IMPACT OF CLIMATE CHANGE ON EXTINCTION RISK OF MONTANE TREE SPECIES

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Abstract

The potential impacts of climate change on many species worldwide remains unknown, especially in those tropical regions that are centers of endemism and are highly biodiverse. This thesis provides an insight into the extinction risk of selected tree species using different species distribution modelling techniques and reviewing the current conservation status on montane forest in the Tropical Andes. Starting with a global analysis, the potential impacts of climate change on montane ecoregions is investigated, by identifying those that are more vulnerable to the expected changes in temperature and precipitation, from global predictions under different climate change scenarios. It then gives an insight on the current and potential threats to biodiversity in the Andean region, including the identification of those that are most likely to be responsible for increasing the extinction risk of the species. With the use of the IUCN Red List Categories and Criteria, selected tree species were assessed to identify their extinction risk. Information on the species' current distribution was collated and used to estimate their potential distribution under climate change, by using different modelling techniques. These results were used to reassess the species using the IUCN Red List and establish the changes in Red List Category. Lastly, it provides a discussion that integrates all the results obtained throughout the thesis, to explore the implications for conservation, in order to highlight the overriding importance of including threatened tree species to target conservation efforts in the region, while considering the uncertainties that surround predictions under climate change scenarios, modelling techniques and the use of the IUCN Red List.

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Author’s declaration

I confirm that this thesis is my own work with the following exceptions of the manuscript below. As senior author on the paper below, I led all aspects, including idea development, data collection, analysis and interpretation. I also led the preparation of the manuscript:

Chapter 3 is published in collaboration with several coauthors of the Tropical Andes Red List expert knowledge group who edited the final manuscript as:

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1 Introduction

1.1 Climate change and forest biodiversity

Climate change is a global scale process but with diverse regional manifestations (Committee on Ecological Impacts of Climate Change 2008; IPCC 2007). In recent decades, evidence for anthropogenic climate change has accumulated. Global average surface temperature has risen some 0.75°C (1.3°F) since 1850 (Crowley 2000; IPCC 2007); there has also been a decrease in extent of mountain glaciers and snow cover in both hemispheres (Kohler and Maselli 2009; Vuille *et al.* 2008). From 1900 to 2005, precipitation has increased significantly in eastern parts of South America (van der Hammen 1974), but has declined in the Sahel (Hulme 1992), the Mediterranean (Sarris *et al.* 2007), southern Africa (Nicholson 1993) and parts of southern Asia (Bradley *et al.* 1987). Global atmospheric concentrations of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) have increased markedly (Matson and Vitousek 1990) and now far exceed pre-industrial values determined from ice cores spanning many thousands of years (Crutzen and Zimmermann 1991). Although the primary causes of natural climate change are suggested to be volcanic activity and fluctuations in solar radiation, much of the recent patterns have been attributed to anthropogenic factors (IPCC 2007; Parmesan *et al.* 2013).

The current rate of induced climate change is of increasing concern with regard to its potential effects on forest systems (Loehle and LeBlanc 1996; McCarty 2001), especially on tropical montane forests. Historically, the global forest biota has been affected by long term climatic fluctuations and has coped through evolutionary changes and the ability to survive and migrate to patches of favourable habitats (Peters 1990). These changes, however, occurred in landscapes that were not as fragmented as they are today and with less pressure from human activities (Secretariat of the Convention on Biological Diversity 2006; Thomas *et al.* 2004; Walther *et al.* 2002). Evidence suggests that climate change affects forest ecosystems by altering the frequency, intensity, duration, and timing of rainfall, fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, icestorms and landslides. This influences forest composition, structure and phenology, the range and distribution of species, as well as the dynamics of communities and functional processes (Beniston 2003; Dale *et al.* 2001; Foster 2001; Walther 2004).

Climate change is shifting vegetation geographically (Rosenzweig 2007), altering the global location of biomes (Gonzalez *et al.* 2010). Understanding the impacts of climate change on plant and animal life requires the development of models that predict future range shifts in species, communities and ecosystems (Kappelle *et al.* 1999). Different approaches to modelling the impact of climate change on forests, such as Dynamic Global Vegetation Models (DGVMs), enable the identification of areas vulnerable to vegetation shifts and potential refugia (Gonzalez *et al.* 2010). Studies using DGVMs have projected the replacement of forests with drier biomes such as grasslands and scrub e.g. (Alo and Wang 2008b, 2008a; Jolly and Haxeltine 1997; Salazar and Nobre 2010; Scholze *et al.* 2006), typically as a result of changes in temperature and precipitation. Vegetation often responds slowly to changes in environmental conditions (Gonzalez *et al.* 2010) and the lags between climate change and vegetation response are quite variable, reflecting limitations on seed dispersal, influence of dispersal barriers (e.g. topography), and slow rates of soil production and biomass accumulation (Gonzalez *et al.* 2010; Masek 2001).

Montane forests

Species can only live in geographical areas where they can tolerate the local temperatures, rainfall and snowfall regimes (Committee on Ecological Impacts of Climate Change 2008). Mountain environments respond strongly even to small changes in temperature. Their vertical (altitudinal) dimension creates steep gradients of temperature, precipitation and solar radiation. Such topography-climate interactions create a multitude of microhabitats over short distances, each with varied micro-climates, ecological conditions and specific sets of organisms. For these reasons mountains are rich repositories of biodiversity, endemism and ecosystem services (Bubb *et al.* 2004; Chaverri-Polini 1998; Hamilton 1995; Körner 2004; Nogués-Bravo *et al.* 2006; Olson and Dinerstein 1997; Sharma *et al.* 2009). They also have tangible economic value as source of energy, tourism, forestry and crop and livestock production (Nogués-Bravo *et al.* 2006).

Some montane forests, such as tropical cloud forests, are high on the list of the world's most threatened ecosystems (Hamilton 1995; Ledo *et al.* 2009; Stadtmüller 1986; Wuethrich 1993). It is widely believed that the majority of those cloud forests that remain are small areas or remnant fragments of their original extent (Aldrich *et al.* 1997; Bubb *et* *al.* 2004; Stadtmüller 1986; Wuethrich 1993). For example, it is estimated that 90 percent of the original cloud forests in the northern Andes have been lost, mainly as a result of deforestation, forest degradation and overexploitation (Hamilton 1995). Climatic and geographic factors, on a local or regional level, influence the formation and elevation of clouds, as well as their water content, thickness, and dynamics (Bruijnzeel 2002; Stadtmüller 1986). Tropical montane cloud forests have the unique additional characteristic of capturing water from the condensation of clouds and fog. Increasing global temperatures and changes in precipitation patterns will have a detrimental impact on the water balance of these forests (Sharma *et al.* 2009), raising the average altitude at the base of the orographic cloud bank (Kappelle *et al.* 1999), affecting the ecosystem integrity and water availability. This will perhaps result in a shift to higher latitudes and altitudes, but one of the key features of upper montane forests is that their scope for such migration is limited geographically (Kappelle *et al.* 1999; Kohler and Maselli 2009; Kreyling *et al.* 2010; Sharma *et al.* 2009; Urrutia and Vuille 2009). As a result, montane forests may be particularly vulnerable to climate change.

Mountain zonation

Biogeographically, mountains are stratified into elevational belts, each with a characteristic flora and fauna. The part below the natural climatic limit of trees (the treeline) is called the 'montane' belt (Körner 2004). Frahm and Gradstein (1991), who classified tropical mountain forests taking into consideration the bryophyte cover, identify five altitudinal belts: lowland forest, submontane forest, lower montane forest, upper montane forest, the subalpine forest. However, the description of the altitudinal zonation differs from place to place (Frahm and Gradstein 1991).

Many of the classifications described above depend on a series of factors such as those described below.

a) Local climatic conditions

Determinant factors in the classification of mountain forests are the frequency of fog (Grubb and Whitmore 1966) and prevailing (trade) winds (Bubb *et al.* 2004; Jarvis and Mulligan 2011). Additional influential factors include: (a) precipitation, which tends to decrease with altitude (Jarvis and Mulligan 2011; Stevens 1992); (b) evapotranspiration; (c) temperature, which tends to decrease by around 0.6°C per 100 m rise (Bach *et al.*

2003; Richards 1952) and (d) solar radiation, which decreases with altitude, as there is an increase of cloud cover as the altitude increases that in turn affects evapotranspiration (Bach *et al.* 2003).

b) Substrate

Zonation may occur due to geological or geomorphological characteristics of the environment. The tree line may in some equatorial areas reach up to 4000 m a.s.l, whilst in others it is considerably lower, e.g. 3200 m a.s.l. in many places in the equatorial Andes and 3400 m on Mt Kinabalu, Borneo. The low tree lines in the Andes are usually considered the result of human influence (wood cutting, burning), however on Mt Kinabalu the upper limit of the forest is determined by the presence above 3400 m a.s.l. of a very steep, granitic slope unsuitable for tree growth (Frahm and Gradstein 1991).

c) Latitude

It is well known that vegetation belts decrease in altitude from the equator towards the tropics of Cancer and Capricorn, and beyond, towards the poles (Frahm and Gradstein 1991).

d) Massenerhebung effect

This has been described as the raising of the limits of a given forest-type to higher altitudes on a large massif than on small, isolated or outlying peaks and coastal mountains (Grubb and Whitmore 1966; Richards 1952). The rate of decrease of temperature with altitude (the lapse rate) is not consistently different on the two types of mountain (Grubb and Whitmore 1966); in other words, this mass-elevation effect causes the occurrence of montane forest conditions at lower altitudes on narrow cordilleras and outlying ridges (Armenteras *et al.* 2003).

Tropical Andes

The Andes is the longest mountain chain on earth, stretching more than seven thousand linear kilometres across tropical, subtropical, and temperate latitudes. These mountains were shaped by local climatic and edaphic factors and by important past events, such as the Pleistocene glaciations, which affected their geology and climate, producing a wide variety of ecosystems with both latitudinal and altitudinal dimensions (Chaverri-Polini 1998; Little 1981). On wet tropical mountains with increasing altitude, the changes in forest structure are principally a decrease in forest stature, and a tendency for the leaves to become smaller, thicker, and harder ("xeromorphic") (Bruijnzeel and Veneklaas 1998). Climatic and geographic factors, on a local or regional level, may additionally influence formation and elevation of clouds, as well as their water content, thickness, and dynamics (Stadtmüller 1986).

Andean forests, even near the tree line at around 3000 m, are more rich in species than are most temperate forests and this high species richness is due to the much greater endemism compared to other parts of the world (Gentry 1982, 1988; Olson and Dinerstein 1997), in which for example, epiphytes (bryophytes, lichens and filmy ferns) contribute largely to the overall floristic diversity (Barthlott *et al.* 2001; Benzing 1998; Wuethrich 1993). One of the reasons for such a high richness and endemism is the complexity and variety of microhabitats available. These microclimates on eastern slopes differ from the western slopes and are different again in the valleys of the interior (Walter 1985). The altitudinal belts on the western slopes become increasingly xerophytic toward the south. In some areas the presence of a warm season pushes the tree line limit up to 4000 m a.s.l., with scattered stands of Polylepis even at 4500 (4900) m a.s.l. (Walter 1985). According to Gentry (1995) Lauraceae is the most species-rich woody plant family in neotropical montane forests (above 1500 m a.s.l.), followed by Melastomataceae and Rubiaceae. Close to the timberline, Compositae and Ericaceae are most prominent.

Montane forest in the tropical Andes are currently a major global conservation priority owing to their biological richness and high level of endemism (Bush *et al.* 2007; Olson and Dinerstein 1997). They are considered as an extremely fragile ecosystem playing an important hydrological and ecological role, and also are considered amongst the least known ecosystems in the tropics (Bubb *et al.* 2004; Gentry 1995; Kessler 2000; La Torre-Cuadros *et al.* 2007; Stadtmüller 1986). Research is very scattered and is commonly undertaken at a national level e.g. (Armenteras *et al.* 2007; Grubb and Whitmore 1966; La Torre-Cuadros *et al.* 2007), and little research has been done at a regional level e.g. (Buytaert *et al.* 2010; Cuesta *et al.* 2009; Urrutia and Vuille 2009). It is believed that warming in the tropical Andes is likely to be of similar magnitude as in the Arctic, and with consequences that may be felt much sooner and which will affect a much larger population (Vuille *et al.* 2008), therefore research on the potential impacts of climate change is urgently needed.

1.2 Extinction risk

Global climate change potentially threatens all ecosystems through temperature and rainfall changes, implying a shift in distribution of optimum habitats. This suggests the replacement of many of the narrow altitude range forests by lower altitude ecosystems. Also, species with low adaptability and/or dispersal ability could be faced with climate-forced range shifts and low chances of finding habitats to colonize in the remaining natural vegetation (Walther *et al.* 2002). In addition, existing cloud forests can be lost altogether (Foster 2001; Kappelle *et al.* 1999; Walther *et al.* 2002). Climate driven changes in the forest structure can lead to a decline of late-successional species, increasing the dominance of early successional species, or leave poorly adapted plant communities that are vulnerable to invasion by species that can thrive in the area's new climate (Dukes and Mooney 1999), acting as a major cause of extinctions in the near future (Malcolm *et al.* 2006; Schwartz *et al.* 2006; Thomas *et al.* 2004), especially on restricted-range endemic species (Thomas *et al.* 2004).

Studies have shown that projected climate change scenarios will have a major impact on biodiversity (Golicher *et al.* 2008; Midgley *et al.* 2002; Thomas *et al.* 2004). Predictive models carried out in the Andes for the year 2020 and 2050 suggest that an estimated 13% to 21% of the pluvial forests will be lost by the end of the studied period (Cuesta *et al.* 2009). There is a need to explore the potential impacts of such changes on extinction risk of species, as forests are already threatened by anthropogenic activities such as of deforestation, degradation and selective logging (Achard *et al.* 2002; Armenteras *et al.* 2011; Carretero *et al.* 2003; Fundación Pachamama 2010; Hansen *et al.* 2010; Hansen *et al.* 2008; Ibisch 2002; Lambin *et al.* 2003; Rodríguez 2005). The IUCN (International Union for Conservation of Nature) Red List is a globally recognised assessment of the extinction risk of species, and has become one of the most effective sources of information for conservation planners (Lamoreux *et al.* 2003; Newton 2010; Rodrigues *et al.* 2006). Although the list includes over 12,000 plant species, the vast majority of the species have yet to be assessed (Newton and Oldfield 2008).

1.3 Predicting the impact of climate change

The quantification of species-environment relationships has gained importance as a tool to assist in decision-making related to nature conservation (Cayuela *et al.* 2006) and for predicting the effects of global change on species distributions and the resulting extinction risks (Feeley and Silman 2010). The availability of data for some areas of the world, however, is still comparatively limited, restricting the parameterization of complex niche models (Feeley and Silman 2011; Golicher *et al.* 2008). In addition, a large number of tree species that are potentially threatened with extinction still await assessment using the IUCN Red List (Newton and Oldfield 2008). To date, the study of tropical forests in mountain areas has emphasized that they are globally significant areas of species richness and endemism (La Torre-Cuadros *et al.* 2007). While there have been reports outlining the extensive biological changes that are ongoing in montane environments because of climate change (Feeley and Silman 2009; Parmesan and Yohe 2003; Wuethrich 1993), few efforts have been made to assess the potential effects of climate change on montane forests at a global scale. Little research has been undertaken focusing on individual tree species and their potential extinction risk using internationally recognised approaches such as the IUCN Red List. Examples of efforts to model extinction risk due to climate change include: Schwartz *et al.*(2006) in the eastern United States, Feeley and Silman (2009, 2010) in Amazonian and Andean forests and Cuesta *et al.*(2009) in selected Andean forests. More accurate predictions of how species and ecosystems will respond to climate change are needed to assist in preparation for future conservation challenges (MacKinnon *et al.* 2008; McCarty 2001). Limitations in the data available and the insufficient studies performed in montane forests are the key gaps in the current knowledge that this research intends to address.

