) report a larger area in Southeast Asia). Compounded with the ambiguity in the definition of what constitutes a TMF (see Chapter 1.1), the neglect of vast world regions leaves open the debate on whether some countries have TMFs. Similarly, the Neotropics are claimed to have the highest plant diversity on Earth (Zizka et al., 2018) and also the greatest diversity of montane specialists thanks to its large and continuous mountain ranges (Laurance et al., 2011). However, African montane regions have been reported to have the highest degree of dissimilarity (i.e., uniqueness) among world regions for amphibians, birds, mammals and tree ferns (Karger et al., 2021).

The geographic trend persists at regional scales. With only five countries in the Neotropics (Colombia, Costa Rica, Ecuador, Mexico and Peru) having more than 5% of the 395 reviewed studies each, the stark contrast with some countries in the region is clear. This is also reflected in the limited data availability in global repositories for many countries in the region, such as Guatemala, Nicaragua and Panama (Ramírez-Barahona et al., 2023), despite neighbouring the most intensely studied ones. Such obvious sampling biases introduce uncertainties that have to be accounted for by studies that rely on such databases (Hughes et al., 2021).

Unfortunately, there is no easy or short-term way to overcome the observed geographic bias. By looking at the global distribution of studies, it is reasonable to assume that it largely responds to socioeconomic or even geopolitical factors. Impoverished regions and countries by definition allocate fewer resources to science and environmental policies. Thus, research in those places often relies on foreign scientists and institutions conducting research sporadically, rather than knowledge being generated locally and continuously (Nuñez et al., 2021).

Thematically and taxonomically, the state of research is not balanced either. Research on vascular plants, especially trees, represents the majority of studies by a wide margin; in contrast, studies at the genetic or species interactions levels are particularly scarce (see Chapter 2.4.1). Thus, a high number of studies on a particular taxonomic group (or growth form in the case of trees) does not necessarily mean their biology and interaction with their environment are well understood. The accumulation of studies on TMF trees allowed for the construction of a somewhat consistent narrative of both observed and predicted significant range losses or due to climate change, but with high local and species-specific variability that remains hard to explain. Novel research looking at genetic diversity and interspecific interactions of TMF biota could shed light on the underlying mechanisms that drive their distributional responses to environmental stressors.

In the short term, it is important to make use of the available data and ecological tools to inform conservation in light of current accelerating climate change (Forster et al., 2024). Therefore, analyses of the literature, such as the one presented in Chapter 2, are useful to identify areas of opportunity to direct research, such as the absence of a species level characterisation of the montane-specialist tree community in Mesoamerica addressed in this thesis.

5.2 Significance of the predicted distributional changes of Mesoamerican montane-specialist trees in response to climate change

Species Distribution Models (SDM) are a crucial tool in conservation biology (Galbraith and Christoffersen, 2015) and their increasing accessibility has made them very popular in TMF research (see Chapter 2). In addition to following as much as possible the best modelling practices (Araújo et al., 2019), it is also important to keep in mind assumptions that are independent of the input data and the choice of parameters for the correct interpretation of their outputs (Soley-Guardia et al., 2024).

The limitations of SDMs are well exemplified by the characterisation of the Mesoamerican montane-specialist tree community presented in Chapter 3. The Bray-Curtis indices, based on the occurrence data of 272 montane-specialist tree species indicated that the community is split latitudinally by the Lake Cocibolca, and is significantly less diverse north of the Isthmus of Tehuantepec. When modelling suitable areas under current climatic conditions, however, 51 species were projected to have climatically suitable areas spanning the most of the study area's latitudinal range. It can thus be inferred that most species are far from occupying their whole potential niche. Although SDM's have been employed specifically to predict the presence of rare species (e.g., Volis and Tojibaev, 2021; Williams et al., 2009), 'stacked' models – i.e., aggregated models of individual species as representatives of a whole community – are known to overestimate potential suitable habitat (Pottier et al., 2013). It is also possible that other non-considered factors (e.g., species interactions, fire dynamics, dispersal barriers, etc.) or simply random chance limit species' realised ranges.

Another caveat is that SDMs seldom incorporate information on the underlying mechanisms that drive plants' distributional changes in response to environmental conditions. By merely correlating recorded or projected environmental conditions to presence (and sometimes absence) records of target species, the predicted influence of each environmental factor is very unidirectional. For instance, a vast majority of modelling studies conclude that species will follow their preferred temperature regimes toward higher latitudes and elevations, but empirical evidence does not support these predictions. There is no evidence of tropical communities successfully tracking changing climatic conditions latitudinally (Colwell and Feeley, 2024). And although upslope migrations have been reported in many montane regions, plant migration significantly lagging behind the rate of change is the norm (Alexander et al., 2018; Corlett, 2015; Corlett and Westcott, 2013). Moreover, contrary to predictions, a global survey of 987 montane animal and plant species found that 28% actually moved downslope (Mamantov et al., 2021).

All these uncertainties call for a rethinking of the aims and utility of modelling studies, which in some instances might even make some modelling steps redundant. For example, it is common practice in modelling studies to compare multiple dispersal assumptions (Corlett, 2015). However, a recent modelling study in a montane region in eastern Mexico found negligible differences between the limited and unlimited dispersal assumptions, except for the most critically endangered species (Ortega et al., 2024), i.e., precisely for the subset of species for which natural dispersion is most unlikely. Therefore, I believe that given the rate of anthropogenic climate change (Forster et al., 2024), modellers should work under the default assumption that montane, rare and range-restricted species will not migrate on their own. This does not mean that the no-dispersal scenario should become the default, but rather that it might be redundant. Instead, SDMs should be seen as indicators of the most suitable areas for complementary active conservation schemes, such as managed recovery and reintroductions (Heywood, 2019) or assisted migration (Sáenz-Romero et al., 2020b, 2016).

Doubts on the efficacy of assisted migration programmes and ethical concerns about the intentional relocation of species in general have been raised (Corlett, 2015), for instance, the potential that some species might become invasive. However, the same characteristics that make a species vulnerable in the first place make this risk highly unlikely, including limited dispersal abilities, slow reproductive rates, specialised habitat and nutritional requirements, narrow physiological tolerances, among others (Pacifici et al., 2015). There is promising evidence that species can successfully establish at higher elevations (García-Hernández et al., 2019; Toledo-Aceves and del-Val, 2021a), where natural seed dispersion rarely reaches (Corlett and Westcott, 2013). Thus, the pertinence and necessity of active conservation measures, such as *ex situ* cultivation, targeted reintroduction and assisted migration, are increasingly acknowledged (Hällfors et al., 2016; Heywood, 2019; Sáenz-Romero et al., 2020b, 2016), prioritising species that have already been identified as endangered or vulnerable, e.g., *Abies guatemalensis*, *A. hickelii*, *Juglans pyriformis*, *Magnolia schiedeana*, *Podocarpus matudae* and *Quercus constaricensis*. Other relevant factors, such as limited pollination (Li et al., 2016) and decline of nurse shrubs (Soto-Correa et al., 2013) have also been explored, although addressing them might be more challenging and effort-intensive.

5.3 Significance of the predicted functional diversity shifts of Mesoamerican montane-specialist trees in response to climate change

The observed spatial patterns of five individual functional traits described in Chapter 4 performed well in describing the functional diversity within the montane-specialist tree community of Mesoamerica. Despite the lack of abundance data to generate accurate site-specific estimates per species, the general latitudinal trends of tree height and leaf area coincide with floristic trends described for the region. For example, the increasing dominance of pines toward the northern part of the study area and their functional space overlap with oaks (Corlett and Hughes, 2015).

More remarkable is the projected relative stability of the spatial patterns throughout the study area, despite the worrying magnitude or suitable area losses projected for most species under future climatic conditions. The fact that merely 272 species (out of the >32,000 plant species estimated for Mesoamerica including all growth forms and elevations; Ramírez-Barahona et al., 2023) exhibit such resilient functional diversity under a severe climate change scenario, suggests a high degree of functional redundancy (Monge-González et al., 2021). This is indicative of the significant contribution of rare and range-restricted species to functional diversity (Leitão et al., 2016; Mi et al., 2021), as well as to vulnerable ecosystem functions (Mouillot et al., 2013) and ecosystem services, such as carbon storage (Kearsley et al., 2019).

Therefore, further research on TMF functional diversity is not only important to better understand the mechanisms that make their communities resilient to environmental change (Ahrens et al., 2020; Kühn et al., 2021), but also to ponder their value as key ecosystems to mitigate the drivers of climate change. It is generally assumed that the carbon storage capacity of TMFs is tiny compared to lowland and temperate forests. However, carbon dynamics in many tropical regions are still not well understood (Cuni-Sanchez et al., 2021) and there is evidence that it has been underestimated (Álvarez-Dávila et al., 2017). Large-sized montane tree species have high carbon storage capacity (Kearsley et al., 2019; Venter et al., 2017), and trees' functional traits also influence other carbon sinks, such as soil organic content (Augusto and Boča, 2022; Iwashita et al., 2013).

Similarly, the observed variability in the height-to-diameter ratio is indicative of the uneven canopy structure characteristic of montane forests (Scatena et al., 2010). Maintaining a complex canopy structure is important, as it is the support of the epiphytic communities that are typical of TMFs and contribute to their water interception capacity (Ah-Peng et al., 2017; Gotsch et al., 2016), but are simultaneously highly threatened by hydric stress caused by rising temperatures and cloud base levels (Gotsch et al., 2018; He et al., 2016; Wagner et al., 2014; Zuleta et al., 2016). The reduction of functional complexity would result in the loss of key microhabitat for montane fauna (Scheffers et al., 2014, 2013a, 2013b) and in decreased water flows on which human populations rely (Viviroli et al., 2011; Zhou et al., 2017).

The lack of obvious spatial patterns for the other traits of the montane-specialist community can in itself be informative. For example, wood density was the least variable trait among the 272 montane-specialist species. The low variation in wood density is indicative of a late successional assemblage (Castillo-Figueroa et al., 2023), possibly indicating that co-occurrence of these montane specialists signals areas with low disturbance. Conversely, high wood density is generally associated with higher resistance to cavitation (Ahrens et al., 2020; Kühn et al., 2021). Thus, the relatively low average wood density of these montane specialists could make them vulnerable to increased vapour pressure deficit (Esperón-Rodríguez and Barradas, 2015b).

The influence of other functional traits cannot be overstated. The functional trait analysis in Chapter 4 was largely constrained by data availability, so some important level of detail is likely missing. For instance, traits that regulate water-use efficiency (e.g., stomatal control) or that influence reproduction (e.g., pollination and seed dispersal syndrome) and establishment (e.g. seed mass, growth rate, etc.) would greatly improve our predictions of which combinations of traits make species more or less likely to successfully cope with the current rates of climate change.

5.4 Implications for the direction of future research

Research on the impacts of climate change on TMFs is far from complete, as shown in Chapter 2, and the single gravest obstacle for this thesis was data availability, yet the resources and time to act are limited. Therefore, our first instinct might be to devote the bulk of research towards the most neglected world regions and thematic gaps. However, it might be more efficient to instead focus on tackling the discrepancies in the existing literature.

Mismatches between theory, prediction and observations are not that uncommon. A clear example is the relatively high and unexpected proportion of montane species that are migrating downslope in response to climate change (Mamantov et al., 2021). What underlying mechanisms cause some species to move in the opposite direction of environmental shifts? What are the implications of these counterintuitive migrations in terms of community composition and interspecific interactions? Which is a better descriptor of the distribution and functional diversity of montane flora: temperature or water availability? Or in which cases one is more determinant than the other, or is it another factor altogether?

Uncertainties like these have been identified in relation to other widespread notions about the effects of climate change on global flora. A review devoted to unexpected or counterintuitive responses of plants to climate change (Parmesan and Hanley, 2015) listed some notable examples, such as (1) erroneous assumptions about plants' phonological responses to changing temperatures in autumn that were assumed to work the same as in spring; (2) negligible physiological responses of plants to increases in eCO_2, despite a widespread notion that it would enhance productivity; and (3) difficulties in establishing a clear link between any one functional trait or combination of traits and responses to climatic shifts. Yet rather than portraying those complex responses as an obstacle, the authors point out that they are testament to plants' inherent adaptability and urge designing research in ways that allow disentangling inconsistent findings, for example, by coordinating manipulation experiments across networks of field sites.

By refining our knowledge of forest dynamics in well-understood proxy regions, the expansion of research into understudied regions would be better informed to specifically identify cases where the same diversity patterns are not applicable and understand why. The accuracy and utility of SDMs could thus be improved to forecast distributional changes and predict functional, ecological, and even genetic-level impacts in response to environmental stressors of tropical montane flora. This consideration is also applicable to other taxa and ecosystem types.

In conservation, a key underrepresented aspect is the link between climate and genetic diversity. Studies on intraspecific responses to environmental stressors are a lot less common than those at the species level (Carvalho et al., 2019). However, the relevance of genetic studies for conservation is increasingly recognised, by improving our understanding of species' demographic histories, population structure and connectivity, adaptive potential, and deleterious variation, among others (Hohenlohe et al., 2021).

More specifically for this thesis, immediate efforts would be to use field surveys to corroborate the characterization of montane-specialist tree communities across the study area. Abundance and population structure could be obtained simultaneously, which in turn could refine the functional trait analysis and allow, for example, for the precise estimation of the aboveground biomass of these species. And additionally, genetic studies of rare montane specialist species could reveal their life histories, connectivity, and adaptive potential amid environmental changes.

5.5 Conclusions

The overarching goal of this thesis was to widen our understanding of the likely impacts of climate change on the montane-specialist tree community in Mesoamerica. My findings highlight the need to broaden the methodological, thematic, taxonomic, and geographical scope of research on tropical montane forests globally, while also emphasising the importance of refining existing approaches to provide immediate and reliable conservation insights. By leveraging moderate to circumstantial evidence, such as remote sensing, palaeoecological data, and advanced forecasting models, we could better address current uncertainties and knowledge gaps, particularly in well-studied regions. Additionally, natural variability along TMFs' altitudinal gradients allows conducting field manipulation experiments to simulate complex environmental conditions more accurately than *ex situ* studies.

This thesis further emphasises the critical role of robust modelling approaches, calling for the development of novel forecasting algorithms that account for ecological factors like species dispersal limitations and biotic interactions. Such enhancements are vital for designing protected areas that accommodate future habitats for entire biotic communities, thus minimising biodiversity loss and preserving ecosystem functions. While trees are central to forest ecosystems, it is also crucial to consider the responses of other taxonomic groups and genetic diversity to climate change, as these factors are key for the long-term viability of TMF populations. In the Mesoamerican bioregion, findings indicate significant reductions in climatically suitable areas for montane-specialist tree species, yet some regions may remain refuges for biodiversity at the highest elevations under a severe climate change scenario. These findings advocate for stringent protection measures, active conservation, reintroduction and assisted migration programmes for key vulnerable species, and immediate transnational collaboration to shield and connect these critical habitats that do not rely on unrealistic assumptions of natural dispersal.

The research also highlights the potential of functional trait analysis, which can provide valuable insights into ecosystem functioning and carbon sequestration potential, even with absence of abundance data. However, to achieve more accurate estimates of carbon stocks and identify key areas for conservation, additional data on abundance, basal area, and phenology are necessary. Incorporating more functional traits, intraspecific variation, genetic and abundance data could further clarify the relationships between functional diversity and environmental factors, helping to identify traits that may confer resilience or vulnerability to climate change and increase the success rate of active conservation strategies.

Overall, this thesis provides a foundational framework for future research and policy design aimed at safeguarding the ecological and biological integrity of montane-specialist trees in Mesoamerica, emphasising the importance of integrating multiple lines of evidence, cross-disciplinary approaches, and international cooperation in addressing the complex challenges posed by climate change.

References

Abera, T.A., Heiskanen, J., Maeda, E.E., Muhammed, M.A., Bhandari, N., Vakkari, V., Hailu, B.T., Pellikka, P.K.E., Hemp, A., Van Zyl, P.G., Zeuss, D., 2024. Deforestation amplifies climate change effects on warming and cloud level rise in African montane forests. Nat. Commun. 15, 6992. https://doi.org/10.1038/s41467-024-51324-7

Adler, P.B., Fajardo, A., Kleinhesselink, A.R., Kraft, N.J., 2013. Trait-based tests of coexistence mechanisms. Ecol. Lett. 16, 1294–1306.

Aguirre-Gutiérrez, J., Berenguer, E., Oliveras Menor, I., Bauman, D., Corral-Rivas, J.J., Nava-Miranda, M.G., Both, S., Ndong, J.E., Ondo, F.E., Bengone, N.N., Mihinhou, V., Dalling, J.W., Heineman, K., Figueiredo, A., González-M, R., Norden, N., Hurtado-M, A.B., González, D., Salgado-Negret, B., Reis, S.M., Moraes De Seixas, M.M., Farfan-Rios, W., Shenkin, A., Riutta, T., Girardin, C.A.J., Moore, S., Abernethy, K., Asner, G.P., Bentley, L.P., Burslem, D.F.R.P., Cernusak, L.A., Enquist, B.J., Ewers, R.M., Ferreira, J., Jeffery, K.J., Joly, C.A., Marimon-Junior, B.H., Martin, R.E., Morandi, P.S., Phillips, O.L., Bennett, A.C., Lewis, S.L., Quesada, C.A., Marimon, B.S., Kissling, W.D., Silman, M., Teh, Y.A., White, L.J.T., Salinas, N., Coomes, D.A., Barlow, J., Adu-Bredu, S., Malhi, Y., 2022. Functional susceptibility of tropical forests to climate change. Nat. Ecol. Evol. 6, 878–889. https://doi.org/10.1038/s41559-022-01747-6

Ah-Peng, C., Cardoso, A.W., Flores, O., West, A., Wilding, N., Strasberg, D., Hedderson, T.A.J., 2017. The role of epiphytic bryophytes in interception, storage, and the regulated release of atmospheric moisture in a tropical montane cloud forest. J. Hydrol. 548, 665–673. https://doi.org/10.1016/j.jhydrol.2017.03.043

Ahrens, C.W., Andrew, M.E., Mazanec, R.A., Ruthrof, K.X., Challis, A., Hardy, G., Byrne, M., Tissue, D.T., Rymer, P.D., 2020. Plant functional traits differ in adaptability and are predicted to be differentially affected by climate change. Ecol. Evol. 10, 232–248. https://doi.org/10.1002/ece3.5890

Aiba, S.-I., Kitayama, K., 2002. Effects of the 1997–98 El Niño drought on rain forests of Mount Kinabalu, Borneo. J. Trop. Ecol. 18, 215–230. https://doi.org/10.1017/S0266467402002146

Aiello-Lammens, M.E., Boria, R.A., Radosavljevic, A., Vilela, B., Anderson, R.P., Bjornson, R., Weston, S., Aiello-Lammens, M.M.E., 2019. Package 'spThin.'

Alexander, J.M., Chalmandrier, L., Lenoir, J., Burgess, T.I., Essl, F., Haider, S., Kueffer, C., McDougall, K., Milbau, A., Nuñez, M.A., Pauchard, A., Rabitsch, W., Rew, L.J., Sanders, N.J., Pellissier, L., 2018. Lags in the response of mountain plant communities to climate change. Glob. Change Biol. 24, 563–579. https://doi.org/10.1111/gcb.13976

Alfaro, R.I., Fady, B., Vendramin, G.G., Dawson, I.K., Fleming, R.A., Sáenz-Romero, C., Lindig-Cisneros, R.A., Murdock, T., Vinceti, B., Navarro, C.M., Skrøppa, T., Baldinelli, G., El-Kassaby, Y.A., Loo, J., 2014. The role of forest genetic resources in responding to biotic and abiotic factors in the context of anthropogenic climate change. For. Ecol. Manag. 333, 76–87. https://doi.org/10.1016/j.foreco.2014.04.006

Alfonso-Corrado, C., Naranjo-Luna, F., Clark-Tapia, R., Campos, J., Rojas-Soto, O., Luna-Krauletz, M., Bodenhorn, B., Gorgonio-Ramírez, M., Pacheco-Cruz, N., 2017. Effects of Environmental Changes on the Occurrence of *Oreomunnea mexicana* (Juglandaceae) in a Biodiversity Hotspot Cloud Forest. Forests 8, 261. https://doi.org/10.3390/f8080261

Allen, C.D., Macalady, A.K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., Kitzberger, T., Rigling, A., Breshears, D.D., Hogg, E.H. (Ted), Gonzalez, P., Fensham, R., Zhang, Z., Castro, J., Demidova, N., Lim, J.-H., Allard, G., Running, S.W., Semerci, A., Cobb, N., 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. For. Ecol. Manag. 259, 660–684. https://doi.org/10.1016/j.foreco.2009.09.001

Allouche, O., Tsoar, A., Kadmon, R., 2006. Assessing the accuracy of species distribution models: prevalence, kappa and the true skill statistic (TSS): Assessing the accuracy of distribution models. J. Appl. Ecol. 43, 1223–1232. https://doi.org/10.1111/j.1365-2664.2006.01214.x

Altamirano-León, S., Ramírez-Martínez, J.C., Díaz-Porras, D.F., Balam-Narváez, R., Luna-Vega, I., Contreras-Medina, R., 2022. Climate change and impact on distribution of tree species in the cloud forest of Oaxaca. Rev. Chapingo Ser. Cienc. For. 28.
Altarez, R.D.D., Apan, A., Maraseni, T., 2022. Spaceborne satellite remote sensing of tropical montane forests: a review of applications and future trends. Geocarto Int. 1–29. https://doi.org/10.1080/10106049.2022.2060330
Álvarez-Dávila, E., Cayuela, L., González-Caro, S., Aldana, A.M., Stevenson, P.R., Phillips, O., Cogollo, Á., Peñuela, M.C., Von Hildebrand, P., Jiménez, E., Melo, O., Londoño-Vega, A.C., Mendoza, I., Velásquez, O., Fernández, F., Serna, M., Velázquez-Rua, C., Benítez, D., Rey-Benayas, J.M., 2017. Forest biomass density across large climate gradients in northern South America is related to water availability but not with temperature. PLOS ONE 12, e0171072. https://doi.org/10.1371/journal.pone.0171072
Anchukaitis, K.J., Evans, M.N., 2010. Tropical cloud forest climate variability and the demise of the Monteverde golden toad. Proc. Natl. Acad. Sci. 107, 5036–5040. https://doi.org/10.1073/pnas.0908572107
Anderson, A.S., Storlie, C.J., Shoo, L.P., Pearson, R.G., Williams, S.E., 2013. Current Analogues of Future Climate Indicate the Likely Response of a Sensitive Montane Tropical Avifauna to a Warming World. PLoS ONE 8, e69393. https://doi.org/10.1371/journal.pone.0069393
Andrade, R., Freitas, L., 2021. Impact of an IUCN national Red List of threatened flora on scientific attention. Endanger. Species Res. 46, 175–184. https://doi.org/10.3354/esr01154
Ankersen, T.T., 2005. 10 Addressing the conservation conundrum in Mesoamerica. Bioregionalism 171.
Apaza-Quevedo, A., Lippok, D., Hensen, I., Schleuning, M., Both, S., 2015. Elevation, topography, and edge effects drive functional composition of woody plant species in tropical montane forests. Biotropica 47, 449–458.
Araújo, M.B., Anderson, R.P., Márcia Barbosa, A., Beale, C.M., Dormann, C.F., Early, R., Garcia, R.A., Guisan, A., Maiorano, L., Naimi, B., O'Hara, R.B., Zimmermann, N.E., Rahbek, C., 2019. Standards for distribution models in biodiversity assessments. Sci. Adv. 5, eaat4858. https://doi.org/10.1126/sciadv.aat4858
Araújo, M.B., Peterson, A.T., 2012. Uses and misuses of bioclimatic envelope modeling 93, 14.
Augusto, L., Boča, A., 2022. Tree functional traits, forest biomass, and tree species diversity interact with site properties to drive forest soil carbon. Nat. Commun. 13, 1097.
Bader, M.Y., Reich, T., Wagner, S., González González, A.S., Zotz, G., 2013. Differences in desiccation tolerance do not explain altitudinal distribution patterns of tropical bryophytes. J. Bryol. 35, 47–56. https://doi.org/10.1179/1743282012Y.0000000033
Báez, S., Homeier, J., 2018. Functional traits determine tree growth and ecosystem productivity of a tropical montane forest: Insights from a long-term nutrient manipulation experiment. Glob. Change Biol. 24, 399–409. https://doi.org/10.1111/gcb.13905
Barker, B.S., Ríos-Franceschi, A., 2015. Population Declines of Mountain Coqui (*Eleutherodactylus portoricensis*) in the Cordillera Central of Puerto Rico 22.
Barquet, K., 2015. Building a bioregion through transboundary conservation in Central America. Nor. Geogr. Tidsskr. - Nor. J. Geogr. 69, 265–276. https://doi.org/10.1080/00291951.2015.1087421
Baselga, A., Orme, D., Villeger, S., De Bortoli, J., Leprieur, F., Baselga, M.A., 2018. Package 'betapart.' Partitioning Beta Divers. Turnover Nestedness Compon. Version 1.
Bawa, K.S., Dayanandan, S., 1998. Global Climate Change and Tropical Forest Genetic Resources, in: Markham, A. (Ed.), Potential Impacts of Climate Change on Tropical Forest Ecosystems. Springer Netherlands, Dordrecht, pp. 333–345. https://doi.org/10.1007/978-94-017-2730-3_16
Becker, J.N., Kuzyakov, Y., 2018. Teatime on Mount Kilimanjaro: Assessing climate and land-use effects on litter decomposition and stabilization using the Tea Bag Index. Land Degrad. Dev. 29, 2321–2329. https://doi.org/10.1002/ldr.2982
Bendix, J., Aguire, N., Beck, E., Bräuning, A., Brandl, R., Breuer, L., Böhning-Gaese, K., de Paula, M.D., Hickler, T., Homeier, J., Inclan, D., Leuschner, C., Neuschulz, E.L., Schleuning, M., Suarez, J.P., Trachte, K., Wilcke, W., Windhorst, D., Farwig, N., 2021. A research framework for projecting

ecosystem change in highly diverse tropical mountain ecosystems. Oecologia 195, 589–600. https://doi.org/10.1007/s00442-021-04852-8
Benning, T.L., LaPointe, D., Atkinson, C.T., Vitousek, P.M., 2002. Interactions of climate change with biological invasions and land use in the Hawaiian Islands: Modeling the fate of endemic birds using a geographic information system. Proc. Natl. Acad. Sci. 99, 14246–14249. https://doi.org/10.1073/pnas.162372399
Bergamin, R.S., Bastazini, V.A.G., Esquivel-Muelbert, A., Bordin, K.M., Klipel, J., Debastiani, V.J., Vibrans, A.C., Loyola, R., Müller, S.C., 2024. Elevational shifts in tree community composition in the Brazilian Atlantic Forest related to climate change. J. Veg. Sci. 35, e13289. https://doi.org/10.1111/jvs.13289
Berrios, H.K., Coronado, I., Marsico, T.D., 2022. High species richness and turnover of vascular epiphytes is associated with water availability along the elevation gradient of Volcán Maderas, Nicaragua. Ecol. Evol. 12. https://doi.org/10.1002/ece3.9501
Biondi, F., 2014. Paleoecology grand challenge. Front. Ecol. Evol. 2. https://doi.org/10.3389/fevo.2014.00050
Bitencourt, C., Rapini, A., Santos Damascena, L., De Marco Junior, P., 2016. The worrying future of the endemic flora of a tropical mountain range under climate change. Flora - Morphol. Distrib. Funct. Ecol. Plants 218, 1–10. https://doi.org/10.1016/j.flora.2015.11.001
Bivand, R., Altman, M., Anselin, L., Assunção, R., Berke, O., Bernat, A., Blanchet, G., 2017. Package 'spdep.' Spat. Depend. Weight. Schemes Stat. R Package Version 1–1.
Blonder, B., 2018. Hypervolume concepts in niche-and trait-based ecology. Ecography 41, 1441–1455.
Blonder, B., 2016. Do Hypervolumes Have Holes? Am. Nat. 187, E93–E105. https://doi.org/10.1086/685444
Blonder, B., Morrow, C.B., Maitner, B., Harris, D.J., Lamanna, C., Violle, C., Enquist, B.J., Kerkhoff, A.J., 2018. New approaches for delineating n-dimensional hypervolumes. Methods Ecol. Evol. 9, 305–319.
Bobrowski, M., Weidinger, J., Schickhoff, U., 2021. Is New Always Better? Frontiers in Global Climate Datasets for Modeling Treeline Species in the Himalayas. Atmosphere 12, 543. https://doi.org/10.3390/atmos12050543
Boege, E., Vidriales-Chan, G., García-Coll, I., Mondragón, M., Rivas, A.J., Lozada, M.P., Soto, F., 2008. El patrimonio biocultural de los pueblos indígenas de México: hacia la conservación in situ de la biodiversidad y agrodiversidad en los territorios indígenas. Instituto Nacional de Antropología e Historia México.
Bonan, G.B., 2008. Forests and Climate Change: Forcings, Feedbacks, and the Climate Benefits of Forests. Science 320, 1444–1449. https://doi.org/10.1126/science.1155121
Brack, D., 2019. Forests and Climate Change.
Brown, A.D., Kappelle, M., n.d. Introducción a los bosques nublados del Neotrópico: Una síntesis regional 17.
Bruijnzeel, L., Kappelle, M., Mulligan, M., Scatena, F., 2010. Tropical montane cloud forests: state of knowledge and sustainability perspectives in a changing world.
Bruijnzeel, L.A., 2004. Hydrological functions of tropical forests: not seeing the soil for the trees? Agric. Ecosyst. Environ. 104, 185–228. https://doi.org/10.1016/j.agee.2004.01.015
Bruijnzeel, L.A., 2001. Hydrology of tropical montane cloud forests: a reassessment. Land Use Water Resour. Res. 1, 1–1.
Bruijnzeel, L.A., Hamilton, L.S., 2000. Decision time for cloud forests.
Bruijnzeel, L.A., Mulligan, M., Scatena, F.N., 2011a. Hydrometeorology of tropical montane cloud forests: emerging patterns. Hydrol. Process. 25, 465–498. https://doi.org/10.1002/hyp.7974
Bruijnzeel, L.A., Scatena, F.N., Hamilton, L.S., 2011b. Tropical Montane Cloud Forests: Science for Conservation and Management. Cambridge University Press, Cambridge. https://doi.org/10.1017/CBO9780511778384
Buytaert, W., Cuesta-Camacho, F., Tobón, C., 2011. Potential impacts of climate change on the environmental services of humid tropical alpine regions: Climate change and environmental services. Glob. Ecol. Biogeogr. 20, 19–33. https://doi.org/10.1111/j.1466-8238.2010.00585.x

Calaway, R., Analytics, R., Weston, S., Tenenbaum, D., Calaway, M.R., 2015. Package “doParallel.”
Camarero, J.J., Gazol, A., Sánchez-Salguero, R., Fajardo, A., McIntire, E.J.B., Gutiérrez, E., Batllori, E., Boudreau, S., Carrer, M., Diez, J., Dufour-Tremblay, G., Gaire, N.P., Hofgaard, A., Jomelli, V., Kirdyanov, A.V., Lévesque, E., Liang, E., Linares, J.C., Mathisen, I.E., Moiseev, P.A., Sangüesa-Barreda, G., Shrestha, K.B., Toivonen, J.M., Tutubalina, O.V., Wilmking, M., 2021. Global fading of the temperature–growth coupling at alpine and polar treelines. Glob. Change Biol. 27, 1879–1889. https://doi.org/10.1111/gcb.15530
Campbell, A., Kapos, V., Scharlemann, J.P.W., Bubb, P., Chenery, A., Coad, L., Dickson, B., Doswald, N., Khan, M.S.I., Kershaw, F., Rashid, M., 2009. Review of the literature on the links between biodiversity and climate change: impacts, adaptation, and mitigation. Secr. Conv. Biol. Divers. Technical Series.
Campos-Cerqueira, M., Aide, T.M., 2017. Lowland extirpation of anuran populations on a tropical mountain. PeerJ 5, e4059. https://doi.org/10.7717/peerj.4059
Carmona, C.P., Bueno, C.G., Toussaint, A., Träger, S., Díaz, S., Moora, M., Munson, A.D., Pärtel, M., Zobel, M., Tamme, R., 2021. Fine-root traits in the global spectrum of plant form and function. Nature 597, 683–687. https://doi.org/10.1038/s41586-021-03871-y
Carmona, C.P., De Bello, F., Mason, N.W.H., Lepš, J., 2016. Traits Without Borders: Integrating Functional Diversity Across Scales. Trends Ecol. Evol. 31, 382–394. https://doi.org/10.1016/j.tree.2016.02.003
Carmona, C.P., Pavanetto, N., Puglielli, G., 2024. funspace: An R package to build, analyse and plot functional trait spaces. Divers. Distrib. e13820. https://doi.org/10.1111/ddi.13820
Carvalho, S.B., Torres, J., Tarroso, P., Velo-Antón, G., 2019. Genes on the edge: A framework to detect genetic diversity imperiled by climate change. Glob. Change Biol. 25, 4034–4047. https://doi.org/10.1111/gcb.14740
Castellanos-Acuña, D., Lindig-Cisneros, R., Sáenz-Romero, C., 2015. Altitudinal assisted migration of Mexican pines as an adaptation to climate change. Ecosphere 6, 1–16. https://doi.org/10.1890/ES14-00375.1
Castillo-Figueroa, D., González-Melo, A., Posada, J.M., 2023. Wood density is related to aboveground biomass and productivity along a successional gradient in upper Andean tropical forests. Front. Plant Sci. 14, 1276424. https://doi.org/10.3389/fpls.2023.1276424
Castro, M.B., Barbosa, A.C.M.C., Pompeu, P.V., Eisenlohr, P.V., de Assis Pereira, G., Apgaua, D.M.G., Pires-Oliveira, J.C., Barbosa, J.P.R.A.D., Fontes, M.A.L., dos Santos, R.M., Tng, D.Y.P., 2020. Will the emblematic southern conifer *Araucaria angustifolia* survive to climate change in Brazil? Biodivers. Conserv. 29, 591–607. https://doi.org/10.1007/s10531-019-01900-x
Catenazzi, A., Lehr, E., Rodriguez, L.O., Vredenburg, V.T., 2010. *Batrachochytrium dendrobatidis* and the Collapse of Anuran Species Richness and Abundance in the Upper Manu National Park, Southeastern Peru: Frog Declines in the Tropical Andes. Conserv. Biol. no-no. https://doi.org/10.1111/j.1523-1739.2010.01604.x
Catenazzi, A., Lehr, E., Vredenburg, V.T., 2014. Thermal Physiology, Disease, and Amphibian Declines on the Eastern Slopes of the Andes: Frog Declines in the Tropical Andes. Conserv. Biol. 28, 509–517. https://doi.org/10.1111/cobi.12194
Center, A., Etterson, J.R., Deacon, N.J., Cavender-Bares, J., 2016. Seed production timing influences seedling fitness in the tropical live oak *Quercus oleoides* of Costa Rican dry forests. Am. J. Bot. 103, 1407–1419. https://doi.org/10.3732/ajb.1500389
Cerano-Paredes, J., Iniguez, J.M., Villanueva-Díaz, J., Vázquez-Selem, L., Cervantes-Martínez, R., Esquivel-Arriaga, G., Franco-Ramos, O., Rodríguez-Trejo, D.A., 2021. Effects of climate on historical fire regimes (1451–2013) in *Pinus hartwegii* forests of Cofre de Perote National Park, Veracruz, Mexico. Dendrochronologia 65, 125784. https://doi.org/10.1016/j.dendro.2020.125784
Chakraborty, A., Joshi, P.K., Ghosh, A., Areendran, G., 2013. Assessing biome boundary shifts under climate change scenarios in India. Ecol. Indic. 34, 536–547. https://doi.org/10.1016/j.ecolind.2013.06.013
Chamberlain, S., Ram, K., Barve, V., Mcglinn, D., Chamberlain, M.S., 2017. Package ‘rgbif.’ Interface Glob. Biodivers. Inf. Facil. ‘API 5, 0–9.