Understanding the potential impacts of climate change on tree diversity in the montane forests, by using projections for future climate scenarios and species' distributions, will provide an assessment of the extinction risk for many species that still remain unknown internationally, and will inform development of conservation priorities. It is acknowledged that uncertainty will arise when predicting extinction risk in relation to the proportionate reduction in area of forest, as stated by Thuiller *et al.* (2004), because species life history and environments change over time. In addition, uncertainty can arise through measurement errors and uncertainty in the status and distribution of species, which has previously been explored using fuzzy approaches such as RAMAS® Red List (Akçakaya and Root 2007) or the Bayesian network described by Newton (2010).

1.4 Thesis aims, objectives and structure

Aims and objectives

The aim of this research is to examine the extinction risk of montane tree species, with a particular focus on the potential impacts of climate change. This will be achieved using spatial analysis and modelling approaches at a range of scales. First, an investigation will be conducted at the global scale, to examine the potential impacts of climate change on montane forests occurring at different altitudes, with the aim of identifying those areas that are most vulnerable to climate change impacts. Second, an investigation will be performed at the regional scale, focusing on the tropical Andes, which will evaluate the risk of extinction of tree species present in montane forests. This will be achieved by analysing spatial data describing the potential change in climate with data on the spatial distribution of tree species, to determine potential changes in distribution patterns and associated extinction risk. The research will therefore increase understanding of the potential role of climate change as a cause of biodiversity loss. The overall objective of this research is to test the hypothesis that climate change is a major potential contributor to biodiversity loss in montane areas. This will be evaluated by testing the following hypotheses:

Hypothesis 1:

Montane forests at the global scale vary in their vulnerability to projected climate change, as defined by the IPCC scenarios.

This hypothesis will be tested by addressing the following objective:

a. To identify the relative vulnerability of different montane forest areas to potential climate change, by analysing the current climatic conditions associated with different montane forests at the global scale, and assessing how they might change under different projected climate scenarios (Chapter 2).

Hypothesis 2:

Projected changes in distributional range resulting from climate change will increase the extinction risk of many tree species, particularly those associated with high rainfall and high elevations, and those with restricted geographical ranges.

This hypothesis will be tested for one selected region (tropical Andes) by addressing the following objectives:

a. Identify the current conservation status of montane forest in the tropical Andes by compiling the evidence available (Chapter 3).
b. Identify the extinction risk for the different tree species that occur in montane forests in the tropical Andes by carrying out an IUCN Red List (RL) assessment, which is checked and validated using expert knowledge from within the region (Chapter 4).
c. Investigate the potential impact of climate change on the extinction risk of selected tree species using different species distribution modelling techniques to support the RL assessment (Chapter 5).
d. Analyse the sources of uncertainty and their impact on the assessment of RL status including: determination of the sample size required for reliable estimation of potential species distributions using species distribution modelling approaches (Chapter 4 and 5).

1.5 Thesis structure

Chapter 1 Introduction and literature review

This comprises an expanded version of the text provided above. This chapter includes a comprehensive literature review of the impact of climate change on forests; the process of evaluating extinction risk including the IUCN Red List; methods for modelling climate change impacts; and the definition and classification of montane forests. The information collated in this chapter has been used to guide the data collection phase of the research and has been referred to throughout the thesis.

Chapter 2 Vulnerability of montane forest to global climate change

This chapter examines how climate change could potentially affect montane forests distributed worldwide. This has been achieved by analysing the current climatic conditions of montane forests and assessing how they might change under different climate scenarios. This part of the research is based on the analysis of spatial data to identify the distribution of montane forests, with their corresponding ecoregions, and by using georeferenced global climate sets to analyse the potential vulnerability of these forests to potential climate change under two climate change scenarios (i.e. A2 and B2 from the Intergovernmental Panel on Climate Change (IPCC)).

Chapter 3 Evaluation of the conservation status of montane forest in the tropical Andes

This chapter is an overview of the conservation status of tropical Andean montane forests and the challenges that they currently face. It also provides information on threats to biodiversity in the region, including the identification of those that are most likely to be responsible for increasing the extinction risk of species. This chapter also highlights the need for more information on the conservation status of species to identify future priorities for conservation in the region. This part of the work has been carried out with the collaboration of a network of experts from throughout the region. This has been published in *Ecosistemas* journal.

Chapter 4 Red List and conservation planning of tree species in montane forests of the tropical Andes

This chapter evaluates the implications of the IUCN Red List and the implementation of conservation planning for the montane forest tree species in the tropical Andes. This was completed in a series of steps. Firstly, a list of tree species present in the region was compiled, to form the basis for subsequent assessment. Secondly, the extinction risk of these was evaluated according to the IUCN Red List Categories and Criteria. This step was achieved by compiling spatial data describing the current distribution of each species. These data was used to produce a Minimum Convex Polygon (MCP) for each of the species present in the tropical Andes, as a measure of geographical range, which enabled the estimation of the species current Extent of Occurrence (EOO) and Area of Occupancy (AOO) in R (R version 2.14.2) (R Development Core Team 2011). These variables were used to identify a preliminary category of the IUCN Red List. This process was supported by the development of a network of experts within the region, who have provided a source of specialist knowledge for the validation of the assessments.

Chapter 5 Modelling potential species distribution under climate change scenarios for the tropical Andes

This was completed through an analysis of the potential species distribution under current and projected climatic variables using the spatial data compiled in Chapter 4. A range of modelling approaches was explored to conduct the analyses, based on a review conducted to date. A reassessment of the species evaluation was then carried out using the IUCN Red List Categories and Criteria, in order to assess the influence of climate change on the species evaluation and potential implications for conservation.

Chapter 6 Discussion

This section discusses the implications of climate change on montane forests and the extinction risk of montane tree species. This section also integrates the results from the previous chapters and includes relevant conclusions and recommendations where possible.

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2 Vulnerability of montane forest to global climate change

2.1 Introduction

Historically, climate change has had a great influence in shaping the distribution of biodiversity (Cárdenas *et al.* 2011; Luterbacher *et al.* 2004), and today it is recognised as one of most influential causes of change in global ecosystems (Parmesan 2006; Rosenzweig *et al.* 2008; Walther *et al.* 2002). Anthropogenic climate change is now an issue of major concern in relation to its potential impacts on biodiversity, as indicated by both projected and observed changes (Root *et al.* 2003; Rosenzweig *et al.* 2008; Travis 2003). Changes such as shifts in ranges in plant and animal species (IPCC 2007; Rosenzweig *et al.* 2008) have already been observed in a range of ecosystems, which could have major implications for their future structure, function and composition (Gottfried *et al.* 2012; Parmesan and Yohe 2003; Sommer *et al.* 2010; Wu *et al.* 2010).

Montane forests are widely recognised to be of high importance for biodiversity conservation. They are often characterised by a relatively high degree of endemism (Bruijnzeel *et al.* 2010; Bubb *et al.* 2004; Foster 2001; Kessler 2000), a wide variety of edaphic conditions and pronounced climate gradients over relatively small geographic areas (Jarvis and Mulligan 2011; Mulligan 2010). They are also important providers of benefits to humans, including water, energy, minerals, timber, fibre and agricultural products (Beniston 2003; Bruijnzeel *et al.* 2010; Tobón 2009). Some montane forests, such as tropical cloud forests, are among the world's most threatened ecosystems (Hamilton 1995; Ledo *et al.* 2009; Stadtmüller 1986; Wuethrich 1993); the majority of the cloud forests that remain are remnant fragments of their original extent (Aldrich *et al.* 1997; Bruijnzeel *et al.* 2010; Bubb *et al.* 2004; Wuethrich 1993). The main threats to these forests come from forest loss and degradation, caused by conversion of forest to agricultural land use, and over-exploitation of tree species for products such as timber and fuelwood. Hansen *et al.*(2010) estimated a global gross forest cover loss of 3.1% between 2000 and 2005. For the same period Hansen *et al.*(2008) estimated that humid tropical forests had an area cleared of 27.2 ±2.28 million hectares, representing a further reduction of forest area to the estimated losses reported by Achard *et al.*(2002) between 1990 and 1997, where the annual forest area lost was of 5.8 ±1.4 million hectares, with a further 2.3 ±0.7 million hectares of forest visibly degraded. Furthermore, there are activities that pose additional threats to these forests, which include the impacts of fire, browsing animals, urban expansion, infrastructural development and mining, as is the case of the tropical Andes (Tejedor Garavito *et al.* 2012). The potential land use changes that result from climate change will interact with current land use change (Lambin *et al.* 2003) and could further increase the level threat to these forests.

Montane forests are the focus of particular concern in relation to the potential impacts of anthropogenic climate change. Increasing global temperatures and changes in precipitation patterns could have a detrimental impact on their water balance (Anderson *et al.* 2011), for example by raising the average altitude of the base of the orographic cloud bank (Ruiz *et al.* 2008), reducing horizontal precipitation (Anderson *et al.* 2011), the amount of foggy days and relative humidity (Ruiz *et al.* 2008). Such changes could negatively affect water cycling and availability, with consequences for both plant and animal communities. Montane forests could be particularly at risk under projected climate change because many species that occur in these forests are characterised by relatively narrow climatic adaptation (Keenan *et al.* 2011). While species are generally expected to respond to climate change by range shifts towards higher latitudes and altitudes (Kappelle *et al.* 1999; Kohler and Maselli 2009; Kreyling *et al.* 2010; Sharma *et al.* 2009; Urrutia and Vuille 2009), there is less scope for such responses in species associated with mountain environments, particularly at higher elevations. Other potential impacts of climate change on montane forest ecosystems related to changes in precipitation include alteration of runoff and filtration patterns, which in turn would increase soil erosion (Anderson *et al.* 2011). Extreme climatic conditions could increase physiological stress, encourage changes in fecundity, phenology, trophic dynamics (including disease), species invasion and migration, and drive habitat loss, causing population declines or extinctions (Larsen *et al.* 2011). Furthermore, climate change could lead to expansion of the agricultural frontier to higher elevations, which could contribute to the further fragmentation and loss of forests (Anderson *et al.* 2011).

Despite such concerns, no systematic analysis of the potential impacts of climate change on montane forests has been conducted previously at the global scale. Recent research carried out on the impacts of climate change on ecoregions (Beaumont *et al.* 2011) and on disappearing climates (Williams *et al.* 2007) have shown that there are ecoregions and climates that are especially vulnerable to predicted changes in temperature and precipitation. However, such analyses do not explicitly consider montane forests, and do not take into consideration the current extent or distribution of such forests. The objective of this research is to identify the relative vulnerability of different montane forest areas to potential climate change, by analysing the current climatic conditions associated with different montane forests at the global scale, and assessing how they might change under different projected climate scenarios.

2.2 Methods

Globally, montane forests are primarily distributed on continuous mountain ridges and can be classified as extratropical, subtropical or tropical, based on the geographic region in which they occur. For the purpose of the research in this chapter, montane forest, at the global scale, is defined as an area with trees where the canopy cover is ≥10%, following Schmitt *et al.*(2009b) and FAO (2010), occurring ≥1000 m a.s.l. as defined by a Digital Elevation Model (DEM) used by WorldClim (Hijmans *et al.* 2005). This threshold allows the exclusion of most lowland vegetation (UNEP-WCMC [United Nations Environment Programme's World Conservation Monitoring Centre] 2002; Young 2006). Although some recent definitions of mountains, such as those of Körner *et al.* (2011) and UNEP-WCMC (2002), use characteristics such as ruggedness or slope to define them, such definitions would exclude most forests occurring in high elevation plateaux. The definitions that were adopted here would explicitly include such forests, which should be borne in mind when interpreting the results. The influence of the different definitions of montane forests on the results was explored by repeating the analyses using different thresholds of forest cover and mountain maps.

To develop a forest distribution map the DEM was masked with a map of global forest cover (GFM) produced by Schmitt *et al.* (2009b). This is based on a satellite-derived 500 m resolution Moderate Resolution Imaging Spectroradiometer (MODIS) Vegetation Continuous Fields Dataset (MODIS05 VCF) for Percent Tree Cover for the year 2005 (Hansen *et al.* 2006), a Global Forest Cover map (UNEP-WCMC 2000) and the Global Land Cover 2000 dataset (GLC 2000) produced by the European Commission Joint Research Centre (Bartholomé and Belward 2005). MODIS05 VCF is the most recent global dataset on tree cover but includes many areas of woody land cover other than natural forests, especially in the lower tree cover classes. Many of these areas, such as shrublands, tree plantations and some types of agro-ecosystems, were identified and excluded from the updated GFM by Schmitt *et al.* (2009b) using the GLC 2000 data. Thus, the GFM is primarily a map of relatively natural forest cover.

Thresholds of 30% and 40% forest cover have previously been used to determine the distribution of montane forests in some previous studies (e.g. Bubb *et al.* (2004)). However, this threshold may underestimate forest extent. To avoid this problem, a threshold of 10% was adopted following (Schmitt *et al.* 2009a; Schmitt *et al.* 2009b) and the FAO, as employed in the most recent Global Forest Assessment (FAO 2010).

Although a number of different approaches have been used to classify and map global vegetation, relatively few of these are primarily based on biogeographic information, rather than climate, and incorporate information on the distribution of both plant and animal species. Therefore, the ecoregion classification described by Olson and Dinerstein (1998) was used, which has been widely used to conduct global and regional conservation assessments. These ecoregions or biogeographic realms represent a development of those presented previously by Pielou (1979) and (Udvardy 1975), and were adapted as a result of extensive literature search and expert consultation. In this investigation the ecoregion classification presented by Olson *et al.* (2001) and the corresponding 'Global 200' priority ecoregions presented by Olson and Dinerstein (2002) were used to identify montane ecoregions. A digital (vector) layer of global ecoregions was accessed from: www.worldwildlife.org/science/data/item6373.html. This was then overlaid with the forest map and the DEM (> 1000 m) in a geographical information system (GIS) enabling the total and the percentage of montane forest within each ecoregion to be calculated. GIS layers were transformed into Mollweide's projection in order to carry out all area calculations. All GIS operations were conducted with ArcGIS v.10 (© 1999-2006 ESRI Inc. California, USA).