Chapman, C.A., Valenta, K., Bonnell, T.R., Brown, K.A., Chapman, L.J., 2018. Solar radiation and ENSO predict fruiting phenology patterns in a 15-year record from Kibale National Park, Uganda. Biotropica 50, 384–395. https://doi.org/10.1111/btp.12559
Chapman, H., Cordeiro, N.J., Dutton, P., Wenny, D., Kitamura, S., Kaplin, B., Melo, F.P.L., Lawes, M.J., 2016. Seed-dispersal ecology of tropical montane forests. J. Trop. Ecol. 32, 437–454. https://doi.org/10.1017/S0266467416000389
Chave, J., Réjou-Méchain, M., Búrquez, A., Chidumayo, E., Colgan, M.S., Delitti, W.B.C., Duque, A., Eid, T., Fearnside, P.M., Goodman, R.C., Henry, M., Martínez-Yrízar, A., Mugasha, W.A., Muller-Landau, H.C., Mencuccini, M., Nelson, B.W., Ngomanda, A., Nogueira, E.M., Ortiz-Malavassi, E., Pélissier, R., Ploton, P., Ryan, C.M., Saldarriaga, J.G., Vieilledent, G., 2014. Improved allometric models to estimate the aboveground biomass of tropical trees. Glob. Change Biol. 20, 3177–3190. https://doi.org/10.1111/gcb.12629
Chen, I.-C., Shiu, H.-J., Benedick, S., Holloway, J.D., Chey, V.K., Barlow, H.S., Hill, J.K., Thomas, C.D., 2009. Elevation increases in moth assemblages over 42 years on a tropical mountain. Proc. Natl. Acad. Sci. 106, 1479–1483. https://doi.org/10.1073/pnas.0809320106
Chirino, E., Ruiz-Yanetti, S., Vilagrosa, A., Mera, X., Espinoza, M., Lozano, P., 2017. Morpho-functional traits and plant response to drought conditions in seedlings of six native species of Ecuadorian Ecosystems. Flora 233, 58–67. https://doi.org/10.1016/j.flora.2017.05.012
Christmann, T., Menor, I.O., 2021. A synthesis and future research directions for tropical mountain ecosystem restoration. Sci. Rep. 11, 23948. https://doi.org/10.1038/s41598-021-03205-y
Climent, J., Tapias, R., Pardos, J.A., Gil, L., 2004. Fire adaptations in the Canary Islands pine (Pinus canariensis). Plant Ecol. Former. Veg. 171, 185–196. https://doi.org/10.1023/B:VEGE.0000029374.64778.68
Cody, S., Richardson, J.E., Rull, V., Ellis, C., Pennington, R.T., 2010. The Great American Biotic Interchange revisited. Ecography no-no. https://doi.org/10.1111/j.1600-0587.2010.06327.x
Collin, P., 2001. L'adaptation au milieu chez les plantes vasculaires. L'Année Biol. 40, 21–42. https://doi.org/10.1016/S0003-5017(01)72083-1
Colwell, R.K., Brehm, G., Cardelus, C.L., Gilman, A.C., Longino, J.T., 2008. Global Warming, Elevational Range Shifts, and Lowland Biotic Attrition in the Wet Tropics. Science 322, 258–261. https://doi.org/10.1126/science.1162547
Colwell, R.K., Feeley, K.J., 2024. Still little evidence of poleward range shifts in the tropics, but lowland biotic attrition may be underway. Biotropica e13358. https://doi.org/10.1111/btp.13358
Colyn, R.B., Ehlers Smith, D.A., Ehlers Smith, Y.C., Smit-Robinson, H., Downs, C.T., 2020. Predicted distributions of avian specialists: A framework for conservation of endangered forests under future climates. Divers. Distrib. 26, 652–667. https://doi.org/10.1111/ddi.13048
Comarazamy, D.E., González, J.E., 2011. Regional long-term climate change (1950-2000) in the midtropical Atlantic and its impacts on the hydrological cycle of Puerto Rico: REGIONAL AND LOCAL CLIMATE CHANGES. J. Geophys. Res. Atmospheres 116. https://doi.org/10.1029/2010JD015414
Comarazamy, D.E., González, J.E., Moshary, F., Piasecki, M., 2015. On the Hydrometeorological Changes of a Tropical Water Basin in the Caribbean and Its Sensitivity to Midterm Changes in Regional Climate. J. Hydrometeorol. 16, 997–1013. https://doi.org/10.1175/JHM-D-14-0083.1
Condit, R., Aguilar, S., Pérez, R., 2020. Trees of Panama: A complete checklist with every geographic range. For. Ecosyst. 7, 1–13.
Cooper, D.L.M., Lewis, S.L., Sullivan, M.J.P., Prado, P.I., Ter Steege, H., Barbier, N., Slik, F., Sonké, B., Ewango, C.E.N., Adu-Bredu, S., Affum-Baffoe, K., De Aguiar, D.P.P., Ahuite Reategui, M.A., Aiba, S.-I., Albuquerque, B.W., De Almeida Matos, F.D., Alonso, A., Amani, C.A., Do Amaral, D.D., Do Amaral, I.L., Andrade, A., De Andrade Miranda, I.P., Angoboy, I.B., Araujo-Murakami, A., Arboleda, N.C., Arroyo, L., Ashton, P., Aymard C, G.A., Baider, C., Baker, T.R., Balinga, M.P.B., Balslev, H., Banin, L.F., Bánki, O.S., Baraloto, C., Barbosa, E.M., Barbosa, F.R., Barlow, J., Bastin, J.-F., Beeckman, H., Begne, S., Bengone, N.N., Berenguer, E., Berry, N., Bitariho, R., Boeckx, P., Bogaert, J., Bonyoma, B., Boundja, P., Bourland, N., Boyemba Bosela, F., Brambach, F., Brienen, R., Burslem, D.F.R.P., Camargo, J.L., Campelo, W., Cano, A., Cárdenas, S., Cárdenas López, D., De Sá Carpanedo, R.,

Carrero Márquez, Y.A., Carvalho, F.A., Casas, L.F., Castellanos, H., Castilho, C.V., Cerón, C., Chapman, C.A., Chave, J., Chhang, P., Chutipong, W., Chuyong, G.B., Cintra, B.B.L., Clark, C.J., Coelho De Souza, F., Comiskey, J.A., Coomes, D.A., Cornejo Valverde, F., Correa, D.F., Costa, F.R.C., Costa, J.B.P., Couteron, P., Culmsee, H., Cuni-Sanchez, A., Dallmeier, F., Damasco, G., Dauby, G., Dávila, N., Dávila Doza, H.P., De Alban, J.D.T., De Assis, R.L., De Canniere, C., De Haulleville, T., De Jesus Veiga Carim, M., Demarchi, L.O., Dexter, K.G., Di Fiore, A., Din, H.H.M., Disney, M.I., Djiofack, B.Y., Djuikouo, M.-N.K., Do, T.V., Doucet, J.-L., Draper, F.C., Droissart, V., Duivenvoorden, J.F., Engel, J., Estienne, V., Farfan-Rios, W., Fauset, S., Feeley, K.J., Feitosa, Y.O., Feldpausch, T.R., Ferreira, C., Ferreira, J., Ferreira, L.V., Fletcher, C.D., Flores, B.M., Fofanah, A., Foli, E.G., Fonty, É., Fredriksson, G.M., Fuentes, A., Galbraith, D., Gallardo Gonzales, G.P., Garcia-Cabrera, K., García-Villacorta, R., Gomes, V.H.F., Gómez, R.Z., Gonzales, T., Gribel, R., Guedes, M.C., Guevara, J.E., Hakeem, K.R., Hall, J.S., Hamer, K.C., Hamilton, A.C., Harris, D.J., Harrison, R.D., Hart, T.B., Hector, A., Henkel, T.W., Herbohn, J., Hockemba, M.B.N., Hoffman, B., Holmgren, M., Honorio Coronado, E.N., Huamantupa-Chuquimaco, I., Hubau, W., Imai, N., Irume, M.V., Jansen, P.A., Jeffery, K.J., Jimenez, E.M., Jucker, T., Junqueira, A.B., Kalamandeen, M., Kamdem, N.G., Kartawinata, K., Kasongo Yakusu, E., Katembo, J.M., Kearsley, E., Kenfack, D., Kessler, M., Khaing, T.T., Killeen, T.J., Kitayama, K., Klitgaard, B., Labrière, N., Laumonier, Y., Laurance, S.G.W., Laurance, W.F., Laurent, F., Le, T.C., Le, T.T., Leal, M.E., Leão De Moraes Novo, E.M., Levesley, A., Libalah, M.B., Licona, J.C., Lima Filho, D.D.A., Lindsell, J.A., Lopes, A., Lopes, M.A., Lovett, J.C., Lowe, R., Lozada, J.R., Lu, X., Luambua, N.K., Luize, B.G., Maas, P., Magalhães, J.L.L., Magnusson, W.E., Mahayani, N.P.D., Makana, J.-R., Malhi, Y., Maniguaje Rincón, L., Mansor, A., Manzatto, A.G., Marimon, B.S., Marimon-Junior, B.H., Marshall, A.R., Martins, M.P., Mbayu, F.M., De Medeiros, M.B., Mesones, I., Metali, F., Mihindou, V., Millet, J., Milliken, W., Mogollón, H.F., Molino, J.-F., Mohd. Said, Mohd.N., Monteagudo Mendoza, A., Montero, J.C., Moore, S., Mostacedo, B., Mozombite Pinto, L.F., Mukul, S.A., Munishi, P.K.T., Nagamasu, H., Nascimento, H.E.M., Nascimento, M.T., Neill, D., Nilus, R., Noronha, J.C., Nsenga, L., Núñez Vargas, P., Ojo, L., Oliveira, A.A., De Oliveira, E.A., Ondo, F.E., Palacios Cuenca, W., Pansini, S., Pansonato, M.P., Paredes, M.R., Paudel, E., Pauletto, D., Pearson, R.G., Pena, J.L.M., Pennington, R.T., Peres, C.A., Permana, A., Petronelli, P., Peñuela Mora, M.C., Phillips, J.F., Phillips, O.L., Pickavance, G., Piedade, M.T.F., Pitman, N.C.A., Ploton, P., Popelier, A., Poulsen, J.R., Prieto, A., Primack, R.B., Priyadi, H., Qie, L., Quaresma, A.C., De Queiroz, H.L., Ramirez-Angulo, H., Ramos, J.F., Reis, N.F.C., Reitsma, J., Revilla, J.D.C., Riutta, T., Rivas-Torres, G., Robiansyah, I., Rocha, M., Rodrigues, D.D.J., Rodriguez-Ronderos, M.E., Rovero, F., Rozak, A.H., Rudas, A., Rutishauser, E., Sabatier, D., Sagang, L.B., Sampaio, A.F., Samsoedin, I., Satdichanh, M., Schietti, J., Schöngart, J., Scudeller, V.V., Seuaturien, N., Sheil, D., Sierra, R., Silman, M.R., Silva, T.S.F., Da Silva Guimarães, J.R., Simo-Droissart, M., Simon, M.F., Sist, P., Sousa, T.R., De Sousa Farias, E., De Souza Coelho, L., Spracklen, D.V., Stas, S.M., Steinmetz, R., Stevenson, P.R., Stropp, J., Sukri, R.S., Sunderland, T.C.H., Suzuki, E., Swaine, M.D., Tang, J., Taplin, J., Taylor, D.M., Tello, J.S., Terborgh, J., Texier, N., Theilade, I., Thomas, D.W., Thomas, R., Thomas, S.C., Tirado, M., Toirambe, B., De Toledo, J.J., Tomlinson, K.W., Torres-Lezama, A., Tran, H.D., Tshibamba Mukendi, J., Tumaneng, R.D., Umaña, M.N., Umunay, P.M., Urrego Giraldo, L.E., Valderrama Sandoval, E.H., Valenzuela Gamarra, L., Van Andel, T.R., Van De Bult, M., Van De Pol, J., Van Der Heijden, G., Vasquez, R., Vela, C.I.A., Venticinque, E.M., Verbeeck, H., Veridiano, R.K.A., Vicentini, A., Vieira, I.C.G., Vilanova Torre, E., Villarroel, D., Villa Zegarra, B.E., Vleminckx, J., Von Hildebrand, P., Vos, V.A., Vriesendorp, C., Webb, E.L., White, L.J.T., Wich, S., Wittmann, F., Zagt, R., Zang, R., Zartman, C.E., Zemagho, L., Zent, E.L., Zent, S., 2024. Consistent patterns of common species across tropical tree communities. Nature 625, 728–734. https://doi.org/10.1038/s41586-023-06820-z

Cordier, J.M., Lescano, J.N., Ríos, N.E., Leynaud, G.C., Nori, J., 2020. Climate change threatens micro-endemic amphibians of an important South American high-altitude center of endemism. Amphib.-Reptil. 41, 233–243. https://doi.org/10.1163/15685381-20191235

Corlett, R.T., 2015. Plant movements in response to rapid climate change, in: Routledge Handbook of Forest Ecology. Routledge, pp. 533–542.

Corlett, R.T., 2013. Where are the Subtropics? Biotropica 45, 273–275. https://doi.org/10.1111/btp.12028

Corlett, R.T., Hughes, A.C., 2015. Subtropical forests, in: Routledge Handbook of Forest Ecology. Routledge, pp. 46–55.
Corlett, R.T., Westcott, D.A., 2013. Will plant movements keep up with climate change? Trends Ecol. Evol. 28, 482–488. https://doi.org/10.1016/j.tree.2013.04.003
Corral-Rosas, V., Morrone, J.J., 2016. Analysing the assembly of cenocrons in the Mexican transition zone through a time-sliced cladistic biogeographic analysis. Aust. Syst. Bot. 29, 489. https://doi.org/10.1071/SB16048
Correa Ayram, C.A., Mendoza, M.E., Etter, A., Pérez Salicrup, D.R., 2017. Potential Distribution of Mountain Cloud Forest in Michoacán, Mexico: Prioritization for Conservation in the Context of Landscape Connectivity. Environ. Manage. 60, 86–103. https://doi.org/10.1007/s00267-017-0871-y
Correa-Díaz, A., Romero-Sánchez, M.E., Villanueva-Díaz, J., 2021. The greening effect characterized by the Normalized Difference Vegetation Index was not coupled with phenological trends and tree growth rates in eight protected mountains of central Mexico. For. Ecol. Manag. 496, 119402. https://doi.org/10.1016/j.foreco.2021.119402
Correa-Díaz, A., Silva, L.C.R., Horwath, W.R., Gómez-Guerrero, A., Vargas-Hernández, J., Villanueva-Díaz, J., Suárez-Espinoza, J., Velázquez-Martínez, A., 2020. From Trees to Ecosystems: Spatiotemporal Scaling of Climatic Impacts on Montane Landscapes Using Dendrochronological, Isotopic, and Remotely Sensed Data. Glob. Biogeochem. Cycles 34. https://doi.org/10.1029/2019GB006325
Costion, C.M., Simpson, L., Pert, P.L., Carlsen, M.M., John Kress, W., Crayn, D., 2015. Will tropical mountaintop plant species survive climate change? Identifying key knowledge gaps using species distribution modelling in Australia. Biol. Conserv. 191, 322–330. https://doi.org/10.1016/j.biocon.2015.07.022
Crain, B.J., Tremblay, R.L., 2017. Hot and Bothered: Changes in Microclimate Alter Chlorophyll Fluorescence Measures and Increase Stress Levels in Tropical Epiphytic Orchids. Int. J. Plant Sci. 178, 503–511. https://doi.org/10.1086/692767
Crausbay, S.D., Frazier, A.G., Giambelluca, T.W., Longman, R.J., Hotchkiss, S.C., 2014. Moisture status during a strong El Niño explains a tropical montane cloud forest's upper limit. Oecologia 175, 273–284. https://doi.org/10.1007/s00442-014-2888-8
Crausbay, S.D., Hotchkiss, S.C., 2010. Strong relationships between vegetation and two perpendicular climate gradients high on a tropical mountain in Hawai'i: Vegetation gradients on a tropical mountain. J. Biogeogr. 37, 1160–1174. https://doi.org/10.1111/j.1365-2699.2010.02277.x
Cruz-Cárdenas, G., Luis Villaseñor, J., López-Mata, L., Ortiz, E., 2012. Potential Distribution of Humid Mountain Forest in Mexico. Bot. Sci. 90, 305. https://doi.org/10.17129/botsci.394
Cruz-Elizalde, R., Magno-Benítez, I., Berriozabal-Islas, C., Ortíz-Pulido, R., Ramírez-Bautista, A., Hernández-Austria, R., 2020. Climatic niche, natural history, and conservation status of the Porthole Treefrog, *Charadrahyla taeniopus* (Günther, 1901) (Anura: Hylidae) in Mexico. Amphib Reptile Conserv 14, 12.
Cuervo-Robayo, A.P., Ureta, C., Gómez-Albores, M.A., Meneses-Mosquera, A.K., Téllez-Valdés, O., Martínez-Meyer, E., 2020. One hundred years of climate change in Mexico. PLOS ONE 15, e0209808. https://doi.org/10.1371/journal.pone.0209808
Cuni-Sanchez, A., Sullivan, M.J.P., Platts, P.J., Lewis, S.L., Marchant, R., Imani, G., Hubau, W., Abiem, I., Adhikari, H., Albrecht, T., Altman, J., Amani, C., Aneseyee, A.B., Avitabile, V., Banin, L., Batumike, R., Bauters, M., Beeckman, H., Begne, S.K., Bennett, A.C., Bitariho, R., Boeckx, P., Bogaert, J., Bräuning, A., Bulonvu, F., Burgess, N.D., Calders, K., Chapman, C., Chapman, H., Comiskey, J., De Haulleville, T., Decuyper, M., DeVries, B., Dolezal, J., Droissart, V., Ewango, C., Feyera, S., Gebrekirstos, A., Gereau, R., Gilpin, M., Hakizimana, D., Hall, J., Hamilton, A., Hardy, O., Hart, T., Heiskanen, J., Hemp, A., Herold, M., Hiltner, U., Horak, D., Kamdem, M.-N., Kayijamahe, C., Kenfack, D., Kinyanjui, M.J., Klein, J., Lisingo, J., Lovett, J., Lung, M., Makana, J.-R., Malhi, Y., Marshall, A., Martin, E.H., Mitchard, E.T.A., Morel, A., Mukendi, J.T., Muller, T., Nchu, F., Nyirambangutse, B., Okello, J., Peh, K.S.-H., Pellikka, P., Phillips, O.L., Plumptre, A., Qie, L., Rovero, F., Sainge, M.N., Schmitt, C.B., Sedlacek, O., Ngute, A.S.K., Sheil, D., Sheleme, D., Simegn, T.Y., Simo-Droissart, M., Sonké, B., Soromessa, T., Sunderland, T., Svoboda, M., Taedoumg, H., Taplin, J., Taylor, D., Thomas, S.C., Timberlake, J.,

Tuagben, D., Umunay, P., Uzabaho, E., Verbeeck, H., Vleminckx, J., Wallin, G., Wheeler, C., Willcock, S., Woods, J.T., Zibera, E., 2021. High aboveground carbon stock of African tropical montane forests. Nature 596, 536–542. https://doi.org/10.1038/s41586-021-03728-4
Dalling, J.W., Heineman, K., González, G., Ostertag, R., 2016. Geographic, environmental and biotic sources of variation in the nutrient relations of tropical montane forests. J. Trop. Ecol. 32, 368–383. https://doi.org/10.1017/S0266467415000619
Dantas de Paula, M., Forrest, M., Langan, L., Bendix, J., Homeier, J., Velescu, A., Wilcke, W., Hickler, T., 2021. Nutrient cycling drives plant community trait assembly and ecosystem functioning in a tropical mountain biodiversity hotspot. New Phytol. 232, 551–566. https://doi.org/10.1111/nph.17600
de Albuquerque, F.S., Benito, B., Beier, P., Assunção-Albuquerque, M.J., Cayuela, L., 2015. Supporting underrepresented forests in Mesoamerica. Nat. Conserv. 13, 152–158. https://doi.org/10.1016/j.ncon.2015.02.001
de Almeida, A., Mikich, S.B., 2018. Combining plant-frugivore networks for describing the structure of neotropical communities. Oikos 127, 184–197. https://doi.org/10.1111/oik.04774
de Araujo-Lira, A.F., Badillo-Montaño, R., Lira-Noriega, A., de Albuquerque, C.M.R., 2020. Potential distribution patterns of scorpions in north-eastern Brazil under scenarios of future climate change: Scorpion distribution on climate change. Austral Ecol. 45, 215–228. https://doi.org/10.1111/aec.12849
de Gasper, A.L. de, Grittz, G.S., Russi, C.H., Schwartz, C.E., Rodrigues, A.V., 2021. Expected impacts of climate change on tree ferns distribution and diversity patterns in subtropical Atlantic Forest. Perspect. Ecol. Conserv. 19, 369–378. https://doi.org/10.1016/j.pecon.2021.03.007
Dettman, S., 2006. The Mesoamerican Biological Corridor in Panama and Costa Rica: Integrating Bioregional Planning and Local Initiatives. J. Sustain. For. 22, 15–34. https://doi.org/10.1300/J091v22n01_02
Dhakal, S., Minx, J.C., Toth, F., Abdel-Aziz, A., Figueroa Meza, M., Hubacek, K., Jonckheere, I.G., Kim, Y.-G., Nemet, G., Pachauri, S., 2022. Emissions Trends and Drivers (Chapter 2).
Díaz, S., Kattge, J., Cornelissen, J.H.C., Wright, I.J., Lavorel, S., Dray, S., Reu, B., Kleyer, M., Wirth, C., Colin Prentice, I., Garnier, E., Bönisch, G., Westoby, M., Poorter, H., Reich, P.B., Moles, A.T., Dickie, J., Gillison, A.N., Zanne, A.E., Chave, J., Joseph Wright, S., Sheremet'ev, S.N., Jactel, H., Baraloto, C., Cerabolini, B., Pierce, S., Shipley, B., Kirkup, D., Casanoves, F., Joswig, J.S., Günther, A., Falczuk, V., Rüger, N., Mahecha, M.D., Gorné, L.D., 2016. The global spectrum of plant form and function. Nature 529, 167–171. https://doi.org/10.1038/nature16489
Dolson, S.J., Loewen, E., Jones, K., Jacobs, S.R., Solis, A., Hallwachs, W., Brunke, A.J., Janzen, D.H., Smith, M.A., 2021. Diversity and phylogenetic community structure across elevation during climate change in a family of hyperdiverse neotropical beetles (Staphylinidae). Ecography 44, 740–752. https://doi.org/10.1111/ecog.05427
Doughty, C.E., Santos-Andrade, P.E., Shenkin, A., Goldsmith, G.R., Bentley, L.P., Blonder, B., Díaz, S., Salinas, N., Enquist, B.J., Martin, R.E., Asner, G.P., Malhi, Y., 2018. Tropical forest leaves may darken in response to climate change. Nat. Ecol. Evol. 2, 1918–1924. https://doi.org/10.1038/s41559-018-0716-y
Dray, S., Siberchicot, M.A., 2017. Package 'ade4.' Univ. Lyon Lyon Fr.
Dunne, J.P., Horowitz, L., Adcroft, A., Ginoux, P., Held, I., John, J., Krasting, J.P., Malyshev, S., Naik, V., Paulot, F., 2020. The GFDL Earth System Model version 4.1 (GFDL-ESM 4.1): Overall coupled model description and simulation characteristics. J. Adv. Model. Earth Syst. 12, e2019MS002015.
Duque, A., Stevenson, P.R., Feeley, K.J., 2015. Thermophilization of adult and juvenile tree communities in the northern tropical Andes. Proc. Natl. Acad. Sci. 112, 10744–10749. https://doi.org/10.1073/pnas.1506570112
Elith, J., H. Graham, C., P. Anderson, R., Dudík, M., Ferrier, S., Guisan, A., J. Hijmans, R., Huettmann, F., R. Leathwick, J., Lehmann, A., Li, J., G. Lohmann, L., A. Loiselle, B., Manion, G., Moritz, C., Nakamura, M., Nakazawa, Y., McC. M. Overton, J., Townsend Peterson, A., J. Phillips, S., Richardson, K., Scachetti-Pereira, R., E. Schapire, R., Soberón, J., Williams, S., S. Wisz, M., E. Zimmermann, N.,

2006. Novel methods improve prediction of species' distributions from occurrence data. Ecography 29, 129–151. https://doi.org/10.1111/j.2006.0906-7590.04596.x
Elith, J., Leathwick, J.R., 2009. Species Distribution Models: Ecological Explanation and Prediction Across Space and Time. Annu. Rev. Ecol. Evol. Syst. 40, 677–697. https://doi.org/10.1146/annurev.ecolsys.110308.120159
Elsen, P.R., Tingley, M.W., 2015. Global mountain topography and the fate of montane species under climate change. Nat. Clim. Change 5, 772–776. https://doi.org/10.1038/nclimate2656
Esperón-Rodríguez, M., Barradas, V., 2014. Potential vulnerability to climate change of four tree species from the central mountain region of Veracruz, Mexico. Clim. Res. 60, 163–174. https://doi.org/10.3354/cr01231
Esperón-Rodríguez, M., Barradas, V.L., 2015a. Ecophysiological vulnerability to climate change: water stress responses in four tree species from the central mountain region of Veracruz, Mexico. Reg. Environ. Change 15, 93–108. https://doi.org/10.1007/s10113-014-0624-x
Esperón-Rodríguez, M., Barradas, V.L., 2015b. Comparing environmental vulnerability in the montane cloud forest of eastern Mexico: A vulnerability index. Ecol. Indic. 52, 300–310. https://doi.org/10.1016/j.ecolind.2014.12.019
Esperón-Rodríguez, M., Beaumont, L.J., Lenoir, J., Baumgartner, J.B., McGowan, J., Correa-Metrio, A., Camac, J.S., 2019. Climate change threatens the most biodiverse regions of Mexico. Biol. Conserv. 240, 108215. https://doi.org/10.1016/j.biocon.2019.108215
Estrada-Contreras, I., Equihua, M., Castillo-Campos, G., Rojas-Soto, O., 2015. Climate change and effects on vegetation in Veracruz, Mexico: an approach using ecological niche modelling. Acta Bot. Mex. 73. https://doi.org/10.21829/abm112.2015.1090
Fadrique, B., Báez, S., Duque, Á., Malizia, A., Blundo, C., Carilla, J., Osinaga-Acosta, O., Malizia, L., Silman, M., Farfán-Ríos, W., Malhi, Y., Young, K.R., Cuesta C., F., Homeier, J., Peralvo, M., Pinto, E., Jadan, O., Aguirre, N., Aguirre, Z., Feeley, K.J., 2018. Widespread but heterogeneous responses of Andean forests to climate change. Nature 564, 207–212. https://doi.org/10.1038/s41586-018-0715-9
Fahey, T.J., Sherman, R.E., Tanner, E.V.J., 2016. Tropical montane cloud forest: environmental drivers of vegetation structure and ecosystem function. J. Trop. Ecol. 32, 355–367. https://doi.org/10.1017/S0266467415000176
Farjon, A., 1996. Biodiversity of Pinus (Pinaceae) in Mexico: speciation and palaeo-endemism. Bot. J. Linn. Soc. 121, 365–384. https://doi.org/10.1111/j.1095-8339.1996.tb00762.x
Feeley, K., Martinez-Villa, J., Perez, T., Silva Duque, A., Triviño Gonzalez, D., Duque, A., 2020. The Thermal Tolerances, Distributions, and Performances of Tropical Montane Tree Species. Front. For. Glob. Change 3, 25. https://doi.org/10.3389/ffgc.2020.00025
Feeley, K.J., Hurtado, J., Saatchi, S., Silman, M.R., Clark, D.B., 2013. Compositional shifts in Costa Rican forests due to climate-driven species migrations. Glob. Change Biol. n/a-n/a. https://doi.org/10.1111/gcb.12300
Feeley, K.J., Silman, M.R., 2010. Biotic attrition from tropical forests correcting for truncated temperature niches: Species loss accounting for truncated niches. Glob. Change Biol. 16, 1830–1836. https://doi.org/10.1111/j.1365-2486.2009.02085.x
Feeley, K.J., Silman, M.R., Bush, M.B., Farfan, W., Cabrera, K.G., Malhi, Y., Meir, P., Revilla, N.S., Quisiyupanqui, M.N.R., Saatchi, S., 2011. Upslope migration of Andean trees: Andean trees migrate upslope. J. Biogeogr. 38, 783–791. https://doi.org/10.1111/j.1365-2699.2010.02444.x
Fick, S.E., Hijmans, R.J., 2017. WorldClim 2: new 1-km spatial resolution climate surfaces for global land areas. Int. J. Climatol. 37, 4302–4315.
Fischer, G., Nachtergaele, F., Prieler, S., Van Velthuizen, H., Verelst, L., Wiberg, D., 2008. Global agro-ecological zones assessment for agriculture (GAEZ 2008). IIASA Laxenburg Austria FAO Rome Italy 10.
Fišer, Ž., Aronne, G., Aavik, T., Akin, M., Alizoti, P., Aravanopoulos, F., Bacchetta, G., Balant, M., Ballian, D., Barazani, O., Bellia, A.F., Bernhardt, N., Bou Dagher Kharrat, M., Bugeja Douglas, A., Burkart, M., Ćalić, D., Carapeto, A., Carlsen, T., Castro, S., Colling, G., Cursach, J., Cvetanoska, S., Cvetkoska, C., Ćušterevska, R., Daco, L., Danova, K., Dervishi, A., Djukanović, G., Dragićević, S., Ensslin, A., Evju, M., Fenu, G., Francisco, A., Gallego, P.P., Galloni, M., Ganea, A., Gemeinholzer, B., Glasnović, P.,

Godefroid, S., Goul Thomsen, M., Halassy, M., Helm, A., Hyvärinen, M., Joshi, J., Kazić, A., Kiehn, M., Klisz, M., Kool, A., Koprowski, M., Kövendi-Jakó, A., Kříž, K., Kropf, M., Kull, T., Lanfranco, S., Lazarević, P., Lazarević, M., Lebel Vine, M., Liepina, L., Loureiro, J., Lukminė, D., Machon, N., Meade, C., Metzing, D., Milanović, Đ., Navarro, L., Orlović, S., Panis, B., Pankova, H., Parpan, T., Pašek, O., Peci, D., Petanidou, T., Plenk, K., Puchałka, R., Radosavljević, I., Rankou, H., Rašomavičius, V., Romanciuc, G., Ruotsalainen, A., Šajna, N., Salaj, T., Sánchez-Romero, C., Sarginci, M., Schäfer, D., Seberg, O., Sharrock, S., Šibík, J., Šibíková, M., Skarpaas, O., Stanković Neđić, M., Stojnic, S., Surina, B., Szitár, K., Teofilovski, A., Thoroddsen, R., Tsvetkov, I., Uogintas, D., Van Meerbeek, K., Van Rooijen, N., Vassiliou, L., Verbylaitė, R., Vergeer, P., Vít, P., Walczak, M., Widmer, A., Wiland-Szymańska, J., Zdunić, G., Zippel, E., 2021. ConservePlants: An integrated approach to conservation of threatened plants for the 21st Century. Res. Ideas Outcomes 7, e62810. https://doi.org/10.3897/rio.7.e62810

Fitzpatrick, M.C., Blois, J.L., Williams, J.W., Nieto-Lugilde, D., Maguire, K.C., Lorenz, D.J., 2018. How will climate novelty influence ecological forecasts? Using the Quaternary to assess future reliability. Glob. Change Biol. 24, 3575–3586. https://doi.org/10.1111/gcb.14138

Flores, O., Coomes, D.A., 2011. Estimating the wood density of species for carbon stock assessments. Methods Ecol. Evol. 2, 214–220. https://doi.org/10.1111/j.2041-210X.2010.00068.x

Forero-Medina, G., Terborgh, J., Socolar, S.J., Pimm, S.L., 2011. Elevational Ranges of Birds on a Tropical Montane Gradient Lag behind Warming Temperatures. PLoS ONE 6, e28535. https://doi.org/10.1371/journal.pone.0028535

Forster, P.M., Smith, C., Walsh, T., Lamb, W.F., Lamboll, R., Hall, B., Hauser, M., Ribes, A., Rosen, D., Gillett, N.P., 2024. Indicators of Global Climate Change 2023: annual update of key indicators of the state of the climate system and human influence. Earth Syst. Sci. Data 16, 2625–2658.

Foster, P., 2001. The potential negative impacts of global climate change on tropical montane cloud forests. Earth-Sci. Rev. 55, 73–106. https://doi.org/10.1016/S0012-8252(01)00056-3

Freed, L.A., Cann, R.L., 2013. Vector movement underlies avian malaria at upper elevation in Hawaii: implications for transmission of human malaria. Parasitol. Res. 112, 3887–3895. https://doi.org/10.1007/s00436-013-3578-x

Freeman, B.G., 2016. Thermal tolerances to cold do not predict upper elevational limits in New Guinean montane birds. Divers. Distrib. 22, 309–317. https://doi.org/10.1111/ddi.12409

Freeman, B.G., Scholer, M.N., Ruiz-Gutierrez, V., Fitzpatrick, J.W., 2018. Climate change causes upslope shifts and mountaintop extirpations in a tropical bird community. Proc. Natl. Acad. Sci. 115, 11982–11987. https://doi.org/10.1073/pnas.1804224115

Freeman, B.G., Song, Y., Feeley, K.J., Zhu, K., 2021. Montane species track rising temperatures better in the tropics than in the temperate zone. Ecol. Lett. 24, 1697–1708. https://doi.org/10.1111/ele.13762

Fricke, E.C., Svenning, J.-C., 2020. Accelerating homogenization of the global plant–frugivore meta-network. Nature 585, 74–78. https://doi.org/10.1038/s41586-020-2640-y

Galbraith, D.R., Christoffersen, B.O., 2015. Modelling Climate Impacts on Forest Ecosystems, in: Routledge Handbook of Forest Ecology. Routledge, pp. 544–555.

García-Hernández, M. de los Á., Toledo-Aceves, T., 2020. Is there potential in elevational assisted migration for the endangered *Magnolia vovidesii*? J. Nat. Conserv. 53, 125782. https://doi.org/10.1016/j.jnc.2019.125782

García-Hernández, M. de los Á., Toledo-Aceves, T., López-Barrera, F., Sosa, V.J., Paz, H., 2019. Effects of environmental filters on early establishment of cloud forest trees along elevation gradients: Implications for assisted migration. For. Ecol. Manag. 432, 427–435. https://doi.org/10.1016/j.foreco.2018.09.042

Gillman, L.N., Wright, S.D., Cusens, J., McBride, P.D., Malhi, Y., Whittaker, R.J., 2015. Latitude, productivity and species richness. Glob. Ecol. Biogeogr. 24, 107–117. https://doi.org/10.1111/geb.12245

Gitay, H., Suárez, A., Watson, R.T., Dokken, D.J., 2002. Climate change and biodiversity.

Godoy-Bürki, A.C., 2016. Efectos del cambio climático sobre especies de plantas vasculares del sur de los Andes Centrales: un estudio en el noroeste de Argentina (NOA). Ecol. Austral 26, 083–094. https://doi.org/10.25260/EA.16.26.1.0.110

Golicher, D.J., Cayuela, L., Alkemade, J.R.M., González-Espinosa, M., Ramírez-Marcial, N., 2008. Applying climatically associated species pools to the modelling of compositional change in tropical montane forests. Glob. Ecol. Biogeogr. 17, 262–273. https://doi.org/10.1111/j.1466-8238.2007.00362.x
Gómez-Pineda, E., Sáenz-Romero, C., Ortega-Rodríguez, J.M., Blanco-García, A., Madrigal-Sánchez, X., Lindig-Cisneros, R., Lopez-Toledo, L., Pedraza-Santos, M.E., Rehfeldt, G.E., 2020. Suitable climatic habitat changes for Mexican conifers along altitudinal gradients under climatic change scenarios. Ecol. Appl. 30. https://doi.org/10.1002/eap.2041
González-del-Pliego, P., Scheffers, B.R., Freckleton, R.P., Basham, E.W., Araújo, M.B., Acosta-Galvis, A.R., Medina Uribe, C.A., Haugaasen, T., Edwards, D.P., 2020. Thermal tolerance and the importance of microhabitats for Andean frogs in the context of land use and climate change. J. Anim. Ecol. 89, 2451–2460. https://doi.org/10.1111/1365-2656.13309
González-Espinosa, M. (Ed.), 2011. The red list of Mexican cloud forest trees. Fauna & Flora International, Cambridge.
Gotsch, S.G., Asbjornsen, H., Holwerda, F., Goldsmith, G.R., Weintraub, A.E., Dawson, T.E., 2014a. Foggy days and dry nights determine crown-level water balance in a seasonal tropical montane cloud forest: Foliar uptake and night-time transpiration determine crown-level water balance. Plant Cell Environ. 37, 261–272. https://doi.org/10.1111/pce.12151
Gotsch, S.G., Crausbay, S.D., Giambelluca, T.W., Weintraub, A.E., Longman, R.J., Asbjornsen, H., Hotchkiss, S.C., Dawson, T.E., 2014b. Water relations and microclimate around the upper limit of a cloud forest in Maui, Hawai'i. Tree Physiol. 34, 766–777. https://doi.org/10.1093/treephys/tpu050
Gotsch, S.G., Dawson, T.E., Draguljić, D., 2018. Variation in the resilience of cloud forest vascular epiphytes to severe drought. New Phytol. 219, 900–913. https://doi.org/10.1111/nph.14866
Gotsch, S.G., Nadkarni, N., Amici, A., 2016. The functional roles of epiphytes and arboreal soils in tropical montane cloud forests. J. Trop. Ecol. 32, 455–468. https://doi.org/10.1017/S026646741600033X
Gotsch, S.G., Nadkarni, N., Darby, A., Glunk, A., Dix, M., Davidson, K., Dawson, T.E., 2015. Life in the treetops: ecophysiological strategies of canopy epiphytes in a tropical montane cloud forest. Ecol. Monogr. 85, 393–412. https://doi.org/10.1890/14-1076.1
Grau, H.R., Veblen, T.T., 2000. Rainfall variability, fire and vegetation dynamics in neotropical montane ecosystems in north-western Argentina. J. Biogeogr. 27, 1107–1121. https://doi.org/10.1046/j.1365-2699.2000.00488.x
Greenhalgh, T., Peacock, R., 2005. Effectiveness and efficiency of search methods in systematic reviews of complex evidence: audit of primary sources. BMJ 331, 1064–1065. https://doi.org/10.1136/bmj.38636.593461.68
Greenwood, S., Chen, J.-C., Chen, C.-T., Jump, A.S., 2014. Strong topographic sheltering effects lead to spatially complex treeline advance and increased forest density in a subtropical mountain region. Glob. Change Biol. 20, 3756–3766. https://doi.org/10.1111/gcb.12710
Griffiths, A.R., Silman, M.R., Farfan Rios, W., Feeley, K.J., Garcia Cabrera, K., Meir, P., Salinas, N., Dexter, K.G., 2021. Evolutionary heritage shapes tree distributions along an Amazon-to-Andes elevation gradient. Biotropica 53, 38–50.
Gual-Díaz, M., Rendón-Correa, A., 2014. Bosques mesófilos de montaña de México: diversidad, ecología y manejo.
Guisan, A., Thuiller, W., Zimmermann, N.E., 2017. Habitat suitability and distribution models: with applications in R. Cambridge University Press.
Gutiérrez-García, G., Ricker, M., 2019. Influencia del clima en el crecimiento radial en cuatro especies de coníferas en la sierra de San Antonio Peña Nevada (Nuevo León, México). Rev. Mex. Biodivers. 90. https://doi.org/10.22201/ib.20078706e.2019.90.2676
Hällfors, M.H., Aikio, S., Fronzek, S., Hellmann, J.J., Ryttäri, T., Heikkinen, R.K., 2016. Assessing the need and potential of assisted migration using species distribution models. Biol. Conserv. 196, 60–68. https://doi.org/10.1016/j.biocon.2016.01.031
Hamilton, L.S., 1995. Mountain Cloud Forest Conservation and Research: A Synopsis. Mt. Res. Dev. 15, 259. https://doi.org/10.2307/3673933

Harrell Jr, F.E., 2019. Package ‘hmisc.’ CRAN2018 2019, 235–236.
Hayes, F.E., Lecourt, P., del Castillo, H., 2018. Rapid southward and upward range expansion of a tropical songbird, the Thrush-like Wren (*Campylorhynchus turdinus*), in South America: a consequence of habitat or climate change? Rev. Bras. Ornitol. 26, 57–64. https://doi.org/10.1007/BF03544416
He, X., He, K.S., Hyvönen, J., 2016. Will bryophytes survive in a warming world? Perspect. Plant Ecol. Evol. Syst. 19, 49–60. https://doi.org/10.1016/j.ppees.2016.02.005
He, X., Lin, Y., Han, G., Guo, P., Tian, X., 2010. The effect of temperature on decomposition of leaf litter from two tropical forests by a microcosm experiment. Eur. J. Soil Biol. 46, 200–207. https://doi.org/10.1016/j.ejsobi.2010.02.001
Heegaard, E., Gjerde, I., Sætersdal, M., 2013. Contribution of rare and common species to richness patterns at local scales. Ecography 36, 937–946.
Heilmeier, H., 2019. Functional traits explaining plant responses to past and future climate changes. Flora 254, 1–11. https://doi.org/10.1016/j.flora.2019.04.004
Helmer, E.H., Gerson, E.A., Baggett, L.S., Bird, B.J., Ruzycki, T.S., Voggesser, S.M., 2019. Neotropical cloud forests and páramo to contract and dry from declines in cloud immersion and frost. PLOS ONE 14, e0213155. https://doi.org/10.1371/journal.pone.0213155
Hermes, C., Keller, K., Nicholas, R.E., Segelbacher, G., Schaefer, H.M., 2018. Projected impacts of climate change on habitat availability for an endangered parakeet. PLOS ONE 13, e0191773. https://doi.org/10.1371/journal.pone.0191773
Hernández-Vargas, G., Perroni, Y., López-Acosta, J.C., Noa-Carrazana, J.C., Sánchez-Velásquez, L.R., 2019. Do the distribution patterns of plant functional traits change during early secondary succession in tropical montane cloud forests? Acta Oecologica 95, 26–35. https://doi.org/10.1016/j.actao.2019.01.003
Heywood, V.H., 2019. Conserving plants within and beyond protected areas – still problematic and future uncertain. Plant Divers. 41, 36–49. https://doi.org/10.1016/j.pld.2018.10.001
Hijmans, R.J., 2020. Introduction to the’raster’package (version 3.0-12).
Hijmans, R.J., Bivand, R., Forner, K., Ooms, J., Pebesma, E., Sumner, M.D., 2022. Package ‘terra.’ Maint. Vienna Austria.
Hillyer, R., Silman, M.R., 2010. Changes in species interactions across a 2.5 km elevation gradient: effects on plant migration in response to climate change: SPECIES INTERACTIONS AND PLANT MIGRATION. Glob. Change Biol. 16, 3205–3214. https://doi.org/10.1111/j.1365-2486.2010.02268.x
Hiltner, U., Bräuning, A., Gebrekirstos, A., Huth, A., Fischer, R., 2016. Impacts of precipitation variability on the dynamics of a dry tropical montane forest. Ecol. Model. 320, 92–101. https://doi.org/10.1016/j.ecolmodel.2015.09.021
Hohenlohe, P.A., Funk, W.C., Rajora, O.P., 2021. Population genomics for wildlife conservation and management. Mol. Ecol. 30, 62–82. https://doi.org/10.1111/mec.15720
Holwerda, F., Bruijnzeel, L.A., Muñoz-Villers, L.E., Equihua, M., Asbjornsen, H., 2010. Rainfall and cloud water interception in mature and secondary lower montane cloud forests of central Veracruz, Mexico. J. Hydrol. 384, 84–96. https://doi.org/10.1016/j.jhydrol.2010.01.012
Hsu, R.C.-C., Oostermeijer, J.G.B., Wolf, J.H.D., 2014. Adaptation of a widespread epiphytic fern to simulated climate change conditions. Plant Ecol. 215, 889–897. https://doi.org/10.1007/s11258-014-0340-0
Hsu, R.C.-C., Tamis, W.L.M., Raes, N., de Snoo, G.R., Wolf, J.H.D., Oostermeijer, G., Lin, S.-H., 2012. Simulating climate change impacts on forests and associated vascular epiphytes in a subtropical island of East Asia: Climate change impacts on forests and epiphytes. Divers. Distrib. 18, 334–347. https://doi.org/10.1111/j.1472-4642.2011.00819.x
Hughes, A.C., Orr, M.C., Ma, K., Costello, M.J., Waller, J., Provoost, P., Yang, Q., Zhu, C., Qiao, H., 2021. Sampling biases shape our view of the natural world. Ecography 44, 1259–1269. https://doi.org/10.1111/ecog.05926
Ivanova, N., Shashkov, M., 2021. The possibilities of GBIF data use in ecological research. Russ. J. Ecol. 52, 1–8.