Current and future climatic data was obtained from the World Data Centre for Climate (http://cera.wdc-climate.de), which is a repository of different Global Circulation Models (GCM) that predict future climatic variables for different spatial and temporal scales. For this research climatic variables were chosen from among those believed to have a significant influence on the growth and distribution of tree species, namely mean temperature (K) and mean precipitation (mm d^{-1}) (Attorre *et al.* 2011). The results of the Hadley Centre Coupled Model for two of the Special Report on Emission Scenarios (SRES), namely HADCM3 for scenario A2 and HADCM3 for scenario B2 were used. These were prepared for the Intergovernmental Panel on Climate Change (IPCC) Third Assessment Report (Cubasch *et al.* 2001), and are considered to be the most policy relevant of the scenarios that have been developed (Johns *et al.* 2003). These scenarios were designed to consider different trajectories of future economic development and energy use and perhaps this choice of scenarios is seen as conservative or optimistic as more fossil fuel intensive scenarios such as A1FI were excluded. The A2 scenario represents a very heterogeneous world, where population continues to increase at a higher rate than the B2 scenario. Fragmented and slower economic growth and technological change characterises this scenario when compared to other scenarios (IPCC 2001; Joos *et al.* 2001; Table 2.1), this scenario is commonly used for 'business as usual' impact studies, as it projects a 3°C increase in surface air temperature by 2100 on average across model (Cubasch *et al.* 2001). The B2 scenario depicts a world that emphasises local solutions to social, economic and environmental sustainability, where human population continues to increase with an intermediate level of economic development, and therefore a less energy-intensive scenario featuring a lower emission path, projecting a 2.2°C temperature increase on average across all models by 2100 (Cubasch *et al.* 2001)**.** The data cover a period of 150 years from 1950 until 2100 at a scale of 3.75^{o} longitude and 2.5^{o}

latitude, which is approximately 417 km x 278 km at the Equator, and 295 km x 278 km at 45° latitude.

Global climate sets were analysed using the Climate Data Operators (CDO) software (available from: https://code.zmaw.de/projects/cdo), which are a collection of many operators for standard processing of climate and model outputs, allowing data splitting, averaging and statistical analysis. The data was split into two sets of 50 years for comparison with a similar length period in the 21st century. Therefore, three sets of data for the following periods were identified: 1951–2000 (baseline climate), 2001–2050 and 2051–2100, for temperature and precipitation variables. All data layers were interpolated to a common resolution of 0.0083° latitude/longitude by using a nearest neighbour method (command RESAMPLE in ArcGIS v10).

Scenario	Temperature range for 2090-2099 (°C)	Sea level rise for 2090-2099 (m)	Characteristics
B1	1.1 – 2.9	0.18 – 0.38	• More environmentally focused • Homogeneous world (globalisation) • Global environmental sustainability • Population growth peaks in mid-century and declines thereafter.
A1T	1.4 – 3.8	0.20 – 0.45	• Non-fossil energy sources • Rapid economic development • Homogeneous world (globalisation).
B2	1.4 – 3.8	0.20 – 0.43	• Local solutions to economic, social and environmental sustainability • Lower population growth than A2 • Regionalisation (heterogeneous world).
A1B	1.7 – 4.4	0.21 – 0.48	• A balance across all sources of energy • Rapid economic development • More economic focus • Homogeneous world (globalisation).
A2	2.0 – 5.4	0.23 – 0.51	• Regionally oriented economic development. • Continuous population growth. • Regionalisation (heterogeneous world).
A1FI	2.4 – 6.4	0.26 – 0.59	• Fossil-intensive • Rapid economic development • More economic focus • Homogeneous world (globalisation).

Table 2.1 Characteristics of climate change scenarios predicted for 2090-2099 relative to 1980-1999. Adapted from Cubasch *et al.*(2001) and Solomon *et al.*(2007).

In order to assess the vulnerability of each ecoregion to climate change the standardized Manhattan Distance (M) was calculated, which is the distance between the mean (μ) values of both temperature and precipitation from projected values for 2051–2100 (100 years) and the mean of the 1951–2000 baseline, standardized by the Standard Deviation (σ) of the baseline climate $\mathbf{M} = \frac{(\mathbf{100year\mu - baseline\mu})}{\mathbf{\sigma baseline}}$ (Beaumont *et al.* 2011). Extreme climatic conditions for a specific ecoregion are considered here as those where M exceeds 2 SDs (σ) (i.e. M>2) departing from the mean (μ) of the 1951–2000 baseline period (Beaumont *et al.* 2011; Luterbacher *et al.* 2004). Differences for precipitation were expressed as ratios (future/baseline) for values that were different from zero. Once mean values were obtained for each ecoregion, these were ranked in descending order to identify the ecoregions with the largest M values.

2.3 Results

518 of the 867 ecoregions and 117 of the 'Global 200' ecoregions were identified, covering an estimated total area of 5,892,529 km^2, in which montane forest was found to be present. However, 307 ecoregions were each found to contain an area <5% of forest cover and were therefore excluded from this analysis. This left a total of 211 ecoregions covering a total area of 5,291,398 km^2. Of these, 141 are part of 68 ecoregions that are considered to be global conservation priorities, as they belong to the 'Global 200' ecoregions defined by Olson and Dinerstein (2002). Africa was found to be the region with the largest forest area (1,744,600 km^2) followed by Asia and North America (with 1,670,875 km^2 and 1,287,313 km^2 respectively).

The ecoregions with the largest areas of montane forests are the Central Zambezian Miombo woodlands, in southern central Africa, followed by the Angolan Miombo woodlands, in Angola and Democratic Republic of Congo, and Sayan montane conifer forests, in the Siberian taiga and the Mongolian steppe (**Figure 2.1**). Percentage montane forest cover within ecoregions was found to be largest in the Angolan montane forest-grassland mosaic followed by the Australian Alps and Eastern Australia Temperate Forests (**Figure 2.2**).

These analyses were repeated using a threshold value of 30% tree cover rather than 10%, to examine the sensitivity of results to this parameter. This would reduce the total area of montane forests to 4,397,664 km^2, and a reduction in the number of ecoregions by 11, as the following would be excluded: from Australia, the Central Ranges xeric scrub; from Africa, West Sudanian savannah and Mediterranean acacia-argania dry woodlands and succulent thickets; from America, Central Canadian Shield forests, Gulf of California xeric scrub, Low Monte; from Asia, Al Hajar montane woodlands, Qilian Mountains subalpine meadows, Northwest Russian-Novaya Zemlya tundra, Red Sea Nubo-Sindian tropical desert and semi-desert; and from Europe, Southeastern Iberian shrubs and woodlands. Also, from the remaining 507 ecoregions, 325 would possess an area <5% forest cover.

When the mountain definition used by UNEP-WCMC (2002) was used as the basis of this analysis, which includes slope and local elevation range, the following ecoregions would be excluded: from America, Nebraska Sand Hills mixed grasslands and Llanos; from Asia, Alashan Plateau semi-desert, Tarim Basin deciduous forests and steppe; and from Australia, Carpentaria tropical savanna. Also, the of forest area in each ecoregion would be reduced, especially in areas with forested plateaux such as these ecoregions from Africa: Central Zambezian Miombo woodlands, Angolan Miombo woodlands, Southern Miombo woodlands, Zambezian Baikiaea woodlands and Northern Congolian forest-savanna mosaic. These would be reduced by up to 99.8% as in the case of Zambezian Baikiaea woodlands.

Analyses of precipitation indicated that no ecoregions possessing montane forest would be projected to experience extreme conditions (i.e. M>2, Beaumont *et al.* 2011) (**Figures 2.3** and **2.4**). However changes in precipitation below the 2 SD (σ) threshold were projected for many of the ecoregions (68.2% under scenario A2 and 49.3% under scenario B2). For example, under scenario A2, some 23.7% of ecoregions would experience a decrease in precipitation whereas 44.6% would experience an increase in precipitation. Similarly, under scenario B2 49.3% of the ecoregions would experience change, 14.7% exhibiting a decrease and 34.6% an increase in precipitation. Analysis of temperature suggested that this might be a more influential driver of change in montane forests (**Figures 2.5** and **2.6**). Under scenario B2, an increase in temperature is projected for all ecoregions, with 99.5% of the ecoregions experiencing extreme conditions (i.e. M>2, Beaumont *et al.* 2011). Under scenario A2 all of the ecoregions will experience extreme conditions, some (such as Northeastern Congolian lowland forests) experiencing M values of >11 (**Table 2.2** and **Figure 2.6**). The difference between the outputs of scenarios A2 and B2, for the projections of precipitation and temperature, suggests that the degree of change in precipitation between the two would be within ±1M (**Figures 2.7**); with 63.7% ecoregions having the no difference in the projected M value; and for temperature the difference ranges from -0.0124M, in ecoregions such as Sierra de la Laguna pine-oak forests in Mexico, to 3.2M, in the Rwenzori-Virunga montane moorlands ecoregion in Central Africa (**Figures 2.8**).

Some ecoregions with montane forest were projected to experience relatively high change in both temperature and precipitation, namely Eastern Java-Bali montane rain forests, Western Java montane rain forests, Northern Triangle subtropical forests and Northern Triangle temperate forests, all of which are in Asia. These are among the top 25 ecoregions with montane forest at the top of the ranking for both temperature and precipitation (**Tables 2.2** and **2.3**).

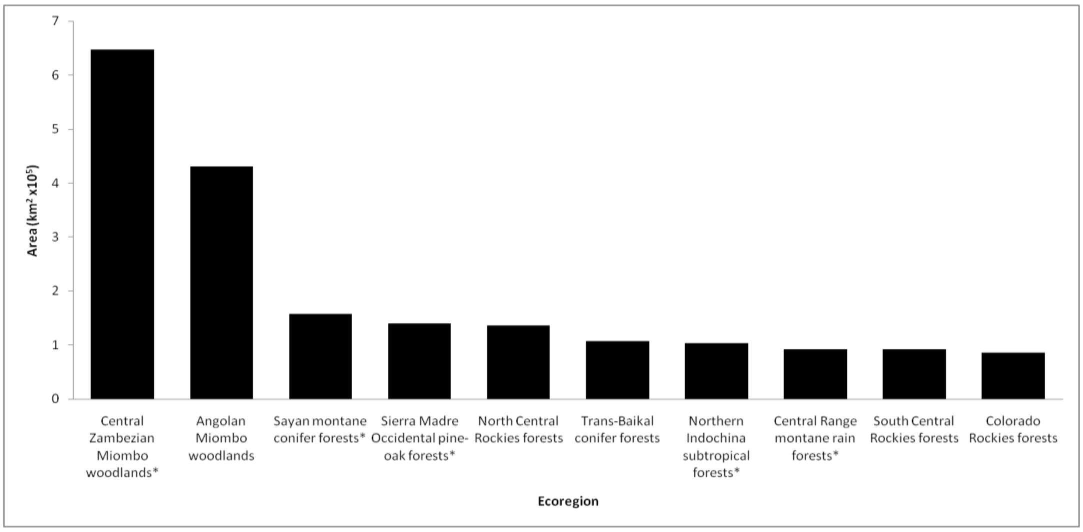

Figure 2.1 Area of montane forest in the ten ecoregions with highest values of forest cover.

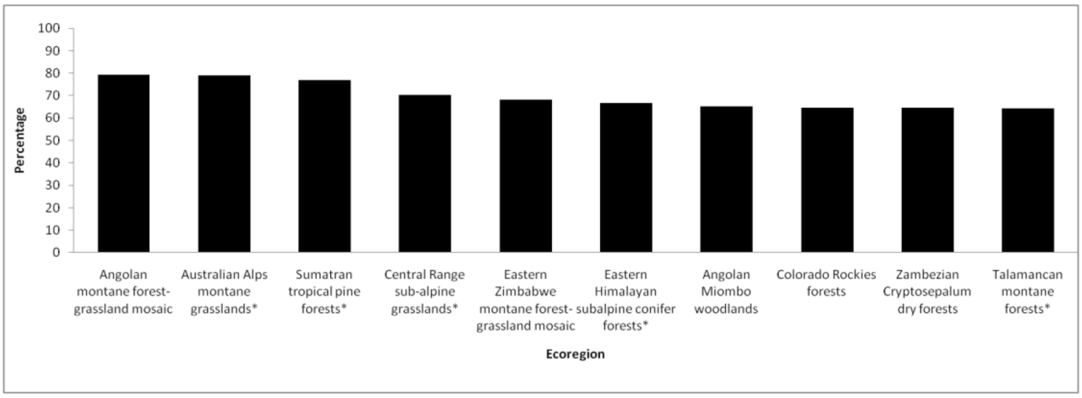

Figure 2.2 Percentage forest cover in the ten ecoregions with highest values of forest cover.

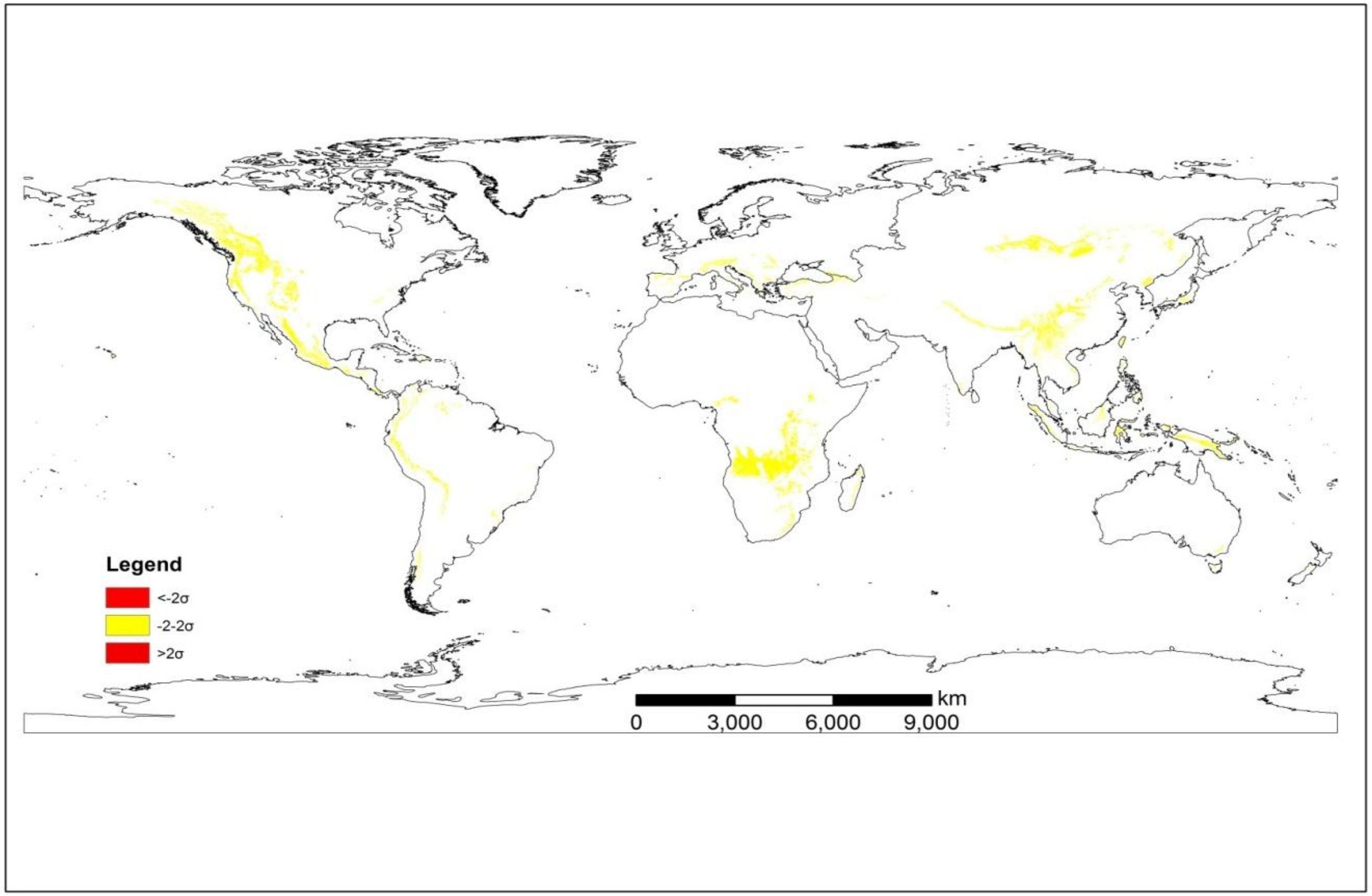

Figure 2.3 Vulnerability of montane forest to projected changes in precipitation. The map illustrates projection of precipitation under the A2 scenario from the HadCM3 model (CERA). Mapped values are Standardized Manhattan Distance (M), which is the distance between the mean (μ) values of precipitation from projected values for 2051–2100 (100 years) and the mean of the 1951–1990 baseline, standardized by the SD (σ) of the baseline climate (for details, see text).

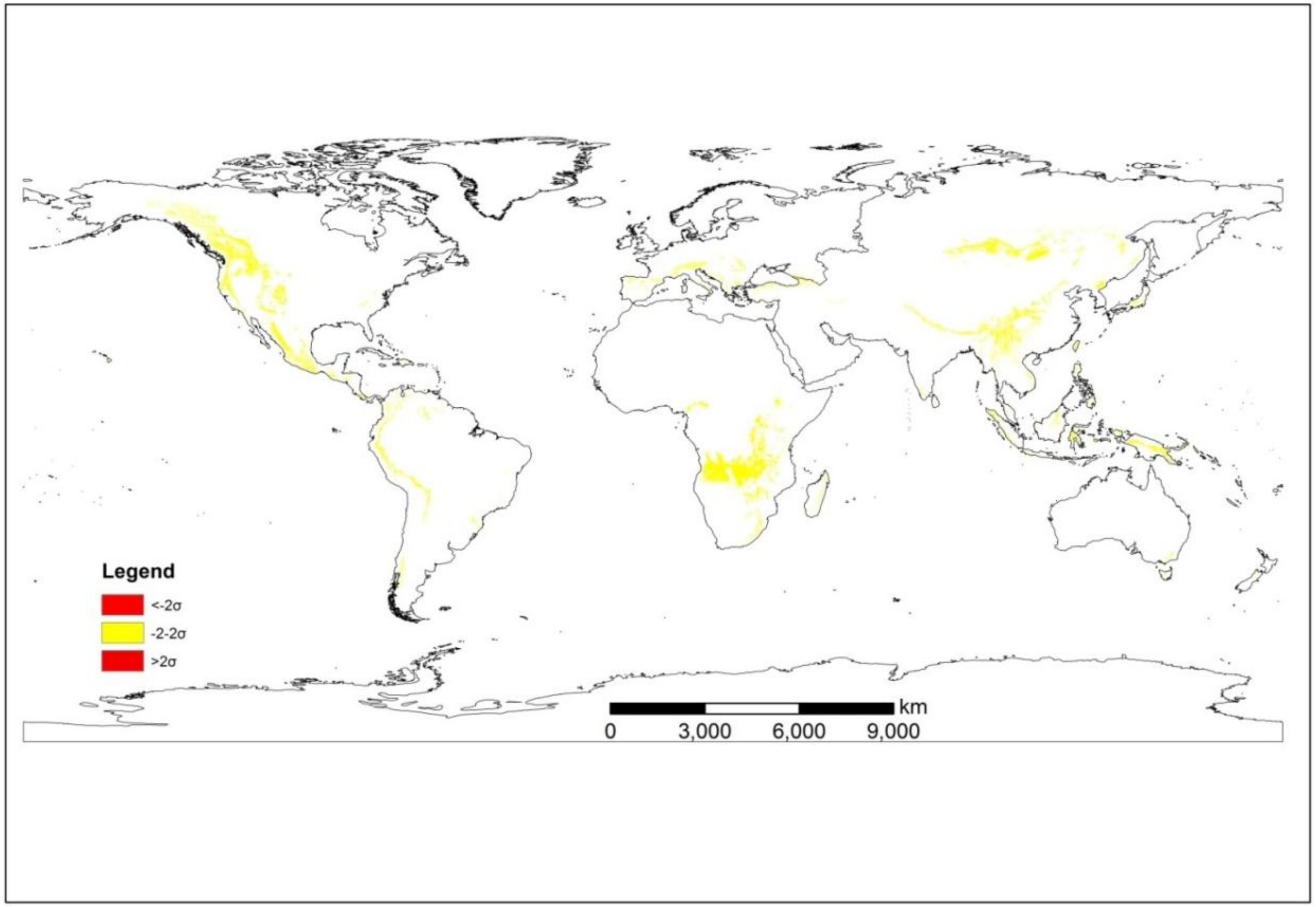

Figure 2.4 Vulnerability of montane forest to projected changes in precipitation. The map illustrates projection of precipitation under the B2 scenario from the HadCM3 model (CERA). Mapped values are Standardized Manhattan Distance (M), which is the distance between the mean (μ) values of precipitation from projected values for 2051–2100 (100 years) and the mean of the 1951–1990 baseline, standardized by the SD (σ) of the baseline climate (for details, see text).

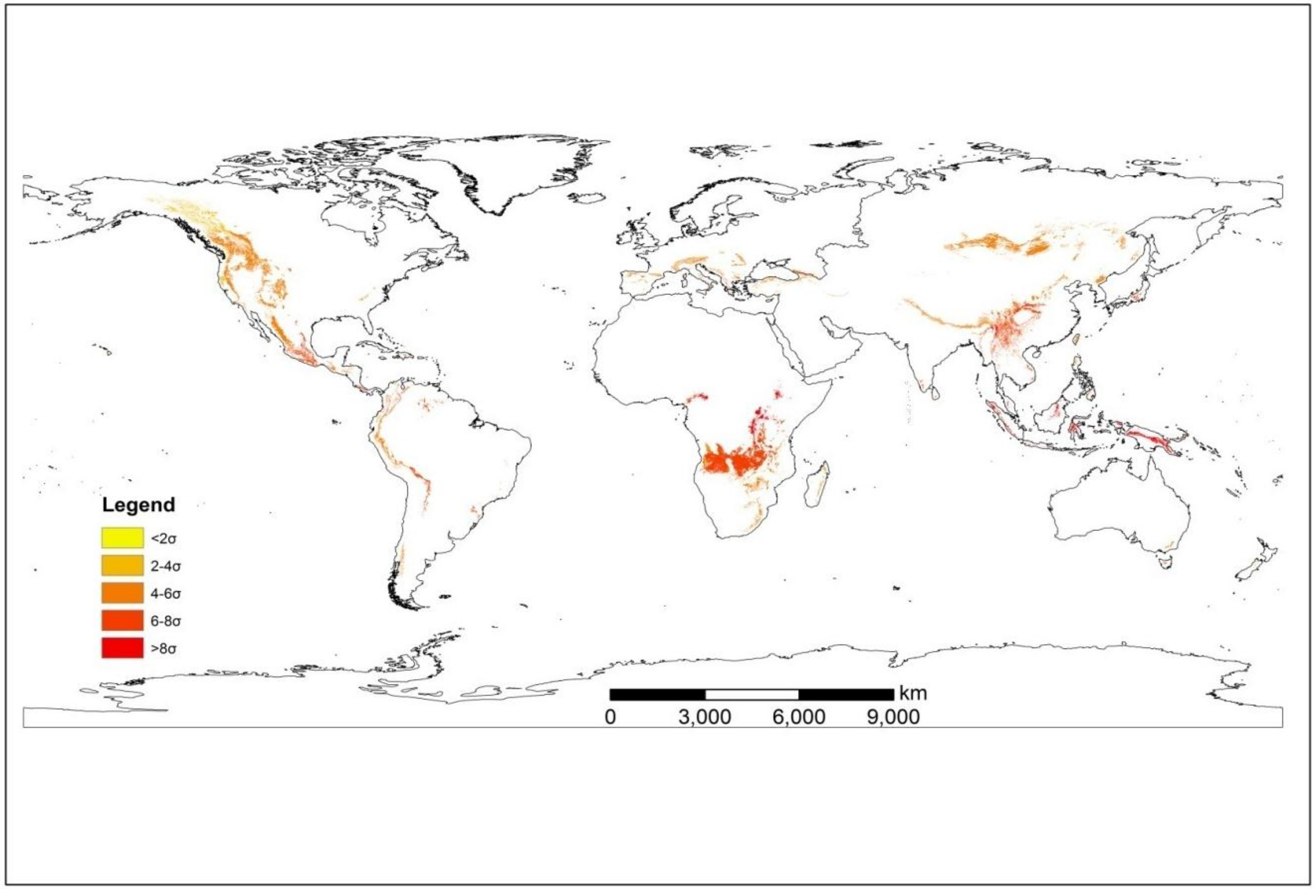

Figure 2.5 Vulnerability of montane forest to projected changes in temperature. The map illustrates projection of temperature under the A2 scenario from the HadCM3 model (CERA). Mapped values are Standardized Manhattan Distance (M), which is the distance between the mean (μ) values of temperature from projected values for 2051–2100 (100 years) and the mean of the 1951–1990 baseline, standardized by the SD (σ) of the baseline climate (for details, see text).

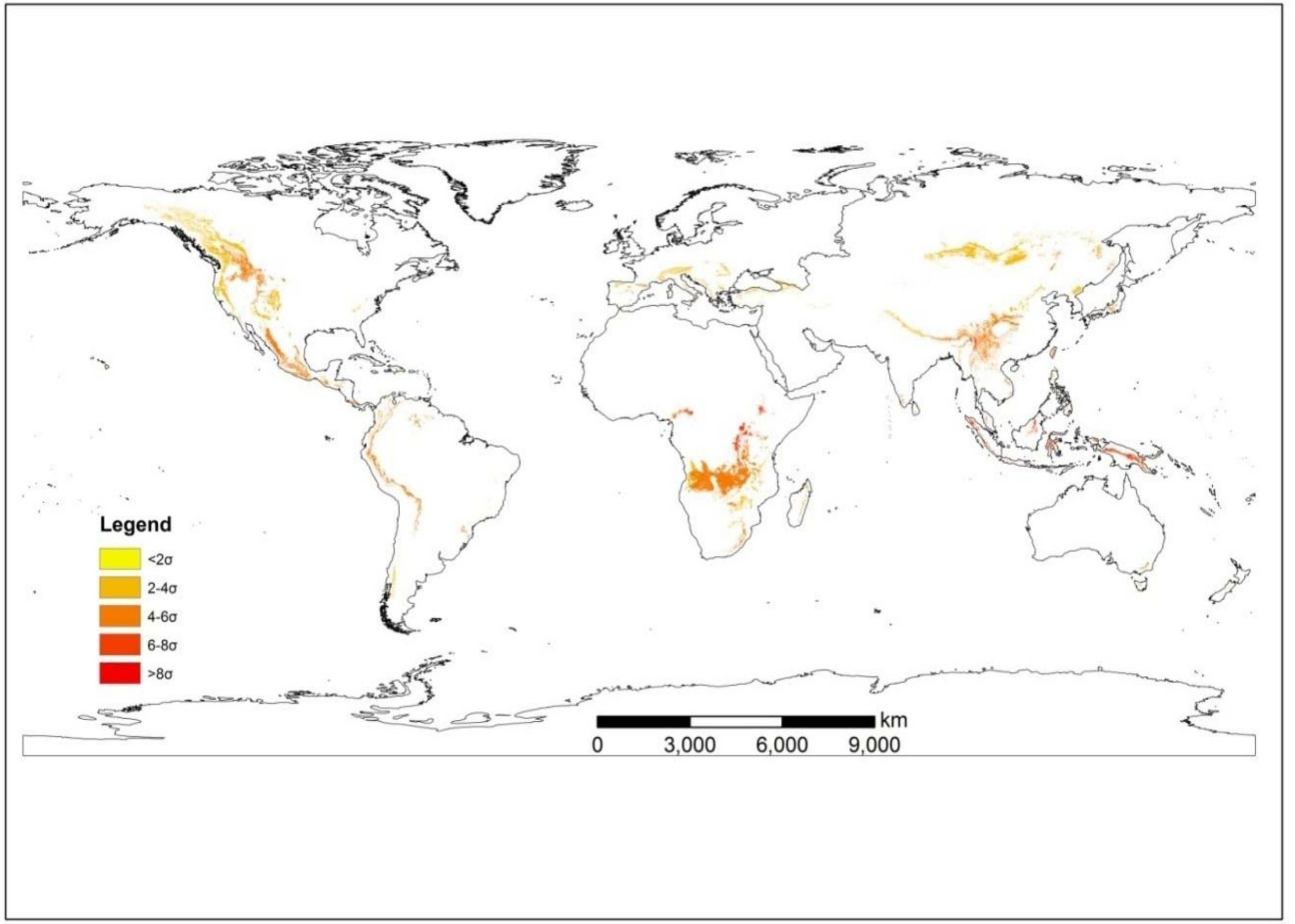

Figure 2.6 Vulnerability of montane forest to projected changes in temperature. The map illustrates projection of temperature under the B2 scenario from the HadCM3 model (CERA). Mapped values are Standardized Manhattan Distance (M), which is the distance between the mean (μ) values of temperature from projected values for 2051–2100 (100 years) and the mean of the 1951–1990 baseline, standardized by the SD (σ) of the baseline climate (for details, see text).