Iwashita, D.K., Litton, C.M., Giardina, C.P., 2013. Coarse woody debris carbon storage across a mean annual temperature gradient in tropical montane wet forest. For. Ecol. Manag. 291, 336–343. https://doi.org/10.1016/j.foreco.2012.11.043
Jacob, M., Annys, S., Frankl, A., De Ridder, M., Beeckman, H., Guyassa, E., Nyssen, J., 2015a. Tree line dynamics in the tropical African highlands - identifying drivers and dynamics. J. Veg. Sci. 26, 9–20. https://doi.org/10.1111/jvs.12215
Jacob, M., Frankl, A., Beeckman, H., Mesfin, G., Hendrickx, M., Guyassa, E., Nyssen, J., 2015b. North Ethiopian Afro-Alpine Tree Line Dynamics and Forest-Cover Change Since the Early 20th Century. Land Degrad. Dev. 26, 654–664. https://doi.org/10.1002/ldr.2320
Jankowski, J.E., Rabenold, K.N., 2007. Endemism and local rarity in birds of neotropical montane rainforest. Biol. Conserv. 138, 453–463. https://doi.org/10.1016/j.biocon.2007.05.015
Jankowski, J.E., Robinson, S.K., Levey, D.J., 2010. Squeezed at the top: Interspecific aggression may constrain elevational ranges in tropical birds. Ecology 91, 1877–1884. https://doi.org/10.1890/09-2063.1
Jiménez-García, D., Li, X., Lira-Noriega, A., Peterson, A.T., 2021. Upward shifts in elevational limits of forest and grassland for Mexican volcanoes over three decades. Biotropica 53, 798–807. https://doi.org/10.1111/btp.12942
Jiménez-García, D., Peterson, A.T., 2019. Climate change impact on endangered cloud forest tree species in Mexico. Rev. Mex. Biodivers. 90. https://doi.org/10.22201/ib.20078706e.2019.90.2781
Jin, Y., Qian, H., 2022. V.PhyloMaker2: An updated and enlarged R package that can generate very large phylogenies for vascular plants. Plant Divers. 44, 335–339. https://doi.org/10.1016/j.pld.2022.05.005
John, E., Bunting, P., Hardy, A., Roberts, O., Giliba, R., Silayo, D.S., 2020. Modelling the impact of climate change on Tanzanian forests. Divers. Distrib. 26, 1663–1686. https://doi.org/10.1111/ddi.13152
Joshi, A.A., Ratnam, J., Sankaran, M., 2020. Frost maintains forests and grasslands as alternate states in a montane tropical forest–grassland mosaic; but alien tree invasion and warming can disrupt this balance. J. Ecol. 108, 122–132. https://doi.org/10.1111/1365-2745.13239
Joswig, J.S., Kattge, J., Kraemer, G., Mahecha, M.D., Rüger, N., Schaepman, M.E., Schrodt, F., Schuman, M.C., 2023. Imputing missing data in plant traits: A guide to improve gap-filling. Glob. Ecol. Biogeogr. 32, 1395–1408.
Karger, D.N., Conrad, O., Böhner, J., Kawohl, T., Kreft, H., Soria-Auza, R.W., Zimmermann, N.E., Linder, H.P., Kessler, M., 2017. Climatologies at high resolution for the earth’s land surface areas. Sci. Data 4, 1–20.
Karger, D.N., Kessler, M., Lehnert, M., Jetz, W., 2021. Limited protection and ongoing loss of tropical cloud forest biodiversity and ecosystems worldwide. Nat. Ecol. Evol. 5, 854–862. https://doi.org/10.1038/s41559-021-01450-y
Karmalkar, A.V., Bradley, R.S., Diaz, H.F., 2008. Climate change scenario for Costa Rican montane forests. Geophys. Res. Lett. 35, L11702. https://doi.org/10.1029/2008GL033940
Kattge, J., Bönisch, G., Díaz, S., Lavorel, S., Prentice, I.C., Leadley, P., Tautenhahn, S., Werner, G.D., Aakala, T., Abedi, M., 2020. TRY plant trait database–enhanced coverage and open access. Glob. Change Biol. 26, 119–188.
Kearsley, E., Hufkens, K., Verbeeck, H., Bauters, M., Beeckman, H., Boeckx, P., Huygens, D., 2019. Large-sized rare tree species contribute disproportionately to functional diversity in resource acquisition in African tropical forest. Ecol. Evol. 9, 4349–4361.
Keenan, R.J., 2016. Forests and Climate Change: Introduction to a special section. For. Ecol. Manag. 360, 353–356. https://doi.org/10.1016/j.foreco.2015.11.039
Klimes, P., Bishop, T.R., Fayle, T.M., Xing, S., 2021. Reported climate change impacts on cloud forest ants are driven by sampling bias: A critical evaluation of Warne et al. (2020). Biotropica 53, 982–986. https://doi.org/10.1111/btp.12952
Koide, D., Yoshida, K., Daehler, C.C., Mueller-Dombois, D., 2017. An upward elevation shift of native and non-native vascular plants over 40 years on the island of Hawai’i. J. Veg. Sci. 28, 939–950. https://doi.org/10.1111/jvs.12549

Kraft, N.J.B., Godoy, O., Levine, J.M., 2015. Plant functional traits and the multidimensional nature of species coexistence. Proc. Natl. Acad. Sci. 112, 797–802. https://doi.org/10.1073/pnas.1413650112
Kremer, A., Potts, B.M., Delzon, S., 2014. Genetic divergence in forest trees: understanding the consequences of climate change. Funct. Ecol. 28, 22–36. https://doi.org/10.1111/1365-2435.12169
Krishnaswamy, J., John, R., Joseph, S., 2014. Consistent response of vegetation dynamics to recent climate change in tropical mountain regions. Glob. Change Biol. 20, 203–215. https://doi.org/10.1111/gcb.12362
Krömer, T., Acebey, A., Kluge, J., Kessler, M., 2013. Effects of altitude and climate in determining elevational plant species richness patterns: A case study from Los Tuxtlas, Mexico. Flora - Morphol. Distrib. Funct. Ecol. Plants 208, 197–210. https://doi.org/10.1016/j.flora.2013.03.003
Kühn, N., Tovar, C., Carretero, J., Vandvik, V., Enquist, B.J., Willis, K.J., 2021. Globally important plant functional traits for coping with climate change. Front. Biogeogr. 13. https://doi.org/10.21425/F5FBG53774
Laliberté, E., Legendre, P., 2010. A distance-based framework for measuring functional diversity from multiple traits. Ecology 91, 299–305.
Laliberté, É., Pierre Legendre, Shipley, B., 2009. FD: Measuring Functional Diversity (FD) from Multiple Traits, and Other Tools for Functional Ecology. https://doi.org/10.32614/CRAN.package.FD
Lamentowicz, M., Słowińska, S., Słowiński, M., 2016. Combining short-term manipulative experiments with long-term palaeoecological investigations at high resolution to assess the response of Sphagnum peatlands to drought, fire and warming. Mires Peat 1–17. https://doi.org/10.19189/MaP.2016.OMB.244
Laurance, W.F., Carolina Useche, D., Shoo, L.P., Herzog, S.K., Kessler, M., Escobar, F., Brehm, G., Axmacher, J.C., Chen, I.-C., Gámez, L.A., Hietz, P., Fiedler, K., Pyrcz, T., Wolf, J., Merkord, C.L., Cardelus, C., Marshall, A.R., Ah-Peng, C., Aplet, G.H., del Coro Arizmendi, M., Baker, W.J., Barone, J., Brühl, C.A., Bussmann, R.W., Cicuzza, D., Eilu, G., Favila, M.E., Hemp, A., Hemp, C., Homeier, J., Hurtado, J., Jankowski, J., Kattán, G., Kluge, J., Krömer, T., Lees, D.C., Lehnert, M., Longino, J.T., Lovett, J., Martin, P.H., Patterson, B.D., Pearson, R.G., Peh, K.S.-H., Richardson, B., Richardson, M., Samways, M.J., Senbeta, F., Smith, T.B., Utteridge, T.M.A., Watkins, J.E., Wilson, R., Williams, S.E., Thomas, C.D., 2011. Global warming, elevational ranges and the vulnerability of tropical biota. Biol. Conserv. 144, 548–557. https://doi.org/10.1016/j.biocon.2010.10.010
LaVal, R.K., 2004. Impact of global warming and locally changing climate on tropical cloud forest bats. J. Mammal. 85, 237–244. https://doi.org/10.1644/BWG-016
Lavorel, S., Grigulis, K., McIntyre, S., Williams, N.S.G., Garden, D., Dorrough, J., Berman, S., Quétier, F., Thébault, A., Bonis, A., 2008. Assessing functional diversity in the field – methodology matters! Funct. Ecol. 22, 134–147. https://doi.org/10.1111/j.1365-2435.2007.01339.x
Ledo, A., Montes, F., Condes, S., 2009. Species dynamics in a montane cloud forest: Identifying factors involved in changes in tree diversity and functional characteristics. For. Ecol. Manag. 258, S75–S84. https://doi.org/10.1016/j.foreco.2009.07.055
Lee, H., Calvin, K., Dasgupta, D., Krinner, G., Mukherji, A., Thorne, P., Trisos, C., Romero, J., Aldunce, P., Ruane, A.C., 2024. CLIMATE CHANGE 2023 synthesis report summary for policymakers. Clim. CHANGE 2023 Synth. Rep. Summ. Policymakers.
Leggett, J.A., 2021. Greenhouse Gas Emissions Scenarios: Background, Issues, and Policy Relevance.
Leitão, R.P., Zuanon, J., Villéger, S., Williams, S.E., Baraloto, C., Fortunel, C., Mendonça, F.P., Mouillot, D., 2016. Rare species contribute disproportionately to the functional structure of species assemblages. Proc. R. Soc. B Biol. Sci. 283, 20160084. https://doi.org/10.1098/rspb.2016.0084
Li, T., Xiong, Q., Luo, P., Zhang, Y., Gu, X., Lin, B., 2020. Direct and indirect effects of environmental factors, spatial constraints, and functional traits on shaping the plant diversity of montane forests. Ecol. Evol. 10, 557–568. https://doi.org/10.1002/ece3.5931
Li, W., Pan, W., Tan, R., Wang, J., 2016. Loss of preferred habitat and pollen limitation threatens reproduction in a rare mountain Paeonia delavayi. J. Mt. Sci. 13, 2147–2154. https://doi.org/10.1007/s11629-016-4030-3

Liao, Z., Zhang, L., Nobis, M.P., Wu, X., Pan, K., Wang, K., Dakhil, M.A., Du, M., Xiong, Q., Pandey, B., Tian, X., 2020. Climate change jointly with migration ability affect future range shifts of dominant fir species in Southwest China. Divers. Distrib. 26, 352–367. https://doi.org/10.1111/ddi.13018

Lippok, D., Beck, S.G., Renison, D., Gallegos, S.C., Saavedra, F.V., Hensen, I., Schleuning, M., 2013. Forest recovery of areas deforested by fire increases with elevation in the tropical Andes. For. Ecol. Manag. 295, 69–76. https://doi.org/10.1016/j.foreco.2013.01.011

Lippok, D., Beck, S.G., Renison, D., Hensen, I., Apaza, A.E., Schleuning, M., 2014. Topography and edge effects are more important than elevation as drivers of vegetation patterns in a neotropical montane forest. J. Veg. Sci. 25, 724–733. https://doi.org/10.1111/jvs.12132

Liu, L.-L., Yang, B., Lu, H.-Z., Wu, Y., Meng, X.-J., Zhang, Y.-J., Song, L., 2021. Dry-Season Fog Water Utilization by Epiphytes in a Subtropical Montane Cloud Forest of Southwest China. Water 13, 3237. https://doi.org/10.3390/w13223237

Liyanage, G.S., Offord, C.A., Crayn, D.M., Guja, L.K., Worboys, S., Sommerville, K.D., 2022. Understanding seed dormancy and germination aids conservation of rainforest species from tropical montane cloud forest: a case study confirming morphophysiological dormancy in the genus. Aust. J. Bot. 70, 399–408. https://doi.org/10.1071/BT22011

Loarie, S.R., Duffy, P.B., Hamilton, H., Asner, G.P., Field, C.B., Ackerly, D.D., 2009. The velocity of climate change. Nature 462, 1052–1055. https://doi.org/10.1038/nature08649

Looby, C.I., Treseder, K.K., 2018. Shifts in soil fungi and extracellular enzyme activity with simulated climate change in a tropical montane cloud forest. Soil Biol. Biochem. 117, 87–96. https://doi.org/10.1016/j.soilbio.2017.11.014

Loope, L.L., Giambelluca, T.W., 1998. Vulnerability of Island Tropical Montane Cloud Forests to Climate Change, with Special Reference to East Maui, Hawaii, in: Markham, A. (Ed.), Potential Impacts of Climate Change on Tropical Forest Ecosystems. Springer Netherlands, Dordrecht, pp. 363–377. https://doi.org/10.1007/978-94-017-2730-3_18

López-Arce, L., Ureta, C., Granados-Sánchez, D., Rodríguez-Esparza, L., Monterroso-Rivas, A., 2019. Identifying cloud forest conservation areas in Mexico from the potential distribution of 19 representative species. Heliyon 5, e01423. https://doi.org/10.1016/j.heliyon.2019.e01423

Los, S.O., Street-Perrott, F.A., Loader, N.J., Froyd, C.A., 2021. Detection of signals linked to climate change, land-cover change and climate oscillators in Tropical Montane Cloud Forests. Remote Sens. Environ. 260, 112431. https://doi.org/10.1016/j.rse.2021.112431

Los, S.O., Street-Perrott, F.A., Loader, N.J., Froyd, C.A., Cuní-Sanchez, A., Marchant, R.A., 2019. Sensitivity of a tropical montane cloud forest to climate change, present, past and future: Mt. Marsabit, N. Kenya. Quat. Sci. Rev. 218, 34–48. https://doi.org/10.1016/j.quascirev.2019.06.016

Lu, X., Liang, E., Wang, Y., Babst, F., Camarero, J.J., 2020. Mountain treelines climb slowly despite rapid climate warming. Glob. Ecol. Biogeogr. geb.13214. https://doi.org/10.1111/geb.13214

Lutz, D.A., Powell, R.L., Silman, M.R., 2013. Four Decades of Andean Timberline Migration and Implications for Biodiversity Loss with Climate Change. PLoS ONE 8, e74496. https://doi.org/10.1371/journal.pone.0074496

Maitner, B.S., Boyle, B., Casler, N., Condit, R., Donoghue, J., Durán, S.M., Guaderrama, D., Hinchliff, C.E., Jørgensen, P.M., Kraft, N.J., 2018. The BIEN R package: A tool to access the Botanical Information and Ecology Network (BIEN) database. Methods Ecol. Evol. 9, 373–379.

Mamantov, M.A., Gibson-Reinemer, D.K., Linck, E.B., Sheldon, K.S., 2021. Climate-driven range shifts of montane species vary with elevation. Glob. Ecol. Biogeogr. 30, 784–794.

Martin, P.H., Bellingham, P.J., 2016. Towards integrated ecological research in tropical montane cloud forests. J. Trop. Ecol. 32, 345–354. https://doi.org/10.1017/S0266467416000432

Martin, P.H., Fahey, T.J., 2014. Mesoclimatic Patterns Shape the Striking Vegetation Mosaic in the Cordillera Central, Dominican Republic. Arct. Antarct. Alp. Res. 46, 755–765. https://doi.org/10.1657/1938-4246-46.4.755

Martin, P.H., Sherman, R.E., Fahey, T.J., 2007. Tropical montane forest ecotones: climate gradients, natural disturbance, and vegetation zonation in the Cordillera Central, Dominican Republic. J. Biogeogr. 34, 1792–1806. https://doi.org/10.1111/j.1365-2699.2007.01726.x

Martin, R.A., Chick, L.D., Garvin, M.L., Diamond, S.E., 2021. In a nutshell, a reciprocal transplant experiment reveals local adaptation and fitness trade-offs in response to urban evolution in an acorn-dwelling ant. Evolution 75, 876–887. https://doi.org/10.1111/evo.14191
Martínez-Ramos, M., González-Espinosa, M., Ramírez-Marcial, N., Negrete-Yankelevich, S., Mena, R.H., 2022. Mid-and long-term ecological changes after enrichment planting with native tree species in Mexican tropical mountain forests. Restor. Ecol. e13847.
Mata-Guel, E.O., Soh, M.C., Butler, C.W., Morris, R.J., Razgour, O., Peh, K.S., 2023. Impacts of anthropogenic climate change on tropical montane forests: an appraisal of the evidence. Biol. Rev.
Maunsell, S.C., Kitching, R.L., Burwell, C.J., Morris, R.J., 2015. Changes in host-parasitoid food web structure with elevation. J. Anim. Ecol. 84, 353–363. https://doi.org/10.1111/1365-2656.12285
McCarty, J.P., 2001. Ecological Consequences of Recent Climate Change. Conserv. Biol. 15, 320–331. https://doi.org/10.1046/j.1523-1739.2001.015002320.x
Menéndez-Guerrero, P.A., Green, D.M., Davies, T.J., 2020. Climate change and the future restructuring of Neotropical anuran biodiversity. Ecography 43, 222–235. https://doi.org/10.1111/ecog.04510
Metcalfe, D.B., Ahlstrand, J.C.M., 2019. Effects of moisture dynamics on bryophyte carbon fluxes in a tropical cloud forest. New Phytol. 222, 1766–1777. https://doi.org/10.1111/nph.15727
Mi, X., Sun, Z., Song, Y., Liu, X., Yang, J., Wu, J., Ci, X., Li, J., Lin, L., Cao, M., Ma, K., 2021. Rare tree species have narrow environmental but not functional niches. Funct. Ecol. 35, 511–520. https://doi.org/10.1111/1365-2435.13714
Miles, L., Grainger, A., Phillips, O., 2004. The impact of global climate change on tropical forest biodiversity in Amazonia. Glob. Ecol. Biogeogr. 13, 553–565. https://doi.org/10.1111/j.1466-822X.2004.00105.x
Miller, P., Mote, T., Ramseyer, C., Van Beusekom, A., Scholl, M., González, G., 2018. A 42 year inference of cloud base height trends in the Luquillo Mountains of northeastern Puerto Rico. Clim. Res. 76, 87–94. https://doi.org/10.3354/cr01529
Ministerio de Medio Ambiente y Recursos Naturales (MARN), 2018. Inventario nacional de bosques de EL Salvador.
Moles, A.T., Perkins, S.E., Laffan, S.W., Flores-Moreno, H., Awasthy, M., Tindall, M.L., Sack, L., Pitman, A., Kattge, J., Aarssen, L.W., Anand, M., Bahn, M., Blonder, B., Cavender-Bares, J., Cornelissen, J.H.C., Cornwell, W.K., Díaz, S., Dickie, J.B., Freschet, G.T., Griffiths, J.G., Gutierrez, A.G., Hemmings, F.A., Hickler, T., Hitchcock, T.D., Keighery, M., Kleyer, M., Kurokawa, H., Leishman, M.R., Liu, K., Niinemets, Ü., Onipchenko, V., Onoda, Y., Penuelas, J., Pillar, V.D., Reich, P.B., Shiodera, S., Siefert, A., Sosinski, E.E., Soudzilovskaia, N.A., Swaine, E.K., Swenson, N.G., Van Bodegom, P.M., Warman, L., Weiher, E., Wright, I.J., Zhang, H., Zobel, M., Bonser, S.P., 2014. Which is a better predictor of plant traits: temperature or precipitation? J. Veg. Sci. 25, 1167–1180. https://doi.org/10.1111/jvs.12190
Monge-González, M.L., Guerrero-Ramírez, N., Krömer, T., Kreft, H., Craven, D., 2021. Functional diversity and redundancy of tropical forests shift with elevation and forest-use intensity. J. Appl. Ecol. 58, 1827–1837.
Montejo-Kovacevich, G., Martin, S.H., Meier, J.I., Bacquet, C.N., Monllor, M., Jiggins, C.D., Nadeau, N.J., 2020. Microclimate buffering and thermal tolerance across elevations in a tropical butterfly. J. Exp. Biol. jeb.220426. https://doi.org/10.1242/jeb.220426
Morrone, J.J., 2020. Biogeographic regionalization of the mexican transition zone, in: The Mexican Transition Zone. Springer, pp. 103–155.
Morton, E.R., Robinson, S.K., Mulindahabi, F., Masozera, M., Singh, A., Oli, M.K., 2022. Spatiotemporal patterns in an Afrotropical montane forest bird community. Glob. Ecol. Conserv. 40, e02333. https://doi.org/10.1016/j.gecco.2022.e02333
Mouillot, D., Bellwood, D.R., Baraloto, C., Chave, J., Galzin, R., Harmelin-Vivien, M., Kulbicki, M., Lavergne, S., Lavorel, S., Mouquet, N., Paine, C.E.T., Renaud, J., Thuiller, W., 2013. Rare Species Support Vulnerable Functions in High-Diversity Ecosystems. PLoS Biol. 11, e1001569. https://doi.org/10.1371/journal.pbio.1001569

Müller, L.-L.B., Albach, D.C., Zotz, G., 2017. 'Are 3 °C too much?': thermal niche breadth in Bromeliaceae and global warming. J. Ecol. 105, 507–516. https://doi.org/10.1111/1365-2745.12681
Müller, W.A., Jungclaus, J.H., Mauritsen, T., Baehr, J., Bittner, M., Budich, R., Bunzel, F., Esch, M., Ghosh, R., Haak, H., 2018. A higher-resolution version of the max planck institute earth system model (MPI-ESM1. 2-HR). J. Adv. Model. Earth Syst. 10, 1383–1413.
Mulligan, M., 2010. Modeling the tropics-wide extent and distribution of cloud forest and cloud forest loss, with implications for conservation priority. Trop. Montane Cloud For. Sci. Conserv. Manag. 740, 16–38.
Muñoz, M.M., Feeley, K.J., Martin, P.H., Farallo, V.R., 2022. The multidimensional (and contrasting) effects of environmental warming on a group of montane tropical lizards. Funct. Ecol. 36, 419–431. https://doi.org/10.1111/1365-2435.13950
Muñoz, M.M., Langham, G.M., Brandley, M.C., Rosauer, D.F., Williams, S.E., Moritz, C., 2016. Basking behavior predicts the evolution of heat tolerance in Australian rainforest lizards: Physiological evolution in Australian skinks. Evolution 70, 2537–2549. https://doi.org/10.1111/evo.13064
Mupepele, A.-C., Walsh, J.C., Sutherland, W.J., Dormann, C.F., 2016. An evidence assessment tool for ecosystem services and conservation studies. Ecol. Appl. 26, 1295–1301. https://doi.org/10.1890/15-0595
Murugan, M., Shetty, P.K., Anandhi, A., Ravi, R., Alappan, S., Vasudevan, M., Gopalan, S., 2009. Rainfall changes over tropical montane cloud forests of southern Western Ghats, India. Curr. Sci. 97, 1755–1760.
Nadkarni, N.M., Solano, R., 2002. Potential effects of climate change on canopy communities in a tropical cloud forest: an experimental approach. Oecologia 131, 580–586. https://doi.org/10.1007/s00442-002-0899-3
Nagel, T.A., Iacopetti, G., Javornik, J., Rozman, A., De Frenne, P., Selvi, F., Verheyen, K., 2019. Cascading effects of canopy mortality drive long-term changes in understorey diversity in temperate old-growth forests of Europe. J. Veg. Sci. 30, 905–916. https://doi.org/10.1111/jvs.12767
Neate-Clegg, M.H.C., Jones, S.E.I., Burdekin, O., Jocque, M., Şekercioğlu, Ç.H., 2018. Elevational changes in the avian community of a Mesoamerican cloud forest park. Biotropica 50, 805–815. https://doi.org/10.1111/btp.12596
Neate-Clegg, M.H.C., O'Brien, T.G., Mulindahabi, F., Şekercioğlu, Ç.H., 2020. A disconnect between upslope shifts and climate change in an Afrotropical bird community. Conserv. Sci. Pract. 2. https://doi.org/10.1111/csp2.291
Neate-Clegg, M.H.C., Stuart, S.N., Mtui, D., Şekercioğlu, Ç.H., Newmark, W.D., 2021. Afrotropical montane birds experience upslope shifts and range contractions along a fragmented elevational gradient in response to global warming. PLOS ONE 16, e0248712. https://doi.org/10.1371/journal.pone.0248712
Neely, W.J., Greenspan, S.E., Ribeiro, L.P., Carvalho, T., Martins, R.A., Rodriguez, D., Rohr, J.R., Haddad, C.F.B., Toledo, L.F., Becker, C.G., 2020. Synergistic effects of warming and disease linked to high mortality in cool-adapted terrestrial frogs. Biol. Conserv. 245, 108521. https://doi.org/10.1016/j.biocon.2020.108521
Neto dos Santos, G., Silva, A.C. da, Higuchi, P., 2020. Impact fo climate change on the geographical distribution of a cloud forest indicator tree species. Rev. Árvore 44, e4432. https://doi.org/10.1590/1806-908820200000032
Noce, S., Caporaso, L., Santini, M., 2020. A new global dataset of bioclimatic indicators. Sci. Data 7, 398. https://doi.org/10.1038/s41597-020-00726-5
Nogués-Bravo, D., 2009. Predicting the past distribution of species climatic niches. Glob. Ecol. Biogeogr. 18, 521–531.
Nooten, S.S., Andrew, N.R., 2016. Transplant Experiments - a Powerful Method to Study Climate Change Impacts, in: Johnson, S.N., Jones, T.H. (Eds.), Global Climate Change and Terrestrial Invertebrates. John Wiley & Sons, Ltd, Chichester, UK, pp. 46–67. https://doi.org/10.1002/9781119070894.ch4
Nottingham, A.T., Turner, B.L., Whitaker, J., Ostle, N., Bardgett, R.D., McNamara, N.P., Salinas, N., Meir, P., 2016. Temperature sensitivity of soil enzymes along an elevation gradient in the Peruvian Andes. Biogeochemistry 127, 217–230. https://doi.org/10.1007/s10533-015-0176-2

Nottingham, A.T., Whitaker, J., Ostle, N.J., Bardgett, R.D., McNamara, N.P., Fierer, N., Salinas, N., Ccahuana, A.J.Q., Turner, B.L., Meir, P., 2019. Microbial responses to warming enhance soil carbon loss following translocation across a tropical forest elevation gradient. Ecol. Lett. 22, 1889–1899. https://doi.org/10.1111/ele.13379
Nowak, L., Schleuning, M., Bender, I.M.A., Böhning-Gaese, K., Dehling, D.M., Fritz, S.A., Kissling, W.D., Mueller, T., Neuschulz, E.L., Pigot, A.L., Sorensen, M.C., Donoso, I., 2022. Avian seed dispersal may be insufficient for plants to track future temperature change on tropical mountains. Glob. Ecol. Biogeogr. 31, 848–860. https://doi.org/10.1111/geb.13456
Nowrouzi, S., Andersen, A.N., Bishop, T.R., Robson, S.K.A., 2018. Is thermal limitation the primary driver of elevational distributions? Not for montane rainforest ants in the Australian Wet Tropics. Oecologia 188, 333–342. https://doi.org/10.1007/s00442-018-4154-y
Nuñez, M.A., Chiuffo, M.C., Pauchard, A., Zenni, R.D., 2021. Making ecology really global. Trends Ecol. Evol. 36, 766–769. https://doi.org/10.1016/j.tree.2021.06.004
Nwe, T., Zomer, R.J., Corlett, R.T., 2020. Projected Impacts of Climate Change on the Protected Areas of Myanmar. Climate 8, 99. https://doi.org/10.3390/cli8090099
Nyongesa, K.W., Olago, D.O., Oguge, N.O., Wandiga, S.O., 2019. An Overview of Impacts of Climate Change on Mountain Biota and Dependant Livelihoods: A Global Perspective with a Focus on Tropical Mountains.
Ohmura, A., 2012. Enhanced temperature variability in high-altitude climate change. Theor. Appl. Climatol. 110, 499–508. https://doi.org/10.1007/s00704-012-0687-x
Oliveira, R.S., Eller, C.B., Bittencourt, P.R.L., Mulligan, M., 2014. The hydroclimatic and ecophysiological basis of cloud forest distributions under current and projected climates. Ann. Bot. 113, 909–920. https://doi.org/10.1093/aob/mcu060
Ordóñez Díaz, J.A.B., Galicia Naranjo, A., Venegas Mancera, N.J., Hernández Tejeda, T., Ordóñez Díaz, M. de J., Dávalos-Sotelo, R., 2015. Densidad de las maderas mexicanas por tipo de vegetación con base en la clasificación de J. Rzedowski: compilación. Madera Bosques 21, 77–216.
Ornelas, J.F., Licona-Vera, Y., Ortiz-Rodriguez, A.E., 2018. Contrasting responses of generalized/specialized mistletoe-host interactions under climate change. Écoscience 25, 223–234. https://doi.org/10.1080/11956860.2018.1439297
Ortega, M.A., Cayuela, L., Griffith, D.M., Camacho, A., Coronado, I.M., Del Castillo, R.F., Figueroa-Rangel, B.L., Fonseca, W., Garibaldi, C., Kelly, D.L., Letcher, S.G., Meave, J.A., Merino-Martín, L., Meza, V.H., Ochoa-Gaona, S., Olvera-Vargas, M., Ramírez-Marcial, N., Tun-Dzul, F.J., Valdez-Hernández, M., Velázquez, E., White, D.A., Williams-Linera, G., Zahawi, R.A., Muñoz, J., 2024. Climate change increases threat to plant diversity in tropical forests of Central America and southern Mexico. PLOS ONE 19, e0297840. https://doi.org/10.1371/journal.pone.0297840
O'Sullivan, K.S.W., Ruiz-Benito, P., Chen, J., Jump, A.S., 2021. Onward but not always upward: individualistic elevational shifts of tree species in subtropical montane forests. Ecography 44, 112–123. https://doi.org/10.1111/ecog.05334
Ovalle-Rivera, O., Läderach, P., Bunn, C., Obersteiner, M., Schroth, G., 2015. Projected Shifts in Coffea arabica Suitability among Major Global Producing Regions Due to Climate Change. PLOS ONE 10, e0124155. https://doi.org/10.1371/journal.pone.0124155
Pacifici, M., Foden, W.B., Visconti, P., Watson, J.E.M., Butchart, S.H.M., Kovacs, K.M., Scheffers, B.R., Hole, D.G., Martin, T.G., Akçakaya, H.R., Corlett, R.T., Huntley, B., Bickford, D., Carr, J.A., Hoffmann, A.A., Midgley, G.F., Pearce-Kelly, P., Pearson, R.G., Williams, S.E., Willis, S.G., Young, B., Rondinini, C., 2015. Assessing species vulnerability to climate change. Nat. Clim. Change 5, 215–224. https://doi.org/10.1038/nclimate2448
Parmesan, C., Hanley, M.E., 2015. Plants and climate change: complexities and surprises. Ann. Bot. 116, 849–864. https://doi.org/10.1093/aob/mcv169
Pau, S., Cordell, S., Ostertag, R., Inman, F., Sack, L., 2020. Climatic sensitivity of species' vegetative and reproductive phenology in a Hawaiian montane wet forest. Biotropica 52, 825–835. https://doi.org/10.1111/btp.12801

Pauls, S.U., Nowak, C., Bálint, M., Pfenninger, M., 2013. The impact of global climate change on genetic diversity within populations and species. Mol. Ecol. 22, 925–946. https://doi.org/10.1111/mec.12152
Pautasso, M., 2009. Geographical genetics and the conservation of forest trees. Perspect. Plant Ecol. Evol. Syst. 11, 157–189. https://doi.org/10.1016/j.ppees.2009.01.003
Peh, K.S.-H., Soh, M.C.K., Sodhi, N.S., Laurance, W.F., Ong, D.J., Clements, R., 2011. Up in the Clouds: Is Sustainable Use of Tropical Montane Cloud Forests Possible in Malaysia? BioScience 61, 27–38. https://doi.org/10.1525/bio.2011.61.1.8
Peñuelas, J., Sardans, J., Estiarte, M., Ogaya, R., Carnicer, J., Coll, M., Barbeta, A., Rivas-Ubach, A., Llusià, J., Garbulsky, M., Filella, I., Jump, A.S., 2013. Evidence of current impact of climate change on life: a walk from genes to the biosphere. Glob. Change Biol. 19, 2303–2338. https://doi.org/10.1111/gcb.12143
Pepin, N.C., Arnone, E., Gobiet, A., Haslinger, K., Kotlarski, S., Notarnicola, C., Palazzi, E., Seibert, P., Serafin, S., Schöner, W., Terzago, S., Thornton, J.M., Vuille, M., Adler, C., 2022. Climate Changes and Their Elevational Patterns in the Mountains of the World. Rev. Geophys. 60. https://doi.org/10.1029/2020RG000730
Pertoldi, C., Bijlsma, R., Loeschcke, V., 2007. Conservation genetics in a globally changing environment: present problems, paradoxes and future challenges. Biodivers. Conserv. 16, 4147–4163. https://doi.org/10.1007/s10531-007-9212-4
Phillips, S.J., Anderson, R.P., Schapire, R.E., 2006. Maximum entropy modeling of species geographic distributions. Ecol. Model. 190, 231–259. https://doi.org/10.1016/j.ecolmodel.2005.03.026
Piantoni, C., Navas, C.A., Ibargüengoytía, N.R., 2016. Vulnerability to climate warming of four genera of New World iguanians based on their thermal ecology. Anim. Conserv. 19, 391–400. https://doi.org/10.1111/acv.12255
Polato, N.R., Gill, B.A., Shah, A.A., Gray, M.M., Casner, K.L., Barthelet, A., Messer, P.W., Simmons, M.P., Guayasamin, J.M., Encalada, A.C., Kondratieff, B.C., Flecker, A.S., Thomas, S.A., Ghalambor, C.K., Poff, N.L., Funk, W.C., Zamudio, K.R., 2018. Narrow thermal tolerance and low dispersal drive higher speciation in tropical mountains. Proc. Natl. Acad. Sci. 115, 12471–12476. https://doi.org/10.1073/pnas.1809326115
Ponce-Reyes, R., Nicholson, E., Baxter, P.W.J., Fuller, R.A., Possingham, H., 2013. Extinction risk in cloud forest fragments under climate change and habitat loss. Divers. Distrib. 19, 518–529. https://doi.org/10.1111/ddi.12064
Ponce-Reyes, R., Plumptre, A.J., Segan, D., Ayebare, S., Fuller, R.A., Possingham, H.P., Watson, J.E.M., 2017. Forecasting ecosystem responses to climate change across Africa's Albertine Rift. Biol. Conserv. 209, 464–472. https://doi.org/10.1016/j.biocon.2017.03.015
Ponce-Reyes, R., Reynoso-Rosales, V.-H., Watson, J.E.M., VanDerWal, J., Fuller, R.A., Pressey, R.L., Possingham, H.P., 2012. Vulnerability of cloud forest reserves in Mexico to climate change. Nat. Clim. Change 2, 448–452. https://doi.org/10.1038/nclimate1453
Pounds, J.A., Fogden, M.P.L., Campbell, J.H., 1999. Biological response to climate change on a tropical mountain. Nature 398, 611–615. https://doi.org/10.1038/19297
Pouteau, R., Meyer, J.-Y., Blanchard, P., Nitta, J.H., Terorotua, M., Taputuarai, R., 2016. Fern species richness and abundance are indicators of climate change on high-elevation islands: evidence from an elevational gradient on Tahiti (French Polynesia). Clim. Change 138, 143–156. https://doi.org/10.1007/s10584-016-1734-x
Qiao, H., Feng, X., Escobar, L.E., Peterson, A.T., Soberón, J., Zhu, G., Papeş, M., 2019. An evaluation of transferability of ecological niche models. Ecography 42, 521–534. https://doi.org/10.1111/ecog.03986
Quiroga, M.P., Premoli, A.C., Kitzberger, T., 2018. Niche squeeze induced by climate change of the cold-tolerant subtropical montane *Podocarpus parlatorei*. R. Soc. Open Sci. 5, 180513. https://doi.org/10.1098/rsos.180513
Raman, S., Shameer, T.T., Charles, B., Sanil, R., 2020a. Habitat suitability model of endangered *Latidens salimalii* and the probable consequences of global warming. Trop. Ecol. 61, 570–582. https://doi.org/10.1007/s42965-020-00114-5

Raman, S., Shameer, T.T., Sanil, R., Usha, P., Kumar, S., 2020b. Protrusive influence of climate change on the ecological niche of endemic brown mongoose (*Herpestes fuscus fuscus*): a MaxEnt approach from Western Ghats, India. Model. Earth Syst. Environ. 6, 1795–1806. https://doi.org/10.1007/s40808-020-00790-1
Ramírez, L.A., Llambí, L.D., Azocar, C.J., Fernandez, M., Torres, J.E., Bader, M.Y., 2022. Patterns in climate and seedling establishment at a dry tropical treeline. Plant Ecol. 223, 1047–1068. https://doi.org/10.1007/s11258-022-01257-2
Ramírez-Amezcua, Y., Steinmann, V.W., Ruiz-Sanchez, E., Rojas-Soto, O.R., 2016. Mexican alpine plants in the face of global warming: potential extinction within a specialized assemblage of narrow endemics. Biodivers. Conserv. 25, 865–885. https://doi.org/10.1007/s10531-016-1094-x
Ramírez-Barahona, S., Cuervo-Robayo, A.P., Magallón, S., 2023. Assessing digital accessible botanical knowledge and priorities for exploration and discovery of plant diversity across Mesoamerica. New Phytol. 240, 1659–1672.
Ramírez-Bautista, A., Thorne, J.H., Schwartz, M.W., Williams, J.N., 2020. Trait-based climate vulnerability of native rodents in southwestern Mexico. Ecol. Evol. 10, 5864–5876. https://doi.org/10.1002/ece3.6323
Ramirez-Villegas, J., Cuesta, F., Devenish, C., Peralvo, M., Jarvis, A., Arnillas, C.A., 2014. Using species distributions models for designing conservation strategies of Tropical Andean biodiversity under climate change. J. Nat. Conserv. 22, 391–404. https://doi.org/10.1016/j.jnc.2014.03.007
Rehm, E.M., Feeley, K.J., 2016. Seedling transplants reveal species-specific responses of high-elevation tropical treeline trees to climate change. Oecologia 181, 1233–1242. https://doi.org/10.1007/s00442-016-3619-0
Rehm, E.M., Feeley, K.J., 2015. Freezing temperatures as a limit to forest recruitment above tropical Andean treelines. Ecology 96, 1856–1865. https://doi.org/10.1890/14-1992.1
Rehm, E.M., Feeley, K.J., 2013. Forest patches and the upward migration of timberline in the southern Peruvian Andes. For. Ecol. Manag. 305, 204–211. https://doi.org/10.1016/j.foreco.2013.05.041
Rehm, E.M., Yelenik, S., D'Antonio, C., 2021. Freezing temperatures restrict woody plant recruitment and restoration efforts in abandoned montane pastures. Glob. Ecol. Conserv. 26, e01462. https://doi.org/10.1016/j.gecco.2021.e01462
Ressl, R., Morales, L.L., 2008. Sistema de información sobre Brosque Mesófilo de Montaña en México para apoyo en programas de restauración (Fase 1). Com. Nac. Para El Conoc. Uso Biodivers. Informe final SNIB-CONABIO proyecto No. EQ007. México D.F.
Reyes-Chávez, J., Quail, M., Tarvin, S., Kessler, M., Batke, S.P., 2021. Nowhere to escape – Diversity and community composition of ferns and lycophytes on the highest mountain in Honduras. J. Trop. Ecol. 37, 72–81. https://doi.org/10.1017/S0266467421000122
Richter, M., 2008. Tropical mountain forests-distribution and general features. Trop. Montane For. Patterns Process. Biodivers. Hotspot 7–24.
Rico, Y., Zurita-Solís, M.A., León-Tapia, M.Á., Miguel-Peñaloza, A., 2023. High genetic diversity but spatially restricted genetic connectivity in a tropical montane cloud forest tree (*Magnolia schiedeana*). Tree Genet. Genomes 19, 1. https://doi.org/10.1007/s11295-022-01578-3
Ripley, B., Venables, B., Bates, D.M., Hornik, K., Gebhardt, A., Firth, D., Ripley, M.B., 2013. Package 'mass.' Cran R 538, 113–120.
Rödder, D., 2009. 'Sleepless in Hawaii' – does anthropogenic climate change enhance ecological and socioeconomic impacts of the alien invasive Eleutherodactylus coqui Thomas 1966 (Anura: Eleutherodactylidae)? 10.
Rodríguez-Ramírez, E.C., Ferrero, M.E., Acevedo-Vega, I., Crispin-DelaCruz, D.B., Ticse-Otarola, G., Requena-Rojas, E.J., 2022. Plastic adjustments in xylem vessel traits to drought events in three Cedrela species from Peruvian Tropical Andean forests. Sci. Rep. 12, 21112. https://doi.org/10.1038/s41598-022-25645-w
Rodríguez-Ramírez, E.C., Terrazas, T., Luna-Vega, I., 2019. The influence of climate on the masting behavior of Mexican beech: growth rings and xylem anatomy. Trees 33, 23–35. https://doi.org/10.1007/s00468-018-1755-3

Rodríguez-Ramírez, E.C., Vázquez-García, J.A., García-González, I., Alcántara-Ayala, O., Luna-Vega, I., 2020. Drought effects on the plasticity in vessel traits of two endemic *Magnolia* species in the tropical montane cloud forests of eastern Mexico. J. Plant Ecol. 13, 331–340. https://doi.org/10.1093/jpe/rtaa019
Rodríguez-Trejo, D.A., Fulé, P.Z., 2003. Fire ecology of Mexican pines and a fire management proposal. Int. J. Wildland Fire 12, 23. https://doi.org/10.1071/WF02040
Rojas Briceño, N.B., Cotrina Sánchez, D.A., Barboza Castillo, E., Barrena Gurbillón, M.Á., Sarmiento, F.O., Sotomayor, D.A., Oliva, M., Salas López, R., 2020. Current and Future Distribution of Five Timber Forest Species in Amazonas, Northeast Peru: Contributions towards a Restoration Strategy. Diversity 12, 305. https://doi.org/10.3390/d12080305
Rojas-Soto, O.R., Sosa, V., Ornelas, J.F., 2012. Forecasting cloud forest in eastern and southern Mexico: conservation insights under future climate change scenarios. Biodivers. Conserv. 21, 2671–2690. https://doi.org/10.1007/s10531-012-0327-x
Rosas Rangel, D.M., Mendoza, M.E., Gómez-Tagle, A., Tobón Marín, C., 2019. Avances y desafíos en el conocimiento de los bosques mesófilos de montaña de México. Madera Bosques 25. https://doi.org/10.21829/myb.2019.2511759
Rubenstein, M.A., Weiskopf, S.R., Carter, S.L., Eaton, M.J., Johnson, C., Lynch, A.J., Miller, B.W., Morelli, T.L., Rodriguez, M.A., Terando, A., Thompson, L.M., 2020. Do empirical observations support commonly-held climate change range shift hypotheses? A systematic review protocol. Environ. Evid. 9, 10. https://doi.org/10.1186/s13750-020-00194-9
Rueda-Solano, L.A., Navas, C.A., Carvajalino-Fernández, J.M., Amézquita, A., 2016. Thermal ecology of montane *Atelopus* (Anura: Bufonidae): A study of intrageneric diversity. J. Therm. Biol. 58, 91–98. https://doi.org/10.1016/j.jtherbio.2016.04.007
Sáenz-Romero, C., Lindig-Cisneros, R.A., Joyce, D.G., Beaulieu, J., St Clair, J.B., Jaquish, B.C., 2016. Assisted migration of forest populations for adapting trees to climate change. Rev. Chapingo Ser. Cienc. For. Ambiente 22, 303–323.
Sáenz-Romero, C., Mendoza-Maya, E., Gómez-Pineda, E., Blanco-García, A., Endara-Agramont, A.R., Lindig-Cisneros, R., López-Upton, J., Trejo-Ramírez, O., Wehenkel, C., Cibrián-Tovar, D., Flores-López, C., Plascencia-González, A., Vargas-Hernández, J.J., 2020a. Recent evidence of Mexican temperate forest decline and the need for ex situ conservation, assisted migration, and translocation of species ensembles as adaptive management to face projected climatic change impacts in a megadiverse country. Can. J. For. Res. 50, 843–854. https://doi.org/10.1139/cjfr-2019-0329
Sáenz-Romero, C., O'Neill, G., Aitken, S.N., Lindig-Cisneros, R., 2020b. Assisted Migration Field Tests in Canada and Mexico: Lessons, Limitations, and Challenges. Forests 12, 9. https://doi.org/10.3390/f12010009
Salinas, N., Cosio, E.G., Silman, M., Meir, P., Nottingham, A.T., Roman-Cuesta, R.M., Malhi, Y., 2021. Editorial: Tropical Montane Forests in a Changing Environment. Front. Plant Sci. 12, 712748. https://doi.org/10.3389/fpls.2021.712748
Sands, P., 1992. The United Nations framework convention on climate change. Rev Eur Comp Intl Envtl L 1, 270.
Sanquetta, C., Dalla Corte, A., Maas, G.B., 2011. The role of forests in climate change. Quebracho-Rev. Cienc. For. 19, 84–96.
Sax, D.F., Early, R., Bellemare, J., 2013. Niche syndromes, species extinction risks, and management under climate change. Trends Ecol. Evol. 28, 517–523. https://doi.org/10.1016/j.tree.2013.05.010
Scatena, F., Bruijnzeel, L., Bubb, P., Das, S., 2010. Setting the stage. Trop. Montane Cloud For. Sci. Conserv. Manag. 3–13.
Scheffers, B.R., Brunner, R.M., Ramirez, S.D., Shoo, L.P., Diesmos, A., Williams, S.E., 2013a. Thermal Buffering of Microhabitats is a Critical Factor Mediating Warming Vulnerability of Frogs in the Philippine Biodiversity Hotspot. Biotropica 45, 628–635. https://doi.org/10.1111/btp.12042