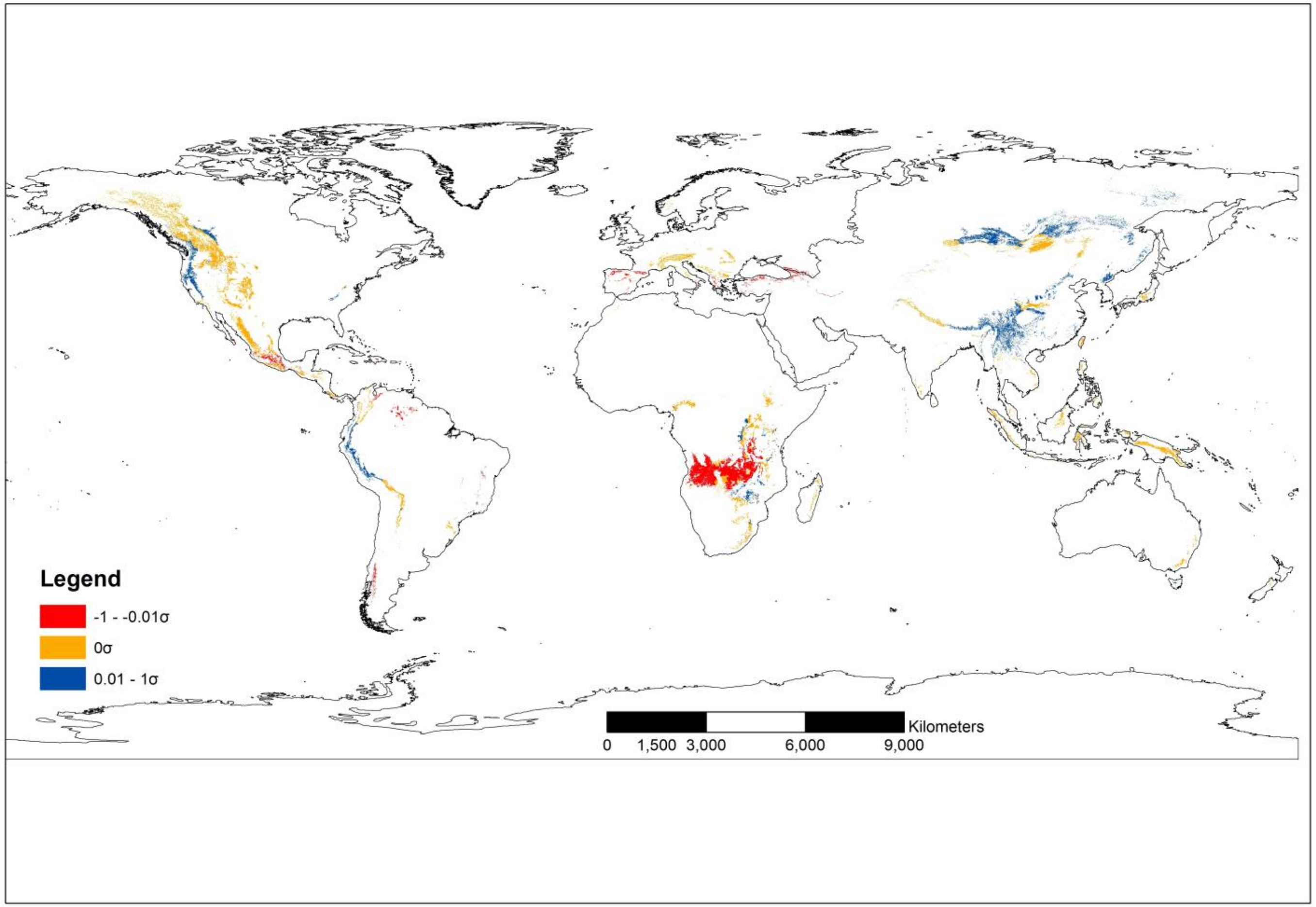

Figure 2.7 Difference in impact of scenario A2 and scenario B2, based on the are Standardized Manhattan Distance (M), which is the distance between the mean (μ) values of precipitation from projected values for 2051–2100 (100 years) and the mean of the 1951–1990 baseline, standardized by the SD (σ) of the baseline climate.

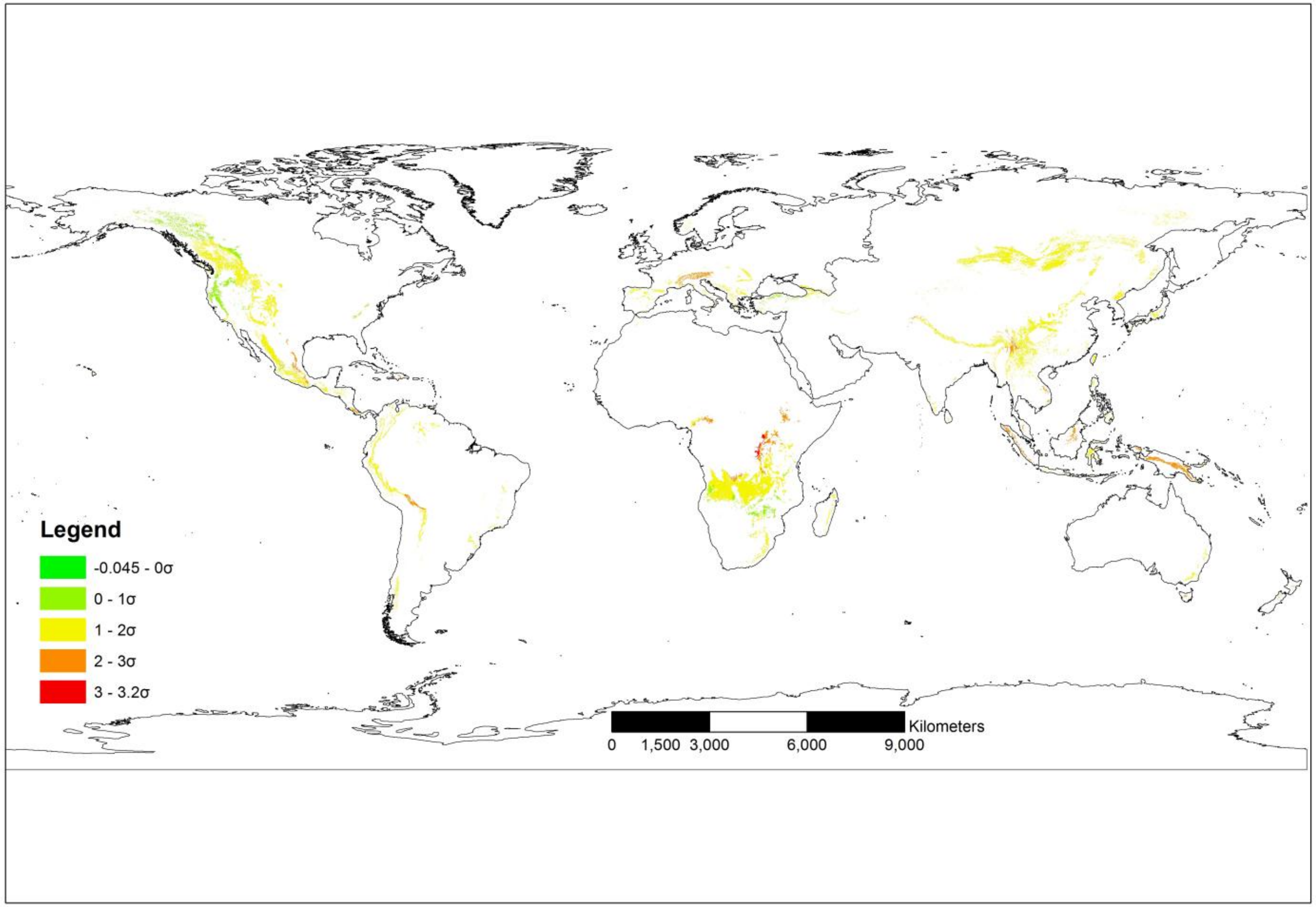

Figure 2.8 Difference in impact of scenario A2 and scenario B2, based on the Standardized Manhattan Distance (M), which is the distance between the mean (μ) values of temperature from projected values for 2051–2100 (100 years) and the mean of the 1951–1990 baseline, standardized by the SD (σ) of the baseline climate.

ID	Ecoregions	Temperature change 2100 A2	Rank A2	Temperature change 2100 B2	Rank B2
30124	Northeastern Congolian lowland forests*	11.07	1	7.87	1
31013	Rwenzori-Virunga montane moorlands*	10.90	2	7.71	2
30101	Albertine Rift montane forests*	10.21	3	7.29	3
30721	Victoria Basin forest-savanna mosaic	9.65	4	6.88	5
30712	Northern Congolian forest-savanna mosaic	9.27	5	6.89	4
40103	Borneo montane rain forests*	9.15	6	6.63	10
10105	Central Range montane rain forests*	8.95	7	6.33	19
10104	Buru rain forests	8.95	8	6.50	13
10107	Huon Peninsula montane rain forests*	8.93	9	6.82	8
40144	Peninsular Malaysian montane rain forests*	8.90	10	6.82	7
40140	Northern Triangle subtropical forests*	8.85	11	6.77	9
40402	Northern Triangle temperate forests*	8.77	12	6.87	6
40112	Eastern Java-Bali montane rain forests	8.69	13	6.57	11
40304	Sumatran tropical pine forests*	8.68	14	6.55	12
31007	Ethiopian montane grasslands and woodlands*	8.64	15	6.44	14

10125	Trobriand Islands rain forests	8.58	16	6.37	16
10127	Vogelkop montane rain forests*	8.56	17	6.15	21
30112	Ethiopian montane forests	8.40	18	6.28	20
11002	Central Range sub-alpine grasslands*	8.39	19	6.07	22
40159	Sumatran montane rain forests*	8.38	20	6.33	18
40167	Western Java montane rain forests*	8.36	21	6.39	15
10124	Sulawesi montane rain forests*	8.27	22	6.36	17
41001	Kinabalu montane alpine meadows*	8.10	23	5.81	31
10120	Southeastern Papuan rain forests	8.08	24	5.62	38
10116	Northern New Guinea montane rain forests*	8.08	24	6.06	23

Table 2.2 List of the Andean ecoregions with montane forest that are most vulnerable to climate change, illustrating the top 25 ecoregions ranked according to the highest M values for projected temperature change under scenario A2. Those ecoregions that belong to the 'Global 200' (Olson and Dinerstein 2002) are indicated by an asterisk.

ID	Ecoregions	Precipitation change 2100 A2	Rank A2	Precipitation change 2100 B2	Rank B2
60169	Pantepui*	-2.00	1	-1.50	1
60124	Guianan Highlands moist forests*	-2.00	1	-1.50	1
51116	Ogilvie-MacKenzie alpine tundra*	1.50	3	1.50	1
51111	Interior Yukon-Alaska alpine tundra*	1.50	3	1.50	1
50617	Yukon Interior dry forests*	1.50	3	1.50	1
60175	Venezuelan Andes montane forests	-1.50	3	-1.00	7
61005	Cordillera de Merida paramo	-1.50	3	-1.00	7
30701	Angolan Miombo woodlands	-1.50	3	-0.50	19
30723	Western Congolian forest-savanna mosaic	-1.50	3	-0.50	19
40140	Northern Triangle subtropical forests*	1.50	3	0.50	19
40112	Eastern Java-Bali montane rain forests	-1.00	11	-1.00	7
40167	Western Java montane rain forests	-1.00	11	-1.00	7
30203	Zambezian Cryptosepalum dry forests	-1.00	11	-1.00	7
30724	Western Zambezian grasslands*	-1.00	11	-1.00	7
61007	Santa Marta paramo*	-1.00	11	-1.00	7
60305	Hispaniolan pine forests*	-1.00	11	-0.50	19
80433	Pyrenees conifer and mixed forests*	-1.00	11	-0.50	19
31001	Angolan montane forest-grassland mosaic*	-1.00	11	-0.50	19
40402	Northern Triangle temperate forests*	1.00	11	0.50	19
80102	Yunnan Plateau subtropical evergreen forests*	1.00	11	0.50	19
40131	Mizoram-Manipur-Kachin rain forests	1.00	11	0.50	19

40137	Northern Indochina subtropical forests	1.00	11	0.50	19
80516	Nujiang Langcang Gorge alpine conifer and mixed forests*	1.00	11	0.00	105
80512	Khangai Mountains conifer forests*	1.00	11	0.00	105
80414	Changbai Mountains mixed forests*	1.00	11	0.00	105

Table 2.3 List of the Andean ecoregions with montane forest that are most vulnerable to climate change, illustrating the top 25 ecoregions ranked according to the highest values of M for projected precipitation change under scenario A2. Those ecoregions that belong to the 'Global 200' (Olson and Dinerstein 2002) are indicated by an asterisk.

2.4 Discussion

These analyses show that montane forests are very likely to be affected by projected changes in climatic conditions, and particularly by increases in temperature. The results indicate that all montane forests are likely to be highly vulnerable to changes in temperature under scenario A2. Although up to 68% of montane forest ecoregions are expected to experience changes in precipitation according to the A2 scenario, none exceeded the 2SD threshold. The results also suggest that in terms of temperature, Africa and Asia are the two continents with the most vulnerable montane forest ecoregions, whereas South America and Asia are the continents with montane forest ecoregions that will experience the largest changes in precipitation. Although precipitation changes are less evident than temperature changes in the analyses presented here, potential interactions between precipitation and temperature could lead to further changes in water balance (Fung *et al.* 2011; Pounds *et al.* 2006).

It should be noted that an M value threshold of 2SD can perhaps best be viewed as an arbitrary threshold, which does not necessarily correspond with anticipated ecological impacts. However, a number of authors have demonstrated that 2SD can correspond to significant ecological impacts (Beaumont *et al.* 2011; Palmer and Raisanen 2002). This value was originally established as the deviation of baseline climatic data for Europe between the years 1500 and 1900 (Luterbacher *et al.* 2004), and from extreme climatic conditions such as the those experienced in Europe during the summer 2003, when temperatures increased by ~3.8°C, corresponding to an excess of up to 5 standard deviations (Schär *et al.* 2004). Schär *et al.* (2004) have shown that exceeding this threshold by 5SD, as experienced in Europe in 2003, ecosystems become vulnerable and changes in the basic performance become affected, such as reduction of primary productivity (Ciais *et al.* 2005).

Many uncertainties are associated with broad-scale analyses such as that presented here, relating to the basic choice of procedures, evaluation metrics, time, scale, global climate models (GCM), emissions scenarios and the amount and character of the predictor variables themselves (Synes and Osborne 2011). Uncertainties in climate modelling are greater when predicting precipitation patterns, as aspects such as wind, clouds, precipitation regimes and small-scale processes cannot always be easily included at the global scale (Randall *et al.* 2007). No single analysis can capture all ecological risks associated with climate change (Williams *et al.* 2007), and using only two climate variables does not provide a complete indication of the projected likelihood of major impacts of climate change on ecosystems. However, over-fitting and variable redundancy can also become an area of concern as more predictor variables are included (Beaumont *et al.* 2005). So, although methods to assess the impacts of climate change vary considerably, a number of studies have suggested that climate change will lead to considerable declines in species survival and perhaps lead to mass extinction (Bellard *et al.* 2012). Although widespread impacts of climate change on montane forest were projected in the current analysis, intra-annual or inter-annual climate variability was not explicitly examined, which may mean that such impacts were underestimated. The current analysis was also limited by the fact that a number of variables that determine the distribution of montane forests were not analysed here, including relative humidity (Foster 2010), length of the dry season (Aiba *et al.* 2010) and mean annual light intensity (Rüger *et al.* 2010).

Studies such as that of Williams *et al.* (2007), that employed the standardized Euclidean distance (SED) to calculate the dissimilarities between 20th- and 21st-century climates for climatic models for A2 and B1 scenarios for mean surface air temperature and precipitation for June–August and December–February, corroborates the fact that projections of climate change will put climates, particularly those occurring in mountains, at risk of disappearing where high SED was found. In particular, they identified the climates occurring in montane ecosystems such as those on the Colombian and Peruvian Andes, in Central America, the African Rift Mountains, the Zambian and Angolan Highlands, the Cape Province of South Africa, southeast Australia, portions of the Himalayas, and the Indonesian and Philippine Archipelagos. Many of these areas were also identified as highly vulnerable in this assessment, as is the case of the Albertine Rift montane forests ecoregion, which had an M value for temperature of 10.21 under scenario A2 and 7.29 under scenario B2 (**Table 2.1**).