Scheffers, B.R., Edwards, D.P., Diesmos, A., Williams, S.E., Evans, T.A., 2014. Microhabitats reduce animal's exposure to climate extremes. Glob. Change Biol. 20, 495–503. https://doi.org/10.1111/gcb.12439
Scheffers, B.R., Phillips, B.L., Laurance, W.F., Sodhi, N.S., Diesmos, A., Williams, S.E., 2013b. Increasing arboreality with altitude: a novel biogeographic dimension. Proc. R. Soc. B Biol. Sci. 280, 20131581. https://doi.org/10.1098/rspb.2013.1581
Schmitt, C.B., Senbeta, F., Woldemariam, T., Rudner, M., Denich, M., 2013. Importance of regional climates for plant species distribution patterns in moist Afromontane forest. J. Veg. Sci. 24, 553–568. https://doi.org/10.1111/j.1654-1103.2012.01477.x
Scholl, M.A., Bassiouni, M., Torres-Sánchez, A.J., 2021. Drought stress and hurricane defoliation influence mountain clouds and moisture recycling in a tropical forest. Proc. Natl. Acad. Sci. 118, e2021646118. https://doi.org/10.1073/pnas.2021646118
Scowcroft, Paul.G., Turner, D.R., Vitousek, P.M., 2000. Decomposition of *Metrosideros polymorpha* leaf litter along elevational gradients in Hawaii. Glob. Change Biol. 6, 73–85. https://doi.org/10.1046/j.1365-2486.2000.00282.x
Sellar, A.A., Jones, C.G., Mulcahy, J.P., Tang, Y., Yool, A., Wiltshire, A., O'Connor, F.M., Stringer, M., Hill, R., Palmieri, J., Woodward, S., De Mora, L., Kuhlbrodt, T., Rumbold, S.T., Kelley, D.I., Ellis, R., Johnson, C.E., Walton, J., Abraham, N.L., Andrews, M.B., Andrews, T., Archibald, A.T., Berthou, S., Burke, E., Blockley, E., Carslaw, K., Dalvi, M., Edwards, J., Folberth, G.A., Gedney, N., Griffiths, P.T., Harper, A.B., Hendry, M.A., Hewitt, A.J., Johnson, B., Jones, A., Jones, C.D., Keeble, J., Liddicoat, S., Morgenstern, O., Parker, R.J., Predoi, V., Robertson, E., Siahaan, A., Smith, R.S., Swaminathan, R., Woodhouse, M.T., Zeng, G., Zerroukat, M., 2019. UKESM1: Description and Evaluation of the U.K. Earth System Model. J. Adv. Model. Earth Syst. 11, 4513–4558. https://doi.org/10.1029/2019MS001739
Setyawan, A.D., Supriatna, J., Nisyawati, Nursamsi, I., Sutarno, S., Sugiyarto, S., Sunarto, S., Pradan, P., Budiharta, S., Pitoyo, A., Suhardono, S., Setyono, P., Indrawan, M., 2020. Anticipated climate changes reveal shifting in habitat suitability of high-altitude selaginellas in Java, Indonesia. Biodiversitas J. Biol. Divers. 21. https://doi.org/10.13057/biodiv/d211157
Shah, A.A., Gill, B.A., Encalada, A.C., Flecker, A.S., Funk, W.C., Guayasamin, J.M., Kondratieff, B.C., Poff, N.L., Thomas, S.A., Zamudio, K.R., Ghalambor, C.K., 2017. Climate variability predicts thermal limits of aquatic insects across elevation and latitude. Funct. Ecol. 31, 2118–2127. https://doi.org/10.1111/1365-2435.12906
Shamseer, L., Moher, D., Clarke, M., Ghersi, D., Liberati, A., Petticrew, M., Shekelle, P., Stewart, L.A., the PRISMA-P Group, 2015. Preferred reporting items for systematic review and meta-analysis protocols (PRISMA-P) 2015: elaboration and explanation. BMJ 349, g7647–g7647. https://doi.org/10.1136/bmj.g7647
Sheldon, K.S., Yang, S., Tewksbury, J.J., 2011. Climate change and community disassembly: impacts of warming on tropical and temperate montane community structure: Climate change and community disassembly. Ecol. Lett. 14, 1191–1200. https://doi.org/10.1111/j.1461-0248.2011.01689.x
Shi, J.P., Zhu, H., 2009. Tree species composition and diversity of tropical mountain cloud forest in the Yunnan, southwestern China. Ecol. Res. 24, 83–92. https://doi.org/10.1007/s11284-008-0484-2
Shiao, M.-T., Chuang, M.-C., Yuan, H.-W., Wang, Y., 2020. Seasonal rainfall in subtropical montane cloud forests drives demographic fluctuations in a Green-backed Tit population. The Condor 122, duaa043. https://doi.org/10.1093/condor/duaa043
Shugart, H.H., French, N.H.F., Kasischke, E.S., Slawski, J.J., Dull, C.W., Shuchman, R.A., Mwangi, J., 2001. Detection of vegetation change using reconnaissance imagery. Glob. Change Biol. 7, 247–252. https://doi.org/10.1046/j.1365-2486.2001.00379.x
Sierra-Morales, P., Rojas-Soto, O., Ríos-Muñoz, C.A., Ochoa-Ochoa, L.M., Flores-Rodríguez, P., Almazán-Núñez, R.C., 2021. Climate change projections suggest severe decreases in the geographic ranges of bird species restricted to Mexican humid mountain forests. Glob. Ecol. Conserv. 30, e01794. https://doi.org/10.1016/j.gecco.2021.e01794

Silveira, F.A.O., Barbosa, M., Beiroz, W., Callisto, M., Macedo, D.R., Morellato, L.P.C., Neves, F.S., Nunes, Y.R.F., Solar, R.R., Fernandes, G.W., 2019. Tropical mountains as natural laboratories to study global changes: A long-term ecological research project in a megadiverse biodiversity hotspot. Perspect. Plant Ecol. Evol. Syst. 38, 64–73. https://doi.org/10.1016/j.ppees.2019.04.001
Sistema Nacional de Áreas de Conservación (Sinac), 2014. Protocolo de campo para la identificación de especies arbóreas: Información taxonómica y dendrológica de las especies arbóreas de Costa Rica. Volumen 3.
Sodhi, N.S., Koh, L.P., Peh, K.S. -H., Tan, H.T.W., Chazdon, R.L., Corlett, R.T., Lee, T.M., Colwell, R.K., Brook, B.W., Sekercioglu, C.H., Bradshaw, C.J.A., 2008. Correlates of extinction proneness in tropical angiosperms. Divers. Distrib. 14, 1–10. https://doi.org/10.1111/j.1472-4642.2007.00398.x
Soh, M.C.K., Mitchell, N.J., Ridley, A.R., Butler, C.W., Puan, C.L., Peh, K.S.-H., 2019. Impacts of Habitat Degradation on Tropical Montane Biodiversity and Ecosystem Services: A Systematic Map for Identifying Future Research Priorities. Front. For. Glob. Change 2, 83. https://doi.org/10.3389/ffgc.2019.00083
Soley-Guardia, M., Alvarado-Serrano, D.F., Anderson, R.P., 2024. Top ten hazards to avoid when modeling species distributions: a didactic guide of assumptions, problems, and recommendations. Ecography 2024, e06852.
Song, L., Liu, W.-Y., Nadkarni, N.M., 2012. Response of non-vascular epiphytes to simulated climate change in a montane moist evergreen broad-leaved forest in southwest China. Biol. Conserv. 152, 127–135. https://doi.org/10.1016/j.biocon.2012.04.002
Song, X., Li, J., Zhang, W., Tang, Y., Sun, Z., Cao, M., 2016a. Variant responses of tree seedling to seasonal drought stress along an elevational transect in tropical montane forests. Sci. Rep. 6, 36438. https://doi.org/10.1038/srep36438
Song, X., Nakamura, A., Sun, Z., Tang, Y., Cao, M., 2016b. Elevational Distribution of Adult Trees and Seedlings in a Tropical Montane Transect, Southwest China. Mt. Res. Dev. 36, 342. https://doi.org/10.1659/MRD-JOURNAL-D-15-00109.1
Soto-Correa, J.C., Sáenz-Romero, C., Lindig-Cisneros, R., de la Barrera, E., 2013. The neotropical shrub *Lupinus elegans* , from temperate forests, may not adapt to climate change. Plant Biol. 15, 607–610. https://doi.org/10.1111/j.1438-8677.2012.00716.x
Sperling, F.N., Washington, R., Whittaker, R.J., 2004. Future Climate Change of the Subtropical North Atlantic: Implications for the Cloud Forests of Tenerife. Clim. Change 65, 103–123. https://doi.org/10.1023/B:CLIM.0000037488.33377.bf
Sterck, F., Markesteijn, L., Schieving, F., Poorter, L., 2011. Functional traits determine trade-offs and niches in a tropical forest community. Proc. Natl. Acad. Sci. 108, 20627–20632. https://doi.org/10.1073/pnas.1106950108
Still, C.J., Foster, P.N., Schneider, S.H., 1999. Simulating the effects of climate change on tropical montane cloud forests. Nature 398, 608–610. https://doi.org/10.1038/19293
Strangas, M.L., Navas, C.A., Rodrigues, M.T., Carnaval, A.C., 2019. Thermophysiology, microclimates, and species distributions of lizards in the mountains of the Brazilian Atlantic Forest. Ecography 42, 354–364. https://doi.org/10.1111/ecog.03330
Su, Y., Mura, M., Zheng, X., Chen, Q., Wei, X., Qiu, Y., Li, M., Ren, Y., 2024. More Accurately Estimating Aboveground Biomass in Tropical Forests With Complex Forest Structures and Regions of High-Aboveground Biomass. J. Geophys. Res. Biogeosciences 129, e2023JG007864. https://doi.org/10.1029/2023JG007864
Sukumar, R., Suresh, H.S., Ramesh, R., 1995. Climate Change and Its Impact on Tropical Montane Ecosystems in Southern India. J. Biogeogr. 22, 533. https://doi.org/10.2307/2845951
Swenson, N.G., Enquist, B.J., 2009. Opposing assembly mechanisms in a Neotropical dry forest: implications for phylogenetic and functional community ecology. Ecology 90, 2161–2170. https://doi.org/10.1890/08-1025.1
Tanner, E.V.J., Bellingham, P.J., Healey, J.R., Feeley, K.J., 2022. Hurricane disturbance accelerated the thermophilization of a Jamaican montane forest. Ecography. https://doi.org/10.1111/ecog.06100

Tao, S., Cheng, K., Li, X., Han, X., Wang, J., Zheng, R., Sun, B., 2021. The Thermal Biology of *Takydromus kuehnei* Indicates Tropical Lizards From High Elevation Have Not Been Severely Threatened by Climate Change. Front. Ecol. Evol. 9, 767102. https://doi.org/10.3389/fevo.2021.767102
Tejedor-Garavito, N., Newton, A.C., Golicher, D., Oldfield, S., 2015. The Relative Impact of Climate Change on the Extinction Risk of Tree Species in the Montane Tropical Andes. PLOS ONE 10, e0131388. https://doi.org/10.1371/journal.pone.0131388
Thierry-Aguilera, R., Perdomo, E.V., Cuate-Moreno, U., Ramírez-Caballero, C.A., Campos-Báez, I., Arriaga-Conde, P.A., 2021. Sustainability and competitiveness of the Inter-Oceanic Corridor of the Isthmus of Tehuantepec. Presented at the IISE ANNUAL CONFERENCE & EXPO 2021.
Thuiller, W., Georges, D., Engler, R., Breiner, F., Georges, M.D., Thuiller, C.W., 2016. Package 'biomod2.' Species Distrib Model Ensemble Forecast Fram.
Thuiller, W., Gueguen, M., Engler, R., Breiner, F., Lafourcade, B., Patin, R., 2023. Package "biomod2": Ensemble Platform for Species Distribution Modeling.
Tito, R., Vasconcelos, H.L., Feeley, K.J., 2021. Multi-population seedling and soil transplants show possible responses of a common tropical montane tree species (*Weinmannia bangii*) to climate change. J. Ecol. 109, 62–73. https://doi.org/10.1111/1365-2745.13443
Tito, R., Vasconcelos, H.L., Feeley, K.J., 2020. Mountain Ecosystems as Natural Laboratories for Climate Change Experiments. Front. For. Glob. Change 3, 38. https://doi.org/10.3389/ffgc.2020.00038
Tobar-Suárez, C., Urbina-Cardona, N., Villalobos, F., Pineda, E., 2022. Amphibian species richness and endemism in tropical montane cloud forests across the Neotropics. Biodivers. Conserv. 31, 295–313. https://doi.org/10.1007/s10531-021-02335-z
Tognetti, S., Aylward, B., Bruijnzeel, L., 2010. Assessment needs to support the development of arrangements for payments for ecosystem services from tropical montane cloud forests. Trop. Montane Cloud For. Sci. Conserv. Manag. Ed. Bruijnzeel Scatena FN Hamilt. Camb. Univ Press Camb. UK 671–685.
Toledo-Aceves, T., del-Val, E., 2021a. Do plant-herbivore interactions persist in assisted migration plantings? Restor. Ecol. 29. https://doi.org/10.1111/rec.13318
Toledo-Aceves, T., del-Val, E., 2021b. Do plant-herbivore interactions persist in assisted migration plantings? Restor. Ecol. 29, e13318.
Toledo-Aceves, T., García-Hernández, M. de los Á., Paz, H., 2019. Leaf functional traits predict cloud forest tree seedling survival along an elevation gradient. Ann. For. Sci. 76, 111. https://doi.org/10.1007/s13595-019-0900-5
Torres, I., González, J.E., Comarazamy, D.E., 2008. Impacts of a changing climate and low land use on a Tropical Montane Cloud Forest, in: Environmental Problems in Coastal Regions VII. Presented at the COASTAL ENVIRONMENT 2008, WIT Press, The New Forest, UK, pp. 59–70. https://doi.org/10.2495/CENV080061
Tovar, C., Carril, A.F., Gutiérrez, A.G., Ahrends, A., Fita, L., Zaninelli, P., Flombaum, P., Abarzúa, A.M., Alarcón, D., Aschero, V., Báez, S., Barros, A., Carilla, J., Ferrero, M.E., Flantua, S.G.A., Gonzáles, P., Menéndez, C.G., Pérez-Escobar, O.A., Pauchard, A., Ruscica, R.C., Särkinen, T., Sörensson, A.A., Srur, A., Villalba, R., Hollingsworth, P.M., 2022. Understanding climate change impacts on biome and plant distributions in the Andes: Challenges and opportunities. J. Biogeogr. 49, 1420–1442. https://doi.org/10.1111/jbi.14389
Trew, B.T., Edwards, D.P., Lees, A.C., Klinges, D.H., Early, R., Svátek, M., Plichta, R., Matula, R., Okello, J., Niessner, A., Barthel, M., Six, J., Maeda, E.E., Barlow, J., Do Nascimento, R.O., Berenguer, E., Ferreira, J., Sallo-Bravo, J., Maclean, I.M.D., 2024. Novel temperatures are already widespread beneath the world's tropical forest canopies. Nat. Clim. Change 14, 753–759. https://doi.org/10.1038/s41558-024-02031-0
Tubiello, F.N., Conchedda, G., Wanner, N., Federici, S., Rossi, S., Grassi, G., 2021. Carbon emissions and removals from forests: new estimates, 1990–2020. Earth Syst. Sci. Data 13, 1681–1691.
Valencia-A., S., 2004. Diversidad del género Quercus (Fagaceae) en México. Bot. Sci. 33–53. https://doi.org/10.17129/botsci.1692

Van Beusekom, A.E., González, G., Rivera, M.M., 2015. Short-Term Precipitation and Temperature Trends along an Elevation Gradient in Northeastern Puerto Rico. Earth Interact. 19, 1–33. https://doi.org/10.1175/EI-D-14-0023.1

Van Beusekom, A.E., González, G., Scholl, M.A., 2017. Analyzing cloud base at local and regional scales to understand tropical montane cloud forest vulnerability to climate change. Atmospheric Chem. Phys. 17, 7245–7259. https://doi.org/10.5194/acp-17-7245-2017

van der Sande, M.T., Bush, M.B., Åkesson, C.M., Berrio, J.C., Correia Metrio, A., Flantua, S.G.A., Hooghiemstra, H., Maezumi, S.Y., McMichael, C.N.H., Montoya, E., Mosblech, N.A.S., De Novaes Nascimento, M., Peña-Claros, M., Poorter, L., Raczka, M.F., Gosling, W.D., 2023. Warming, drought, and disturbances lead to shifts in functional composition: A millennial-scale analysis for Amazonian and Andean sites. Glob. Change Biol. 29, 4775–4792. https://doi.org/10.1111/gcb.16818

Vásquez-Morales, S.G., Téllez-Valdés, O., Pineda-López, M. del R., Sánchez-Velásquez, L.R., Flores-Estevez, N., Viveros-Viveros, H., 2014. Effect of climate change on the distribution of *Magnolia schiedeana*: a threatened species. Bot. Sci. 92, 575. https://doi.org/10.17129/botsci.116

Véliz, M., Gallardo, N., Luarca, R., Vásquez, M., 2000. La Vegetación Montana de Guatemala (Protecto FODECYT 35-99).

Veloz, S.D., Williams, J.W., Blois, J.L., He, F., Otto-Bliesner, B., Liu, Z., 2012. No-analog climates and shifting realized niches during the late quaternary: implications for 21st-century predictions by species distribution models. Glob. Change Biol. 18, 1698–1713.

Venter, M., Dwyer, J., Dieleman, W., Ramachandra, A., Gillieson, D., Laurance, S., Cernusak, L.A., Beehler, B., Jensen, R., Bird, M.I., 2017. Optimal climate for large trees at high elevations drives patterns of biomass in remote forests of Papua New Guinea. Glob. Change Biol. 23, 4873–4883. https://doi.org/10.1111/gcb.13741

Vilanova, E., Ramírez-Angulo, H., Torres-Lezama, A., Aymard, G., Gámez, L., Durán, C., Hernández, L., Herrera, R., Van Der Heijden, G., Phillips, O.L., Ettl, G.J., 2018. Environmental drivers of forest structure and stem turnover across Venezuelan tropical forests. PLOS ONE 13, e0198489. https://doi.org/10.1371/journal.pone.0198489

Villaseñor, J.L., 2010. Bosque húmedo de montaña en México 42.

Violle, C., Navas, M., Vile, D., Kazakou, E., Fortunel, C., Hummel, I., Garnier, E., 2007. Let the concept of trait be functional! Oikos 116, 882–892. https://doi.org/10.1111/j.0030-1299.2007.15559.x

Violle, C., Reich, P.B., Pacala, S.W., Enquist, B.J., Kattge, J., 2014. The emergence and promise of functional biogeography. Proc. Natl. Acad. Sci. 111, 13690–13696. https://doi.org/10.1073/pnas.1415442111

Viviroli, D., Archer, D.R., Buytaert, W., Fowler, H.J., Greenwood, G.B., Hamlet, A.F., Huang, Y., Koboltschnig, G., Litaor, M.I., López-Moreno, J.I., Lorentz, S., Schädler, B., Schreier, H., Schwaiger, K., Vuille, M., Woods, R., 2011. Climate change and mountain water resources: overview and recommendations for research, management and policy. Hydrol. Earth Syst. Sci. 15, 471–504. https://doi.org/10.5194/hess-15-471-2011

Volis, S., Tojibaev, K., 2021. Defining critical habitat for plant species with poor occurrence knowledge and identification of critical habitat networks. Biodivers. Conserv. 30, 3603–3611. https://doi.org/10.1007/s10531-021-02265-w

von May, R., Catenazzi, A., Corl, A., Santa-Cruz, R., Carnaval, A.C., Moritz, C., 2017. Divergence of thermal physiological traits in terrestrial breeding frogs along a tropical elevational gradient. Ecol. Evol. 7, 3257–3267. https://doi.org/10.1002/ece3.2929

Vuille, M., Bradley, R.S., Werner, M., Keimig, F., 2003. 20th Century Climate Change in the Tropical Andes: Observations and Model Results, in: Diaz, H.F. (Ed.), Climate Variability and Change in High Elevation Regions: Past, Present & Future, Advances in Global Change Research. Springer Netherlands, Dordrecht, pp. 75–99. https://doi.org/10.1007/978-94-015-1252-7_5

Wagner, S., Zotz, G., Bader, M.Y., 2014. The temperature acclimation potential of tropical bryophytes. Plant Biol. 16, 117–124. https://doi.org/10.1111/plb.12037

Wang, L., He, Y., Umer, M., Guo, Y., Tan, Q., Kang, L., Fang, Z., Shen, K., Xia, T., Wu, P., Liu, Y., Zang, L., Liu, Q., Zhao, Yan, Chen, H., Zhao, Ying, 2023. Strategic differentiation of subcommunities composed

of evergreen and deciduous woody species associated with leaf functional traits in the subtropical mixed forest. Ecol. Indic. 150, 110281. https://doi.org/10.1016/j.ecolind.2023.110281
Wang, L.-Q., Ali, A., 2021. Climate regulates the functional traits – aboveground biomass relationships at a community-level in forests: A global meta-analysis. Sci. Total Environ. 761, 143238. https://doi.org/10.1016/j.scitotenv.2020.143238
Warne, C.P.K., Hallwachs, W., Janzen, D.H., Smith, M.A., 2020. Functional and genetic diversity changes through time in a cloud forest ant assemblage. Biotropica 52, 1084–1091. https://doi.org/10.1111/btp.12882
Warren, D., Matzke, N., Cardillo, M., Baumgartner, J., Beaumont, L., Huron, N., Simões, M., Dinnage, R., 2019. ENMTools (software package). URL Httpsgithub ComdanlwarrenENMTools Doi 10.
Wauchope, H.S., Amano, T., Geldmann, J., Johnston, A., Simmons, B.I., Sutherland, W.J., Jones, J.P.G., 2021. Evaluating Impact Using Time-Series Data. Trends Ecol. Evol. 36, 196–205. https://doi.org/10.1016/j.tree.2020.11.001
Wei, T., Simko, V., Levy, M., Xie, Y., Jin, Y., Zemla, J., 2017. Package 'corrplot.' Statistician 56, e24.
Weston, S., 2019. Using the foreach package.
Whitman, M., Beaman, R.S., Repin, R., Kitayama, K., Aiba, S., Russo, S.E., 2021. Edaphic specialization and vegetation zones define elevational range-sizes for Mt Kinabalu regional flora. Ecography 44, 1698–1709.
Wickham, H., 2011. ggplot2. Wiley Interdiscip. Rev. Comput. Stat. 3, 180–185.
Wieczynski, D.J., Boyle, B., Buzzard, V., Duran, S.M., Henderson, A.N., Hulshof, C.M., Kerkhoff, A.J., McCarthy, M.C., Michaletz, S.T., Swenson, N.G., Asner, G.P., Bentley, L.P., Enquist, B.J., Savage, V.M., 2019. Climate shapes and shifts functional biodiversity in forests worldwide. Proc. Natl. Acad. Sci. 116, 587–592. https://doi.org/10.1073/pnas.1813723116
Williams, J.N., Rivera, R., Choe, H., Schwartz, M.W., Thorne, J.H., 2018. Climate risk on two vegetation axes—Tropical wet-to-dry and temperate arid-to-moist forests. J. Biogeogr. 45, 2361–2374. https://doi.org/10.1111/jbi.13413
Williams, J.N., Seo, C., Thorne, J., Nelson, J.K., Erwin, S., O'Brien, J.M., Schwartz, M.W., 2009. Using species distribution models to predict new occurrences for rare plants. Divers. Distrib. 15, 565–576. https://doi.org/10.1111/j.1472-4642.2009.00567.x
Williams, S.E., Bolitho, E.E., Fox, S., 2003. Climate change in Australian tropical rainforests: an impending environmental catastrophe. Proc. R. Soc. Lond. B Biol. Sci. 270, 1887–1892. https://doi.org/10.1098/rspb.2003.2464
Williams, W., 2009. Resultados del inventario nacional forestal: Nicaragua 2007-2008.
Wright, S.J., Muller-Landau, H.C., Schipper, J., 2009. The Future of Tropical Species on a Warmer Planet. Conserv. Biol. 23, 1418–1426. https://doi.org/10.1111/j.1523-1739.2009.01337.x
Yocom, L.L., Fulé, P.Z., 2012. Human and climate influences on frequent fire in a high-elevation tropical forest. J. Appl. Ecol. 49, 1356–1364. https://doi.org/10.1111/j.1365-2664.2012.02216.x
Zanne, A.E., 2009. Global wood density database. Dryad.
Zea Ramírez, C., 2006. Resultados del inventario de bosques y árboles 2005-2006: Evaluación nacional forestal. SAG, COHDEFOR.
Zeng, Z., Wang, D., Yang, L., Wu, J., Ziegler, A.D., Liu, M., Ciais, P., Searchinger, T.D., Yang, Z.-L., Chen, D., Chen, A., Li, L.Z.X., Piao, S., Taylor, D., Cai, X., Pan, M., Peng, L., Lin, P., Gower, D., Feng, Y., Zheng, C., Guan, K., Lian, X., Wang, T., Wang, L., Jeong, S.-J., Wei, Z., Sheffield, J., Caylor, K., Wood, E.F., 2021. Deforestation-induced warming over tropical mountain regions regulated by elevation. Nat. Geosci. 14, 23–29. https://doi.org/10.1038/s41561-020-00666-0
Zhou, Z., Ouyang, Y., Qiu, Z., Zhou, G., Lin, M., Li, Y., 2017. Evidence of climate change impact on stream low flow from the tropical mountain rainforest watershed in Hainan Island, China. J. Water Clim. Change 8, 293–302. https://doi.org/10.2166/wcc.2016.149
Zizka, A., Steege, H.T., Pessoa, M.D.C.R., Antonelli, A., 2018. Finding needles in the haystack: where to look for rare species in the American tropics. Ecography 41, 321–330. https://doi.org/10.1111/ecog.02192

Zotz, G., Bogusch, W., Hietz, P., Ketteler, N., 2010. Growth of epiphytic bromeliads in a changing world: The effects of CO2, water and nutrient supply. Acta Oecologica 36, 659–665. https://doi.org/10.1016/j.actao.2010.10.003

Zuleta, D., Benavides, A.M., López-Rios, V., Duque, A., 2016. Local and regional determinants of vascular epiphyte mortality in the Andean mountains of Colombia. J. Ecol. 104, 841–849. https://doi.org/10.1111/1365-2745.12563

Appendices

Table of Supplementary Tables

Table S1. Details of 395 studies published between 1994 and 2021 included in the assessment of evidence strength. 140

Table S2. National inventories used to build the list of tropical montane tree in this study. 157

Table S3. Species omitted due to low number (<20) of usable records (shown in brackets). This step was carried out before verifying growth habit and taxonomy, so this list may contain non-arboreal forms and lineages that do not occur as trees (e.g., pteridophytes or monocots) 158

Table S4. List of the 272 species included in the study and number of records per species. Classification according to the Angiosperm Phylogeny Website, URL www.mobot.org/MOBOT/research/APweb/), and current IUCN status (URL https://iconicspecies.iucnredlist.org/). 165

Table S5. Number of species and records per country and subnational division. Most species extent over more than one jurisdiction, thus the total number of species does not correspond to the addition of species per country. 37 records extended beyond our study region into Colombia, which we retained to avoid creating an artificial boundary along political lines. 173

Table S6. Post-SDM Analysis – Species Classification 174

Table S7. Bray-Curtis indices at the topographic discontinuities of the Isthmus of Tehuantepec and Lake Cocibolca. 175

Table S8. SDM current and future projections in pixels and km^2. Projected number of pixels (30 arc-seconds resolution) and equivalent area (km^2) suitable for TMF tree species under current (1981–2010 baseline) and future (2071–2100) climate conditions under RCP8.5. 176

Table S9. Functional trait data for the 272 species included in the analysis before imputation. Height, maximum height, diameter at breast height (DBH), leaf area (LA) and specific leaf area (SLA) were obtained from the TRY (Kattge et al., 2020) and BIEN v4.2 databases (Maitner et al., 2018). Data for height and DBH were complemented with records from scanned herbaria specimens available in GBIF (www.gbif.org; consulted March-April 2024). Wood density (WD) data was obtained from the Global Wood Density Database (Zanne, 2009) and Ordóñez Díaz et al. (2015). For LA, SLA and WD, species-specific data is shown in black, whereas data available at genus level is shown in red; this approach was necessary before imputation to reduce the number of genera without a single datum for any of those traits. 181

Table S10. Imputed functional traits for 272 species included in the analysis, following the data imputation process described in (Carmona et al., 2024) with the *V.PhyloMaker* package (Jin and Qian, 2022). Leaf mass per area (LMA) was obtained as the inverse of specific leaf area (SLA). 187

Table S11. Moran's I coefficients of five functional traits (H, DBH, WD, LA and LMA) and AGB throughout the study area. 195

Table S12. PCA stats of five functional traits (H, DBH, WD, LA and LA) of 272 montane-specialist tree species. Significance of the PCA axes calculated with Monte-Carlo tests, number of repetitions = 999, number of tests = 6. 195

Table of Supplementary Figures

Fig. S1. Linear regression of projected range losses (%) in response to current extent by the end of the century under RCP8.5 180

Fig. S2. Community-weighted means of diameter at breast height (m) under current (1981-2010; left) and future (2071-2100; right) climatic conditions. 193

Fig. S3. Community-weighted means of wood density (g cm^{-3}) under current (1981-2010; left) and future (2071-2100; right) climatic conditions. 193

Fig. S4. Community-weighted means of leaf mass per area (g cm^{-2}) under current (1981-2010; left) and future (2071-2100; right) climatic conditions. 194

Appendix A. Supplementary Information for Chapter 2

Table S1. Details of 395 studies published between 1994 and 2021 included in the assessment of evidence strength.

LoE, Level of Evidence (see Table **2.1** and Figure **2.2** for definitions of categories); N/A, not available; SDM, species distribution model.

Research topic: AE = abiotic environment; BD = biodiversity; CS = conservation strategies; DE = distributional effects; EF = ecosystem functions; ES, ecosystem services; S = synergies.

Ecological level: bi = biosphere; co = community; ec = ecosystem or landscape; ge = genetic; hu = human society; si = species interactions; sp = species.

Taxa: amp = amphibians; bac = bacteria; bir = birds; bry = bryophytes; fun = fungi; ins = insects; lic = lichens; mam = mammals; oar = non-insect arthropods; oin = non-arthropod invertebrates; oth = other taxa; rep = reptiles; vpe = vascular plants (epiphytes); vpl =vascular plants (general); vpt = vascular plants (trees).

Authors	Year	Region	Country	Study type	Research topic	Ecological level	Taxa	LoE
Aiba & Kitayama	2002	Southeast Asia	Malaysia	Survey (longitudinal)	AE	co, ec	vpt	2a
Ainsworth & Drake	2020	Pacific Ocean	Hawaii	Survey (cross-sectional)	CS	co	vpl	2c
Alfonso-Corrado *et al.*	2017	North America	Mexico	Modelling (SDM)	DE	sp	vpt	2b
Altamirano-León *et al.*	2022	North America	Mexico	Modelling (SDM)	DE	sp, ec	vpt	2b
Alvarado-Barrientos *et al.*	2015	North America	Mexico	Survey (longitudinal)	EF	ec	vpt	2a
Alvarado-Barrientos *et al.*	2014	North America	Mexico	Survey (longitudinal)	EF	ec	vpt	2a
Amici *et al.*	2020	Central America	Costa Rica	Survey (cross-sectional)	BD	co, ec	vpe	2c
Anchukaitis & Evans	2010	Central America	Costa Rica	Historic data analysis	S	sp, si, ec	vpt, fun, amp	1b
Anderson *et al.*	2013	Oceania	Australia	Survey (longitudinal)	BD	co	bir	1b
Anderson *et al.*	2012	Oceania	Australia	Modelling (SDM)	BD	co, ec	bir	2b
Arias-Aguilar *et al.*	2020	Central America	Costa Rica	Survey (cross-sectional)	BD	sp	mam	2c
Arzac *et al.*	2019	South America	Venezuela	Modelling (SDM)	DE	co	vpl	2b
Ashton *et al.*	2011	Oceania	Australia	Survey (cross-sectional)	BD	sp, co	ins	2c
Bach *et al.*	2007	South America	Bolivia	Survey (cross-sectional), modelling (SDM)	BD	co	vpe	3
Bader *et al.*	2013	Central America	Panama	Experiment (*ex situ*)	BD	sp, ec	bry	1b
Barker & Ríos-Franceschi	2015	Caribbean	Puerto Rico	Survey (longitudinal)	BD	sp	amp	2a
Barnes *et al.*	2017	Pacific Ocean	Hawaii	Survey (cross-sectional)	EF	ec	vpl	2c
Batumike *et al.*	2022	Africa	DRC	Survey (cross-sectional)	AE, ES	ec, hu	vpl	2b
Baumbach *et al.*	2021	North America, Central America,	Mexico, Guatemala, Belize, El Salvador,	Modelling (SDM)	DE	ec	vpt	2b

		South America	Honduras, Nicaragua, Costa Rica, Panama, Colombia, Venezuela					
Bax *et al.*	2021	South America	Peru	Modelling (SDM)	DE	ec	vpt	2b
Becker & Kuzyakov	2018	Africa	Tanzania	Experiment (field)	EF	ec	bac, vpl, fun	1a
Becker *et al.*	2015	Africa	Tanzania	Survey (longitudinal)	EF	ec	vpt	2a
Bender *et al.*	2019	South America	Peru	Modelling (SDM, other)	BD	co	bir	2b
Bendix *et al.*	2021	South America	Ecuador	Survey (cross-sectional)	EF	ec	vpt	2c
Benning *et al.*	2002	Pacific Ocean	Hawaii	Modelling (other)	S	co	bir	2b
Berriozabal-Islas *et al.*	2018	North America, Central America	Mexico, Guatemala	Modelling (SDM)	BD	sp	rep	2b
Berry *et al.*	2016	North America	Mexico	Survey (cross-sectional)	EF	sp, ec	vpt	2c
Bertoncello *et al.*	2011	South America	Brazil	Survey (cross-sectional)	BD	co	vpt	2c
Bitencourt *et al.*	2016	South America	Brazil	Modelling (SDM)	DE	ec	vpl	2b
Blundo *et al.*	2012	South America	Argentina	Survey (longitudinal)	DE	ec	vpt	2a
Bogawski *et al.*	2019	Africa	Ethiopia, Tanzania, Kenya, DRC, Burundi, Rwanda	Modelling (SDM)	DE	sp	vpl	2b
Borchert *et al.*	2005	North America, Central America	Mexico, Costa Rica	Survey (longitudinal)	EF	sp	vpt	2a
Bothwell *et al.*	2014	Pacific Ocean	Hawaii	Experiment (field)	EF	ec	N/A	1a
Bouskill *et al.*	2013	Caribbean	Puerto Rico	Experiment (field)	EF	co, ec	bac	1a
Caballero-Villalobos *et al.*	2021	South America	Colombia	Modelling (SDM)	DE	ec	vpt	2b
Camarero *et al.*	2021	Global	Peru	Historic data analysis, modelling (climate)	EF	bi	vpt	1b
Campos	2014	North America	Mexico	Survey (longitudinal)	EF	ec	N/A	2a
Campos-Cerqueira & Aide	2017	Caribbean	Puerto Rico	Survey (longitudinal)	BD	sp, co	amp	1b
Campos-Cerqueira & Aide	2021	Caribbean	Puerto Rico	Survey (longitudinal)	DE	co	amp, bir	2a
Campos-Cerqueira *et al.*	2017	Caribbean	Puerto Rico	Surveys (sequential)	BD	co	bir	2b

Cárdenas	2016	South America	Ecuador	Survey (cross-sectional)	BD	co	ins	2c
Castro *et al.*	2020	South America	Brazil, Argentina, Paraguay	Modelling (SDM)	DE	sp	vpt	2b
Catenazzi *et al.*	2014	South America	Peru	Experiment (*ex situ*)	BD	co	amp	1b
Catenazzi *et al.*	2010	South America	Peru	Surveys (sequential)	S	co	amp	2b
Ceballos *et al.*	2021	South America	Argentina	Survey (longitudinal)	EF	co	vpt	1b
Center *et al.*	2016	Central America	Costa Rica	Experiment (*ex situ*)	EF	sp	vpt	1b
Cerano-Paredes *et al.*	2021	North America	Mexico	Historic data analysis	S	ec	vpt	1b
Chakraborty *et al.*	2013	South Asia	India	Modelling (life zones)	DE	bi	N/A	2c
Chang *et al.*	2008	East Asia	Taiwan	Survey (longitudinal), experiment (field)	EF	ec	N/A	1a
Chapman *et al.*	2018	Africa	Uganda	Survey (longitudinal)	EF	sp, si, co	vpt	1b
Chen *et al.*	2017	Caribbean	Puerto Rico	Experiment (field)	EF	ec	N/A	1a
Chen *et al.*	2009	Southeast Asia	Malaysia	Surveys (sequential)	DE	co	ins	1b
Chen *et al.*	2011	Southeast Asia	Malaysia	Surveys (sequential)	BD	co	ins	2b
Chirino *et al.*	2017	South America	Ecuador	Experiment (*ex situ*)	EF	sp	vpt	1b
Chu *et al.*	2014	East Asia	Taiwan	Survey (longitudinal)	EF	sp, ec	vpt	2a
Colares *et al.*	2021	South America	Brazil	Survey (cross-sectional)	BD	co	ins	2c
Colwell *et al.*	2008	Central America	Costa Rica	Survey (cross-sectional), modelling (SDM)	BD	ec, bi	vpl, vpe, ins	2c
Colyn *et al.*	2020	Africa	South Africa	Modelling (SDM)	BD, CS	co, ec	bir	2b
Comarazamy & González	2011	Caribbean	Puerto Rico	Modelling (climate)	AE	ec	N/A	2c
Comarazamy *et al.*	2015	Caribbean	Haiti, Dominican Republic	Historic data analysis, survey (longitudinal), modelling (other)	AE	ec	N/A	1b
Cordier *et al.*	2020	South America	Argentina	Modelling (SDM)	DE. BD	sp, co	amp	2b
Correa-Ayram *et al.*	2017	North America	Mexico	Modelling (SDM)	DE	ec	vpt	2b
Correa-Díaz *et al.*	2021	North America	Mexico	Historic data analysis	EF	sp, ec	vpt	1b
Correa-Díaz *et al.*	2020	North America	Mexico	Historic data analysis	EF	sp	vpt	1b
Costion *et al.*	2015	Oceania	Australia	Modelling (SDM)	BD	sp	vpl	2b

Crain & Tremblay	2017	Caribbean	Puerto Rico	Experiment (*ex situ*)	EF	sp	vpe	1b
Crausbay & Hotchkiss	2010	Pacific Ocean	Hawaii	Survey (cross-sectional)	AE	co, ec	bry, vpe, lic	2c
Crausbay *et al.*	2014	Pacific Ocean	Hawaii	Survey (longitudinal)	AE	co, ec	vpt	2a
Cruz-Elizalde *et al.*	2020	North America	Mexico	Survey (cross-sectional), modelling (SDM)	BD	sp	amp	2b
Cuervo-Robayo *et al.*	2020	North America	Mexico	Historic data analysis, modelling (climate)	AE	bi	N/A	1b
Cuesta *et al.*	2020	South America	N/A	Survey (longitudinal), climate projection	DE	co, ec	vpl	2a
Cuni-Sanchez *et al.*	2019	Africa	Kenya	Survey (longitudinal), historic data analysis	ES	ec, hu	N/A	1b
Cusack *et al.*	2010	Caribbean	Puerto Rico	Experiment (*ex situ*)	EF	ec	N/A	1b
Dangles *et al.*	2008	South America	Ecuador	Survey (cross-sectional)	S	si, ec	ins	2c
Dantas de Paula *et al.*	2021	South America	Ecuador	Survey (longitudinal), modelling (other)	EF	ec	vpt, fun	1b
de Araujo-Lira *et al.*	2020	South America	Brazil	Modelling (SDM)	BD, DE	sp	oar	2b
de la Cruz-Amo *et al.*	2020	South America	Ecuador, Peru	Survey (cross-sectional)	EF	ec	vpl	2c
del Castillo *et al.*	2009	North America, Central America	Mexico, Guatemala	Survey (cross-sectional), modelling (climate, other)	CS	sp, co, ec	vpt	2c
de Gasper *et al.*	2021	South America	Brazil	Modelling (SDM)	DE, BD	sp, ec	vpl	2b
Delsinne *et al.*	2013	South America	Ecuador	Experiment (field)	BD	co	ins	1a
Dialynas & Bras	2019	Caribbean	Puerto Rico	Modelling (climate, other)	EF	ec	N/A	2b
Dolson *et al.*	2021	Central America	Costa Rica	Survey (cross-sectional)	BD	ge, co	ins	2c
Doughty *et al.*	2018	South America	Peru	Survey (cross-sectional), modelling (climate, other)	EF	ec	vpt	2b
Dulle *et al.*	2016	Africa	Tanzania	Survey (cross-sectional)	BD	co	bir	2b
Duque *et al.*	2015	South America	Colombia	Historic data analysis	BD	co, ec	vpt	1b
Dusenge *et al.*	2021	Africa	Rwanda	Experiment (field)	EF	sp	vpt	1a

Eaton *et al.*	2012	Central America	Costa Rica	Survey (cross-sectional)	EF	ec	bac, fun	2c
Eguiguren-Velepucha *et al.*	2016	South America	Ecuador	Survey (cross-sectional), modelling (climate, other)	AE	ec	N/A	2b
Eller *et al.*	2016	South America	Brazil	Experiment (*ex situ*)	EF	ec	vpt	1b
Elsen & Tingley	2015	Global	N/A	Modelling (other)	AE	bi	N/A	2a
Enquist	2002	Central America	Costa Rica	Modelling (life zones)	DE	ec	vpl (mainly)	2b
Enquist & Enquist	2011	Central America	Costa Rica	Surveys (sequential)	EF	co, ec	vpl	2b
Esperón-Rodríguez & Barradas	2015*b*	North America	Mexico	Experiment (*ex situ*), modelling (SDM)	EF	ec	vpt	1b
Esperón-Rodríguez & Barradas	2014	North America	Mexico	Experiment (*ex situ*), modelling (SDM)	DE	co, ec	vpt	1b
Esperón-Rodríguez & Barradas	2015*a*	North America	Mexico	Survey (cross-sectional)	EF	co, ec	vpt	2c
Esperón-Rodríguez *et al.*	2019	North America	Mexico	Modelling (other)	AE	ec	N/A	2b
Estrada-Contreras *et al.*	2015	North America	Mexico	Modelling (SDM)	DE	co, ec	vpt	2b
Fadrique *et al.*	2018	South America	Colombia, Ecuador, Peru, Argentina	Survey (longitudinal) (meta-analysis?)	DE	co, ec	vpt	1b
Feeley & Silman	2010*c*	South America	N/A	Modelling (SDM)	DE	ec	vpl	2c
Feeley & Silman	2010*b*	South America	Colombia, Ecuador, Peru, Bolivia	Modelling (SDM)	DE	ec	vpl	2b
Feeley *et al.*	2020	South America	Colombia	Experiment (*ex situ*)	EF	sp	vpt	1b
Feeley *et al.*	2013	Central America	Costa Rica	Survey (longitudinal)	DE	co, ec	vpt	2a
Feeley *et al.*	2011	South America	Peru	Surveys (sequential)	DE	co	vpt	2b
Forero-Medina *et al.*	2011*a*	South America	Colombia	Modelling (SDM)	DE, BD	sp	amp	2b
Forero-Medina *et al.*	2011*b*	South America	Peru	Surveys (sequential)	BD	co	bir	2b
Franklin *et al.*	2019	Caribbean	Dominican Republic	Survey (cross-sectional)	DE	co	vpl, bir	2c
Freed & Cann	2013	Pacific Ocean	Hawaii	Survey (longitudinal)	S	sp, si	bir, oth	1b
Freeman & Class Freeman	2014	Oceania	Papua New Guinea	Surveys (sequential)	BD	co	bir	2b
Freeman *et al.*	2018	South America	Peru	Survey (cross-sectional)	BD	co	bir	2c
García-Hernández & Toledo-Aceves	2020	North America	Mexico	Experiment (field)	CS	sp	vpt	1a

García-Hernández *et al.*	2019	North America	Mexico	Experiment (field)	CS	sp, si, co	vpt	1a
García-Robledo *et al.*	2018	Central America	Costa Rica	Experiment (*ex situ*)	BD	sp	ins	1b
Gasner *et al.*	2010	Central America	Costa Rica	Surveys (sequential), modelling (other)	BD	sp, co	bir	2b
Gavrutenko *et al.*	2021	Africa	Madagascar	Modelling (SDM)	DE	sp	mam	2b
Geml *et al.*	2014	South America	Argentina	Survey (cross-sectional)	AE	co, ec	fun	2c
Gerold *et al.*	2008	South America	Bolivia	Survey (longitudinal)	EF	ec	vpt	2a
Giamminola	2020	South America	Argentina	Modelling (SDM)	DE	sp	vpt	3
Giardina *et al.*	2014	Pacific Ocean	Hawaii	Survey (longitudinal)	EF	ec	N/A	2a
Girardin *et al.*	2014	South America	Ecuador, Peru, Bolivia	Survey (cross-sectional)	EF	co, ec	vpt	2c
Godoy-Bürki	2016	South America	Argentina	Modelling (SDM)	BD	ec	vpl (?)	2b
Goldsmith at al.	2013	Central America	Costa Rica	Survey (longitudinal), experiment (field)	EF	sp, ec	vpt	2a
Golicher *et al.*	2008	North America	Mexico	Modelling (SDM)	DE	co, ec	vpt	2a
Golicher *et al.*	2012	North America, Central America	Mexico, Guatemala, Belize, El Salvador, Honduras, Nicaragua, Costa Rica, Panama	Modelling (SDM)	BD	ec	vpt	2b
Gómez-Pineda *et al.*	2020	North America	Mexico	Modelling (SDM)	DE	sp, ec	vpt	2b
Gontijo *et al.*	2018	South America	Brazil	Survey (cross-sectional)	BD	sp	rep	2c
González-del-Pliego *et al.*	2020	South America	Colombia	Experiment (*ex situ*)	BD	sp, ec	amp	1b
González-Fernández *et al.*	2018	North America	Mexico	Modelling (SDM)	BD	sp	rep	2b
Gotsch *et al.*	2014*a*	North America	Mexico	Survey (longitudinal), experiment (field)	EF	sp, ec	lic	1a
Gotsch *et al.*	2014*b*	Pacific Ocean	Hawaii	Survey (longitudinal)	EF	sp, ec	vpt	2a
Gotsch *et al.*	2015	Central America	Costa Rica	Experiment (*ex situ*)	EF, BD	sp,ec	vpe	1b
Gotsch *et al.*	2018	Central America	Costa Rica	Survey (longitudinal)	BD	sp, co	vpe	2a
Gotsch *et al.*	2017	Central America	Costa Rica	Survey (cross-sectional)	BD	co	bry, vpe	2c
Grau & Veblen	2000	South America	Argentina	Survey (longitudinal)	DE	co, ec	vpt	1b

Gray *et al.*	2017	Southeast Asia	Malaysia, Brunei, Indonesia	Modelling (SDM)	DE	sp	vpl	2b
Greenwood *et al.*	2014	East Asia	Taiwan	Historic data analysis, Survey (longitudinal)	DE	ec	vpt	1b
Gu *et al.*	2021	East Asia	Taiwan	Surveys (comparative, longitudinal), modelling (other)	EF	ec	vpt	2c
Guillaumet *et al.*	2017	Pacific Ocean	Hawaii	Modelling (SDM, other)	S	sp, si	vpt, bir, oth	2b
Güizado-Rodríguez *et al.*	2012	North America	Mexico	Modelling (SDM)	BD	sp	rep	2b
Gurdak *et al.*	2014	South America	Peru	Survey (cross-sectional)	EF	ec	vpt	2c
Gutiérrez-García & Ricker	2019	North America	Mexico	Historic data analysis	EF	sp, ec	vpt	1b
Gütlein *et al.*	2018	Africa	Tanzania	Survey (longitudinal), experiment (*ex situ*, field)	S	ec	N/A	1a
Harris *et al.*	2014	Southeast Asia	Indonesia	Survey (longitudinal), modelling (climate, other)	BD	co	bir	2b
Hayes *et al.*	2018	South America	Colombia, Ecuador, Peru, Bolivia, Brazil, Argentina, Paraguay	Historic data analysis	BD	sp	bir	1b
He *et al.*	2009	East Asia	China	Experiment (field)	EF	ec	bac, fun	1a
He *et al.*	2010	East Asia	China	Experiment (*ex situ*)	EF	ec	fun	1b
Helmer *et al.*	2019	North America, Central America, South America, Caribbean	N/A	Modelling (SDM)	DE	bi	N/A	2b
Hemp	2009	Africa	Tanzania	Survey (longitudinal)	S	ec	vpt	1b
Hemp	2005	Africa	Tanzania	Survey (longitudinal)	S	co, ec	vpt	2a
Henareh-Khalyani *et al.*	2019	Caribbean	Puerto Rico	Modelling (SDM)	DE	sp, si, ec	vpl	2b
Hermes *et al.*	2018*a*	South America	Ecuador	Modelling (SDM)	BD	sp	bir	1b
Hermes *et al.*	2018*b*	South America	Ecuador	Modelling (SDM. other)	CS	ec	bir	2c
Hernández-Calderón *et al.*	2013	North America	Mexico	Survey (longitudinal)	EF	sp, ec	vpt	2a
Hernández-Gordillo *et al.*	2021	Central America	Costa Rica	Survey (longitudinal)	EF	ec	vpt	2a

Hillyer & Silman	2010	South America	Peru	Experiment (field)	EF	si	vpt	1a
Hiltner *et al.*	2016	Africa	Ethiopia	Modelling (other)	AE	ec	vpt	2b
Horwath *et al.*	2019	South America	Peru	Survey (cross-sectional)	BD	ec	bry	2c
Hsu *et al.*	2014	East Asia	Taiwan	Experiment (field)	BD	sp	vpe	1a
Hsu *et al.*	2012	East Asia	Taiwan	Modelling (SDM)	BD	co, ec	vpe	2b
Iturraide-Pólit *et al.*	2017	South America	Ecuador	Modelling (SDM)	BD	sp, ec	mam	2b
Iwashita *et al.*	2013	Pacific Ocean	Hawaii	Survey (cross-sectional)	EF	ec	vpt	2c
Jacob *et al.*	2015*b*	Africa	Ethiopia	Historic data analysis, Surveys (sequential)	DE	ec	vpt	1b
Jankowski & Rabenold	2007	Central America	Costa Rica	Surveys (sequential)	BD	co	bir	2b
Jankowski *et al.*	2010	Central America	Costa Rica	Experiment (field)	BD	si	bir	1a
Jankowski *et al.*	2013	South America	Peru	Survey (longitudinal)	BD	sp, co	vpt, bir	2a
Jankowski *et al.*	2009	Central America	Costa Rica	Survey (cross-sectional)	BD	co, ec	bir	2c
Jasso-Flores *et al.*	2020	North America	Mexico	Survey (cross-sectional)	EF	ec	vpt	2c
Jiménez-García & Peterson	2019	North America	Mexico	Modelling (SDM)	DE	sp, ec	vpt	2b
Jiménez-García *et al.*	2021	North America	Mexico	Historic data analysis	DE	ec	vpt	1b
Jiménez-López *et al.*	2020	North America	Mexico	Survey (cross-sectional)	BD	co	vpe	2c
Jiménez-Rodríguez *et al.*	2015	Central America	Costa Rica	Survey (longitudinal), modelling (other)	ES	ec	N/A	2c
Jin *et al.*	2012	Southeast Asia	Malaysia	Survey (cross-sectional)	S	co	vpe	2c
Johansson *et al.*	2018	Africa	Ethiopia	Survey (cross-sectional)	CS	co, ec	vpl	2c
John *et al.*	2020	Africa	Tanzania	Modelling (SDM)	DE	ec	vpl	2b
Jones *et al.*	2008	North America	Mexico	Survey (cross-sectional)	BD	co	ins	2c
Joshi *et al.*	2020	South Asia	India	Experiment (field)	S	si, co	vpt	1a
Karger *et al.*	2021	Global	N/A	Modelling (SDM)	DE, BD	bi	vpl, amp, bir, mam	2b
Karmalkar *et al.*	2008	Central America	Costa Rica	Modelling (climate)	AE	ec	N/A	2c
Keyimu *et al.*	2020	East Asia	China	Historic data analysis	EF	sp	vpt	1b
Klauke *et al.*	2016	South America	Ecuador	Survey (cross-sectional)	BD	ge, sp	bir	2c
Klorvuttimontara *et al.*	2011	Southeast Asia	Thailand	Modelling (SDM)	BD	co, bi	ins	2b

Koide *et al.*	2017	Pacific Ocean	Hawaii	Surveys (sequential)	DE	co	vpl	2b
Kreyling *et al.*	2010	Africa	Ethiopia	Modelling (SDM)	DE	co, ec	vpl	2b
Krishnaswamy *et al.*	2014	Global	N/A	Historic data analysis, modelling (other)	EF	bi	vpl	1b
Krömer *et al.*	2013	North America	Mexico	Survey (cross-sectional)	DE	co	vpl, vpe	2c
Kuo *et al.*	2021	East Asia	Taiwan	Surveys (sequential)	AE, BD	co, ec	vpl	2b
La Marca *et al.*	2005	Central America, South America	Costa Rica, Ecuador, Venezuela	Surveys (sequential)	BD	sp	amp	2c
La Sorte *et al.*	2014	Global	N/A	Modelling (climate, SDM)	BD	bi	bir	2b
Larsen	2012	South America	Peru	Survey (cross-sectional)	S	co, ec	ins	2c
Latta *et al.*	2011	South America	Ecuador	Surveys (sequential)	BD	co	bir	2b
LaVal	2004	Central America	Costa Rica	Survey (longitudinal)	BD	co	mam	1b
Lawton *et al.*	2001	Central America	Costa Rica	Modelling (climate)	S	ec	vpl	2b
Ledo *et al.*	2009	South America	Peru	Modelling (SDM)	DE	co, ec	vpt	2b
Li *et al.*	2020	East Asia	China	Experiment (field)	EF	sp	vpt	1a
Li *et al.*	2009	Oceania	Australia	Survey (longitudinal), modelling (SDM)	BD	sp	bir	2b
Liao *et al.*	2020	East Asia	China	Modelling (SDM)	DE	sp	vpt	2b
Lippok *et al.*	2014	South America	Bolivia	Modelling (SDM)	DE	co, ec	vpt	2c
Lips	1998	Central America	Costa Rica	Survey (longitudinal)	BD	co	amp	2a
Liptzin *et al.*	2011	Caribbean	Puerto Rico	Survey (longitudinal)	EF	ec	N/A	2a
Litton *et al.*	2011	Pacific Ocean	Hawaii	Survey (longitudinal)	EF	ec	vpt, fun	2a
Litton *et al.*	2020	Pacific Ocean	Hawaii	Survey (longitudinal)	EF	ec	vpt	2c
Liu *et al.*	2021	East Asia	China	Survey (cross-sectional)	EF	sp, ec	bry, vpe, lic	2c
Lloret & González-Mancebo	2011	Macaronesia	Canary Islands	Modelling (SDM)	BD	co, ec	bry	2b
Loarie *et al.*	2009	Global	Global	Modelling (climate)	AE	bi	N/A	2b
Looby & Treseder	2018	Central America	Costa Rica	Experiment (field)	EF	ec	fun	1a
Looby *et al.*	2016	Central America	Costa Rica	Surveys (sequential)	EF	co, ec	lic, fun	2b
Loope & Giambelluca	1998	Pacific Ocean	Hawaii	Modelling (climate)	AE	co, ec		2b

López-Arce *et al.*	2019	North America	Mexico	Modelling (SDM)	CS	sp, ec	vpl	2b
Los *et al.*	2021	Global	N/A	Historic data analysis, modelling (SDM)	AE, DE	bi	N/A	1b
Lutz *et al.*	2013	South America	Peru	Survey (longitudinal)	DE	ec	vpt	1b
Lyu *et al.*	2021	Pacific Ocean	Hawaii	Survey (longitudinal)	EF, EF	ec	vpt	2a
Macedo *et al.*	2018	South America	Brazil	Survey (cross-sectional)	BD	co	ins	2c
Martin & Fahey	2014	Caribbean	Dominican Republic	Survey (longitudinal)	EF	co, ec	vpt	1b
Martin *et al.*	2007	Caribbean	Dominican Republic	Survey (longitudinal)	AE	ec	vpt	2a
Maul *et al.*	2020	Africa	Uganda	Survey (cross-sectional)	BD	co	bry	2c
Menéndez-Guerrero *et al.*	2020	North America, Central America, South America	N/A	Modelling (SDM)	BD	bi	amp	2b
Metcalfe & Ahlstrand	2019	South America	Peru	Experiment (*ex situ*)	EF	sp, ec	bry	1b
Miles *et al.*	2004	South America	Bolivia, Brazil, Colombia, Ecuador, Peru	Modelling (SDM)	DE	sp,ec	vpl	3
Miller at al.	2018	Caribbean	Puerto Rico	Survey (longitudinal), historic data analysis	AE	ec	N/A	1b
Molina-Venegas *et al.*	2020	Africa	Tanzania	Survey (cross-sectional)	DE	ec	vpl	2c
Montejo-Kovacevich *et al.*	2020	South America	Ecuador	Surveys (comparative, longitudinal), experiment (*in situ*)	BD	sp	ins	1b
Monterroso-Rivas *et al.*	2016	North America	Mexico	Modelling (other)	ES	ec, hu	vpt	2b
Monterroso-Rivas *et al.*	2013	North America	Mexico	Modelling (SDM)	DE	co, ec	vpt	2b
Morales *et al.*	2004	South America	Argentina	Historic data analysis	EF	sp	vpt	1b
Mujawamariya *et al.*	2018	Africa	Rwanda	Survey (longitudinal)	EF	sp	vpt	2a
Müller *et al.*	2017	Global	N/A	Experiment (*ex situ*)	BD	sp	vpe	1b
Muñoz *et al.*	2016	Oceania	Australia	Survey (cross-sectional), experiment (*ex situ*)	BD	sp	rep	1b
Murugan *et al.*	2009	South Asia	India	Survey (longitudinal)	AE	ec	N/A	1b
Nadkarni & Solano	2002	Central America	Costa Rica	Experiment (field)	BD	co	vpe	1a

Navas *et al.*	2013	Central America, South America	Panama, Colombia	Surveys (sequential)	BD	sp, co	amp	2b
Neate-Clegg *et al.*	2021*a*	Africa	Tanzania	Survey (longitudinal)	BD	co	bir	1b
Neate-Clegg *et al.*	2020	Africa	Rwanda	Survey (longitudinal)	BD	co	bir	1b
Neate-Clegg *et al.*	2018	Central America	Honduras	Survey (longitudinal)	BD	co	bir	1b
Neate-Clegg *et al.*	2021*b*	Africa	Tanzania	Surveys (sequential)	DE	co	bir	2b
Neely *et al.*	2020	South America	Brazil	Experiment (*ex situ*)	S	si	fun, amp	1b
Neto dos Santos *et al.*	2020	South America	Brazil	Modelling (SDM)	DE	sp	vpt	2b
Niessner *et al.*	2020	South America	Peru	Survey (longitudinal)	EF	sp	vpt	2a
Notter *et al.*	2007	Africa	Kenya	Modelling (other)	S	ec	N/A	2a
Nottingham *et al.*	2019*a*	South America	Peru	Survey (cross-sectional), experiment (*ex situ*)	EF	ec	bac, fun	1b
Nottingham *et al.*	2019*b*	South America	Peru	Experiment (field)	EF	ec	bac, fun	1a
Nottingham *et al.*	2016	South America	Peru	Experiment (*ex situ*)	EF	ec	N/A	1b
Nowrouzi *et al.*	2018	Oceania	Australia	Survey (cross-sectional), experiment (*ex situ*)	BD	sp, co	ins	1b
Nowrouzi *et al.*	2016	Oceania	Australia	Survey (cross-sectional)	BD	sp, co	ins	2c
Nwe at al.	2020	Southeast Asia	Myanmar	Modelling (SDM)	DE	ec	N/A	2b
O'Donell & Kumar	2006	Central America	Costa Rica	Survey (cross-sectional)	BD	co, ec	ins	2c
Odell *et al.*	2016	Oceania	Australia	Surveys (sequential)	BD	co	ins	2b
Ornelas *et al.*	2018	North America	Mexico	Modelling (SDM)	S	sp, si	vpl, vpe	2a
Ortega-Andrade *et al.*	2015	South America	Ecuador, Colombia, Peru	Modelling (SDM)	BD	sp	mam	2b
O'Sullivan *et al.*	2021	East Asia	Taiwan	Historic data analysis	DE, EF	sp, ec	vpt	2a
Pau *et al.*	2020	Pacific Ocean	Hawaii	Survey (longitudinal)	EF	sp	vpt	1b
Phillips	1996	Global	N/A	Survey (longitudinal)	BD	ec, bi	vpt	2a
Piantoni *et al.*	2016	North America, South America, Caribbean	N/A	Experiment (*ex situ*)	BD	co, ec	rep	1b
Platts *et al.*	2013	Africa	Kenya, Tanzania	Modelling (SDM)	DE	co, ec	vpl	2b
Polato *et al.*	2018	South America	Ecuador	Experiment (*ex situ*)	BD	ge, sp, ec	ins	1b

Ponce-Reyes *et al.*	2017	Africa	Uganda, Rwanda, Burundi, Tanzania, DRC	Modelling (SDM)	DE	ec	N/A	2b
Ponce-Reyes *et al.*	2013	North America	Mexico	Modelling (SDM)	DE	sp, co, ec	vpl, amp, bir, mam	2b
Ponce-Reyes *et al.*	2012	North America	Mexico	Modelling (SDM)	DE	ec	vpl, amp, rep, bir, mam	2b
Ponette-González *et al.*	2010	North America	Mexico	Survey (longitudinal)	S	ec	vpt	2a
Pounds & Crump	1994	Central America	Costa Rica	Survey (longitudinal)	BD	sp	amp	2c
Pounds *et al.*	2006	Central America, South America	Costa Rica, Colombia, Venezuela	Survey (longitudinal)	S	si	fun, amp	1b
Pounds *et al.*	1999	Central America	Costa Rica	Survey (longitudinal)	BD	co	amp, rep, bir	1b
Pouteau & Birnbaum	2016	Pacific Ocean	New Caledonia	Modelling (SDM)	BD	co, ec	vpt	2b
Pouteau *et al.*	2016	Pacific Ocean	Tahiti	Modelling (SDM)	BD	co, ec	vpl, vpe	2b
Puschendorf *et al.*	2009	Central America	Costa Rica	Modelling (SDM)	S	ec	amp	2b
Quadri *et al.*	2021	North America	Mexico	Historic data analysis	EF	sp, ec	vpt	1b
Quiroga *et al.*	2018	South America	Bolivia, Argentina	Modelling (SDM)	DE	sp, ec	vpt	2b
Raman *et al.*	2020*a*	South Asia	India	Survey (cross-sectional), modelling (SDM)	DE	sp	mam	2b
Raman *et al.*	2020*b*	South Asia	India	Survey (cross-sectional), modelling (SDM)	DE	sp	mam	2b
Rambal *et al.*	2020	Central America	Costa Rica, Honduras	Survey (cross-sectional), modelling (other)	EF	sp, ec	vpt	2b
Ramírez-Amezcua *et al.*	2016	North America	Mexico	Modelling (SDM)	DE	sp, ec	vpl	2b
Ramírez-Bautista *et al.*	2020	North America	Mexico	Modelling (other)	BD	sp, co	mam	2b
Ramírez-Villegas *et al.*	2014	South America	Venezuela, Colombia, Ecuador, Peru, Bolivia	Modelling (SDM)	DE	sp, co, ec	vpl, bir	2b
Rapp *et al.*	2012	South America	Peru	Survey (cross-sectional)	EF	sp, co	vpt	2c
Ravindranath & Sukumar	1998	South Asia	India	Modelling (climate)	AE	co, ec	N/A	3
Raxworthy *et al.*	2008	Africa	Madagascar	Surveys (sequential),	BD	sp, co	amp, rep	2b

				historic data analysis				
Regalado & Ritter	2021	Macaronesia	Canary Islands	Survey (longitudinal)	AE, EF	sp, ec	vpt	2c
Rehm & Feeley	2016	South America	Peru	Experiment (field)	DE	ec	vpt	1b
Rehm & Feeley	2015	South America	Peru	Survey (longitudinal), experiment (*ex situ*)	DE	ec	vpt	1b
Rehm & Feeley	2013	South America	Peru	Survey (longitudinal)	DE	co	vpt	2a
Rehm *et al.*	2021	Pacific Ocean	Hawaii	Experiment (*ex situ*), historic data analysis	AE, CS	sp, ec	vpl	1b
Reyes-Chávez *et al.*	2021	Central America	Honduras	Survey (cross-sectional)	BD	co	vpl, vpe	2c
Richards	2021	Central America	Nicaragua	Surveys (comparative, cross-sectional), modelling (other)	BD	co	bry, vpl, lic	2b
Ricker *et al.*	2007	North America	Mexico	Modelling (other)	EF	sp	vpt	2b
Ritter *et al.*	2019	Macaronesia	Canary Islands	Survey (longitudinal), modelling (climate)	EF	ec	vpt	2a
Ritter *et al.*	2009	Macaronesia	Canary Islands	Survey (longitudinal)	EF	ec	vpt	2a
Robertson *et al.*	2010	South America	Peru	Survey (cross-sectional)	EF	sp	vpl	2c
Rödder	2009	Global, Pacific Ocean	Hawaii	Modelling (SDM)	S	sp	amp	2b
Rodríguez-Quiel *et al.*	2019	Central America	Panama	Survey (longitudinal)	BD	co	bry, lic	2a
Rodríguez-Ramírez *et al.*	2020	North America	Mexico	Historic data analysis	EF	sp	vpt	1b
Rodríguez-Ramírez *et al.*	2018	North America	Mexico	Historic data analysis	EF	sp	vpt	1b
Rodríguez-Ramìrez *et al.*	2019	North America	Mexico	Historic data analysis	EF	sp	vpt	1b
Rojas-Briceño *et al.*	2020	South America	Peru	Modelling (SDM)	DE	co	vpt	2b
Rojas-Soto *et al.*	2012	North America	Mexico	Modelling (SDM)	DE	ec	vpl, bir	2b
Román-Cuesta *et al.*	2011	South America	Peru	Historic data analysis, Survey (cross-sectional)	S	ec	vpt	2a
Rosselli *et al.*	2017	South America	Colombia	Surveys (sequential)	BD	co	bir	2b
Round & Gale	2008	Southeast Asia	Thailand	Survey (longitudinal)	BD	sp	bir	1b
Rueda-Solano *et al.*	2016	South America	Colombia	Experiment (field)	BD	sp	amp	1a
Ruiz-Benito *et al.*	2015	East Asia	Taiwan	Historic data analysis	EF	sp, ec	vpt	1b

Salinas *et al.*	2011	South America	Peru	Experiment (field)	EF	ec	vpt	1b
Santillán *et al.*	2020	South America	Ecuador	Survey (cross-sectional)	BD	co	bir	2c
Sarmiento & Kooperman	2019	South America	N/A	Modelling (climate)	AE	ec, hu	N/A	2b
Scheffers & Williams	2018	Africa, Southeast Asia, Oceania	Madagascar, Philippines, Australia	Surveys (comparative, cross-sectional)	BD	co, ec	ins, amp, rep, bir, mam	2c
Scheffers *et al.*	2013a	Southeast Asia	Philippines	Experiment (field)	BD	sp	amp	1a
Scheffers *et al.*	2013b	Southeast Asia	Philippines	Survey (cross-sectional)	BD	sp, co	amp	2c
Scheffers *et al.*	2014	Southeast Asia	Philippines	Survey (longitudinal)	BD	ec	amp	2a
Schmitt *et al.*	2013	Africa	Ethiopia	Survey (longitudinal)	AE	co	vpl	2a
Scholl *et al.*	2021	Caribbean	Puerto Rico	Survey (longitudinal)	AE	ec	vpl	2a
Schuur	2001	Pacific Ocean	Hawaii	Experiment (field)	EF	ec	vpt	1a
Scowcroft *et al.*	2000	Pacific Ocean	Hawaii	Experiment (field)	EF	ec	vpt	1a
Şekercioğlu *et al.*	2008	Global	N/A	Modelling (SDM)	BD	bi	bir	2b
Selmants *et al.*	2014	Pacific Ocean	Hawaii	Survey (longitudinal)	EF	ec	vpl	1b
Selmants *et al.*	2016	Pacific ocean	Hawaii	Survey (cross-sectional)	EF	co	bac	2c
Setyawan *et al.*	2020	Southeast Asia	Indonesia	Modelling (SDM)	DE, BD	sp	vpl	2b
Shah *et al.*	2017*a*	South America	Ecuador	Experiment (*ex situ*)	BD	sp, ec	ins	1b
Shah *et al.*	2017*b*	South America	Ecuador	Experiment (*ex situ*)	BD	sp, ec	ins	1b
Sheldon *et al.*	2011	Global	N/A	Modelling (SDM)	DE	co, ec, bi	ins, amp, rep, bir, mam	2b
Shi & Zhu	2009	East Asia	China	Survey (cross-sectional)	BD	co, ec	vpt	2c
Shiao *et al.*	2020	East Asia	Taiwan	Survey (longitudinal), modelling (SDM)	BD	sp	bir	1b
Shiao *et al.*	2015	East Asia	Taiwan	Survey (longitudinal)	BD	sp	bir	2a
Shoo *et al.*	2010	Oceania	Australia	Survey (longitudinal)	S	ec	amp	2a
Shoo *et al.*	2011	Oceania	Australia	Modelling (SDM)	CS	ec	amp, rep, bir, mam	2b
Shugart *et al.*	2001	Africa	Kenya, Tanzania	Survey (longitudinal)	DE	co, ec	vpl	2a
Sierra-Morales *et al.*	2021	North America	Mexico	Modelling (SDM)	BD, DE	sp, ec	bir	2b
Silva *et al.*	2020	South America	Brazil	Survey (longitudinal),	BD	ge, sp, co	ins	1b

				experiment (*ex situ*)				
Singh *et al.*	2021	Southeast Asia	Malaysia, Brunei, Indonesia	Modelling (SDM)	BD	ec	bir	2b
Song *et al.*	2012	East Asia	China	Experiment (field)	BD	co, ec	bry, lic	1a
Song *et al.*	2016*a*	East Asia	China	Survey (longitudinal)	EF	co	vpt	2a
Song *et al.*	2016*b*	East Asia	China	Survey (longitudinal)	DE	co	vpt	2a
Song *et al.*	2019	Southeast Asia	China, Vietnam	Modelling (SDM)	DE	sp	vpt	2b
Soto-Correa *et al.*	2013	North America	Mexico	Experiment (*ex situ*), modelling (climate)	BD	sp	vpt	1b
Sperling *et al.*	2004	Macaronesia	Canary Islands	Modelling (future climate)	AE	ec	vpt	3
Sreekumar & Nameer	2021	South Asia	India	Modelling (SDM)	DE, BD	sp	bir	2b
Staunton *et al.*	2016	Oceania	Australia	Survey (cross-sectional)	BD	sp, co	ins	2c
Still *et al.*	1999	Global	Costa Rica, Colombia, Malaysia and Rwanda	Modelling (climate)	AE	co	vpl	3
Strangas *et al.*	2019	South America	Brazil	Experiment (*ex situ*), modelling (SDM)	BD	sp, si	rep	1b
Strauch *et al.*	2017	Pacific Ocean	Hawaii	Modelling (other)	ES	ec	vpt	2b
Sukumar *et al.*	1995	South Asia	India	Modelling (climate)	AE	co		2b
Sun *et al.*	2020	East Asia	China	Survey (cross-sectional)	EF	co	bac, vpl	2c
Tagliari *et al.*	2021	South America	Brazil	Modelling (SDM)	DE	ec	vpt	2b
Tao *et al.*	2021	East Asia	China	Experiment (*ex situ*)	BD	sp	rep	1b
Tejedor-Garavito *et al.*	2015	South America	Argentina, Bolivia, Colombia, Ecuador, Peru, Venezuela	Modelling (SDM)	BD	ec	vpt	2b
Téllez-Valdés *et al.*	2006	North America	Mexico	Modelling (SDM)	DE	sp, ec	vpt	2b
Thang & Thu	2021	Southeast Asia	Myanmar	Survey (cross-sectional)	EF	so, co	vpt	2c
Thang *et al.*	2020	Southeast Asia	Myanmar	Survey (cross-sectional), modelling (SDM)	EF	sp, co	vpt	2c
Tito *et al.*	2021	South America	Peru	Experiment (field)	DE	sp, ec	vpt	1a
Toledo-Aceves & del Val	2021	North America	Mexico	Experiment (field)	EF	si	vpt, ins	1a

Toledo-Aceves *et al.*	2019	North America	Mexico	Experiment (field)	EF	ec	vpt	1a
Toledo-Aceves *et al.*	2011	North America	Mexico	Survey (cross-sectional)	CS	ec, hu	N/A	2c
Torres *et al.*	2008	Caribbean	Puerto Rico	Modelling (climate)	AE	ec	N/A	2b
Tovar *et al.*	2013	South America	Venezuela, Colombia, Ecuador, Peru and Bolivia	Modelling (SDM)	DE	ec	N/A	2b
Townsend & Masters	2015	Central America	Costa Rica	Conservation strategy proposal	CS	ec, hu	N/A	3
Tsai *et al.*	2015	East Asia	Taiwan	Surveys (sequential)	BD	sp, co	bir	2b
Urrutia & Vuille	2009	South America	N/A	Modelling (climate)	AE	ec	N/A	2b
Valtonen *et al.*	2013	Africa	Uganda	Survey (longitudinal)	BD	co, ec	ins	1b
Van Beusekom *et al.*	2017	Caribbean	Puerto Rico	Survey (longitudinal)	AE	ec	N/A	1b
Van Beusekom *et al.*	2015	Caribbean	Puerto Rico	Survey (longitudinal)	AE	ec	N/A	2a
Vasquez-Morales *et al.*	2014	North America	Mexico	Modelling (SDM)	DE	sp	vpt	2b
Velásquez-Restrepo *et al.*	2012	South America	Colombia	Survey (longitudinal)	AE	ec	vpt	2a
Velásquez-Tibatá *et al.*	2013	South America	Colombia	Modelling (SDM)	BD	co	bir	2b
Velo-Antón *et al.*	2013	North America	Mexico	Modelling (SDM)	BD	ge, sp	amp	2b
Venter *et al.*	2017	Oceania	Papua New Guinea	Survey (cross-sectional)	EF	ec	vpt	2c
Vieira *et al.*	2011	South America	Brazil	Survey (cross-sectional)	EF	ec	vpl	2c
Villa *et al.*	2019	South America	Venezuela	Survey (longitudinal)	BD	sp, ec	amp	2a
Villers-Ruiz & Trejo-Vásquez	1998	North America	Mexico	Modelling (climate, life zones)	DE	ec	vpt	2b
Vollstädt *et al.*	2020	South America, Africa	Ecuador, Tanzania	Surveys (comparative, cross-sectional)	EF	si	vpt, bir	2c
von May *et al.*	2017	South America	Peru	Experiment (*ex situ*)	BD	sp	amp	1b
Wagner *et al.*	2014	Central America	Panama	Experiment (field)	BD	sp, co	bry	1a
Wallace & McJannet	2013	Oceania	Australia	Survey (longitudinal)	AE	ec	vpl	2a
Wang *et al.*	2003	Caribbean	Puerto Rico	Modelling (climate)	EF	ec	vpt	2a
Wang *et al.*	2002	Caribbean	Puerto Rico	Modelling (other)	EF	ec	N/A	2b
Warne *et al.*	2020	Central America	Costa Rica	Surveys (sequential)	BD	ge, co	ins	2b
Weintraub *et al.*	2016	Central America	Costa Rica, Panama	Survey (cross-sectional)	EF	ec	bac, fun	2c
Whitaker *et al.*	2014	South America	Peru	Survey (cross-sectional),	EF	ec	bac, fun	1b

				experiment (*ex situ*)				
Wicaksono *et al.*	2017	South America	Bolivia, Argentina	Modelling (SDM)	DE	sp, co	vpt, fun	2b
Wilcke *et al.*	2020	South America	Ecuador	Survey (longitudinal)	EF, EF	ec	N/A	1b
Wilcke *et al.*	2013	South America	Ecuador	Survey (longitudinal)	EF	ec	N/A	2a
Williams *et al.*	2020	Central America	Costa Rica	Experiment (*ex situ*)	BD, EF	sp	vpe	1b
Williams *et al.*	2018	North America	Mexico	Modelling (climate, other)	AE	ec	N/A	2b
Williams *et al.*	2003	Oceania	Australia	Modelling (SDM)	BD	ec	amp, rep, bir, mam	2b
Willig *et al.*	2021	Caribbean	Puerto Rico	Survey (longitudinal)	BD	co	oin	1b
Wilson *et al.*	2007	Oceania	Australia	Surveys (sequential)	BD	sp, co	ins	2b
Woodhams *et al.*	2008	Oceania	Australia	Experiment (*ex situ*)	S	sp, si	fun, amp	1b
Wright *et al.*	2009	Global	N/A	Modelling (climate, other)	BD	bi	N/A	2b
Yocom & Fulé	2012	North America	Mexico	Survey (longitudinal)	S	ec	vpt	1b
Young & Lipton	2006	South America	Peru	Survey (cross-sectional)	CS	ec, hu	N/A	2c
Young *et al.*	2017	South America	Peru	Historic data analysis	AE	ec	vpt	2b
Yuste *et al.*	2017	South America	Colombia	Experiment (*ex situ*)	EF	ec	N/A	1b
Zach *et al.*	2010	South America	Ecuador	Survey (longitudinal)	EF	sp	vpt	2a
Zamora-Vilchis *et al.*	2012	Oceania	Australia	Survey (cross-sectional)	S	si, co	bir, oth	2c
Zhang	2010	East Asia	China	Surveys (comparative, longitudinal)	EF	ec	vpt	2a
Zhang *et al.*	1996	Global	N/A	Modelling (climate)	S	ec, bi	vpt	3
Zhou *et al.*	2017	East Asia	China	Historic data analysis	ES	ec	vpt	1b
Zhou *et al.*	2013	East Asia	China	Survey (longitudinal)	EF	ec	vpt	2a
Zimmermann *et al.*	2010	South America	Peru	Experiment (field)	EF	ec	bac	1a
Zotz *et al.*	2010	North America, Central America	Mexico, Panama	Experiment (*ex situ*)	BD	co	vpe	1b
Zuleta *et al.*	2016	South America	Colombia	Surveys (sequential)	BD	co, ec	vpe	2b

Appendix B. Supplementary Information for Chapter 3

Table S2. National inventories used to build the list of tropical montane tree in this study.

Country*	Authors and year	Title	Description
Mexico	González-Espinosa, 2011	Red List of Mexican Cloud Forest Trees	List of 962 plant species recorded in Mexican TMFs, of which 754 are trees, and includes their description, common names and conservation status.
Guatemala	Véliz et al., 2000	La Vegetación Montana de Guatemala	Inventory of montane flora of Guatemala, listing 608 plant species and their habit, of which 62 are trees.
Honduras	Zea Ramírez, 2006	Resultados del inventario de bosques y árboles 2005-2006. Evaluación Nacional Forestal (No. Nc.860). SAG, COHDEFOR	Report on the conservation status of Honduran forests, listing 314 species, distinguishing only those recorded in forested and non-forested areas.
El Salvador	Ministerio de Medio Ambiente y Recursos Naturales (MARN), 2018	Inventario nacional de bosques de El Salvador	Forest inventory which lists a 140 tree species in the country, without distinction of forest types.
Nicaragua	Williams, 2009	Resultados del Inventario Nacional Forestal 2007-2008	National inventory listing 370 tree species without distinction of forest types.
Costa Rica	Sistema Nacional de Áreas de Conservación (Sinac), 2014	Protocolo de campo para la identificación de especies arbóreas: Información taxonómica y dendrológica de las especies arbóreas de Costa Rica. Volumen 3.	Master list of tree species for the country, containing 2,032 species without distinction of forest types.
Panama	Condit et al., 2020	Trees of Panama: A complete checklist with every geographic range	Natonal inventory of trees, listing 3,035 species without forest type distinction.

* Of the seven countries, only the inventories from Mexico and Guatemala listed montane species separately, and despite their focus on arboreal species, some of these sources included other life forms, such as ferns, climbers, epiphytes, etc.

Table S3. Species omitted (N = 1,212) due to low number (<20) of usable records (shown in brackets). This step was carried out before verifying growth habit and taxonomy, so this list may contain non-arboreal forms and lineages that do not occur as trees (e.g., pteridophytes or monocots).