Of the 211 ecoregions considered in this study, 141 of them are considered a global conservation priority (the Global 200) and 151 ecoregions are part of 27 biodiversity ‘hotspots’ identified by Myers *et al.* (2000). Ecoregions scoring high in the ranks for vulnerability to climate change for being associated with the some of the largest M values (i.e. >9) under projections of the scenario A2 for temperature, such as Rwenzori-Virunga montane moorlands and Albertine Rift montane forests, are also part of the Eastern Afromontane biodiversity ‘hotspot’. The projected climate change impacts documented here could therefore have significant implications for conservation of biodiversity at the global scale.

Generally the direct effects of climate change on forests are related to phenology and physiological processes that affect shifts in the boundary of the geographical range and population dynamics of tree species (Abbott and Le Maitre 2010). These shifts may be more likely to occur in montane forests as many of the tree species occurring in these forests do not have long range dispersal; are situated at the edge of their ranges; are geographically localised; are genetically impoverished; are slow reproducers; or are highly specialised (Price and Neville 2003). Prolonged high temperatures may expose species to increased water stress, leading to increased tree mortality (Allen *et al.* 2010) and the reduction of population sizes, threatening those species with few and small populations (Abbott and Le Maitre 2010). Increasing temperature increases evaporation, which in turn increases water loss through transpiration, increasing the risk of hydraulic failure in specific tree species (Allen *et al.* 2010). Also the carbon storage of trees is affected as the carbohydrate consumption required for respiration is closely associated with temperature (Norby *et al.* 1992); root mortality also increases as soil temperature rises (Majdi and öhrvik 2004).

The potential impact of increased temperatures could also lead to widespread increases in the extent and frequency of vegetation die-off (Adams *et al.* 2009). The occurrence and frequency of fires will not only be increased by high temperatures but also by elevated CO_2 concentration, as woody tissue is generated at a higher rate. As a result, dry woody tissue and leaf litter will increase, leading to increased occurrence of fires, which in turn will increase the abundance of those species that are tolerant of fire (Abbott and Le Maitre 2010) and limit the persistence of plant populations dependent on seed banks (Ooi 2012) or have thinner protective layers of bark (Cochrane 2003). Tree mortality could increase by changes in temperature alone independently of other changes in the ecosystem water balance (Adams *et al.* 2009), as increases in temperature will be directly related to heat exhaustion and the inhibition of the abilities of the species to cope with the heat. Carbon storage in soils will also be likely to change due to fires, particularly in tropical montane forests, Montane forest peat soils are reserves of terrestrial organic carbon, maintained by conditions occurring in montane ecosystems such as cold temperatures, slow decomposition, high humidity, and large litterfall (Román-Cuesta *et al.* 2011). Fires directly release a large proportion of carbon into the atmosphere. Furthermore, carbon losses continue to appear through time as a result of the decay process from the biomass of killed trees (Cochrane 2003). Potential carbon recovery would take place after a fire as species resprout, but many of the species may never recover as are not adapted to withstand fire and have a slow recovery rate (Román-Cuesta *et al.* 2011).

Changes in climate could potentially interact with other factors influencing montane forest. Montane forests could be particularly vulnerable to invasion by non-native species (Pauchard *et al.* 2009), and interactions between climate change and species invasion could increase the likelihood of the establishment of these species, by enhancing their reproductive means, their survival and their competitive power against the native species (Thomas *et al.* 2004; Thuiller 2007). Increased habitat loss has been forecasted for cooler environments, such as those present in montane forests, as species niches in colder regions show a linear relationship with temperature gradients (Thuiller *et al.* 2005). Although vegetation in montane ecosystems will be constrained by effects of temperature, there are also other factors that will influence their distribution such as radiation, wind, storminess and water availability (Beniston 2003), which could also influence their vulnerability to invasive species.

In addition to affecting trees directly, climate change could also affect the communities of other species associated with forest habitats. Climate change may be detrimental for tropical organisms, particularly for those animals that are physiologically specialised with respect to temperature and have poor acclimation capacity. Many such species are particularly vulnerable as they rely on constant shade and are not adapted to warmer temperatures, and may display few behavioural options to evade increases in temperature (Tewksbury *et al.* 2008). Birds and mammals that are highland specialists might be particularly vulnerable to global warming because of their small geographic ranges and high energetic and area requirements (Laurance *et al.* 2011). Also, ectothermic vertebrates such as amphibians and reptiles that have high species richness may also be vulnerable to increases in temperature (Laurance *et al.* 2011). Pounds *et al.* (2006) also concluded that the potential outbreak of pathogens that are implicated with the extinction of amphibians will be encouraged by warmer conditions.

Although most of the consequences mentioned above reflect the potential negative impacts of climate change, interactions between temperature, precipitation and seasonality can lead to the creation of novel climates (Williams *et al.* 2007), where the species may be able to migrate to and thrive in, for example in cool and dry temperate regions which will be likely to experience increased moisture availability are expected to exhibit increases in richness (Holzinger *et al.* 2008), but this depend on the dispersal capability and the patchiness of the species in the landscape (Menéndez *et al.* 2006).

2.5 Conclusions and conservation implications

These results show that montane forests are at high risk of extinction under projected climate change scenarios. Climate change is now regarded as a current threat to biodiversity along with deforestation, land use change, habitat fragmentation, pollution and invasion by non-native plant and animal species (Thuiller 2007). The potential migration and adaptation of species will not only be jeopardised by dispersal barriers but by the lack of suitable habitats which they could occupy, as a result of climate change (Williams *et al.* 2007). Montane forest species could potentially have no place to go.

Some of the ecoregions in this study are characterised by having a naturally occurring low canopy cover and fragmented forest structures. Others are meant to have a high canopy cover but may have shown low canopy cover in this assessment, due to the consequences of human activities mentioned above. This may imply that these latter ecoregions may be further affected by the potential effects of climate change. Therefore, the relative impact of climate change on montane forests is likely to vary in relation to geographic location and the forest composition, as well as with the intensity of land use change.

As suggested by Scheffer *et al.* (2001), in order to reduce the risk of changes in the state of ecosystems in the face of increasing climate change, it is necessary to build and maintain resilience by addressing the gradual changes that affect ecosystem stability, such as land use, nutrient stocks, soil properties and biomass of long-lived organisms. This can potentially be achieved by designating new protected areas and undertaking low-level habitat management to reinforce species' intrinsic dispersal and migration mechanisms (Dawson *et al.* 2011). The encouragement and participation of schemes such REDD+, which encourages afforestation and diversification with locally valued tree species, could be of value to encourage forest protection at the same time as providing local developments for humans.

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3 Evaluation of the conservation status of montane forest in the tropical Andes

3.1 Abstract

The tropical Andes is a unique region with high habitat diversity, resulting from broad altitudinal and latitudinal gradients. Andean montane forests are currently a major global conservation priority owing to their high species richness and high level of endemism, and are considered one of the least known ecosystems in the tropics. These forests are fundamental for the provision of a variety of ecosystem services, including the regulation of regional climate and the capture and storage of carbon. This article provides an overview of the conservation status of tropical Andean montane forests and the challenges it entails. It also provides information on threats, including the identification of those that are most likely to be responsible for increasing the extinction risk of species. It highlights the need for more information on the conservation status of species to identify future priorities for conservation in the region. The recent collaborative initiative "Red List and conservation planning for montane tree species of the tropical Andes," with delegates from several countries in the region, will provide a solid basis for the development of policies and management responses aimed to reduce deforestation and species loss in these forests, including actions to promote the creation of protected areas, forest restoration and sustainable forest management.

3.2 Introduction

Montane forests in the tropical Andes are a global conservation priority, because of their high levels of biodiversity and endemism (Bush *et al.* 2007; Olson and Dinerstein 1997; Pennington *et al.* 2010), and their important role in providing different ecosystem services to the region (Anderson *et al.* 2011; Balvanera 2012). Furthermore, these ecosystems are considered among the least known and most threatened in the tropics (Ataroff and Rada 2000; Bubb *et al.* 2004; Gentry 1995; Kessler 2000; Price *et al.* 2011; Stadtmüller 1986), associated with high deforestation rates, mainly as a result of conversion to agriculture and logging. Research is sparse, often carried out at national level (Armenteras *et al.* 2007;

Grubb and Whitmore 1966; La Torre-Cuadros *et al.* 2007), with few studies undertaken at the regional level (Cuesta *et al.* 2009; Herzog *et al.* 2011). This document provides an overview of the current state of knowledge of montane forest in the tropical Andes, and the importance of biodiversity at regional and global scales, together with identification of the main threats to which biodiversity is subjected in the region. The paper also highlights the need for more information on the conservation status of species, to identify future conservation priorities in the region.

3.3 Distribution and status of knowledge of montane forests

The Andes are the longest mountain chain in the world, stretching over seven thousand linear kilometres through tropical, subtropical and temperate latitudes. This is a unique region with multiple ecosystems types, reflecting high species and ecosystem diversity as a result of large altitudinal and latitudinal gradients (Josse *et al.* 2003; Young *et al.* 2002; Young 2007). The tropical Andes cover an area of approximately 1,542,644 km^2 in Colombia, Venezuela, Ecuador, Peru, Bolivia, part of northern Argentina and Chile, and contain most of the montane forests of the Andean region (Josse *et al.* 2009; Young *et al.* 2002). In Venezuela, the Andes extend from the depression of Barquisimeto in Lara state, to the state of Tachira on the border with Colombia. In Colombia the Andes are divided into three ranges (Oriental, Central and Occidental), corresponding to 24.52% of the area of the country, 60.47% of which are ecosystems that have been transformed by human activity (Cabrera and Ramírez 2007; Rodríguez *et al.* 2006). Throughout Ecuador, the Andes consists of two parallel mountain ranges (Occidental and Oriental or Real) and lose altitude as they cross Peru until the Huancabamba depression, where the elevation of the Andes again increases, forming the Cordillera Occidental and Central until Junin. Further south, these converge again as the main Andean chain extends into Chile and Argentina (Josse *et al.* 2009). In Bolivia, the Andes have two chains: Occidental and Oriental.

The term forest is defined here as an area with trees where the canopy cover is 10% or more, following Schmitt *et al.* (2009b). Montane forests of the tropical Andes typically have very dense vegetation, with tree heights between 10 and 35 m, often with abundant lichens and mosses. Seasonal forests have species adapted for 3 to 5 months of drought (Josse *et al.* 2009). Montane forests comprise two major landscapes recognized in the North and Centre of the Andes by Josse *et al.* (2009): cloud forests and seasonal forests, which are each characterized by the presence of multiple ecosystems types, mostly rainforest and/or pluviseasonal forest. Montane forests are also distributed among three of the five phytoregions specified by Josse *et al.* (2009): the Northern Andes, the Yungas and Bolivian-Tucuman forests, which differ in their floristic and biogeographic characteristics. For example the Yungas are biogeographically similar to the forest formations of Peru but are very different from the Bolivian-Tucuman forests of the eastern slopes of the region (Ibisch *et al.* 2004), which are also known as *las Yungas* (Cabrera and Willink 1973). Within these three phytoregions there is a wide variety of forested ecological communities as shown in **Table 3.1**. The montane forests of the tropical Andes reach their southernmost limit in the Northwest of Argentina (22°- 29° S), located above 1,500 m a.s.l. and occupying about 2.1 million ha in the country.

The main montane forest ecoregions in the tropical Andes, according to Olson *et al.* (2001) are: Bolivian Yungas, Cauca Valley montane forests, Cordillera Oriental montane forests, Cordillera Real Oriental montane forests, Magdalena Valley montane forests, Northwestern Andean montane forests, Peruvian Yungas, Southern Andean Yungas and Venezuelan Andes montane forests (**Figure 3.1**). The threshold elevation of montane forests varies geographically. In the high mountains of the interior this transition typically occurs at an altitude of 1200-1500 m a.s.l., but it can occur at much lower altitudes (Bruijnzeel 2002). The constant moisture on both external sides of the Andes is due to winds from the Pacific Ocean and the Atlantic, while on internal slopes, the bimodal patterns that characterize the wet-dry inter-Andean valleys is attributable to the rain shadow effect (Kattan *et al.* 2004) and daily cycles of atmospheric circulation (Killeen *et al.* 2007b).

Northern Andes	Central Andes	Southern
Andes del Norte	Bolivian-Tucuman	Yungas
Eastern sub-Andean ridge montane pluvial forest	Bolivian-Tucuman upper montane seasonal evergreen forest	Yungas upper montane seasonal evergreen forest
Northern Andes upper montane evergreen forest	Bolivian-Tucuman inter-Andean foothill wash woodland	Yungas upper montane pluvial forest
Northern Andes upper montane Polylepis pluvial forest	Bolivian-Tucuman montane Alder forest	Yungas ridge seasonal evergreen dwarf forest
Cordillera del Cóndor lower montane pluvial forest	Bolivian-Tucuman montane Podocarpus forest	Yungas lower montane seasonal evergreen forest
Northern Andes lower montane pluvial forest	Bolivian-Tucuman montane semideciduous forest	Northern Yungas lower montane semideciduous forest
Northern Andes lower montane seasonal evergreen forest	Bolivian-Tucuman inter-Andean foothill riparian forest	Southern Yungas lower montane semideciduous forest
Northern Andes lower montane semideciduous forest	Transitional sub-Andean Bolivian-Tucuman Yungas forest	Southern Yungas lower montane xeric forest
Eastern sub-Andean ridge montane pluvial forest	Sub-Andean Bolivian-Tucuman moist forest	Yungas upper montane Polylepis pluvial forest
Northern Andes montane pluvial forest	Lower sub-Andean Bolivian-Tucuman semideciduous forest	Yungas upper montane Polylepis seasonal evergreen forest
Northern Andes montane seasonal evergreen forest	Upper sub-Andean Bolivian-Tucuman semideciduous forest	High Andean Yungas Polylepis pluvial forest
Northern Andes foothill semideciduous forest	Bolivian-Tucuman dry montane riparian forest	Yungas montane pluvial forest
Sub-Andean transitional forest of the Llanos del Orinoco		Yungas montane seasonal evergreen forest
Cordillera del Condor sandstone plateau pluvial forest		Yungas montane semideciduous forest
Andean lowland forests and shrublands paramunos		Yungas lower montane pluvial (palm dominated) forest
Premontane rain forest of the Choco-Darien		

Table 3.1 Examples of ecological communities occurring in the humid and pluvi-seasonal forests of northern and central Andes. Adapted from Josse *et al.* (2011). Argentinean forests are excluded.