Abarema racemiflora (17)
Abarema zolleriana (16)
Acanthocladus dukei (1)
Actinostemon caribaeus (17)
Aegiphila hirsutissima (12)
Aegiphila valerii (14)
Agave hurteri (11)
Ageratina ovilla (12)
Agrostis exserta (6)
Agrostis laxissima (8)
Aiouea obscura (17)
Aiouea vexatrix (2)
Alchornea guatemalensis (6)
Alfaroa manningii (8)
Alibertia utleyorum (17)
Allophylus gentryi (11)
Amaioua magnicarpa (8)
Amphitecna gentryi (14)
Amphitecna haberi (3)
Amphitecna macrophylla (12)
Amphitecna parviflora (5)
Amphitecna spathicalyx (3)
Amphitecna steyermarkii (11)
Anaxagorea panamensis (10)
Aniba bracteata (4)
Annona caesia (6)
Annona danforthii (8)
Annona hayesii (4)
Annona inconformis (3)
Annona pruinosa (18)
Aphelandra laxa (3)
Aphelandra leonardii (5)
Aphelandra micans (7)
Aphelandra panamensis (2)
Aphelandra scolnikiae (5)
Arachnothryx calycosa (15)
Arachnothryx chaconii (6)
Arachnothryx chiriquiana (6)
Arachnothryx darienensis (1)
Arachnothryx dwyeri (8)
Arachnothryx guerrerensis (5)
Arachnothryx manantlanensis (8)
Arachnothryx monticola (2)
Arachnothryx purpurea (4)
Arachnothryx secunda (5)
Arachnothryx tayloriae (5)
Arachnothryx tuxtlensis (19)
Arawakia caputmonsia (4)
Arawakia panamaea (2)
Archibaccharis corymbosa (8)
Ardisia albisepala (1)
Ardisia amanuensis (9)
Ardisia atropurpurea (1)
Ardisia blepharodes (10)
Ardisia breedlovei (6)
Ardisia brevis (10)
Ardisia capitellata (18)
Ardisia cartagoana (5)
Ardisia colombiana (6)
Ardisia coloradoana (3)
Ardisia conglomerata (2)
Ardisia crassipes (4)
Ardisia crassiramea (19)
Ardisia darienensis (3)
Ardisia dodgei (17)
Ardisia dukei (1)
Ardisia dwyeri (6)
Ardisia folsomii (7)
Ardisia fruticosa (10)
Ardisia furfuracea (11)
Ardisia furfuracella (16)
Ardisia geniculata (4)
Ardisia glomerata (3)
Ardisia gordonii (1)
Ardisia granatensis (9)
Ardisia hagenii (9)
Ardisia hammelii (1)
Ardisia herrerana (1)
Ardisia jefeana (12)
Ardisia knappiae (4)
Ardisia liesneri (16)
Ardisia maxonii (9)
Ardisia megistophylla (16)
Ardisia mexicana (11)
Ardisia nervosissima (9)
Ardisia opaca (2)
Ardisia perinsignis (6)
Ardisia pseudocuspidata (1)
Ardisia pulverulenta (2)
Ardisia rigidifolia (7)
Ardisia scheryi (16)
Ardisia storkii (1)
Ardisia subsessilifolia (10)
Ardisia tenuis (3)
Ardisia tysonii (7)
Ardisia unguiensis (4)
Ardisia ursina (7)
Ardisia vesca (1)
Arthrostemma primaevum (13)
Aspidosperma crypticum (5)
Astrocaryum confertum (5)
Bactris caudata (11)
Bactris charnleyae (5)
Bactris kunorum (5)
Bactris panamensis (8)
Balmea stormae (3)
Bartlettina maxonii (6)
Bartlettina prionophylla (4)
Bauhinia chapulhuacania (19)
Bauhinia proboscidea (6)
Beilschmiedia angustielliptica (3)
Beilschmiedia immersinervis (11)
Beilschmiedia manantlanensis (7)
Beilschmiedia ovalioides (9)
Beilschmiedia sulcata (17)
Beilschmiedia tilaranensis (9)
Bernardia fonsecae (1)
Bernardia macrocarpa (4)
Bernardia macrophylla (8)
Bernardia mollis (8)
Bernardia oblanceolata (7)
Besleria arborescens (12)
Bidens chrysanthemifolia (13)
Bidens holwayi (2)
Blakea crinita (11)
Blakea darcyana (1)
Blakea fragrantissima (9)
Blakea gregii (6)
Blakea hammelii (5)
Blakea parvifolia (8)
Blakea pluvialis (4)
Blakea purpusii (12)
Blakea wilsoniorum (16)
Bocconia glaucifolia (14)
Bocconia vulcanica (19)
Bourreria grandicalyx (11)
Bourreria grayumii (11)
Bourreria rinconensis (5)
Brachistus knappiae (5)
Brownea leucantha (6)
Browneopsis excelsa (19)
Brunellia hygrothermica (13)
Brunellia morii (3)
Brunellia standleyana (15)
Brunfelsia chocoensis (12)
Brunfelsia dwyeri (12)
Buddleja megalocephala (17)
Bunchosia brevisurcularis (7)
Bunchosia grayumii (12)
Bunchosia stipulacea (2)
Bunchosia tutensis (1)
Bunchosia veluticarpa (17)
Bunchosia volcanica (8)
Bursera glabra (18)
Bursera inversa (7)
Buxus moctezumae (10)
Byrsonima herrerae (11)
Calamagrostis guatemalensis (6)
Calamagrostis vulcanica (8)
Calliandra erythrocephala (14)
Calophyllum nubicola (5)
Calyptranthes chytraculia (3)
Calyptrogyne tutensis (12)
Campnosperma panamense (18)
Canavalia dura (17)
Caphexandra heydeana (19)
Cardamine eremita (1)
Carex cuchumatanensis (4)
Carex tojquianensis (4)
Caryodendron angustifolium (3)
Casearia atlantica (7)

Casearia cajambrensis (17)
Casearia standleyana (16)
Castilleja tapeinoclada (9)
Cavendishia panamensis (8)
Cavendishia revoluta (4)
Cavendishia subfasciculata (15)
Cavendishia tenella (12)
Ceanothus coeruleus (10)
Cecropia garciae (17)
Cecropia heterochroma (16)
Cecropia longipes (18)
Cecropia pittieri (6)
Cecropia virgusa (15)
Cedrela monroensis (1)
Centrolobium yavizanum (9)
Cerastium guatemalense (12)
Cerastium juniperorum (3)
Ceratozamia matudae (19)
Cestrum acuminatum (12)
Cestrum dasyanthum (5)
Cestrum formosum (3)
Cestrum knappiae (10)
Cestrum langeanum (9)
Cestrum pacayense (4)
Chamaedorea alternans (19)
Chamaedorea anemophila (11)
Chamaedorea guntheriana (3)
Chamaedorea ponderosa (7)
Chamaedorea subjectifolia (10)
Chomelia grandicarpa (2)
Chomelia rubra (1)
Chrysochlamys angustifolia (14)
Chrysochlamys membrillensis (5)
Chrysophyllum striatum (1)
Cirsium nigriceps (9)
Cirsium skutchii (4)
Citharexylum bourgeauianum (7)
Citharexylum gentryi (6)
Citharexylum hintonii (8)
Citharexylum steyermarkii (7)
Clavija fusca (8)
Clethra chiapensis (5)
Clethra consimilis (13)
Clethra formosa (18)
Clethra fragrans (8)
Clethra gelida (17)
Clethra licanioides (7)
Clethra luzmariae (5)
Clethra pachecoana (14)
Clethra purpusii (7)
Clethra pyrogena (15)
Clethra tutensis (1)
Clethra tuxtlensis (5)
Cleyera cernua (5)
Cleyera velutina (13)
Clibadium sessile (2)
Clitoria glaberrima (14)
Clusia pentandra (12)
Clusia talamancana (7)
Clusia veneralensis (7)
Coaxana purpurea (9)
Cobaea pachysepala (18)
Coccoloba darienensis (5)
Coccoloba gentryi (19)
Coccoloba johnstonii (7)
Coccoloba lasseri (10)
Colpothrinax aphanopetala (10)
Colpothrinax cookii (11)
Conceveiba parvifolia (14)
Conchocarpus toxicarius (16)
Connarus silvanensis (7)
Connarus williamsii (13)
Copaifera panamensis (19)
Cordia anisophylla (14)
Cordia correae (2)
Cordia croatii (18)
Cordia lasiocalyx (14)
Cordia leslieae (4)
Cordia nitida (1)
Cordia tacarcunensis (1)
Couepia janzenii (3)
Couepia scottmorii (3)
Coussapoa glaberrima (11)
Coussapoa macerrima (16)
Coussarea brevipedunculata (3)
Coussarea curvigemmia (7)
Coussarea durifolia (3)
Coussarea garciae (13)
Coussarea veraguensis (8)
Cremastosperma westrae (4)
Croizatia panamensis (12)
Crossopetalum gomezii (14)
Croton jimenezii (13)
Croton rosarianus (3)
Croton speciosus (12)
Croton tonduzii (7)
Crudia acuminata (18)
Cryosophila cookii (2)
Cryosophila grayumii (15)
Cryosophila guagara (13)
Cryptocarya panamensis (1)
Cuatresia amistadensis (12)
Cuatresia morii (12)
Cuchumatanea steyermarkii (1)
Cupania grandiflora (6)
Cupania largifolia (6)
Cuphea schumannii (9)
Cyathea cervantesiana (4)
Cyathea conformis (2)
Cyathea darienensis (1)
Cyathea impar (8)
Cyathea povedae (17)
Cyathea rojasiana (2)
Cybianthus montanus (16)
Cyclanthera steyermarkii (9)
Cymbopetalum rugulosum (9)
Dacryodes patrona (4)
Dahlstedtia calcarata (12)
Dalbergia tilarana (9)
Damburneya bicolor (6)
Daphnopsis correae (4)
Daphnopsis flavida (10)
Daphnopsis folsomii (8)
Daphnopsis hammelii (14)
Daphnopsis mexiae (16)
Daphnopsis morii (9)
Daphnopsis tuerckheimiana (12)
Decazyx macrophyllus (16)
Dendropanax alberti-smithii (8)
Dendropanax bracteatus (3)
Dendropanax hondurensis (14)
Dendropanax pachypedunculatus (1)
Dendropanax pallidus (15)
Dendropanax praestans (13)
Dendropanax punctatus (14)
Deppea guerrerensis (11)
Desmopsis biseriata (3)
Desmopsis brachypoda (5)
Desmopsis confusa (14)
Desmopsis dukei (10)
Desmopsis oerstedii (10)
Desmotes incomparabilis (5)
Dichapetalum gentryi (3)
Dichapetalum nevermannianum (8)
Disocactus phyllanthoides (6)
Disterigma trimerum (11)
Drymaria hypericifolia (5)
Duguetia gentryi (5)
Duguetia tuberculata (4)
Duguetia vallicola (9)
Dussia martinicensis (1)
Echeveria nuda (8)
Elaeagia glossostipula (4)
Elaeagia nitidifolia (11)
Elaphoglossum leporinum (16)
Elaphoglossum tenuifolium (12)
Endlicheria jefensis (1)
Endlicheria tschudyana (11)
Epidendrum microcharis (12)
Erisma blancoa (4)
Erythrina chiriquensis (15)
Erythrina cochleata (11)
Erythrina florenciae (11)
Erythrina globocalyx (13)
Erythrina hondurensis (11)
Erythroxylum brennae (10)
Erythroxylum oxycarpum (16)
Eschweilera biflava (17)
Eschweilera correae (18)
Eschweilera jacquelyniae (16)
Eschweilera jefensis (9)
Eschweilera longipedicellata (10)
Eschweilera pachyderma (10)
Esenbeckia panamensis (14)
Eugenia arrhaphocalyx (4)
Eugenia basilaris (2)
Eugenia belloi (4)
Eugenia brachyblastiflora (1)
Eugenia cerrocacaoensis (7)
Eugenia chavarriae (5)
Eugenia chepensis (18)
Eugenia chiapensis (4)
Eugenia citroides (13)
Eugenia cocosensis (1)
Eugenia coibensis (12)
Eugenia coloradoensis (15)
Eugenia corusca (9)
Eugenia cricamolensis (11)
Eugenia darcyi (12)
Eugenia earthiana (5)
Eugenia glandulosopunctata (19)
Eugenia grayumii (3)
Eugenia haberi (4)
Eugenia hammelii (7)
Eugenia hartshornii (16)
Eugenia herrerae (4)
Eugenia jambos (3)
Eugenia lepidota (12)

Eugenia letreroana (5)
Eugenia leucadendron (2)
Eugenia lithosperma (6)
Eugenia magniflora (1)
Eugenia mcphersonii (5)
Eugenia monteverdensis (14)
Eugenia nesiotica (6)
Eugenia ovandensis (1)
Eugenia pacifica (9)
Eugenia paloverdensis (4)
Eugenia ravenii (10)
Eugenia riosiae (10)
Eugenia roseola (1)
Eugenia roseopetala (2)
Eugenia sancarlosensis (4)
Eugenia sanjuanensis (7)
Eugenia sarapiquensis (7)
Eugenia selvana (8)
Eugenia skutchii (4)
Eugenia teresae (15)
Eugenia tilarana (5)
Eugenia tonii (1)
Eugenia trunciflora (15)
Eugenia veraguensis (1)
Eugenia verruculata (4)
Eugenia zuchowskiae (7)
Euphorbia macropodoides (9)
Euphorbia zierioides (3)
Faramea accumulans (2)
Faramea areolata (15)
Faramea capulifolia (13)
Faramea cobana (6)
Faramea correae (3)
Faramea lehmannii (8)
Faramea liesneri (10)
Faramea macrura (6)
Faramea papillata (4)
Faramea papirifolia (11)
Faramea permagnifolia (10)
Faramea scalaris (3)
Festuca amplissima (10)
Ficus carchiana (6)
Ficus davidsoniae (13)
Ficus francoae (2)
Ficus osensis (2)
Ficus richteri (3)
Forchhammeria iltisii (9)
Frangula darienensis (1)
Freziera forerorum (1)
Furcraea quicheensis (17)
Galipea panamensis (5)
Garrya corvorum (3)
Gaultheria odorata (2)
Geissanthus perpuncticulosus (11)
Geonoma mooreana (17)
Geonoma triandra (13)
Gibsoniothamnus truncatus (5)
Gloeospermum blakeanum (1)
Gloeospermum dichotomum (8)
Gloeospermum pauciflorum (8)
Gloeospermum portobelense (5)
Gonzalagunia longithyrsa (17)
Graffenrieda jefensis (1)
Gratiola oresbia (5)
Guapira standleyana (9)
Guarea adenophylla (13)
Guarea aguilarii (5)
Guarea caulobotrys (11)
Guarea constricta (3)
Guarea corticosa (5)
Guarea donnell-smithii (7)
Guarea gentryi (16)
Guarea inesiana (3)
Guarea macrocalyx (2)
Guarea montana (16)
Guarea pilosa (12)
Guarea pyriformis (17)
Guarea subsessilifolia (5)
Guarea tafae-malekui (1)
Guarea talamancana (1)
Guarea zarceroensis (9)
Guatteria aberrans (10)
Guatteria acrantha (1)
Guatteria allenii (10)
Guatteria jefensis (13)
Guatteria panamensis (2)
Guatteria rostrata (19)
Guatteria rotundata (9)
Guatteria rubiginosa (7)
Guatteria sessilicarpa (13)
Guatteria tacarcunae (1)
Guatteria zamorae (6)
Guettarda brenesii (8)
Guettarda ramuliflora (15)
Gustavia fosteri (2)
Gustavia monocaulis (4)
Gymnanthes dressleri (2)
Gyranthera darienensis (8)
Hackelia skutchii (9)
Halenia alata (9)
Halenia schiedeana (17)
Hamelia barbata (9)
Hamelia sanguinea (16)
Hampea albipetala (11)
Hampea longipes (18)
Hampea micrantha (19)
Hampea montebellensis (10)
Hampea punctulata (16)
Hasseltia allenii (1)
Hedyosmum burgerianum (3)
Hedyosmum correanum (5)
Hedyosmum gentryi (16)
Heisteria skutchii (2)
Helenium integrifolium (1)
Heliopsis buphthalmoides (16)
Henriettea strigosa (3)
Herrania pulcherrima (7)
Hibiscus spathulatus (3)
Hieracium guatemalense (1)
Hippotis stellata (3)
Hirtella papillata (11)
Hirtella trichotoma (4)
Hoffmannia hondurensis (6)
Hoffmannia manussatani (2)
Hordeum guatemalense (2)
Huberodendron allenii (18)
Huertea cubensis (15)
Hymenandra crosbyi (4)
Hymenopus costaricensis (9)
Hyperbaena allenii (1)
Hyperbaena eladioana (18)
Hypericum calcicola (7)
Hypericum epigeium (4)
Ilex condensata (18)
Ilex dugesii (6)
Ilex haberi (19)
Ilex mexicana (1)
Ilex servinii (3)
Ilex stellata (1)
Ilex tectonica (7)
Inga bracteifera (4)
Inga cabrerae (1)
Inga calderonii (18)
Inga canonegrensis (4)
Inga colimana (3)
Inga cuspidata (2)
Inga dasycarpa (7)
Inga dwyeri (3)
Inga filiformis (16)
Inga golfodulcensis (17)
Inga herrerae (9)
Inga huastecana (6)
Inga involucrata (2)
Inga jefensis (19)
Inga latipes (10)
Inga portobellensis (5)
Inga pseudoinvolucrata (1)
Inga saffordiana (6)
Inga spiralis (7)
Inga stenophylla (19)
Inga tenuiloba (4)
Inga urceolata (6)
Isertia scorpioides (6)
Ixchelia uxpanapana (4)
Ixora knappiae (4)
Jaltomata confinis (3)
Juglans boliviana (19)
Juniperus standleyi (15)
Karwinskia pluvialis (3)
Klarobelia stipitata (17)
Koanophyllon panamense (11)
Koanophyllon wetmorei (6)
Krugiodendron acuminatum (5)
Ladenbergia dwyeri (14)
Ladenbergia laurifolia (4)
Leptobalanus morii (1)
Leucaena salvadorensis (16)
Licania cruegeriana (7)
Licaria brenesii (9)
Licaria caribaea (3)
Licaria chinanteca (11)
Licaria cogolloi (3)
Licaria glaberrima (19)
Licaria mexicana (3)
Licaria multinervis (13)
Licaria nitida (10)
Licaria pergamentacea (12)
Licaria phymatosa (7)
Licaria sarapiquensis (11)
Licaria siphonantha (1)
Licaria velutina (17)
Liparis fantastica (5)
Lisianthius aurantiacus (13)
Lisianthius habuensis (2)
Lithospermum mediale (5)
Lobelia stolonifera (9)
Lobelia umbellifera (9)
Lonchocarpus hidalgensis (14)

Lonchocarpus lasiotropis (6)
Lonchocarpus megacarpus (1)
Lonchocarpus nebularis (2)
Lonchocarpus purpureus (8)
Lonicera guatemalensis (1)
Lopezia lopezioides (10)
Lubaria aroensis (9)
Lycianthes howardiana (3)
Lycianthes rzedowskii (17)
Lycoseris crocata (18)
Lysimachia steyermarkii (4)
Mabea tenorioi (7)
Macoubea mesoamericana (8)
Macrolobium dressleri (19)
Macrolobium herrerae (11)
Macrolobium modicopetalum (13)
Macrolobium pittieri (16)
Magnolia chiriquiensis (1)
Magnolia guerrerensis (4)
Magnolia krusei (10)
Magnolia morii (3)
Magnolia panamensis (2)
Magnolia sambuensis (8)
Magnolia vazquezii (2)
Magnolia yoroconte (12)
Malmea dimera (11)
Malpighia verruculosa (14)
Mammea immansueta (3)
Manilkara spectabilis (1)
Mappia multiflora (19)
Mappia racemosa (19)
Marila domingensis (4)
Marila lactogena (12)
Matisia arteagensis (16)
Matisia dolichopoda (17)
Matisia dolichosiphon (1)
Matisia exalata (11)
Matisia jefensis (11)
Matisia pacifica (6)
Matisia sanblasensis (12)
Matisia tinamastiana (6)
Maytenus grisea (4)
Meliosma chiriquensis (2)
Meliosma clandestina (18)
Meliosma cordata (1)
Meliosma isthmensis (9)
Meliosma linearifolia (4)
Meliosma mexicana (3)
Meliosma nesites (6)
Meliosma oligantha (3)
Meliosma schlimii (12)
Meliosma subcordata (12)
Meriania odorata (6)
Meriania panamensis (15)
Miconia aurantiaca (19)
Miconia calocoma (9)
Miconia cionotricha (13)
Miconia colliculosa (2)
Miconia commutata (13)
Miconia coriacea (14)
Miconia crocata (13)
Miconia cuspidatissima (16)
Miconia dissita (15)
Miconia gentryi (3)
Miconia heterothrix (9)
Miconia hildeana (3)
Miconia incurva (18)
Miconia latidecurrens (16)
Miconia mcphersonii (2)
Miconia morii (5)
Miconia nutans (10)
Miconia quadridomius (12)
Miconia santaritensis (7)
Miconia saxicola (16)
Miconia sparrei (13)
Miconia talamancensis (12)
Miconia teotepecensis (8)
Miconia umbriensis (7)
Monnina crepinii (1)
Monnina sylvicola (10)
Monochaetum alpestre (18)
Monochaetum cordatum (18)
Monochaetum exaltatum (12)
Montanoa echinacea (4)
Montanoa pteropoda (13)
Monteverdia jefeana (5)
Monteverdia sieberiana (17)
Montia calcicola (2)
Moquilea chiriquiensis (2)
Moquilea dodsonii (6)
Moquilea fasciculata (3)
Mortoniodendron apetalum (12)
Mortoniodendron hirsutum (16)
Mortoniodendron ruizii (2)
Mosannona costaricensis (13)
Mosannona hypoglauca (8)
Mosannona maculata (5)
Mouriri completens (13)
Mouriri osaensis (14)
Mouriri panamensis (5)
Mouriri tuberculata (15)
Muehlenbeckia vulcanica (5)
Muhlenbergia breviculmis (5)
Muhlenbergia orophila (7)
Myosotis scorpioides (16)
Myrcia concinna (4)
Myrcia fosteri (17)
Myrcia fusca (5)
Myrcia grandis (2)
Myrcia lapidulosa (4)
Myrcia panamensis (2)
Myrcia zetekiana (10)
Myrrhidendron maxonii (10)
Naucleopsis straminea (4)
Nectandra belizensis (11)
Nectandra hypoleuca (19)
Nectandra longipetiolata (5)
Nectandra ramonensis (11)
Neea amplexicaulis (9)
Neea darienensis (13)
Neomirandea burgeri (5)
Neomirandea folsomiana (4)
Neomirandea homogama (16)
Neosprucea wilburiana (10)
Neurolaena oaxacana (7)
Nyssa talamancana (9)
Ocotea adela (8)
Ocotea arcuata (6)
Ocotea atacta (1)
Ocotea atlantica (11)
Ocotea candidovillosa (7)
Ocotea darcyi (4)
Ocotea disjuncta (6)
Ocotea fendleri (7)
Ocotea glaucosericea (18)
Ocotea gordonii (7)
Ocotea guatemalensis (10)
Ocotea haberi (11)
Ocotea hartshorniana (14)
Ocotea holdridgeana (14)
Ocotea iridescens (6)
Ocotea jefensis (2)
Ocotea jorge-escobarii (14)
Ocotea lentii (13)
Ocotea matudae (3)
Ocotea monteverdensis (19)
Ocotea multiflora (15)
Ocotea parvula (7)
Ocotea patula (6)
Ocotea pausiaca (3)
Ocotea pentagona (7)
Ocotea pharomachrosorum (6)
Ocotea pittieri (10)
Ocotea platyphylla (12)
Ocotea producta (4)
Ocotea rivularis (18)
Ocotea rubrinervis (14)
Ocotea rufescens (5)
Ocotea salvinii (14)
Ocotea sarcodes (2)
Ocotea sauroderma (6)
Ocotea standleyi (10)
Ocotea subalata (3)
Ocotea tonduzii (12)
Ocotea tonii (6)
Ocotea truncata (11)
Ocotea valerioides (18)
Ocotea viridiflora (10)
Ocotea wedeliana (2)
Onoseris silvatica (19)
Oreomunnea pterocarpa (17)
Oreopanax arcanus (10)
Oreopanax compactus (4)
Oreopanax costaricensis (16)
Oreopanax flaccidus (12)
Oreopanax nubigenus (19)
Oreopanax paramicola (13)
Oreopanax platyphyllus (7)
Oreopanax spathulatus (1)
Oreopanax striatus (10)
Oreopanax superoerstedianus (11)
Ormosia cruenta (15)
Ormosia intermedia (2)
Ormosia panamensis (7)
Ormosia tovarensis (13)
Ormosia velutina (8)
Osa pulchra (6)
Osmanthus americanus (17)
Ouratea darienensis (1)
Ouratea flexipedicellata (1)
Ouratea jefensis (10)
Ouratea knappiae (9)
Ouratea oblita (2)
Ouratea stenobasis (4)
Ouratea sulcatinervia (6)
Ouratea tristis (1)
Oxalis calcicola (2)
Oxylobus glandulifer (2)

PRESERVED_SPECIMEN (1)
Palicourea bella (6)
Palicourea boraginoides (12)
Palicourea chiriquina (14)
Palicourea discolor (9)
Palicourea hammelii (11)
Palicourea macrantha (18)
Palicourea montensis (10)
Palicourea ochnoides (6)
Palicourea orosiana (12)
Palicourea pereziana (6)
Palicourea roseocremea (12)
Palicourea roseofaucis (5)
Palicourea tubuliflora (7)
Palicourea tumidonodosa (1)
Palicourea tutensis (1)
Palicourea vestita (13)
Panopsis cinnamomea (4)
Paramachaerium gruberi (7)
Parathesis acostensis (9)
Parathesis cartagoana (3)
Parathesis cintalapana (3)
Parathesis columnaris (17)
Parathesis crassiramea (9)
Parathesis croatii (1)
Parathesis fusca (17)
Parathesis glaberrima (1)
Parathesis glendae (2)
Parathesis lanceolata (6)
Parathesis leptopa (19)
Parathesis montana (2)
Parathesis panamensis (3)
Parathesis seibertii (10)
Parathesis subcoriacea (3)
Parathesis subulata (8)
Parietaria macrophylla (11)
Parinari chocoensis (7)
Parinari parvifolia (15)
Parmentiera cereifera (12)
Parmentiera dressleri (1)
Passiflora podadenia (16)
Patinoa almirajo (9)
Peltogyne mexicana (15)
Pentagonia angustifolia (6)
Pentagonia dwyeriana (1)
Pentagonia gymnopoda (11)
Pentagonia lobata (14)
Pentagonia nuciformis (15)
Pentagonia osaensis (9)
Pentagonia parvifolia (10)
Pentagonia sanblasensis (4)
Pentaplaris doroteae (9)
Peperomia cuchumatanica (4)
Peperomia donaguiana (15)
Pera aperta (2)
Pera oppositifolia (2)
Perrottetia excelsa (1)
Persea albida (15)
Persea albiramea (2)
Persea brenesii (6)
Persea chrysantha (1)
Persea laevifolia (3)
Persea obscura (2)
Persea obtusifolia (13)
Persea pallescens (13)
Persea rufescens (9)
Persea silvatica (3)
Perymenium gracile (6)
Phanerophlebia pumila (8)
Phaseolus chiapasanus (15)
Pholidostachys pulchra (12)
Photinia mexicana (16)
Phyllanthus gentryi (3)
Phyllanthus purpusii (9)
Phyllanthus tuerckheimii (6)
Phytelephas seemannii (17)
Picramnia guerrerensis (8)
Pilea quercifolia (13)
Pinus rudis (3)
Piper affectans (5)
Piper albopunctulatissimum (4)
Piper asymmetricum (1)
Piper auritifolium (17)
Piper cativalense (2)
Piper chiriquinum (17)
Piper clavuligerum (7)
Piper corozalanum (14)
Piper daguanum (19)
Piper decurrens (19)
Piper dunlapii (5)
Piper euryphyllum (18)
Piper gibbosum (9)
Piper gonocarpum (2)
Piper hartwegianum (5)
Piper hirtellipetiolum (10)
Piper melastomoides (16)
Piper paulowniifolium (1)
Piper pinoganense (16)
Piper sanctum (16)
Piper subnudispicum (18)
Piper xanthostachyum (18)
Pisonia silvatica (14)
Pithecellobium bipinnatum (6)
Pithecellobium furcatum (18)
Pithecellobium johansenii (14)
Platymiscium curuense (13)
Platymiscium darienense (14)
Pleurothyrium glabritepalum (13)
Pleurothyrium guindonii (8)
Pleurothyrium hexaglandulosum (4)
Pleurothyrium immersum (3)
Pleurothyrium oblongum (3)
Pleurothyrium pauciflorum (5)
Pleurothyrium racemosum (1)
Pleurothyrium triflorum (6)
Plinia cerrocampanensis (3)
Plinia coclensis (5)
Plinia cuspidata (4)
Plinia darienensis (3)
Plinia gentryi (3)
Plinia guanacastensis (5)
Plinia moralesii (2)
Plinia panamensis (7)
Plinia povedae (11)
Plinia puriscalensis (8)
Plinia salamancana (4)
Poa seleri (8)
Podocarpus costaricensis (5)
Podocarpus magnifolius (12)
Polystichum furfuraceum (1)
Polystichum ordinatum (10)
Porcelia magnifructa (6)
Posoqueria chocoana (12)
Posoqueria correana (6)
Posoqueria costaricensis (6)
Posoqueria grandifructa (19)
Posoqueria laevis (15)
Posoqueria robusta (14)
Potalia turbinata (14)
Potentilla goldmanii (2)
Potentilla heterosepala (13)
Pouteria austin-smithii (14)
Pouteria belizensis (16)
Pouteria bulliformis (13)
Pouteria calistophylla (18)
Pouteria chiricana (11)
Pouteria filiformis (9)
Pouteria foveolata (19)
Pouteria lecythidicarpa (9)
Pouteria sclerocarpa (5)
Pouteria silvestris (10)
Pouteria simulans (19)
Pouteria stipitata (13)
Pouteria triplarifolia (2)
Preslianthus panamensis (4)
Prestoea pubens (7)
Prockia costaricensis (8)
Prunus annularis (11)
Prunus guatemalensis (1)
Prunus lundelliana (10)
Pseudomalmea darienensis (1)
Psychotria cascajalensis (2)
Psychotria convergens (19)
Psychotria durilancifolia (6)
Psychotria insignis (1)
Psychotria jefensis (4)
Psychotria liesneri (2)
Psychotria olgae (5)
Psychotria pacorensis (2)
Psychotria philacra (2)
Psychotria sixaolensis (10)
Pterandra isthmica (2)
Pterandra mcphersonii (1)
Pterocarpus michelianus (18)
Pterygota excelsa (13)
Pyrola angustifolia (13)
Quadrella antonensis (17)
Quadrella dressleri (1)
Quadrella mirifica (4)
Quadrella morenoi (6)
Qualea cymulosa (4)
Qualea panamensis (3)
Quararibea ciroana (2)
Quararibea gomeziana (13)
Quararibea parviflora (1)
Quararibea pendula (7)
Quararibea platyphylla (18)
Quararibea pumila (7)
Quararibea santaritensis (1)
Quararibea tulekunae (3)
Quercus acuta (1)
Quercus brenesii (7)
Quercus delgadoana (2)
Quercus hirtifolia (3)
Quercus macdougallii (4)
Quercus nixoniana (2)
Quercus sarahmariae (5)
Quetzalia contracta (10)

Quetzalia guatemalensis (16)
Randia cookii (17)
Randia genipoides (19)
Randia grayumii (12)
Randia lasiantha (11)
Randia mira (13)
Randia tomatillo (15)
Raphia taedigera (15)
Raputiarana subsigmoidea (11)
Rauvolfia purpurascens (19)
Rhamnus serrata (19)
Rhynchosia amabilis (11)
Rhynchostele ehrenbergii (14)
Rinorea brachythrix (1)
Rinorea hirsuta (10)
Rinorea zygomorpha (10)
Rojasianthe superba (14)
Roldana anisophylla (5)
Roldana robinsoniana (11)
Romanschulzia alpina (1)
Romanschulzia guatemalensis (2)
Rondeletia hameliifolia (18)
Rondeletia panamensis (19)
Roupala percoriacea (7)
Rubus cymosus (18)
Rubus hadrocarpus (16)
Rubus philyrophyllus (5)
Rudgea hemisphaerica (7)
Rudgea isthmensis (2)
Rudgea laevis (9)
Rudgea mcphersonii (5)
Rudgea panamensis (2)
Rudgea sanblasensis (2)
Rudgea trifurcata (18)
Russelia syringifolia (8)
Rustia costaricensis (15)
Rustia dressleri (3)
Sabazia pinetorum (1)
Sabinaria magnifica (1)
Sacoglottis holdridgei (9)
Salacia macrocremastra (2)
Salacia panamensis (11)
Salix aeruginosa (15)
Salvia atropaenulata (11)
Salvia disjuncta (16)
Salvia langlassei (18)
Salvia sanctae-luciae (12)
Salvia tonalensis (16)
Sapium allenii (4)
Sarcomphalus strychnifolius (6)
Saurauia conzattii (19)
Saurauia matudae (9)
Saurauia pustulata (10)
Saurauia seibertii (7)
Saurauia waldheimia (2)
Saurauia zahlbruckneri (4)
Schefflera albocapitata (1)
Schefflera aquaverensis (2)
Schefflera cartagoensis (2)
Schefflera cicatricata (3)
Schefflera instita (4)
Schefflera jefensis (6)
Schefflera macphersonii (4)
Schefflera panamensis (12)
Schefflera pubens (4)
Schefflera seibertii (5)
Schefflera whitefoordiae (1)
Schistocarpha liebmannii (12)
Schizocalyx veraguensis (15)
Schizocarpum dieterleae (8)
Schoepfia macrophylla (8)
Schradera neeoides (2)
Schradera rotundata (14)
Schultesianthus crosbyanus (11)
Sebastiania jaliscensis (10)
Sedum australe (10)
Sedum guatemalense (13)
Senecio godmanii (13)
Senna cajamarcae (6)
Senna caudata (18)
Sigesbeckia nudicaulis (1)
Simira klugei (10)
Simira panamensis (3)
Sisyrinchium johnstonii (13)
Sloanea brenesii (18)
Sloanea garcia-cossioi (9)
Sloanea geniculata (8)
Sloanea guapilensis (4)
Sloanea laevigata (8)
Sloanea ligulata (2)
Sloanea longipes (10)
Sloanea megaphylla (1)
Sloanea paucinervia (3)
Sloanea rugosa (14)
Smallanthus mcvaughii (10)
Sobralia galeottiana (17)
Solanum agrimoniifolium (2)
Solanum armentalis (12)
Solanum dimorphandrum (8)
Solanum fortunense (17)
Solanum fosbergianum (10)
Solanum incomptum (12)
Solanum muenscheri (5)
Solanum narcoticosmum (11)
Solanum pastillum (15)
Solanum pluviale (8)
Solanum roblense (19)
Solanum sotobosquense (9)
Sommera chiapensis (8)
Sorocea ruminata (14)
Spathacanthus parviflorus (15)
Sphaeropteris brunei (19)
Stachyarrhena heterochroa (14)
Stachys calcicola (13)
Stachys nubilorum (6)
Stanmarkia medialis (2)
Stelis oaxacana (6)
Stellaria irazuensis (14)
Stenanona panamensis (6)
Stenanona tubiflora (1)
Stenostomum turrialbanum (14)
Stephanopodium gentryi (7)
Sterculia allenii (4)
Steriphoma paradoxum (9)
Strychnos croatii (17)
Stylogyne hayesii (7)
Styphnolobium monteviridis (5)
Styphnolobium parviflorum (5)
Styphnolobium sporadicum (8)
Styrax austromexicanus (17)
Styrax conterminus (13)
Styrax magnus (18)
Styrax peruvianus (9)
Swartzia maquenqueana (5)
Swartzia nuda (6)
Swartzia picramnioides (13)
Swartzia robiniifolia (13)
Swartzia zeledonensis (10)
Symphoricarpos guatemalensis (2)
Symplocos elliptica (7)
Symplocos excelsa (10)
Symplocos morii (4)
Symplocos naniflora (16)
Symplocos oreophila (1)
Symplocos panamensis (11)
Symplocos povedae (18)
Symplocos retusa (5)
Symplocos sousae (14)
Symplocos striata (6)
Symplocos tacanensis (3)
Tabebuia impetiginosa (14)
Tabebuia striata (7)
Tacarcuna gentryi (2)
Tachigali panamensis (12)
Talisia equatoriensis (10)
Talisia morii (1)
Talisia princeps (14)
Tapirira rubrinervis (12)
Tapura colombiana (10)
Tapura cubensis (6)
Tapura panamensis (2)
Tauschia steyermarkii (3)
Telanthophora liebmannii (18)
Telanthophora standleyi (19)
Ternstroemia sylvatica (9)
Tessmannianthus cereifolius (2)
Tessmannianthus gordonii (2)
Tetrorchidium costaricense (18)
Tetrorchidium robledoanum (15)
Tetrorchidium trichotocarpum (16)
Theobroma mammosum (16)
Tigridia immaculata (11)
Tournefortia multiflora (5)
Tournefortia ramonensis (11)
Tovomita morii (3)
Trichilia pittieri (19)
Trisetum rosei (13)
Trisetum tonduzii (8)
Triumfetta arborescens (1)
Trophis noraminervae (14)
Unonopsis bullata (18)
Unonopsis costaricensis (9)
Unonopsis darienensis (15)
Unonopsis hammelii (4)
Unonopsis longipes (13)
Unonopsis macrocarpa (2)
Unonopsis megalosperma (1)
Unonopsis osae (15)
Unonopsis penduliflora (7)
Unonopsis stevensii (6)
Urera martiniana (5)
Vaccinium luteynii (5)
Vaccinium santafeense (2)
Vachellia melanoceras (15)
Vachellia ruddiae (16)
Valeriana deltoidea (18)
Valeriana scandens (16)
Vallesia spectabilis (1)

Vantanea barbourii (13)
Vantanea occidentalis (6)
Vatairea lundellii (12)
Verbesina baruensis (2)
Verbesina calciphila (5)
Verbesina fuscasiccans (7)
Verbesina hidalgoana (6)
Verbesina oerstediana (3)
Verbesina tapantiana (3)
Viburnum discolor (11)
Viburnum disjunctum (13)
Viburnum euryphyllum (1)
Viburnum obtusatum (12)
Virola albidiflora (13)
Virola amistadensis (6)
Virola chrysocarpa (14)
Virola fosteri (7)
Virola megacarpa (10)
Virola montana (15)
Virola otobifolia (7)
Vismia jefensis (19)
Vismia latisepala (14)
Vitex floridula (5)
Vitex masoniana (13)
Vochysia allenii (13)
Vochysia gentryi (8)
Vochysia jefensis (13)
Volkameria pittieri (13)
Votomita cupuliformis (2)
Votschia nemophila (1)
Weinmannia horrida (4)
Weinmannia karsteniana (14)
Weinmannia trianae (2)
Weinmannia vulcanicola (12)
Wercklea cocleana (10)
Wercklea grandiflora (2)
Wercklea lutea (12)
Wercklea woodsonii (19)
Wettinia donosoensis (1)
Wimmeria montana (16)
Wimmeria sternii (14)
Xylopia panamensis (6)
Zanthoxylum apiculatum (8)
Zanthoxylum harmsianum (8)
Zanthoxylum pucro (8)
Zapoteca mollis (9)
Ziziphus chloroxylon (4)
Zygia biflora (6)
Zygia brenesii (19)
Zygia confusa (9)
Zygia rubiginosa (7)

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Table S4. List of the 272 species included in the study and number of records per species. Classification according to the Angiosperm Phylogeny Website, URL www.mobot.org/MOBOT/research/APweb/), and current IUCN status (URL https://iconicspecies.iucnredlist.org/).

Order	Family	Species	Records	IUCN Status*
Clade: Conifers				
Pinales	Pinaceae	*Abies guatemalensis*	81	EN
		Abies hickelii	27	EN
		Pinus ayacahuite	104	LC
		Pinus hartwegii	160	LC
		Pinus lawsonii	48	LC
		Pinus pringlei	52	LC
		Pinus pseudostrobus	355	LC
		Pinus strobus	51	LC
	Podocarpaceae	*Podocarpus matudae*	106	VU
		Prumnopitys standleyi	20	EN
Clade: Basal Angiosperms				
Chloranthales	Chloranthaceae	*Hedyosmum mexicanum*	106	LC
Clade: Magnoliids				
Laurales	Lauraceae	*Aiouea brenesii*	23	NT
		Aiouea hammeliana	29	LC
		Beilschmiedia brenesii	32	NA
		Beilschmiedia mexicana	21	LC
		Beilschmiedia ovalis	23	NT
		Damburneya cufodontisii	60	LC
		Damburneya salicina	41	LC
		Ocotea acuminatissima	41	VU
		Ocotea austinii	39	LC
		Ocotea brenesii	26	LC
		Ocotea endresiana	35	LC
		Ocotea gomezii	20	LC
		Ocotea helicterifolia	98	NA
		Ocotea mollicella	23	NT
		Ocotea praetermissa	33	LC
		Ocotea psychotrioides	33	VU
		Ocotea purpurea	37	LC
		Persea donnell-smithii	33	VU
		Persea liebmannii	70	LC
		Persea schiedeana	82	EN

		Persea veraguasensis	23	DD
	Monimiaceae	*Mollinedia costaricensis*	61	LC
		Mollinedia viridiflora	236	LC
Magnoliales	Annonaceae	*Guatteria costaricensis*	42	LC
		Guatteria dolichopoda	77	LC
		Guatteria oliviformis	52	LC
		Guatteria slateri	22	LC
		Guatteria verrucosa	36	LC
		Stenanona costaricensis	31	LC
	Magnoliaceae	*Magnolia poasana*	36	NT
		Magnolia schiedeana	36	VU
		Magnolia sororum	29	NT
Piperales	Piperaceae	*Piper irazuanum*	47	NA
		Piper maxonii	20	NT
Clade: Basal Eudicots				
Proteales	Proteaceae	*Panopsis costaricensis*	46	LC
	Sabiaceae	*Meliosma alba*	29	LC
		Meliosma brenesii	32	LC
		Meliosma dentata	67	LC
		Meliosma idiopoda	89	LC
Clade: Pentapetalae				
Saxifragales	Hamamelidaceae	*Matudaea trinervia*	47	LC
Clade: Rosid I / Fabids				
Celastrales	Celastraceae	*Euonymus costaricensis*	50	LC
		Maytenus woodsonii	24	NT
		Monteverdia recondita	26	NT
		Zinowiewia integerrima	115	LC
		Zinowiewia rubra	21	VU
Malpighiales	Clusiaceae	*Chrysochlamys allenii*	40	LC
		Chrysochlamys psychotriifolia	45	LC
		Clusia croatii	88	LC
		Clusia palmana	46	LC
		Clusia salvinii	121	LC
		Clusia torresii	40	LC
	Dichapetalaceae	*Dichapetalum brenesii*	26	LC
	Euphorbiaceae	*Croton megistocarpus*	30	LC
	Hypericaceae	*Hypericum irazuense*	36	NT
	Malpighiaceae	*Bunchosia ternata*	25	NT
	Passifloraceae	*Passiflora tica*	30	LC
	Salicaceae	*Hasseltia guatemalensis*	38	LC
		Hasseltiopsis dioica	24	NT

		Macrohasseltia macroterantha	41	LC
Oxalidales	Brunelliaceae	*Brunellia costaricensis*	45	VU
		Brunellia mexicana	66	LC
	Cunoniaceae	*Weinmannia burserifolia*	35	LC
		Weinmannia wercklei	35	LC
	Elaeocarpaceae	*Sloanea ampla*	27	LC
Sapindales	Meliaceae	*Cedrela tonduzii*	41	LC
	Rutaceae	*Peltostigma guatemalense*	32	LC
		Peltostigma pteleoides	30	LC
		Stauranthus perforatus	24	NT
Clade: N-fixing clade				
Fabales	Fabaceae	*Cojoba costaricensis*	49	NT
		Inga exalata	21	LC
		Inga flexuosa	67	LC
		Inga leonis	32	LC
		Inga longispica	24	LC
		Inga mortoniana	44	LC
		Inga sierrae	29	LC
		Inga tonduzii	32	LC
		Inga xalapensis	27	LC
		Senna multifoliolata	38	VU
		Zygia palmana	37	NA
Fagales	Betulaceae	*Ostrya virginiana*	156	LC
	Fagaceae	*Quercus affinis*	53	LC
		Quercus benthamii	64	NT
		Quercus candicans	158	NA
		Quercus costaricensis	61	VU
		Quercus crispipilis	36	NT
		Quercus germana	36	LC
		Quercus lancifolia	109	LC
		Quercus martinezii	21	LC
		Quercus salicifolia	82	LC
		Quercus scytophylla	77	LC
		Quercus seemannii	72	LC
		Quercus xalapensis	124	LC
	Juglandaceae	*Alfaroa costaricensis*	47	LC
		Juglans pyriformis	29	EN
		Oreomunnea mexicana	44	LC
	Ticodendraceae	*Ticodendron incognitum*	66	NT
Myrtales	Melastomataceae	*Axinaea costaricensis*	36	LC
		Blakea storkii	54	LC

		Meriania phlomoides	38	LC
		Miconia albertobrenesii	60	LC
		Miconia biperulifera	21	LC
		Miconia brenesii	42	LC
		Miconia hemenostigma	33	VU
		Miconia oligotricha	31	LC
		Miconia pittieri	27	LC
		Miconia schnellii	30	LC
		Miconia tonduzii	138	LC
	Myrtaceae	*Eugenia austin-smithii*	51	LC
		Eugenia cartagensis	34	DD
		Eugenia siggersii	47	LC
		Myrcianthes storkii	21	LC
		Pimenta guatemalensis	21	LC
	Onagraceae	*Fuchsia paniculata*	343	LC
Rosales	Cannabaceae	*Lozanella enantiophylla*	93	LC
	Rhamnaceae	*Frangula discolor*	128	LC
		Frangula oreodendron	72	LC
		Frangula pendula	40	LC
	Rosaceae	*Prunus brachybotrya*	126	LC
		Prunus fortunensis	27	VU
		Prunus rhamnoides	72	LC
		Prunus tetradenia	25	LC
	Ulmaceae	*Ulmus mexicana*	79	LC
Clade: Rosid II / Malvids				
Caryophyllales	Polygonaceae	*Coccoloba liportizii*	24	NT
Huerteales	Dipentodontaceae	*Perrottetia longistylis*	74	LC
		Perrottetia multiflora	53	LC
		Perrottetia ovata	34	LC
Malvales	Malvaceae	*Chiranthodendron pentadactylon*	63	LC
		Malvaviscus palmanus	25	LC
		Phymosia rosea	25	LC
		Wercklea insignis	26	LC
		Wercklea woodsonii	20	LC
	Thymelaeaceae	*Daphnopsis mollis*	22	VU
Picramniales	Picramniaceae	*Picramnia teapensis*	94	LC
		Picramnia xalapensis	22	VU
Clade: Basal Asterids				
Cornales	Cornaceae	*Cornus disciflora*	259	LC
		Cornus excelsa	125	LC
Ericales	Actinidiaceae	*Saurauia leucocarpa*	28	VU

	Saurauia montana	134	LC
	Saurauia oreophila	25	LC
	Saurauia pittieri	46	LC
	Saurauia rubiformis	49	LC
	Saurauia scabrida	93	NT
	Saurauia serrata	44	NT
	Saurauia villosa	27	VU
Clethraceae	*Clethra gelida*	20	LC
	Clethra hartwegii	37	LC
	Clethra macrophylla	45	LC
	Clethra pringlei	28	LC
	Clethra suaveolens	60	LC
Ericaceae	*Comarostaphylis arbutoides*	88	LC
	Comarostaphylis longifolia	26	NT
	Gaultheria acuminata	88	LC
	Lyonia squamulosa	59	LC
	Vaccinium leucanthum	96	LC
Pentaphylacaceae	*Cleyera integrifolia*	70	LC
	Cleyera theaeoides	145	LC
	Freziera guatemalensis	22	LC
	Symplococarpon purpusii	89	LC
	Ternstroemia lineata	120	LC
Primulaceae	*Ardisia glandulosomarginata*	71	LC
	Ardisia palmana	59	LC
	Ardisia pleurobotrya	30	VU
	Ardisia verapazensis	30	LC
	Parathesis chiapensis	31	LC
	Parathesis glabra	40	LC
	Parathesis leptopa	22	LC
	Parathesis melanosticta	44	LC
	Parathesis villosa	26	LC
Sapotaceae	*Pouteria exfoliata*	20	NT
Styracaceae	*Styrax glabrescens*	145	LC
	Styrax radians	24	NT
	Styrax ramirezii	74	NA
	Styrax warscewiczii	74	LC
Symplocaceae	*Symplocos citrea*	72	LC
	Symplocos costaricana	33	LC
	Symplocos limoncillo	77	LC
	Symplocos pycnantha	58	LC
	Symplocos serrulata	60	LC

Clade: Asterid I / Lamids				
Aquifoliales	Aquifoliaceae	*Ilex discolor*	117	LC
		Ilex lamprophylla	65	LC
		Ilex maxima	24	LC
		Ilex pallida	60	LC
	Phyllonomaceae	*Phyllonoma laticuspis*	105	LC
Boraginales	Heliotropiaceae	*Tournefortia johnstonii*	30	LC
Gentianales	Apocynaceae	*Lacmellea zamorae*	20	NT
		Rauvolfia aphlebia	42	LC
		Tabernaemontana alfaroi	66	LC
		Vallesia aurantiaca	31	LC
	Rubiaceae	*Amaioua pedicellata*	29	NT
		Arachnothryx aspera	20	NT
		Arachnothryx brenesii	20	NT
		Arachnothryx costaricensis	30	NT
		Arachnothryx laniflora	28	LC
		Arachnothryx monteverdensis	27	NT
		Chiococca phaenostemon	70	LC
		Cosmibuena valerii	48	LC
		Coussarea caroliana	66	LC
		Deppea grandiflora	101	LC
		Faramea ovalis	29	LC
		Glossostipula concinna	58	LC
		Gonzalagunia rosea	103	LC
		Gonzalagunia stenostachya	32	LC
		Hoffmannia amplexifolia	27	NA
		Hoffmannia arborescens	65	LC
		Ladenbergia brenesii	51	LC
		Palicourea adusta	55	NA
		Palicourea alajuelensis	28	LC
		Palicourea eurycarpa	37	LC
		Palicourea galeottiana	145	LC
		Palicourea lasiorrhachis	136	LC
		Palicourea purpurea	70	LC
		Palicourea salicifolia	34	VU
		Psychotria jimenezii	31	LC
		Psychotria nubiphila	60	LC
		Psychotria orosiana	41	LC
		Rogiera amoena	152	LC
		Rogiera cordata	54	LC
		Rudgea reducticalyx	45	LC

		Sommera arborescens	61	LC
		Sommera donnell-smithii	108	LC
Lamiales	Bignoniaceae	*Amphitecna sessilifolia*	41	LC
	Lamiaceae	*Aegiphila odontophylla*	56	LC
	Scrophulariaceae	*Buddleja nitida*	70	LC
		Buddleja skutchii	44	LC
	Verbenaceae	*Citharexylum macradenium*	31	LC
		Duranta costaricensis	30	LC
Metteniusales	Metteniusaceae	*Oecopetalum mexicanum*	25	LC
Solanales	Solanaceae	*Cestrum poasanum*	40	NT
		Cestrum rugulosum	47	LC
		Solanum nigricans	158	LC
		Solanum storkii	39	NA
Clade: Asterid II / Campanulids				
Apiales	Araliaceae	*Dendropanax capillaris*	26	LC
		Dendropanax globosus	23	LC
		Dendropanax gonatopodus	41	LC
		Dendropanax leptopodus	24	LC
		Dendropanax querceti	67	LC
		Oreopanax echinops	47	LC
		Oreopanax oerstedianus	61	LC
		Oreopanax pycnocarpus	23	NT
		Oreopanax sanderianus	31	LC
		Oreopanax vestitus	32	LC
		Oreopanax xalapensis	304	LC
		Schefflera rodriguesiana	50	LC
Asterales	Asteraceae	*Clibadium leiocarpum*	96	NA
		Jessea multivenia	50	LC
		Lepidaploa polypleura	31	NT
		Montanoa guatemalensis	34	LC
		Roldana lanicaulis	30	NA
		Stevia microchaeta	43	NA
		Telanthophora cobanensis	35	NT
		Telanthophora grandifolia	188	LC
		Telanthophora uspantanensis	37	NT
		Verbesina apleura	29	LC
		Verbesina perymenioides	57	LC
Cardiopteridales	Cardiopteridaceae	*Citronella costaricensis*	28	LC
Dipsacales	Viburnaceae	*Viburnum acutifolium*	69	LC
		Viburnum costaricanum	179	LC
		Viburnum elatum	42	LC

	Viburnum jucundum	27	VU
	Viburnum microcarpum	20	LC
	Viburnum venustum	42	LC

* Applicable IUCN categories: DD (data deficient), LC (least concern), NT (near threatened), VU (vulnerable) and EN (endangered). The conservation status was not available (NA) for 12 species.

Table S5. Number of species and records per country and subnational division. Most species extent over more than one jurisdiction, thus the total number of species does not correspond to the addition of species per country. 37 records extended beyond our study region into Colombia, which we retained to avoid creating an artificial boundary along political lines.

Country	State	No. spp.	No. records
Costa Rica	N/A	193	6235
El Salvador	N/A	48	125
Guatemala	N/A	97	469
Honduras	N/A	82	343
Nicaragua	N/A	69	423
Panama	N/A	164	1400
Mexico	ALL STATES	155	6523
	Ciudad de México	*15*	*37*
	Chiapas	*132*	*1834*
	Durango	*13*	*32*
	(Estado de) México	*44*	*307*
	Guerrero	*68*	*304*
	Guanajuato	*6*	*9*
	Hidalgo	*54*	*332*
	Jalisco	*49*	*245*
	Michoacán	*43*	*424*
	Morelos	*15*	*22*
	Nayarit	*22*	*76*
	Nuevo León	*12*	*65*
	Oaxaca	*120*	*1215*
	Puebla	*72*	*362*
	Queretaro	*24*	*56*
	Sinaloa	*19*	*40*
	San Luis Potosí	*26*	*78*
	Sonora	*7*	*38*
	Tamaulipas	*26*	*98*
	Tlaxcala	*5*	*16*
	Veracruz	*92*	*795*
	Other states	*60*	*138*
Other (Colombia)*	N/A	N/A	37
TOTAL	N/A	272	15,555

* Colombia is not included in our study, but records that fell within the southern buffer of my study area on montane areas in its territory were kept to avoid introducing unnatural political barriers to our analysis.

Table S6. Post-SDM Analysis – Species Classification

	Broad altitudinal classes	
Class name	**Class code**	**Altitudinal criteria**
Very low	**VL**	<1,000 m asl
Low	**L**	median <1,000 m asl, but interquartile range extending above 1,000 m asl
Mid-low	**ML**	1,000–1,500 m asl
Middle*	**M**	1,000–2,000 m asl
Mid-high	**MH**	1,500–2,000 m asl
High	**H**	median >2,000 m asl, but interquartile range extending below 2,000 m asl
Very high	**VH**	>2,000 m asl

* Only 6 species had projected suitable habitat neatly within the 1,000–2,000 m asl; most species in the mid-elevation categories were skewed toward lower or upper elevations.

Table S7. Bray-Curtis indices at the topographic discontinuities of the Isthmus of Tehuantepec and Lake Cocibolca.

Discontinuity	**Isthmus of Tehuantepec**		**Lake Cocibolca**	
Latitudinal break (° N)	>16.5	<16.5	>12	<12
Number of records	5,581	9,974	7,880	7,675
Total No. of species	151	264	178	199
Exclusive species	8	121	73	94
Bray-Curtis index	*0.690*		*0.799*	

Table S8. SDM current and future projections in pixels and km². Projected number of pixels (30 arc-seconds resolution) and equivalent area (km²) suitable for TMF tree species under current (1981–2010 baseline) and future (2071–2100) climate conditions under RCP8.5.

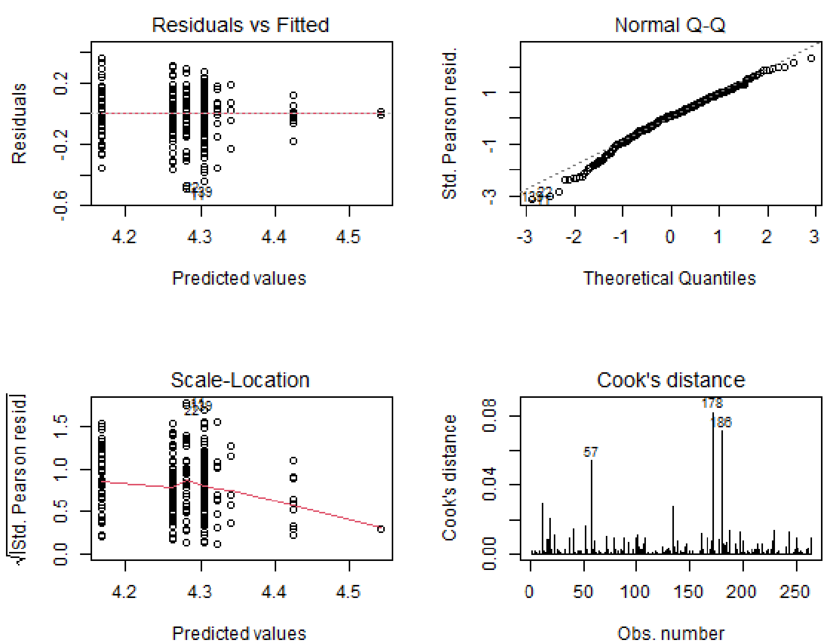
Number of species	Number of pixels			Area (km²)		
	Current projection	Future projection (RCP8.5)	Change (%)	Current projection	Future projection (RCP8.5)	Change (%)
1 to 5	654,024	768,470	17.5	548,061	648,145	18.3
6 to 10	214,302	136,563	-36.3	180,001	115,277	-36.0
11 to 15	127,403	65,227	-48.8	107,675	54,626	-49.3
16 to 20	84,506	36,508	-56.8	71,095	30,695	-56.8
21 to 30	105,358	45,299	-57.0	88,590	38,299	-56.8
31 to 40	70,293	28,349	-59.7	58,884	24,094	-59.1
41 to 50	53,737	21,166	-60.6	45,253	17,695	-60.9
51 to 75	91,030	35,479	-61.0	76,179	29,650	-61.1
76 to 100	48,281	8,489	-82.4	39,960	6,963	-82.6
101 to 150	21,140	5,170	-75.5	18,217	4,420	-75.7
>150	2,264	252	-88.9	1,843	177	-90.4
Total	1,472,338	1,150,972	-21.8	1,235,758	970,041	-21.5

Post-SDM Analysis - Results

1. *GLM: Percent losses (‘PercLoss’) by latitude (‘Dist2’) and elevation (‘Elev2’), Gamma family. Outliers removed n = 8.*

```
Call:
glm(formula = PercLoss ~ Dist2 + Elev2 + Dist2:Elev2, family = Gamma(link = "log"),
    data = postSDM[-c(77, 105, 108, 115, 146, 148, 227, 256),])

Deviance Residuals:
     Min        1Q    Median        3Q       Max
-0.60927  -0.09119   0.01589   0.10303   0.32660

Coefficients: (1 not defined because of singularities)
                 Estimate Std. Error t value Pr(>|t|)
(Intercept)      4.364864   0.045592  95.737   <2e-16 ***
Dist2.L         -0.027952   0.072496  -0.386   0.7001
Dist2.Q         -0.021375   0.021128  -1.012   0.3126
Elev2.L          0.006898   0.069539   0.099   0.9211
Elev2.Q          0.146779   0.056745   2.587   0.0102 *
Dist2.L:Elev2.L  0.229374   0.151534   1.514   0.1313
Dist2.Q:Elev2.L -0.165785   0.138158  -1.200   0.2313
Dist2.L:Elev2.Q -0.153452   0.088436  -1.735   0.0839 .
Dist2.Q:Elev2.Q        NA         NA      NA       NA
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

(Dispersion parameter for Gamma family taken to be 0.02491416)

    Null deviance: 8.0871  on 263  degrees of freedom
Residual deviance: 7.0322  on 256  degrees of freedom
AIC: 2061.8

Number of Fisher Scoring iterations: 4
```

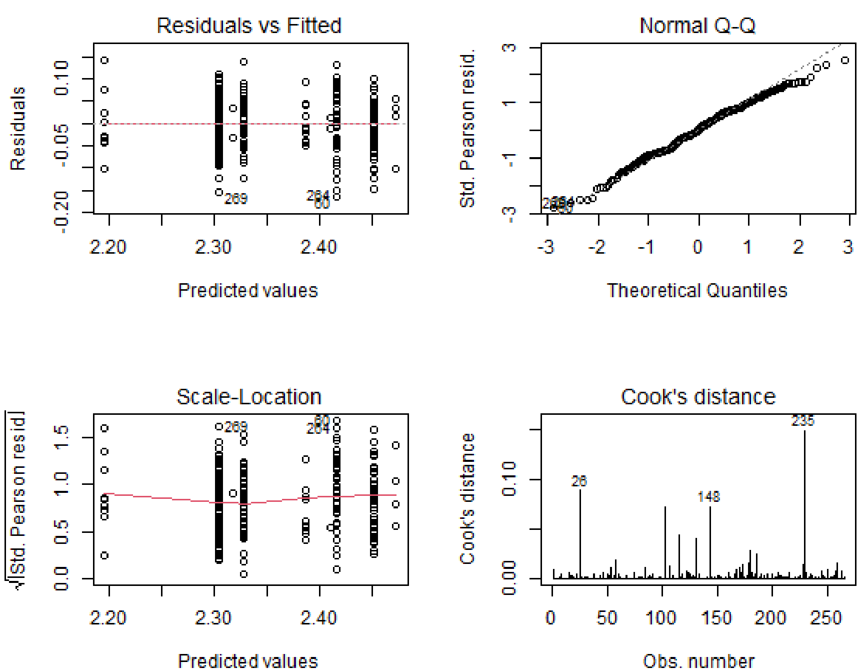
```
Bartlett test of homogeneity of variances

data:  residuals(model.gm1) by fitted(model.gm1)
Bartlett's K-squared = 12.201, df = 7, p-value = 0.09413
```



2. *GLM: Absolute losses ('log(Loss)') by latitude ('Dist2') and elevation ('Elev2'), Gamma family. Outliers removed n = 7.*

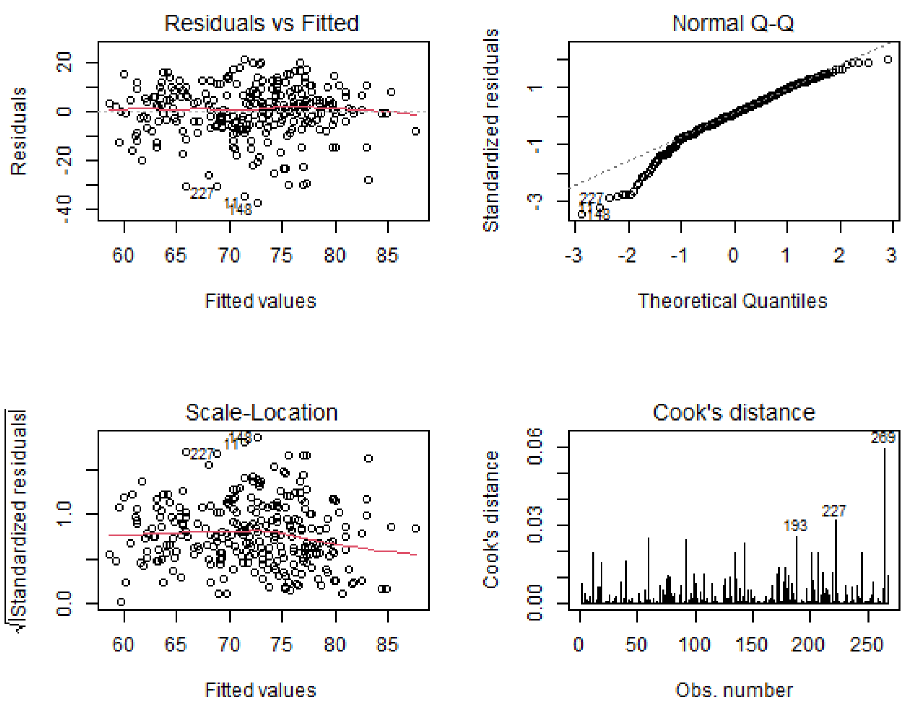
```
Call:
glm(formula = log(Loss) ~ Dist2 + Elev2 + Dist2 + Dist2:Elev2,
    family = Gamma(link = "log"), data = postSDM[-c(25, 57, 108, 115, 146, 207, 271),])

Deviance Residuals:
      Min         1Q     Median         3Q        Max
-0.176540  -0.043774   0.002867   0.042320   0.136304

Coefficients:
                 Estimate Std. Error t value Pr(>|t|)
(Intercept)      2.365127   0.008197 288.542  < 2e-16 ***
Dist2.L         -0.068581   0.012247  -5.600 5.52e-08 ***
Dist2.Q         -0.097465   0.015910  -6.126 3.38e-09 ***
Elev2.L         -0.029808   0.017098  -1.743 0.082477 .
Elev2.Q         -0.031700   0.010525  -3.012 0.002857 **
Dist2.L:Elev2.L -0.100109   0.025431  -3.936 0.000107 ***
Dist2.Q:Elev2.L  0.017574   0.033276   0.528 0.597865
Dist2.L:Elev2.Q  0.011787   0.015910   0.741 0.459463
Dist2.Q:Elev2.Q -0.028316   0.020286  -1.396 0.163986
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

(Dispersion parameter for Gamma family taken to be 0.003566953)

    Null deviance: 2.14678  on 264  degrees of freedom
Residual deviance: 0.92643  on 256  degrees of freedom
AIC: 524

Number of Fisher Scoring iterations: 3
```

```
Bartlett test of homogeneity of variances

data:  residuals(model.gm2) by fitted(model.gm2)
Bartlett's K-squared = 10.387, df = 8, p-value = 0.2389
```



3. Linear regression: Percent loss ('PercLoss') by projected current suitable areas ('log(CurrentSize)'). Outliers removed n = 5.

```
Call:
lm(formula = PercLoss ~ log(CurrentSize), data = postSDM[-c(22, 105, 108, 115, 146), ])

Residuals:
    Min      1Q  Median      3Q     Max 
-37.480  -5.052   0.793   7.167  21.569 

Coefficients:
                 Estimate Std. Error t value Pr(>|t|)    
(Intercept)      136.2097     7.1485  19.054   <2e-16 ***
log(CurrentSize)  -5.8485     0.6484  -9.019   <2e-16 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 10.8 on 265 degrees of freedom
Multiple R-squared:  0.2349,	Adjusted R-squared:  0.232 
F-statistic: 81.35 on 1 and 265 DF,  p-value: < 2.2e-16
```



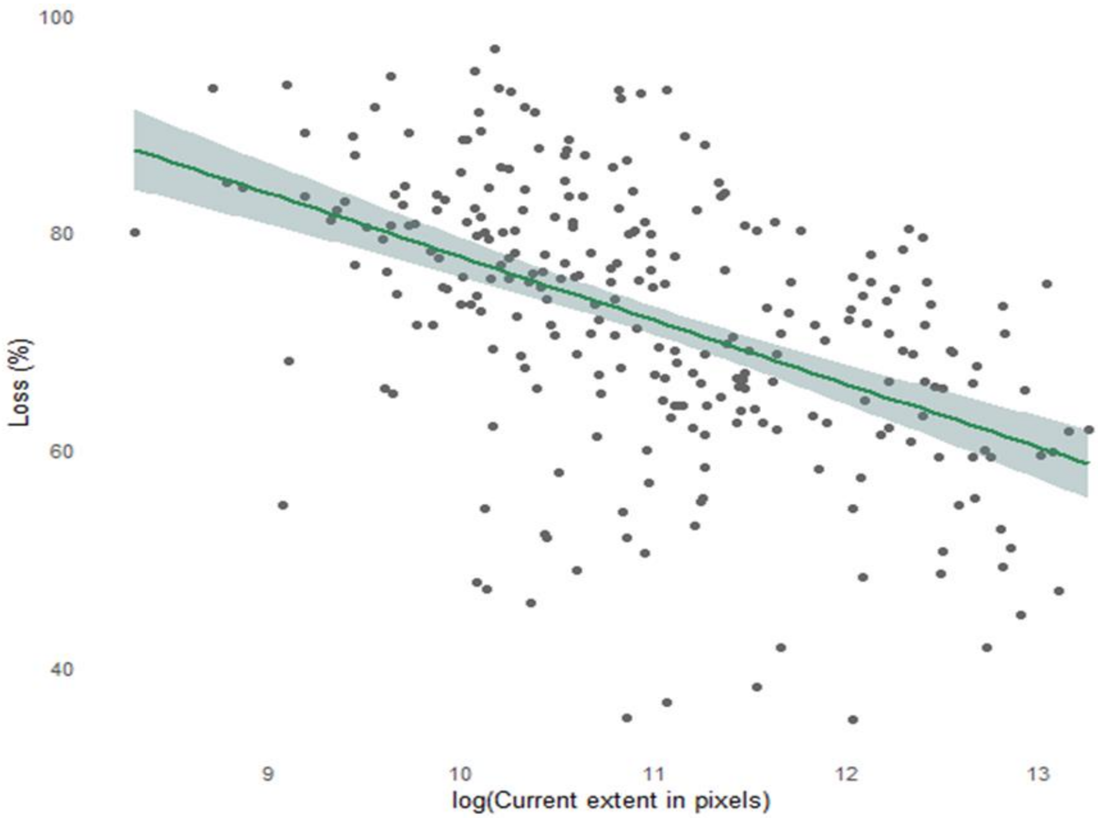


Fig. S1. Linear regression of projected range losses (%) in response to current extent by the end of the century under RCP8.5.

Appendix C. Supplementary Information for Chapter 4

Table S9. Functional trait data for the 272 species included in the analysis before imputation. Height, maximum height, diameter at breast height (DBH), leaf area (LA) and specific leaf area (SLA) were obtained from the TRY (Kattge et al., 2020) and BIEN v4.2 databases (Maitner et al., 2018). Data for height and DBH were complemented with records from scanned herbaria specimens available in GBIF (www.gbif.org; consulted March-April 2024). Wood density (WD) data was obtained from the Global Wood Density Database (Zanne, 2009) and Ordóñez Díaz et al. (2015). For LA, SLA and WD, species-specific data is shown in black, whereas data available at genus level is shown in red; this approach was necessary before imputation to reduce the number of genera without a single datum for any of those traits.

Species	Height (m)	Max. height (m)	DBH (cm)	LA (mm^2)	SLA (mm^2 mg^{-1})	Wood density (g cm^{-3})
Abies guatemalensis	30 to 45	45	100 to 150	59.3	8.4	0.43
Abies hickelii	20 to 30	30	100 to 130	59.3	8.4	0.38
Aegiphila odontophylla	8	8	20	8,645.4	22.6	0.66
Aiouea brenesii	8	20	30	10,661.7	13.6	0.37
Aiouea hammeliana	12 to 15	15	45	10,661.7	13.6	0.37
Alfaroa costaricensis	12 to 27	27	77	NA	NA	0.51
Amaioua pedicellata	10 to 12	12	22	NA	NA	0.62
Amphitecna sessilifolia	9 to 12	12	20	6,200.4	10.8	0.43
Arachnothryx aspera	3 to 8	8	NA	NA	NA	0.56
Arachnothryx brenesii	4 to 6	18	12	NA	NA	0.56
Arachnothryx costaricensis	10 to 15	15	NA	NA	NA	0.56
Arachnothryx laniflora	2 to 8	8	NA	NA	NA	0.56
Arachnothryx monteverdensis	5	5	NA	NA	NA	0.56
Ardisia glandulosomarginata	4 to 8	8	NA	4,651.8	15.1	0.565
Ardisia palmana	5 to 15	15	NA	4,651.8	15.1	0.565
Ardisia pleurobotrya	6 to 8	8	20 to 35	4,651.8	15.1	0.565
Ardisia verapazensis	5 to 15	20	NA	4,651.8	15.1	0.565
Axinaea costaricensis	9 to 12	12	15	NA	13.0	0.53
Beilschmiedia brenesii	18 to 25	25	100	13,773.4	17.1	0.54
Beilschmiedia mexicana	10 to 20	20	20 to 30	13,773.4	17.1	0.54
Beilschmiedia ovalis	8 to 30	30	25 to 50	8,560.0	8.6	0.54
Blakea storkii	9	9	15	7,263.0	12.8	0.71
Brunellia costaricensis	6 to 10	10	35 to 47	NA	NA	0.32
Brunellia mexicana	20 to 25	25	40	NA	NA	0.32
Buddleja nitida	4.5	4.5	NA	1,067.1	5.7	0.52
Buddleja skutchii	3 to 10	15	30	1,951.7	7.4	0.52
Bunchosia ternata	2.5 to 7	7	NA	11,549.9	15.9	0.65
Cedrela tonduzii	6 to 8	8	16 to 25	2,443.0	28.5	0.36
Cestrum poasanum	2 to 5	5	NA	4,511.0	27.5	0.5
Cestrum rugulosum	3	3	NA	4,511.0	27.5	0.5
Chiococca phaenostemon	5 to 8	8	NA	NA	NA	NA
Chiranthodendron pentadactylon	30	30	200	NA	NA	0.44
Chrysochlamys allenii	10 to 12	12	20	13,397.1	17.0	0.43
Chrysochlamys psychotriifolia	5 to 10	10	NA	13,397.1	17.0	0.43

Citharexylum macradenium	9 to 15	15	NA	4,700.0	9.6	0.645
Citronella costaricensis	9 to 18	18	40 to 50	NA	5.6	0.49
Clethra gelida	6 to 10	10	25	2,453.1	4.7	0.53
Clethra hartwegii	10	30	30 to 80	4,535.0	8.3	0.53
Clethra macrophylla	12 to 25	25	NA	1,805.8	6.4	0.53
Clethra pringlei	15 to 22	22	30 to 50	1,805.8	6.4	0.53
Clethra suaveolens	20 to 30	30	100	1,805.8	6.4	0.53
Cleyera integrifolia	15	15	NA	1,992.5	8.9	0.64
Cleyera theaeoides	6 to 12	12	NA	2,746.2	8.2	0.63
Clibadium leiocarpum	6	12	NA	NA	NA	NA
Clusia croatii	5 to 25	25	NA	6,681.3	11.1	0.65
Clusia palmana	10	10	NA	8,615.0	9.2	0.65
Clusia salvinii	3 to 12	12	NA	6,681.3	4.6	0.65
Clusia torresii	8 to 12	12	NA	6,681.3	11.1	0.65
Coccoloba liportizii	4 to 6	6	15	16,040.9	21.3	0.62
Cojoba costaricensis	15 to 20	20	15	NA	23.5	NA
Comarostaphylis arbutoides	1 to 4	20	NA	1,040.7	3.3	NA
Comarostaphylis longifolia	1 to 5	5	NA	NA	3.4	NA
Cornus disciflora	23	23	25 to 50	2,722.0	28.8	0.58
Cornus excelsa	12	12	15	2,722.0	28.8	0.57
Cosmibuena valerii	5 to 6	6	15	NA	4.8	NA
Coussarea caroliana	3 to 5	5	NA	4,645.0	18.0	0.66
Croton megistocarpus	11 to 18	18	25 to 45	5,570.4	20.3	0.76
Damburneya cufodontisii	10 to 15	15	NA	NA	NA	NA
Damburneya salicina	7	7	38	NA	NA	NA
Daphnopsis mollis	5 to 8	15	15	NA	17.8	0.52
Dendropanax capillaris	4 to 8	30	NA	10,004.6	17.4	0.41
Dendropanax globosus	7 to 13	13	10	10,004.6	17.4	0.41
Dendropanax gonatopodus	10 to 30	30	12 to 58	10,004.6	17.4	0.41
Dendropanax leptopodus	5 to 9	9	10	10,004.6	17.4	0.41
Dendropanax querceti	5 to 8	8	10	3,166.0	19.3	0.41
Deppea grandiflora	3 to 5	5	NA	NA	NA	NA
Dichapetalum brenesii	4 to 9	9	20	3,573.0	18.3	0.761
Duranta costaricensis	3 to 4	4	NA	NA	11.7	0.56
Eugenia austin-smithii	8	8	12	4,545.8	14.4	0.86
Eugenia cartagensis	6 to 7	7	25	4,545.8	14.4	0.86
Eugenia siggersii	4 to 12	12	25	4,545.8	14.4	0.86
Euonymus costaricensis	3 to 10	10	NA	1,103.6	13.4	0.521
Faramea ovalis	6	6	NA	13,284.0	17.7	0.62
Frangula discolor	4 to 15	15	NA	4,157.2	26.1	NA
Frangula oreodendron	3 to 6	6	NA	2,759.2	11.0	NA
Frangula pendula	6 to 10	10	NA	4,157.2	26.1	NA
Freziera guatemalensis	12	12	NA	NA	NA	0.58
Fuchsia paniculata	10	10	NA	NA	21.6	0.56
Gaultheria acuminata	0.5 to 7	7	NA	2,877.5	6.9	NA
Glossostipula concinna	3 to 15	30	NA	NA	NA	NA
Gonzalagunia rosea	2 to 4	4	3	NA	NA	NA
Gonzalagunia stenostachya	2 to 3	9	NA	NA	NA	NA
Guatteria costaricensis	6 to 10	10	20	10,538.1	13.4	0.495
Guatteria dolichopoda	5 to 15	15	10	10,538.1	13.4	0.495
Guatteria oliviformis	9	9	15	10,538.1	13.4	0.495
Guatteria slateri	12	12	30	10,538.1	13.4	0.495
Guatteria verrucosa	8 to 12	12	16 to 35	10,538.1	13.4	0.495

Hasseltia guatemalensis	20	20	NA	6,880.7	14.2	0.575
Hasseltiopsis dioica	10	10	NA	NA	NA	NA
Hedyosmum mexicanum	20	20	9	5,753.0	21.6	0.45
Hoffmannia amplexifolia	1.5 to 4	4	NA	NA	NA	NA
Hoffmannia arborescens	4	4	NA	NA	NA	NA
Hypericum irazuense	2.5	2.5	NA	6,580.5	16.7	0.98
Ilex discolor	2 to 20	30	NA	5,096.9	11.2	0.575
Ilex lamprophylla	25	25	16 to 25	5,096.9	11.2	0.575
Ilex maxima	5 to 12	12	NA	5,096.9	11.2	0.575
Ilex pallida	6 to 10	10	NA	5,096.9	11.2	0.575
Inga exalata	9 to 12	12	10	19,076.2	14.7	0.58
Inga flexuosa	6 to 20	20	12 to 25	19,076.2	14.7	0.58
Inga leonis	10	10	25	19,076.2	14.7	0.58
Inga longispica	10 to 20	20	20 to 53	19,076.2	14.7	0.58
Inga mortoniana	7 to 15	15	10 to 25	19,076.2	14.7	0.58
Inga sierrae	20	20	NA	19,076.2	14.7	0.58
Inga tonduzii	8	8	20	19,076.2	14.7	0.58
Inga xalapensis	5 to 20	20	NA	19,076.2	14.7	0.58
Jessea multivenia	2.5 to 5	5	NA	NA	NA	NA
Juglans pyriformis	25 to 30	30	90	3,147.1	37.5	0.5
Lacmellea zamorae	3 to 8	8	25	1,968.3	19.7	0.5
Ladenbergia brenesii	6 to 12	12	8 to 28	NA	NA	0.49
Lepidaploa polypleura	5 to 10	10	NA	NA	NA	NA
Lozanella enantiophylla	6 to 10	10	12	NA	NA	NA
Lyonia squamulosa	3 to 6	6	NA	1,014.4	11.4	NA
Macrohasseltia macroterantha	13 to 37	37	70 to 90	NA	NA	NA
Magnolia poasana	7 to 15	15	20	6,040.4	10.6	0.45
Magnolia schiedeana	15 to 20	20	35 to 55	7,010.0	10.2	0.505
Magnolia sororum	6 to 20	20	25	6,040.4	10.6	0.47
Malvaviscus palmanus	4 to 20	20	NA	NA	NA	NA
Matudaea trinervia	8 to 35	35	5 to 40	NA	NA	NA
Maytenus woodsonii	2 to 4	4	NA	2,260.7	11.0	0.72
Meliosma alba	6 to 15	15	50	25,729.5	42.8	0.48
Meliosma brenesii	8 to 10	10	10 to 40	25,729.5	16.9	0.48
Meliosma dentata	8 to 22	22	NA	3,860.0	8.6	0.48
Meliosma idiopoda	5 to 20	20	NA	25,729.5	16.9	0.48
Meriania phlomoides	6 to 10	10	NA	NA	NA	0.49
Miconia albertobrenesii	3 to 5	5	NA	14,599.9	12.9	0.655
Miconia biperulifera	4	4	NA	14,599.9	12.9	0.655
Miconia brenesii	6 to 10	10	NA	14,599.9	12.9	0.655
Miconia hemenostigma	1 to 9	9	NA	14,599.9	12.9	0.655
Miconia oligotricha	2 to 4	4	NA	14,599.9	12.9	0.655
Miconia pittieri	2 to 5	5	NA	14,599.9	12.9	0.655
Miconia schnellii	4 to 15	15	32	2,022.5	8.2	0.655
Miconia tonduzii	2 to 10	10	NA	14,599.9	12.9	0.655
Mollinedia costaricensis	3 to 8	8	NA	12,269.6	18.3	NA
Mollinedia viridiflora	3 to 10	10	NA	4,711.9	21.5	NA
Montanoa guatemalensis	6 to 15	15	NA	NA	NA	NA
Monteverdia recondita	7 to 15	25	5 to 20	NA	NA	NA
Myrcianthes storkii	3 to 25	25	NA	640.0	14.2	0.76
Ocotea acuminatissima	5 to 10	20	NA	19,848.2	13.8	0.52
Ocotea austinii	15 to 16	16	35 to 49	19,848.2	13.8	0.48
Ocotea brenesii	4 to 8	8	10	19,848.2	13.8	0.51