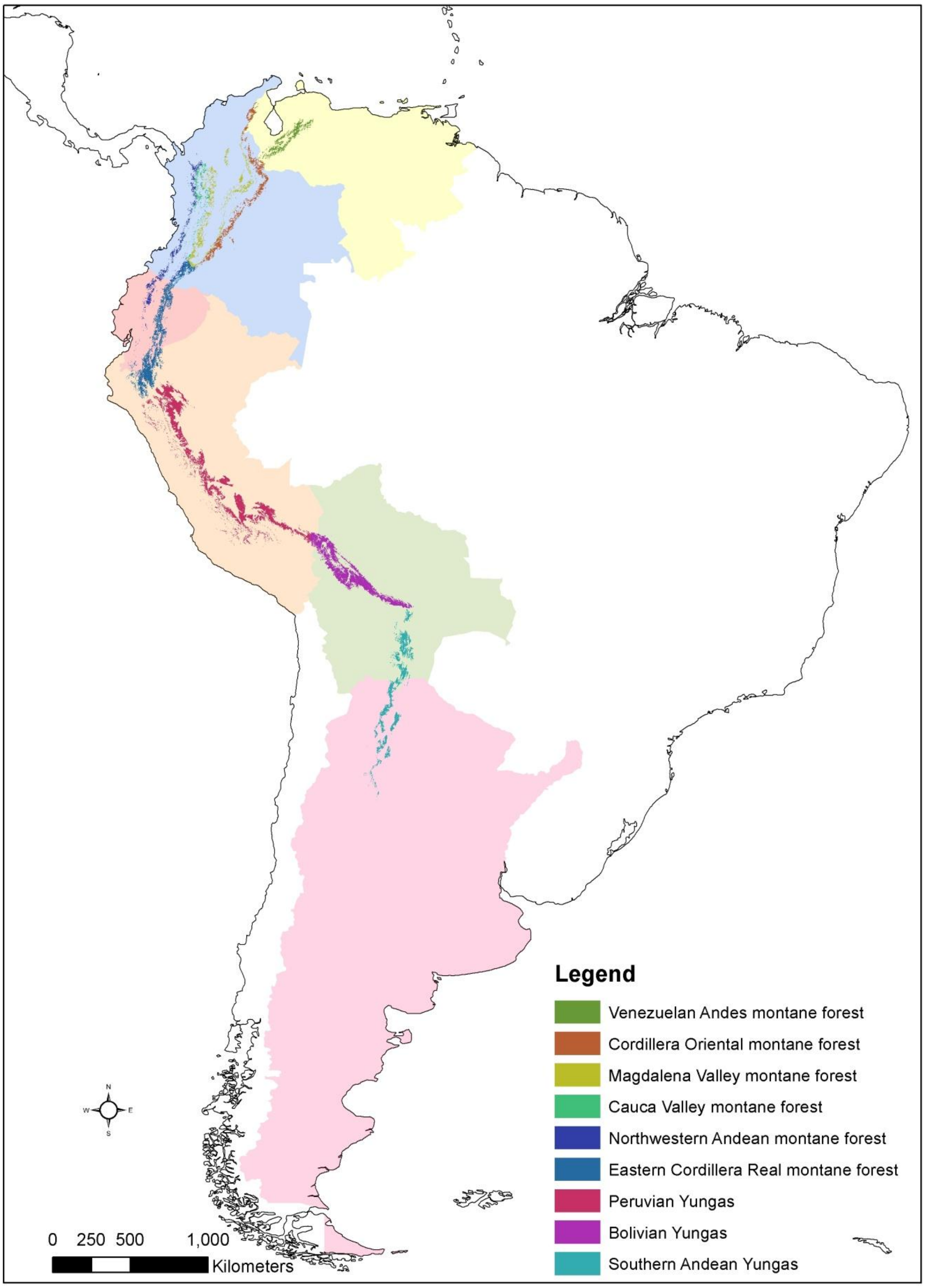

Figure 3.1 Montane forest in the Andean region with more than 10% canopy cover in 2005, as identified by MODIS imagery (Schmitt *et al.* 2009a) limited to altitudes ≥ 1,500 m a.s.l., and the different ecoregions present in the area (Olson *et al.* 2001) (see 'Legend' on Figure).

3.4 Importance of biodiversity

Montane forests of the tropical Andes contain the largest concentration of species with restricted distribution in South America (Dinerstein *et al.* 1995; Latta *et al.* 2011), which is manifested in the high number of endemic species of fauna and flora (Jørgensen 2011; Myers *et al.* 2000; WWF and IUCN 1997). Birds (Fjeldså and Irestedt 2009; Latta *et al.* 2011), mammals (Grenyer *et al.* 2006), amphibians (Grenyer *et al.* 2006), insects (Brehm *et al.* 2005), bryophytes (Churchill 1996, 2009) and vascular plants (Brooks *et al.* 2002; Joppa *et al.* 2013; Kier *et al.* 2005; Myers *et al.* 2000; Pennington *et al.* 2010) are each characterised by high levels of species richness and endemism (**Table 3.2**). This exceptionally rich biodiversity has been attributed to three m0061jor historical factors: 1) the lifting of the Andean ridges in a complex series of orogenic processes, 2) the connection with North America through the Isthmus of Panama, which promoted biotic exchange, and 3) climatic fluctuations during the Pleistocene, which led to fragmentation and isolation of populations, with later events of speciation and radiation (Hughes and Eastwood 2006).

Secies	Plants	Mammals	Birds	Reptiles	Amphibians	Total
Total	45000	414	1666	479	830	48389
Total endemics	20000	68	677	218	604	21567

Table 3.2 Estimated number of species within the "*hotspot*" of the tropical Andes identified by Myers *et al.* (2000). This "*hotspot*" includes all ecosystems of the Andes: paramos, puna, grasslands, montane forests and dry forests.

Collectively, the montane forests of the tropical Andes are considered to be global conservation priorities, both as 'hotspots' of global biodiversity (Myers *et al.* 2000) and as priority ecoregions (Olson and Dinerstein 1997). The tropical Andes contain has been identified as a priority for the conservation of birds, defined in terms of species richness, threat and endemism (Orme *et al.* 2005). These forests also are considered as a Centre of Plant Diversity (Davis *et al.* 1997). As shown in **Table 3.2**, more than 20,000 plant species are believed to be endemic to the tropical Andes, and although efforts are continuing to produce a complete checklist of plant species (e.g León *et al.* 2006), there are still many species that remain undescribed

(Honorio and Reynel 2011). Knowledge gaps are substantial, even at a basic taxonomic level. Interactions between species and their contribution to ecosystem functioning have been little explored in the region. Factors determining vulnerability, such as population density, biological, ecological and physiological needs are poorly understood (Tiessen 2011). Although a number of research initiatives conducted in individual countries have enabled the development of knowledge of these forests in terms of the composition of their flora and fauna, in terms of dynamics (e.g. impacts of human disturbance and climate change) they are less known and there are still many gaps to fill. Therefore the montane forests of the tropical Andes are still considered among the least known ecosystems in the tropics (Ataroff and Rada 2000; Bubb *et al.* 2004; Gentry 1995; Kessler 2000; Stadtmüller 1986). Particular conservation attention has been paid to cloud forests, which represent a subset of montane forests occurring in sites of exceptional humidity, associated with high cloud cover. In general, most of the cloud forests that still exist are small areas or remnants (Aldrich *et al.* 1997; Bubb *et al.* 2004; Wuethrich 1993). It is estimated that a large part of the cloud forests in the northern Andes have been lost (Hamilton 1995; WWF and IUCN 1997) as a result of anthropogenic activity. Other specific forest types such as those of *Polylepis* spp. are also considered to be highly threatened; 15 species of this genus have been classified as Vulnerable under the IUCN Red List criteria (Jameson and Ramsay 2007), and *Polylepis* forests also contains several species of birds that are threatened worldwide.

3.5 Threats and impacts

Several factors have contributed to the loss and degradation of these forests, which continue to be subjected to processes of exploitation, colonization, deforestation, fragmentation and resource extraction. Agricultural expansion, opening of new roads, lack of planning in mining and oil extraction, the establishment of illicit crops (Dávalos *et al.* 2011), social inequality (poverty), population growth and new settlements, have all contributed to their loss. One of the principal threats to Andean forests is deforestation, which is attributable to the complex interaction of different social, cultural, political, technological, economic and ecological factors (Geist and Lambin 2002) that occur in the region.

In Colombia, between 1985 and 2005 there was a change in forest cover in the montane forests from 7,335,125 to 6,405,591 ha (Armenteras *et al.* 2011), representing an annual deforestation rate of 0.63%, which is higher than reported nationally (**Table 3.3**). In Venezuela, estimates of deforestation have primarily focused on lowland forests; however, it is estimated that by 2005 the cloud forests had been reduced by 32% (Rodríguez 2005). In Ecuador, although there are no specific figures available for montane forest, the country as a whole exhibits the highest deforestation rates of the region (**Table 3.3**; Fundacion Pachamama (2010)). The deforested area of the Peruvian Yungas owing to agricultural expansion amounted to 1,452,955 ha by 2001, representing 9.65% of the area of the ecoregion (Tovar *et al.* 2010). In Bolivia, Carretero *et al.* (2003) considered that the Yungas are predominantly in relatively good condition, although it has also been reported that a high percentage of Andean forests in Bolivia have been severely affected by human impacts (Ibisch 2002).

Country	Forest area (1000 ha)		Annual change rate	
	2005	2010	2005-2010 1000ha/yr	%
Argentina	30599	29400	-240	-0.80
Bolivia	58734	57196	-308	-0.53
Colombia	61004	60499	-101	-0.17
Ecuador	10853	9865	-198	-1.89
Perú	68742	67992	-150	-0.22
Venezuela	47713	46275	-288	-0.61

Table 3.3 Percentage of forest cover change at the national scale, adapted from FAO (2010), based on national statistics. The rate of loss in percentages is given with respect to the total forest remaining in each country in each year within the period. These rates include new forests established through planting or seeding (not necessarily with native species) and natural expansion of existing forests, so they underestimate the total losses of natural forest that have occurred.

Other research studies that have estimated deforestation rates, as shown in **Table 3.4**, indicate that many of these forests are being subjected to high rates of deforestation. These are not always reflected in national statistics, such as those presented by FAO (2010), because the latter often include afforestation, which in most cases it undertaken with non-native species. Also, deforestation rates do not provide an indication of the extent of degradation of the forests that remain, for example that caused by the selective exploitation of species.

Country	Annual deforestation rate (%)	Study area	Year	Reference
Argentina	-0.32	Bolivian-Tucuman forest	1998-2002	(Montenegro *et al.* 2005)
Bolivia	-0.49	Forests (wet, semi-wet, semi-deciduous and deciduous, up to 3000 m a.s.l.)	1976-2004	(Killeen *et al.* 2007a)
Colombia	-0.9	*Quercus humboldtii* Bonpl. Forest (Andes)	1985-1993	(Moncada Rasmussen 2010)
	-0.63	Montane forest	1985-2005	(Armenteras *et al.* 2011)
	-0.49	Andean forest (humid, sub-humid, dry)	1970-2000	(Etter *et al.* 2008)
	-0.54	Andean forest	2000-2005	(Cabrera and Ramírez 2007)
Ecuador	-0.6 to -0.9	Loja and Zamora (montane moist forest (Parque Nacional Podocarpus), grasslands, abandoned grasslands, succession	1985-2001	(Goerner *et al.* 2007)
Perú	-0.5 to -1.0	Peruvian Andes	1990-1997	(Achard *et al.* 2002)
Venezuela	-0.3	National	1920-2008	(Pacheco Angulo *et al.* 2011)
	-1.3- -2.4	Cloud forest Capaz river	2005	(Rodríguez 2005)

Table 3.4 Deforestation rates from different sources that include montane forest. Estimates obtained with analysis of LANDSAT images, maps and reconstruction of historical documents.

At least two national experts from each country associated with this project participated in an exercise that aimed to identify and score the relative importance of the different threats affecting forests in each country of the region, to identify those that are having the greatest impact on montane forests (**Table 3.5**). Although the relevance of the threats varies across countries, there is some congruence in that livestock, deforestation for land use change to agriculture, logging and fragmentation are widely considered to be the major threats to these forests. In Bolivia and Colombia, cultivation of illicit crops are considered to be a major threat to montane forests.

Threats to montane forests	Argentina	Bolivia	Colombia	Ecuador	Perú	Venezuela
Livestock	1	1	1	1	3	1
Deforestation, land use change to agriculture	4	1	1	1	1	1
Wood extraction	1	3	2	2	2	1
Fragmentation	4	3	1	3	1	3
Extraction of minerals / mining activities including hydrocarbons	4	3	1	3	2	5
Illicit crops	n.a.	2	2	5	4	5
Wood and charcoal collection	5	3	2	3	2	4
Urbanisation and infrastructure, including hydro-electrical plants	4	5	2	3	3	2
Fires	2	2	3	5	5	2
Invasion by exotics	3	5	3	4	4	2
Climate change	3	5	1	5	4	4
Plantation of exotics	3	5	3	4	5	2
Diseases and plagues	5	5	3	4	4	5
Landslides	5	5	3	4	4	5
Non timber products	5	5	3	4	4	5

Table 3.5 Results of an assessment based on expert knowledge of the importance of different threats to montane forests in different countries of the tropical Andes. Classification according to level of importance: (1) the most important (5) the least important in the country. Based on responses from 14 experts.

Regionally, other potential threats include climate change, about which very little is known, although some research results suggest that the effects of climate change may be highly significant (Feeley *et al.* 2011; Herzog *et al.* 2011; Román-Cuesta *et al.* 2011; Urrutia and Vuille 2009; Table 3.6). Also, the use of exotic herbaceous species such as the Kikuyo (*Pennisetum clandestimum*), have major impacts on the biodiversity associated with forest ecosystems (Etter *et al.* 2008), as they reduce local species richness and interfere with water flow and runoff (Ataroff 2003; Ataroff and Rada 2000).

Ecoregion	Area Km^2	Temperature A2 (σ)	Temperature B2 (σ)	Precipitation A2 (σ)	Precipitation B2 (σ)
Venezuelan Andes montane forest	12091	7 .1	5 .4	-1 .5	-1
Cordillera Oriental montane forest	18433	6 .4	4 .9	0 .5	0 .5
Cordillera Real Oriental montane forest	47720	5 .4	4 .1	1	0 .5
Northwestern Andes montane forest	16926	5	3 .8	1	0 .5
Cauca Valley montane forest	8225	5 .2	4	0 .5	0 .5
Magdalena Valley montane forest	22399	6 .1	4 .7	0 .5	0 .5
Bolivian Yungas	50536	7 .5	5 .5	0	0
Peruvian Yungas	88803	5 .8	4 .3	1	0 .5
Southern Andes Yungas	36396	6 .9	5 .4	0	0

Table 3.6 Vulnerability of different ecoregions of Andean montane forests to climate change. Values for climate change scenarios A2 and B2 of the HadCM3 model for the period between 2051 and 2100. The values presented are the number of standard deviations (σ) from the mean (μ) for the years 1961-1990. Values greater than 2σ are considered as extreme weather conditions. Methodology adapted from Beaumont *et al.* (2011). Estimates of remaining forest area in each ecoregion for 2005 are derived from MODIS data (Schmitt *et al.* 2009a) and refer to areas with ≥ 10% forest canopy cover and occurring at elevations ≥ 1.500 m a.s.l.

3.6 Policy responses

All Andean countries have ratified a series of international treaties that promote the protection and conservation of natural areas, including the Convention on Biological Diversity (CBD), the Framework Convention on Climate Change (UNFCCC), Convention on International Trade of Endangered Species (CITES) and the Global Strategy for Plant Conservation (GSPC). One objective of targets of the CBD is that terrestrial ecosystems should have a minimum of 15% protection by 2020.