Ocotea endresiana	12 to 23	23	10 to 80	19,848.2	13.8	0.52
Ocotea gomezii	7 to 8	8	5 to 20	19,848.2	13.8	0.52
Ocotea helicterifolia	3 to 8	15	8	5,470.0	11.2	0.52
Ocotea mollicella	3 to 20	20	50	19,848.2	13.8	0.52
Ocotea praetermissa	4 to 15	15	30	2,804.6	19.6	0.52
Ocotea psychotrioides	3 to 8	8	NA	19,848.2	13.8	0.52
Ocotea purpurea	18	18	38	19,848.2	13.8	0.52
Oecopetalum mexicanum	25 to 30	30	30	NA	NA	0.7
Oreomunnea mexicana	25 to 40	40	40 to 75	1,090.0	20.2	NA
Oreopanax echinops	15	15	17	6,150.7	22.3	0.5
Oreopanax oerstedianus	20	20	70	6,150.7	22.3	0.59
Oreopanax pycnocarpus	2 to 6	6	10 to 60	6,150.7	22.3	0.59
Oreopanax sanderianus	3 to 18	18	NA	6,150.7	22.3	0.59
Oreopanax vestitus	20	20	70	6,150.7	22.3	0.59
Oreopanax xalapensis	30	30	NA	6,050.9	20.8	0.59
Ostrya virginiana	25	25	50	2,311.1	37.1	0.5
Palicourea adusta	2	2	NA	11,245.8	19.5	0.55
Palicourea alajuelensis	5 to 8	8	10 to 25	11,245.8	19.5	0.55
Palicourea eurycarpa	6 to 7	7	15	10,756.0	19.2	0.55
Palicourea galeottiana	1 to 8	8	NA	11,245.8	19.5	0.55
Palicourea lasiorrhachis	2 to 4	4	NA	11,245.8	19.5	0.55
Palicourea purpurea	2 to 8	8	NA	11,245.8	19.5	0.55
Palicourea salicifolia	2 to 4	4	NA	11,245.8	19.5	0.55
Panopsis costaricensis	15	15	40	NA	NA	0.51
Parathesis chiapensis	3 to 6	6	NA	NA	NA	0.61
Parathesis glabra	3 to 5	5	NA	NA	NA	0.61
Parathesis leptopa	6 to 9	9	NA	NA	NA	0.61
Parathesis melanosticta	1 to 7	7	NA	NA	NA	0.61
Parathesis villosa	4 to 8	8	NA	NA	NA	0.61
Passiflora tica	2 to 6	6	NA	NA	NA	NA
Peltostigma guatemalense	4 to 6	6	8	NA	NA	NA
Peltostigma pteleoides	3 to 10	20	NA	NA	NA	NA
Perrottetia longistylis	5 to 15	15	NA	NA	17.6	0.71
Perrottetia multiflora	5 to 8	8	NA	NA	17.6	0.71
Perrottetia ovata	6 to 10	10	15	NA	17.6	0.71
Persea donnell-smithii	6 to 15	20	NA	1,856.7	8.0	0.52
Persea liebmannii	3 to 15	27	50	1,856.7	8.0	0.52
Persea schiedeana	4 to 15	25	NA	1,856.7	8.0	0.5
Persea veraguasensis	3 to 15	15	40	1,856.7	8.0	0.52
Phyllonoma laticuspis	4 to 15	15	NA	NA	NA	NA
Phymosia rosea	4 to 10	10	NA	NA	NA	NA
Picramnia teapensis	3 to 7	7	15	10,223.5	17.3	NA
Picramnia xalapensis	3 to 10	10	NA	10,223.5	17.3	NA
Pimenta guatemalensis	9 to 12	12	15 to 20	NA	NA	0.82
Pinus ayacahuite	50	50	200	NA	NA	0.45
Pinus hartwegii	32	32	145	NA	NA	0.44
Pinus lawsonii	10 to 25	25	25 to 40	NA	NA	0.48
Pinus pringlei	20 to 27	27	45 to 54	NA	NA	0.46
Pinus pseudostrobus	45	45	100	NA	NA	0.55
Pinus strobus	18 to 46	46	20 to 102	227.5	11.6	0.38
Piper irazuanum	2 to 7	7	NA	15,477.8	22.6	0.39
Piper maxonii	3 to 5	5	NA	15,477.8	22.6	0.39
Podocarpus matudae	30	30	150	NA	5.6	0.48

Pouteria exfoliata	15 to 35	35	70 to 80	23,814.5	14.0	0.78
Prumnopitys standleyi	22	22	80	NA	6.8	0.53
Prunus brachybotrya	35	35	40	2,650.0	8.2	0.665
Prunus fortunensis	14	14	NA	5,700.8	12.6	0.665
Prunus rhamnoides	8 to 20	20	14	5,700.8	23.3	0.665
Prunus tetradenia	3 to 15	25	NA	5,700.8	12.6	0.665
Psychotria jimenezii	2 to 7	7	NA	7,368.2	16.2	0.53
Psychotria nubiphila	2 to 8	8	NA	7,368.2	16.2	0.53
Psychotria orosiana	2 to 3	3	NA	7,368.2	16.2	0.53
Quercus affinis	16	16	NA	1,312.5	8.9	0.58
Quercus benthamii	50	50	NA	2,045.5	10.2	0.575
Quercus candicans	25	25	NA	1,360.8	11.8	0.64
Quercus costaricensis	10 to 20	20	35 to 100	2,749.9	6.8	0.61
Quercus crispipilis	27	27	30 to 60	1,360.8	11.8	0.66
Quercus germana	12	12	NA	3,509.0	16.4	0.56
Quercus lancifolia	25	25	NA	1,360.8	11.8	0.575
Quercus martinezii	30	30	NA	1,360.8	11.8	0.575
Quercus salicifolia	6 to 25	25	100	1,360.8	19.4	0.67
Quercus scytophylla	20	20	NA	1,360.8	11.8	0.64
Quercus seemannii	10 to 15	15	12 to 18	1,990.0	8.7	0.575
Quercus xalapensis	30	30	NA	2,525.0	14.3	0.575
Rauvolfia aphlebia	5 to 9	9	NA	6,245.0	48.9	0.48
Rogiera amoena	1.5 to 10	10	18 to 20	NA	NA	NA
Rogiera cordata	2 to 10	10	NA	NA	NA	NA
Roldana lanicaulis	3	3	NA	NA	NA	NA
Rudgea reducticalyx	2.5	2.5	NA	7,099.0	20.9	0.57
Saurauia leucocarpa	10	10	NA	NA	NA	0.435
Saurauia montana	4 to 17	17	15	6,875.0	17.5	0.435
Saurauia oreophila	15	15	25	NA	17.5	0.435
Saurauia pittieri	5 to 15	15	NA	NA	17.5	0.435
Saurauia rubiformis	3 to 9	9	NA	NA	17.5	0.435
Saurauia scabrida	5 to 20	20	NA	NA	17.5	0.435
Saurauia serrata	6 to 15	15	NA	NA	17.5	0.435
Saurauia villosa	15	15	NA	NA	NA	0.435
Schefflera rodriguesiana	8 to 12	12	30	2,037.2	12.8	0.43
Senna multifoliolata	2 to 8	8	NA	4,050.5	22.7	0.56
Sloanea ampla	15 to 18	18	NA	18,590.5	13.4	0.81
Solanum nigricans	1 to 7	10	NA	10,224.4	20.7	0.42
Solanum storkii	2.5 to 10	10	NA	10,224.4	20.7	0.42
Sommera arborescens	2 to 10	10	NA	NA	NA	NA
Sommera donnell-smithii	7 to 15	15	10	NA	NA	NA
Stauranthus perforatus	5 to 6.5	6.5	NA	NA	NA	NA
Stenanona costaricensis	5 to 7	7	5 to 8	NA	NA	NA
Stevia microchaeta	3 to 4	4	NA	NA	NA	0.59
Styrax glabrescens	7	7	NA	9,218.0	13.7	0.44
Styrax radians	20	20	30	9,218.0	13.7	0.44
Styrax ramirezii	20 to 27	27	NA	9,218.0	13.7	0.44
Styrax warscewiczii	20 to 30	30	25	NA	NA	0.44
Symplococarpon purpusii	10 to 25	25	35 to 75	NA	20.1	NA
Symplocos citrea	15	15	NA	2,892.0	14.4	0.64
Symplocos costaricana	3 to 12	12	15 to 20	2,892.0	14.4	0.64
Symplocos limoncillo	12 to 20	20	NA	2,892.0	14.4	0.64
Symplocos pycnantha	9 to 12	12	10 to 35	2,892.0	14.4	0.64

Symplocos serrulata	7 to 10	10	39	4,184.2	6.7	0.64
Tabernaemontana alfaroi	2 to 5	5	NA	6,544.8	34.1	0.56
Telanthophora cobanensis	1 to 9	9	NA	NA	NA	NA
Telanthophora grandifolia	3 to 7	7	NA	NA	NA	NA
Telanthophora uspantanensis	2 to 5	5	10	NA	NA	NA
Ternstroemia lineata	10 to 20	20	NA	NA	NA	0.62
Ticodendron incognitum	25	25	35	2,230.0	8.8	NA
Tournefortia johnstonii	3 to 5	5	NA	NA	NA	0.47
Ulmus mexicana	50 to 70	80	NA	4,216.2	27.4	0.38
Vaccinium leucanthum	4 to 12	12	NA	456.6	14.9	0.291
Vallesia aurantiaca	2 to 6	20	NA	NA	NA	NA
Verbesina apleura	3 to 6	6	NA	NA	NA	0.437
Verbesina perymenioides	1 to 7	7	NA	NA	NA	0.437
Viburnum acutifolium	3 to 6	6	NA	2,485.4	13.3	0.54
Viburnum costaricanum	7	7	NA	1,133.7	11.4	0.54
Viburnum elatum	2 to 8	8	NA	2,485.4	17.5	0.54
Viburnum jucundum	3 to 6	6	NA	2,485.4	17.5	0.54
Viburnum microcarpum	3 to 7	7	NA	2,485.4	17.5	0.54
Viburnum venustum	2 to 7	7	8 to 10	2,002.0	16.0	0.54
Weinmannia burserifolia	12	12	60	NA	9.5	0.615
Weinmannia wercklei	10 to 15	15	NA	NA	NA	0.49
Wercklea insignis	10	10	30 to 40	NA	NA	0.24
Wercklea woodsonii	6 to 20	20	NA	NA	NA	0.24
Zinowiewia integerrima	13	13	40	1,525.0	9.1	0.71
Zinowiewia rubra	10 to 20	20	30 to 60	1,525.0	9.1	0.71
Zygia palmana	8	8	20	17,707.0	14.9	0.83

Table S10. Imputed functional traits for 272 species included in the analysis, following the data imputation process described in (Carmona et al., 2024) with the *V.PhyloMaker* package (Jin and Qian, 2022). Leaf mass per area (LMA) was obtained as the inverse of specific leaf area (SLA).

Species	Mean height (m)	DBH (cm)	LA (mm^2)	LMA (mg^1 mm^2)	Wood density (g cm^{-3})
Abies guatemalensis	37.5	125	59.26	0.12	0.43
Abies hickelii	25.0	115	59.26	0.12	0.38
Aegiphila odontophylla	8.0	20	8,645.43	0.04	0.66
Aiouea brenesii	14.0	30	10,661.67	0.07	0.37
Aiouea hammeliana	13.5	45	10,661.67	0.07	0.37
Alfaroa costaricensis	19.5	77	3,222.43	0.05	0.51
Amaioua pedicellata	11.0	22	8,596.68	0.06	0.62
Amphitecna sessilifolia	10.5	20	6,200.40	0.09	0.43
Arachnothryx aspera	5.5	4	6,731.03	0.14	0.56
Arachnothryx brenesii	9.3	12	6,945.85	0.14	0.56
Arachnothryx costaricensis	12.5	4	6,786.89	0.14	0.56
Arachnothryx laniflora	5.0	4	6,731.03	0.14	0.56
Arachnothryx monteverdensis	5.0	4	6,731.03	0.14	0.56
Ardisia glandulosomarginata	6.0	12	4,651.78	0.07	0.57
Ardisia palmana	10.0	12	4,651.78	0.07	0.57
Ardisia pleurobotrya	7.0	27.5	4,651.78	0.07	0.57
Ardisia verapazensis	13.3	12	4,651.78	0.07	0.57
Axinaea costaricensis	10.5	15	8,930.29	0.08	0.53
Beilschmiedia brenesii	21.5	100	13,773.42	0.06	0.54
Beilschmiedia mexicana	15.0	25	13,773.42	0.06	0.54
Beilschmiedia ovalis	19.0	37.5	8,560.00	0.12	0.54
Blakea storkii	9.0	15	7,263.00	0.08	0.71
Brunellia costaricensis	8.0	41	5,825.59	0.07	0.32
Brunellia mexicana	22.5	40	6,176.70	0.07	0.32
Buddleja nitida	4.5	2.5	1,067.13	0.18	0.52
Buddleja skutchii	9.3	30	1,951.69	0.13	0.52
Bunchosia ternata	4.8	8.5	11,549.92	0.06	0.65
Cedrela tonduzii	7.0	20.5	2,443.00	0.04	0.36
Cestrum poasanum	3.5	4.9	4,511.00	0.04	0.50
Cestrum rugulosum	3.0	4.9	4,511.00	0.04	0.50
Chiococca phaenostemon	6.5	3.7	6,960.95	0.09	0.54
Chiranthodendron pentadactylon	30.0	200	7,630.41	0.08	0.44
Chrysochlamys allenii	15.0	20	13,397.11	0.06	0.43
Chrysochlamys psychotriifolia	7.5	11.4	13,397.11	0.06	0.43
Citharexylum macradenium	12.0	11.8	4,700.00	0.10	0.65
Citronella costaricensis	13.5	30	4,441.09	0.18	0.49
Clethra gelida	8.0	25	2,453.06	0.21	0.53
Clethra hartwegii	20.0	55	4,535.00	0.12	0.53
Clethra macrophylla	18.5	7.9	1,805.79	0.16	0.53
Clethra pringlei	18.5	40	1,805.79	0.16	0.53
Clethra suaveolens	36.7	100	1,805.79	0.16	0.53
Cleyera integrifolia	15.0	8.2	1,992.50	0.11	0.64
Cleyera theaeoides	9.0	8.2	2,746.23	0.12	0.63
Clibadium leiocarpum	6.0	4.5	4,213.99	0.07	0.53
Clusia croatii	15.0	10	6,681.26	0.09	0.65
Clusia palmana	10.0	10	8,614.99	0.11	0.65

Clusia salvinii	7.5	10	6,681.26	0.22	0.65
Clusia torresii	10.0	10	6,681.26	0.09	0.65
Coccoloba liportizii	5.0	15	16,040.85	0.05	0.62
Cojoba costaricensis	13.7	15	15,355.66	0.04	0.68
Comarostaphylis arbutoides	8.3	6.1	1,040.74	0.30	0.55
Comarostaphylis longifolia	3.0	6.1	1,639.02	0.30	0.58
Cornus disciflora	23.0	37.5	2,722.00	0.03	0.58
Cornus excelsa	12.0	15	2,722.00	0.03	0.57
Cosmibuena valerii	5.5	15	6,857.29	0.21	0.55
Coussarea caroliana	4.0	7.8	4,644.98	0.06	0.66
Croton megistocarpus	14.5	35	5,570.43	0.05	0.76
Damburneya cufodontisii	12.5	33.8	19,166.46	0.07	0.50
Damburneya salicina	7.0	38	19,008.31	0.07	0.51
Daphnopsis mollis	9.3	15	8,036.85	0.06	0.52
Dendropanax capillaris	14.0	17.5	10,004.58	0.06	0.41
Dendropanax globosus	10.0	10	10,004.58	0.06	0.41
Dendropanax gonatopodus	20.0	30	10,004.58	0.06	0.41
Dendropanax leptopodus	7.0	10	10,004.58	0.06	0.41
Dendropanax querceti	6.5	10	3,166.00	0.05	0.41
Deppea grandiflora	4.0	17.5	6,846.48	0.15	0.55
Dichapetalum brenesii	6.5	20	3,573.04	0.05	0.76
Duranta costaricensis	3.5	2.5	3,507.17	0.09	0.56
Eugenia austin-smithii	8.0	12	4,545.76	0.07	0.86
Eugenia cartagensis	6.5	25	4,545.76	0.07	0.86
Eugenia siggersii	8.0	25	4,545.76	0.07	0.86
Euonymus costaricensis	6.5	3.6	1,103.63	0.07	0.52
Faramea ovalis	6.0	10.3	13,283.96	0.06	0.62
Frangula discolor	9.5	4.9	4,157.23	0.04	0.48
Frangula oreodendron	4.5	4.9	2,759.24	0.09	0.48
Frangula pendula	8.0	4.9	4,157.23	0.04	0.48
Freziera guatemalensis	12.0	21	6,050.55	0.06	0.58
Fuchsia paniculata	10.0	3.6	4,159.47	0.05	0.56
Gaultheria acuminata	3.8	3	2,877.49	0.14	0.55
Glossostipula concinna	16.0	21.7	7,986.22	0.06	0.58
Gonzalagunia rosea	3.0	3	6,723.28	0.14	0.55
Gonzalagunia stenostachya	2.5	4.2	6,705.24	0.14	0.56
Guatteria costaricensis	8.0	20	10,538.08	0.07	0.50
Guatteria dolichopoda	10.0	10	10,538.08	0.07	0.50
Guatteria oliviformis	9.0	15	10,538.08	0.07	0.50
Guatteria slateri	12.0	30	10,538.08	0.07	0.50
Guatteria verrucosa	10.0	25.5	10,538.08	0.07	0.50
Hasseltia guatemalensis	20.0	19	6,880.67	0.07	0.58
Hasseltiopsis dioica	10.0	23.7	7,444.00	0.07	0.61
Hedyosmum mexicanum	20.0	9	5,753.00	0.05	0.45
Hoffmannia amplexifolia	2.8	3.3	6,679.12	0.15	0.55
Hoffmannia arborescens	4.0	3.3	6,723.28	0.15	0.55
Hypericum irazuense	2.5	1	6,580.50	0.06	0.98
Ilex discolor	17.3	8.8	5,096.95	0.09	0.58
Ilex lamprophylla	25.0	20.5	5,096.95	0.09	0.58
Ilex maxima	8.5	8.8	5,096.95	0.09	0.58
Ilex pallida	8.0	8.8	5,096.95	0.09	0.58
Inga exalata	10.5	10	19,076.18	0.07	0.58
Inga flexuosa	13.0	18.5	19,076.18	0.07	0.58

Inga leonis	10.0	25	19,076.18	0.07	0.58
Inga longispica	15.0	36.5	19,076.18	0.07	0.58
Inga mortoniana	11.0	17.5	19,076.18	0.07	0.58
Inga sierrae	20.0	32.5	19,076.18	0.07	0.58
Inga tonduzii	8.0	20	19,076.18	0.07	0.58
Inga xalapensis	12.5	32.5	19,076.18	0.07	0.58
Jessea multivenia	3.8	8.5	4,500.82	0.07	0.55
Juglans pyriformis	27.5	90	3,147.10	0.03	0.50
Lacmellea zamorae	5.5	25	1,968.27	0.05	0.50
Ladenbergia brenesii	9.0	18	8,646.89	0.08	0.49
Lepidaploa polypleura	7.5	2.8	3,506.38	0.08	0.50
Lozanella enantiophylla	8.0	12	4,950.97	0.05	0.48
Lyonia squamulosa	4.5	1.6	1,014.40	0.09	0.48
Macrohasseltia macroterantha	25.0	80	8,049.67	0.07	0.60
Magnolia poasana	11.0	20	6,040.43	0.09	0.45
Magnolia schiedeana	17.5	45	7,010.00	0.10	0.51
Magnolia sororum	13.0	25	6,040.43	0.09	0.47
Malvaviscus palmanus	12.0	3.2	8,889.88	0.06	0.40
Matudaea trinervia	21.5	22.5	8,071.52	0.05	0.50
Maytenus woodsonii	3.0	17.9	2,260.73	0.09	0.72
Meliosma alba	10.5	50	25,729.50	0.02	0.48
Meliosma brenesii	9.0	25	25,729.50	0.06	0.48
Meliosma dentata	15.0	37	3,860.00	0.12	0.48
Meliosma idiopoda	12.5	37	25,729.50	0.06	0.48
Meriania phlomoides	8.0	9.8	9,080.68	0.08	0.49
Miconia albertobrenesii	4.0	12.1	14,599.93	0.08	0.66
Miconia biperulifera	4.0	12.1	14,599.93	0.08	0.66
Miconia brenesii	8.0	12.1	14,599.93	0.08	0.66
Miconia hemenostigma	5.0	12.1	14,599.93	0.08	0.66
Miconia oligotricha	3.0	12.1	14,599.93	0.08	0.66
Miconia pittieri	3.5	12.1	14,599.93	0.08	0.66
Miconia schnellii	9.5	32	2,022.48	0.12	0.66
Miconia tonduzii	6.0	12.1	14,599.93	0.08	0.66
Mollinedia costaricensis	5.5	6.7	12,269.60	0.05	0.49
Mollinedia viridiflora	6.5	6.7	4,711.93	0.05	0.52
Montanoa guatemalensis	10.5	6.5	3,032.46	0.08	0.49
Monteverdia recondita	15.7	12.5	5,877.14	0.07	0.57
Myrcianthes storkii	14.0	11.2	640.00	0.07	0.76
Ocotea acuminatissima	11.7	33.7	19,848.21	0.07	0.52
Ocotea austinii	15.5	42	19,848.21	0.07	0.48
Ocotea brenesii	6.0	10	19,848.21	0.07	0.51
Ocotea endresiana	17.5	45	19,848.21	0.07	0.52
Ocotea gomezii	7.5	12.5	19,848.21	0.07	0.52
Ocotea helicterifolia	8.7	8	5,470.00	0.09	0.52
Ocotea mollicella	11.5	50	19,848.21	0.07	0.52
Ocotea praetermissa	9.5	30	2,804.58	0.05	0.52
Ocotea psychotrioides	5.5	33.7	19,848.21	0.07	0.52
Ocotea purpurea	18.0	38	19,848.21	0.07	0.52
Oecopetalum mexicanum	27.5	30	4,515.08	0.08	0.70
Oreomunnea mexicana	32.5	57.5	1,090.00	0.05	0.53
Oreopanax echinops	15.0	17	6,150.70	0.04	0.50
Oreopanax oerstedianus	20.0	70	6,150.70	0.04	0.59
Oreopanax pycnocarpus	4.0	35	6,150.70	0.04	0.59

Oreopanax sanderianus	10.5	7	6,150.70	0.04	0.59
Oreopanax vestitus	20.0	70	6,150.70	0.04	0.59
Oreopanax xalapensis	30.0	7	6,050.94	0.05	0.59
Ostrya virginiana	25.0	50	2,311.05	0.03	0.50
Palicourea adusta	2.0	6.1	11,245.83	0.05	0.55
Palicourea alajuelensis	6.5	17.5	11,245.83	0.05	0.55
Palicourea eurycarpa	6.5	15	10,756.00	0.05	0.55
Palicourea galeottiana	4.5	6.1	11,245.83	0.05	0.55
Palicourea lasiorrhachis	3.0	6.1	11,245.83	0.05	0.55
Palicourea purpurea	5.0	6.1	11,245.83	0.05	0.55
Palicourea salicifolia	3.0	6.1	11,245.83	0.05	0.55
Panopsis costaricensis	15.0	40	13,779.23	0.06	0.51
Parathesis chiapensis	4.5	6.2	2,526.22	0.11	0.61
Parathesis glabra	4.0	6.2	2,553.43	0.11	0.61
Parathesis leptopa	7.5	6.2	2,633.46	0.12	0.61
Parathesis melanosticta	4.0	6.2	2,553.43	0.11	0.61
Parathesis villosa	6.0	6.2	2,519.32	0.11	0.61
Passiflora tica	4.0	4.8	10,345.29	0.06	0.64
Peltostigma guatemalense	5.0	8	6,458.64	0.05	0.49
Peltostigma pteleoides	11.0	15. 8	6,221.35	0.05	0.49
Perrottetia longistylis	10.0	8.22	7,303.01	0.06	0.71
Perrottetia multiflora	6.5	18	6,929.39	0.06	0.71
Perrottetia ovata	8.0	11.61	7,125.74	0.06	0.71
Persea donnell-smithii	13.7	8.6	1,856.67	0.13	0.52
Persea liebmannii	15.0	50	1,856.67	0.13	0.52
Persea schiedeana	14.7	8.6	1,856.67	0.13	0.50
Persea veraguasensis	9.0	40	1,856.67	0.13	0.52
Phyllonoma laticuspis	9.5	3.3	3,952.23	0.10	0.54
Phymosia rosea	7.0	19.9	7,869.87	0.06	0.45
Picramnia teapensis	5.0	15	10,223.50	0.06	0.48
Picramnia xalapensis	6.5	14.6	10,223.50	0.06	0.48
Pimenta guatemalensis	10.5	17.5	3,812.80	0.07	0.82
Pinus ayacahuite	50.0	200	742.70	0.11	0.45
Pinus hartwegii	32.0	145	763.04	0.11	0.44
Pinus lawsonii	17.5	32.5	1,965.85	0.10	0.48
Pinus pringlei	23.5	49.5	1,192.47	0.10	0.46
Pinus pseudostrobus	45.0	100	1,263.16	0.10	0.55
Pinus strobus	32.0	61	227.45	0.09	0.38
Piper irazuanum	4.5	3.4	15,477.80	0.04	0.39
Piper maxonii	4.0	3.4	15,477.80	0.04	0.39
Podocarpus matudae	30.0	150	2,403.39	0.18	0.48
Pouteria exfoliata	25.0	75	23,814.47	0.07	0.78
Prumnopitys standleyi	22.0	80	3,314.63	0.15	0.53
Prunus brachybotrya	35.0	50	2,650.00	0.12	0.67
Prunus fortunensis	14.0	15.4	5,700.80	0.08	0.67
Prunus rhamnoides	14.0	14	5,700.80	0.04	0.67
Prunus tetradenia	14.3	15.4	5,700.80	0.08	0.67
Psychotria jimenezii	4.5	7.6	7,368.21	0.06	0.53
Psychotria nubiphila	5.0	7.6	7,368.21	0.06	0.53
Psychotria orosiana	2.5	7.6	7,368.21	0.06	0.53
Quercus affinis	16.0	23.7	1,312.49	0.11	0.58
Quercus benthamii	50.0	23.7	2,045.53	0.10	0.58
Quercus candicans	25.0	23.7	1,360.77	0.09	0.64

Quercus costaricensis	15.0	67.5	2,749.95	0.15	0.61
Quercus crispipilis	27.0	45	1,360.77	0.09	0.66
Quercus germana	12.0	23.7	3,508.97	0.06	0.56
Quercus lancifolia	25.0	23.7	1,360.77	0.09	0.58
Quercus martinezii	30.0	23.7	1,360.77	0.09	0.58
Quercus salicifolia	15.5	100	1,360.77	0.05	0.67
Quercus scytophylla	20.0	23.7	1,360.77	0.09	0.64
Quercus seemannii	12.5	15	1,990.00	0.11	0.58
Quercus xalapensis	30.0	23.7	2,525.00	0.07	0.58
Rauvolfia aphlebia	7.0	25.2	6,245.00	0.02	0.48
Rogiera amoena	5.8	19	7,878.25	0.06	0.59
Rogiera cordata	6.0	16	8,520.34	0.06	0.60
Roldana lanicaulis	2.0	7.7	4,469.11	0.07	0.55
Rudgea reducticalyx	2.5	12.8	7,099.00	0.05	0.57
Saurauia leucocarpa	10.0	16.5	5,876.19	0.06	0.44
Saurauia montana	10.5	15	6,875.00	0.06	0.44
Saurauia oreophila	15.0	25	5,937.58	0.06	0.44
Saurauia pittieri	10.0	16.5	6,086.54	0.06	0.44
Saurauia rubiformis	6.0	16.5	5,763.32	0.06	0.44
Saurauia scabrida	12.5	16.5	6,056.42	0.06	0.44
Saurauia serrata	10.5	16.5	5,870.07	0.06	0.44
Saurauia villosa	15.0	16.5	5,803.90	0.06	0.44
Schefflera rodriguesiana	10.0	30	2,037.18	0.08	0.43
Senna multifoliolata	5.0	19	4,050.54	0.04	0.56
Sloanea ampla	16.5	41	18,590.50	0.07	0.81
Solanum nigricans	6.0	6.1	10,224.40	0.05	0.42
Solanum storkii	6.3	6.1	10,224.40	0.05	0.42
Sommera arborescens	6.0	3	8,053.96	0.06	0.57
Sommera donnell-smithii	11.0	10	8,132.56	0.06	0.58
Stauranthus perforatus	5.8	12.8	6,424.79	0.05	0.47
Stenanona costaricensis	6.0	6.5	11,290.29	0.08	0.48
Stevia microchaeta	3.5	7.8	4,751.05	0.07	0.59
Styrax glabrescens	7.0	29.9	9,218.00	0.07	0.44
Styrax radians	20.0	30	9,218.00	0.07	0.44
Styrax ramirezii	18.3	29.9	9,218.00	0.07	0.44
Styrax warscewiczii	25.0	25	8,276.31	0.07	0.44
Symplococarpon purpusii	17.5	55	6,255.43	0.05	0.60
Symplocos citrea	15.0	12.8	2,892.00	0.07	0.64
Symplocos costaricana	7.5	17.5	2,892.00	0.07	0.64
Symplocos limoncillo	16.0	12.8	2,892.00	0.07	0.64
Symplocos pycnantha	10.5	22.5	2,892.00	0.07	0.64
Symplocos serrulata	8.5	39	4,184.22	0.15	0.64
Tabernaemontana alfaroi	3.5	16.4	6,544.75	0.03	0.56
Telanthophora cobanensis	5.0	8.7	4,468.32	0.07	0.54
Telanthophora grandifolia	5.0	8.7	4,468.32	0.07	0.54
Telanthophora uspantanensis	3.5	10	4,574.59	0.07	0.55
Ternstroemia lineata	15.0	52.7	7,272.23	0.07	0.62
Ticodendron incognitum	25.0	35	2,230.00	0.11	0.55
Tournefortia johnstonii	4.0	4.2	7,330.52	0.06	0.47
Ulmus mexicana	69.0	16.2	4,216.22	0.04	0.38
Vaccinium leucanthum	8.0	2.2	456.58	0.07	0.29
Vallesia aurantiaca	9.3	19.9	5,697.34	0.04	0.52
Verbesina apleura	4.5	1.2	3,740.19	0.08	0.44

Verbesina perymenioides	4.0	1.2	3,758.55	0.08	0.44
Viburnum acutifolium	4.5	7.6	2,485.44	0.08	0.54
Viburnum costaricanum	7.0	7.6	1,133.67	0.09	0.54
Viburnum elatum	5.0	7.6	2,485.44	0.06	0.54
Viburnum jucundum	4.5	7.6	2,485.44	0.06	0.54
Viburnum microcarpum	5.0	7.6	2,485.44	0.06	0.54
Viburnum venustum	4.5	9	2,001.98	0.06	0.54
Weinmannia burserifolia	12.0	60	2,787.48	0.10	0.62
Weinmannia wercklei	12.5	22.4	5,522.80	0.07	0.49
Wercklea insignis	10.0	35	8,089.06	0.06	0.24
Wercklea woodsonii	13.0	42.7	8,664.04	0.07	0.24
Zinowiewia integerrima	13.0	40	1,525.00	0.11	0.71
Zinowiewia rubra	15.0	45	1,525.00	0.11	0.71
Zygia palmana	8.0	20	17,707.00	0.07	0.83

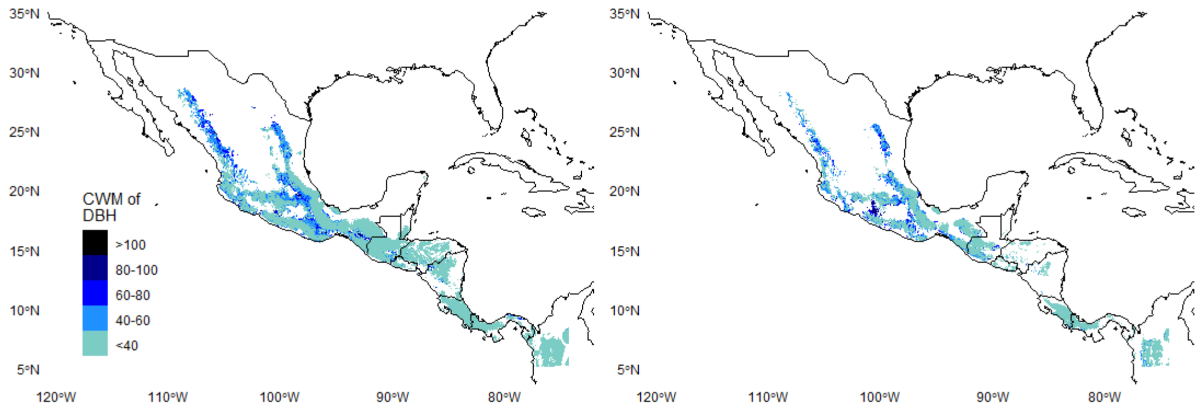


Fig. S2. Community-weighted means of diameter at breast height (cm) under current (1981-2010; left) and future (2071-2100; right) climatic conditions.

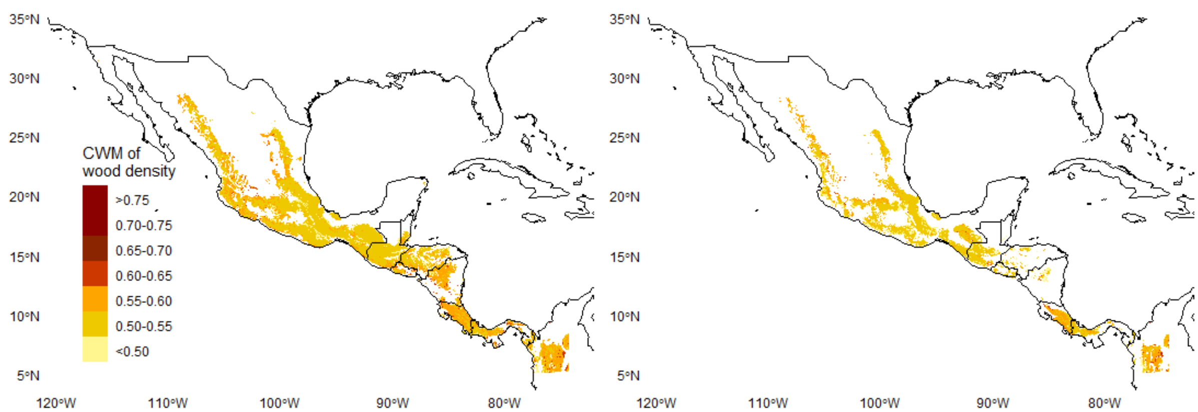


Fig. S3. Community-weighted means of wood density (g cm^{-3}) under current (1981-2010; left) and future (2071-2100; right) climatic conditions.

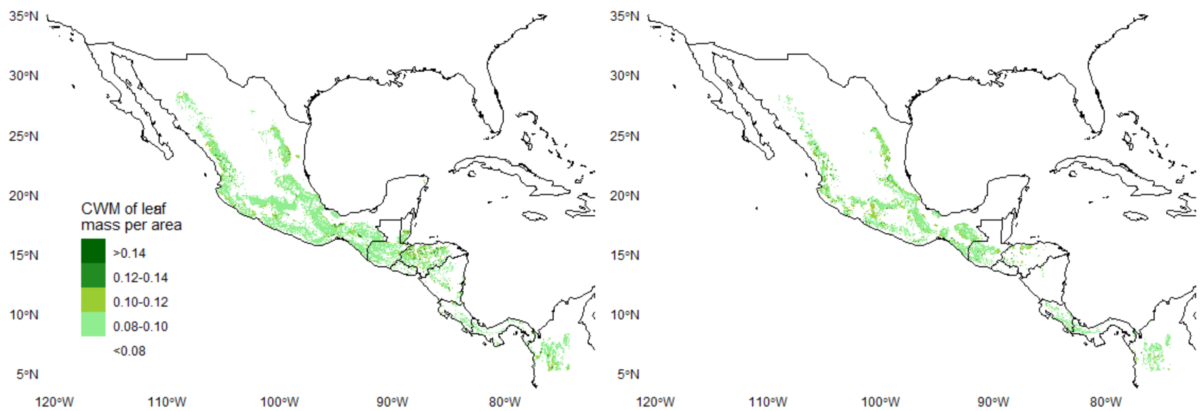


Fig. S4. Community-weighted means of leaf mass per area (g cm^{-2}) under current (1981-2010; left) and future (2071-2100; right) climatic conditions.

Table S11. Moran's I coefficients of five functional traits (H, DBH, WD, LA and LMA) and AGB throughout the study area.

Variable	Time	Moran I Statistic	p-value
H	Current	0.606	<0.0001
	Future	0.407	<0.0001
DBH	Current	0.493	<0.0001
	Future	0.347	<0.0001
WD	Current	0.330	<0.0001
	Future	0.371	<0.0001
LA	Current	0.670	<0.0001
	Future	0.520	<0.0001
LMA	Current	0.277	<0.0001
	Future	0.258	<0.0001
AGB	Current	0.476	<0.0001
	Future	0.343	<0.0001

Table S12. PCA stats of five functional traits (H, DBH, WD, LA and LA) of 272 montane-specialist tree species. Significance of the PCA axes calculated with Monte-Carlo tests, number of repetitions = 999, number of tests = 6.

PC	Eigenvalue	Proportion of Variance	Cumulative proportion	Obs.	Std. Obs.	Alternative	P-value
1	1.86	37.30	37.30	0.682	11.134	greater	0.001
2	1.27	25.34	62.63	0.631	2.866	greater	0.009
3	1.02	20.38	83.01	0.673	-2.761	greater	1
4	0.55	11.00	94.01	0.843	22.482	greater	0.001
5	0.30	5.99	100.00	0.876	14.028	greater	0.001

GAM model outputs

```
$H_current

Family: gaussian
Link function: identity

Formula:
H_current ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept) 15.51398    0.04904   316.4   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
               edf Ref.df      F p-value
s(lat)       7.903  8.643 285.77  <2e-16 ***
s(elev)      7.239  8.258  95.16  <2e-16 ***
ti(lat,elev) 9.812 11.560  22.36  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.555   Deviance explained = 55.7%
-REML =  12672  Scale est. = 9.1674    n = 5000
```

```
$H_future

Family: gaussian
Link function: identity

Formula:
H_future ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept) 15.46532    0.09072   170.5   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
                edf Ref.df     F p-value
s(lat)        8.565  8.933 49.19  <2e-16 ***
s(elev)       6.148  7.343 10.14  <2e-16 ***
ti(lat,elev) 12.198 13.399 41.36  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.214   Deviance explained = 21.9%
-REML =  15657  Scale est. = 30.165    n = 5000

$DBH_current

Family: gaussian
Link function: identity

Formula:
DBH_current ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept)  31.9306     0.2236   142.8   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
               edf Ref.df      F p-value
s(lat)       8.125  8.756 116.60  <2e-16 ***
s(elev)      7.842  8.648  22.79  <2e-16 ***
ti(lat,elev) 9.521 11.209   7.93  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.309   Deviance explained = 31.3%
-REML =  20270  Scale est. = 191.72    n = 5000

$DBH_future

Family: gaussian
Link function: identity

Formula:
DBH_future ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept)  34.3664     0.3604   95.36   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
                edf Ref.df     F p-value
s(lat)        8.228  8.812 48.56  <2e-16 ***
s(elev)       7.215  8.240 10.34  <2e-16 ***
ti(lat,elev) 10.301 11.963 17.23  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.159   Deviance explained = 16.4%
-REML =  22622  Scale est. = 491.46    n = 5000
```

```
$WD_current

Family: gaussian
Link function: identity

Formula:
WD_current ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
             Estimate Std. Error t value Pr(>|t|)
(Intercept) 0.5370691  0.0004425    1214   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
                edf Ref.df     F p-value
s(lat)        8.256  8.830 43.30  <2e-16 ***
s(elev)       6.619  7.774 11.48  <2e-16 ***
ti(lat,elev) 13.132 14.061 26.75  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.178   Deviance explained = 18.3%
-REML = -10979  Scale est. = 0.00070472  n = 5000

$WD_future

Family: gaussian
Link function: identity

Formula:
WD_future ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
             Estimate Std. Error t value Pr(>|t|)
(Intercept) 0.5244384  0.0006344   826.7   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
               edf Ref.df      F  p-value
s(lat)       8.510  8.909 75.867  < 2e-16 ***
s(elev)      7.456  8.410 32.701  < 2e-16 ***
ti(lat,elev) 9.403 11.244  4.308 3.89e-06 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =   0.18   Deviance explained = 18.5%
-REML = -9047.7  Scale est. = 0.001533  n = 5000

$LA_current

Family: gaussian
Link function: identity

Formula:
LA_current ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept)  5382.42      26.25   205.1   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
                edf Ref.df      F p-value
s(lat)        8.611  8.943 342.37  <2e-16 ***
s(elev)       5.775  6.969  37.38  <2e-16 ***
ti(lat,elev) 10.192 11.869  18.30  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.524   Deviance explained = 52.6%
-REML =  44040  Scale est. = 2.602e+06  n = 5000

$LA_future
```

```
Family: gaussian
Link function: identity

Formula:
LA_future ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept)  5498.24      37.43   146.9   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
               edf Ref.df      F p-value
s(lat)       8.585  8.935 152.87  <2e-16 ***
s(elev)      3.711  4.661  47.87  <2e-16 ***
ti(lat,elev) 9.962 11.670  10.74  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.354   Deviance explained = 35.7%
-REML =  45826  Scale est. = 5.3293e+06  n = 5000

$LMA_current

Family: gaussian
Link function: identity

Formula:
LMA_current ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
             Estimate Std. Error t value Pr(>|t|)
(Intercept) 0.0843336  0.0002619     322   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
                edf Ref.df     F p-value
s(lat)        8.684  8.962 13.80  <2e-16 ***
s(elev)       6.229  7.421 10.26  <2e-16 ***
ti(lat,elev) 12.694 13.684 25.46  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.119   Deviance explained = 12.4%
-REML = -13597  Scale est. = 0.00024689  n = 5000

$LMA_future

Family: gaussian
Link function: identity

Formula:
LMA_future ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
             Estimate Std. Error t value Pr(>|t|)
(Intercept) 0.0865418  0.0003499   247.3   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
                edf Ref.df     F p-value
s(lat)        8.676  8.962 13.42  <2e-16 ***
s(elev)       7.139  8.192 30.52  <2e-16 ***
ti(lat,elev) 12.925 13.935 19.70  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.133   Deviance explained = 13.8%
-REML = -12150  Scale est. = 0.00044024  n = 5000

$AGB_current
```

```
Family: gaussian
Link function: identity

Formula:
AGB_current ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept)  1796.51      28.88   62.21   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
               edf Ref.df     F p-value
s(lat)       8.532  8.915 81.92  <2e-16 ***
s(elev)      8.385  8.891 54.46  <2e-16 ***
ti(lat,elev) 9.732 11.323 24.95  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.345   Deviance explained = 34.8%
-REML =  44552  Scale est. = 3.1836e+06  n = 5000

$AGB_future

Family: gaussian
Link function: identity

Formula:
AGB_future ~ s(lat) + s(elev) + ti(lat, elev)

Parametric coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept)  1779.55      40.34   44.12   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
               edf Ref.df      F p-value
s(lat)       8.651  8.947 45.263  <2e-16 ***
s(elev)      5.804  6.995 20.900  <2e-16 ***
ti(lat,elev) 7.819  9.836  9.452  <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.129   Deviance explained = 13.3%
-REML =  46252  Scale est. = 6.3221e+06  n = 5000
```