Within the tropical Andes, Josse *et al.* (2009) identified those ecosystems that meet these criteria and concluded that although protected areas vary considerably between countries and ecosystems, (e.g. Venezuela (67%), Colombia (54%), Ecuador (52%), Bolivia (45%) and Peru (30%)), there are still many areas where there is little or no formal protection, such as pluviseasonal areas of Colombia and Venezuela, montane ecosystems of Ecuador and the Bolivian-Tucuman ecosystem in Bolivia (Carretero *et al.* 2011). The Yungas ecoregion corresponds to the best preserved (56%) area in the region, with areas of mixed forest with *Dictyocaryum lamarckianum* and *Nectandra laurel*, mixed forest with *Juglans boliviana*, and mixed forest with *Prumnopitys harmsiana* and *Weinnmannia pinnata*. In contrast, forests such as the Cauca Valley in Colombia are considered to have a critical conservation status because they are severely fragmented and have little protection (**Table 3.7**).

Ecoregion	Habitat loss[a]	Fragmentation[b]	Conversion[c]	Conservation status[d]	Final conservation status	Biodiversity priority [e]	Biodiversity distictivness[f]	Protection[g]
Venezuelan Andes montane forest	10	16	6	51	Threatened	I	1	4
Northwestern Andes montane forest	20	12	6	54	Threatened	I	1	6
Cordillera Oriental montane forest	20	12	9	47	Vulnerable	I	1	4
Cordillera Real Oriental montane forest	20	12	8	50	Vulnerable	I	1	8
Cauca valley montane forest	32	20	6	88	Critical	I	1	10
Magdalena valley montane forest	32	20	6	88	Critical	II	3	10
Bolivan Yungas	20	12	8	50	Threatened	I	2	8
Peruvian Yungas	20	12	8	51	Threatened	I	1	8
Southern Andes Yungas	20	12	6	41	Vulnerable	III	3	1

Table 3.7 Evaluation of the ecoregions at the landscape level, with their conservation status, adapted from (Dinerstein *et al.* 1995). [a]Habitat loss: index from 0 (least loss) to 40 (most) [b]Fragmentation: index from 0 (least fragmented) to 20 (most). [c]Conversion: index from 0 (lowest annual conversion rate of natural habitat) to 10 (highest rate), [d]Conservation status: index from 0 (best) to 100 (worst), [e]Biodiversity priority: I = highest priority at regional scale, II = High priority regional scale, III = Moderate regional priority and VI = important at national level; [f]Biological distinctiveness: 1= Globally outstanding, 2 = Regionally outstanding, 3 = Bioregionally outstanding, and 4= Locally important; [g]protection: 1 (best protection) to 10 (least).

Colombia currently has a number of laws in place, such as Act 299 of 1996, Act 464 of 1998 and Act 599 of 2000, which provide for the protection of flora in the country, ratifies international conventions for wood extraction and specifies the penalty for the illegal exploitation of non-timber forest products. Most recently, resolution 383 of 2010 declares the wildlife species that are endangered in the country. In Venezuela discussion is ongoing for a new national strategy for biological conservation which would include protection and conservation measures adapted to each ecoregion of the country, including the Andean ecoregion, to be in the context of the new Law on Biodiversity Management (Venezuela 2008). In Peru conservation jurisdiction is divided: on the one hand, the protection of the biota in the system of protected areas, including buffer zones, is part of the Ministry of Environment, while areas outside of these correspond to the Ministry of Agriculture. In Bolivia current laws such as 1333 (Bolivia 1992), 1700 (Bolivia 1996) and 3525 (Bolivia 2006), that support the conservation and sustainable management of forests and their resources, and the regulation and promotion of the production of agricultural and non-timber forest products; in the Bolivian montane forests there are eight national protected areas and a biosphere reserve. In Argentina, around 320,000 ha (15% of total) of cloud forests are protected in different forms and levels of restriction of use, including a biosphere reserve and 18 protected areas designated nationally, provincially and municipally. Also, the law 'Minimum budgets for Environmental Protection of Native Forests' (Presupuestos Mínimos de Protección Ambiental de los Bosques Nativos) (Argentina 2007) establishes the zoning of forest areas in all the provinces of Argentina. In this context, tropical Andean montane forests have been mostly categorized as areas of medium conservation value that can be used under sustainable schemes, tourism, collection and research, and as areas of high conservation value not to be transformed (mainly protected areas already established). One of the mayor problems in terms of protected areas is that in countries like Colombia the highest proportion of protected areas is preferentially located in areas with low deforestation threats (i.e. far from roads and urban settlements, at high elevations and on steep slopes, and on less suitable land for agriculture).

3.7 Tree species conservation efforts

The identification and assessment of conservation status and threat of trees species in montane forests of the tropical Andes has not been undertaken previously at the regional scale. However, national efforts to assess species have been undertaken in some countries through the use of the IUCN Red List (International Union for Conservation of Nature) categories and criteria (Zamin *et al.* 2010). Some examples where tree species are included are: the Red Books of endangered flora of Colombia, with a total of 1,853 species of which 36% are threatened with extinction (García *et al.* 2010). In Peru, 5509 taxa restricted to Peru, of which 33% are classified as Endangered, 18% Critically Endangered and 10% as Vulnerable (León *et al.* 2006). In Ecuador, the Red Book of endemic plants, evaluated 4500 species, 0.07% are Extinct, 0.02% are Extinct in the Wild, 7.84% are Critically Endangered, 23.8% Endangered, 46,22% Vulnerable and 8,5% Near Threatened (León-Yánez *et al.* 2011). In Bolivia, the Red Book of wild relatives of cultivated species where 152 species have been identified as threatened (Moraes *et al.* 2009), some of them represented in the Andes, and in early 2012, the first chapter of the Red Book of Plants of Bolivia in the Andean region is expected to be completed (Navarro *et al.* in preparation). In Venezuela, 1598 vascular plant species were evaluated in 2003, of which 341 were considered to be the most threatened (Llamozas *et al.* 2003). Although Argentina does not yet have a Red Book of endangered species, it is estimated that for the montane forests there are about 130 tree species in total, which are currently being evaluated.

To assess the conservation status of tree species in the montane forests of the tropical Andes, a collaborative initiative with experts from Venezuela, Colombia, Ecuador, Peru, Bolivia and Argentina has been established, which aims to evaluate about 3750 species of trees, using the IUCN Red List categories and criteria. This initiative intends to identify priorities for conservation action in accordance with objectives of the Global Strategy for Plant Conservation (EGCEV/GSPC) 2020 (http://www.cbd.int/gspc/objectives.shtml). The main objectives addressed are: Objective I: Understand, document and recognize the diversity of plant species; Objective IV: Promote education and awareness of the diversity of plant species, their role in sustainable livelihoods and importance for life on Earth; and Objective V: Capacity building and public engagement necessary to implement the Strategy. A series of workshops have been carried out to begin the evaluation. The use of georeferenced information of species distributions, previously published information and expert knowledge are the main sources of information being used in this evaluation.

Once the most threatened species at the regional level are identified, this will provide a strong basis for the development and targeting of policy and management responses aimed at reducing deforestation and species loss, including actions such as promoting the creation of protected areas, forest restoration and sustainable forest management. It should also emphasize the need to implement land management policies aimed at the conservation of biodiversity in productive rural landscapes. Recently, several authors (e.g. Herrera 2012; Perfecto and Vandermeer 2008, 2010; Perfecto *et al.* 2010; Vandermeer and Perfecto 2005) have discussed the importance of preserving and managing territories which integrate conservation, productive systems (agriculture and livestock) and human population. For example, Perfecto *et al.* (2010) propose the concept of "Nature’s matrix" and argue for the importance of managing the rural landscape, improving habitat conditions for the conservation of biodiversity and ecosystem services (e.g. improving connectivity between forests remnants and protected areas), while ensuring the sustainability of food production and distribution. Harvey and González Villalobos (2007), Bhagwat *et al.* (2008) and Mendenhall *et al.* (2011) present a series of case studies showing the importance of rural landscapes for biodiversity conservation. Perfecto and Vandermeer (2010; 2012) make an analysis from the recent development of ecological theory (showing the importance of migration between fragments and local extinction) supplemented with empirical evidence to show that a model that incorporates the agricultural matrix as an integral component of conservation programs can be successful in the context of small-scale agro-ecological production in tropical regions. It is encouraging that in some regions of the tropical Andes progress has been made in the implementation of these models. For example, a paper published on “Management tools for biodiversity conservation in rural landscapes" in Colombia, by Lozano-Zambrano (2009), recommended management practices aimed at increasing the quality of habitat in productive territories. Considering the high level of land use change of the tropical Andes, it is urgent to train politicians, planners and local communities throughout the region, so they can implement these alternative models of production and conservation, and take into consideration threatened tree species.

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4 Red List assessment of tree species in montane forests of the tropical Andes

4.1 Introduction

The IUCN (International Union for Conservation of Nature) Red List (RL) is an authoritative approach for assessing the extinction risk of species (Lamoreux *et al.* 2003; Newton and Oldfield 2008; Rodrigues *et al.* 2006; Zamin *et al.* 2010), which uses clear and comprehensive categories and criteria that are flexible enough to allow uncertainty (Akçakaya *et al.* 2006). The quantitative rules for this approach are based on population sizes and rate of population decline, and the extent and decline of geographic ranges (**Table 4.1**; Akçakaya *et al.* 2006; Lamoreux *et al.* 2003; Newton and Oldfield 2008; Rodrigues *et al.* 2006; Zamin *et al.* 2010). The RL has been used by governments and other institutions to inform conservation policies and influence legislation, and has also been applied to support the identification of priority areas for conservation, to guide conservation investment, and to encourage species-based conservation and management, biodiversity evaluation and monitoring (Hoffmann *et al.* 2008).

Criterion	Critically endangered	Endangered	Vulnerable	Subcriteria
A1: reduction in population size	≥90%	≥70%	≥50%	Over 10 years/3 generations in the past, where causes are reversible, understood and have ceased
A2–4: reduction in population size	≥80%	≥50%	≥30%	Over 10 years/3 generations in past, future or combination
B1: small range (extent of occurrence)	<100km^2	<5000km^2	<20000km^2	Plus two of (a) severe fragmentation/few localities (1, ≤5, ≤10), (b) continuing decline, (c) extreme fluctuation
B2: small range (area of occupancy)	<10km^2	<500km^2	<2000km^2	Plus two of (a) severe fragmentation/few localities (1, ≤5, ≤10), (b) continuing decline, (c) extreme fluctuation
C: small and declining population	<250	<2500	<10 000	Mature individuals. Continuing decline either (1) over specified rates and time periods or (2) with (a) specified population structure or (b) extreme fluctuation
D1: very small population	<50	<250	<1000	Mature individuals
D2: very small range locations	N/A	N/A	<20 km^2 or ≤5	Capable of becoming critically endangered or extinct within a very short time
E: quantitative analysis	50% in 10 years/3 generations	20% in 20 years/5 generations	10% in 100 years	Estimated extinction-risk using quantitative models, e.g. population viability analyses

Table 4.1 Summary of IUCN Red List Categories and Criteria (IUCN 2001).

A recent review of the progress on the Red Listing of tree species by Newton and Oldfield (2008) identified that by 2008, the IUCN RL database included a total of 8324 tree species of the estimated total of over 24000 tree species in the world. These mostly comprised the species assessed more than 10 years previously in the *World List of Threatened Trees* (Oldfield *et al.* 1998), together with assessments of the endemic tree species of Ecuador, carried out by Valencia *et al.* (2000) and with an update by León-Yánez *et al.* (2011), and the assessment of endemic tree species of Peru by León *et al.* (2006). Since 1998, more than 2500 tree taxa have been evaluated, primarily by the IUCN/SSC Global Tree Specialist Group (GTSG), but only a fraction of these have so far been added to the RL database owing to a lack of resources or capacity to perform the evaluation work required (Hoffmann *et al.* 2008; Newton and Oldfield 2008; Nic Lughadha *et al.* 2005). Overall, it is clear that many tree species still await assessment. Progress has been limited by a number of factors, including the lack of appropriate data on the status and distribution of many species. Many assessments that have been undertaken depend on the use of herbarium records (Rivers *et al.* 2011; Willis *et al.* 2003) and supporting GIS data (Bachman *et al.* 2011; Cicuzza *et al.* 2007; Nic Lughadha *et al.* 2005) to identify the potential distribution of the species and the potential threats, which are mainly of anthropogenic origin (Mace *et al.* 2008). However, compilation and analysis of such data can be a time consuming task. Assessments have been further hindered by taxonomic confusion surrounding many taxa, and a lack of resources to support the assessment process, which is primarily undertaken by volunteers (Hoffmann *et al.* 2008; Newton and Oldfield 2008; Nic Lughadha *et al.* 2005).

RL assessments provide fundamental information of both scientific and political relevance regarding the state of biodiversity and for reporting trends in biodiversity loss. They also contribute to the development of national level strategies for species and ecosystem adaptation to climate change (Zamin *et al.* 2010). Although significant national RL assessments of vascular plants have been undertaken previously in the Andean region (e.g. Valencia *et al.* 2000), no systematic RL assessment of montane tree species has previously been undertaken in this area. This is a unique region, with around 133 different types of ecosystem (Josse *et al.* 2009a, 2009b), and high habitat diversity, resulting from broad altitudinal and latitudinal gradients (Josse *et al.* 2003). Andean montane forests are currently a major global conservation priority owing to their biological richness and high level of endemism (Bush *et al.* 2007; Olson and Dinerstein 1997). These forests are of high value for the provision of ecosystem services related to water, the regulation of regional climate and the capture and storage of carbon (Cuesta *et al.* 2009); they are also considered to be amongst the least known ecosystems in the tropics (Ataroff and Rada 2000; Bubb *et al.* 2004; Gentry 1995; Kessler 2000; Stadtmüller 1986). Andean montane forests are considered to be highly threatened by the continuing rates of deforestation, fragmentation, degradation (Chapter 3; Cabrera and Ramírez 2007; Tejedor Garavito *et al.* 2012) and the potential effects of climate change (Cuesta *et al.* 2009; Herzog *et al.* 2011; Urrutia and Vuille 2009).

The objective of this chapter is to evaluate the extinction risk of the tree species that occur in montane forests in the tropical Andes by carrying out an IUCN RL assessment, which is checked and validated using expert knowledge from within the region. This process was implemented as a series of steps: (i) compilation of a list of tree species occurring within the montane forests of the region; (ii) compilation of spatial data indicating the geographical distribution of each species; (iii) production of distribution maps for each species, and use of these to estimate the extent of geographical range of each species, according to the RL guidelines; (iv) validating the distribution maps using expert knowledge from within the region; (v) performing a preliminary RL assessment of each taxon according to the RL categories and criteria, in collaboration with experts within the region, and employing the distribution maps produced. The assessment focused on taxa with regional distributions (i.e. present in more than one country), in order to complement previous or ongoing RL assessments of vascular plants undertaken within individual countries.

4.2 Scope and study area

The scope of this assessment is the tropical Andes occurring in Argentina, Bolivia, Colombia, Ecuador, Peru and Venezuela. Geographically, the tropical Andes represent most of the montane forests in the Andean region. The definition of upper montane forest for the purposes of the research in this chapter includes cloud forest (Northern Andean forests, Yungas forests and Bolivia-Tucuman forests) and seasonal (wet) forest above 1500 m a.s.l., with temperatures between 6-18°C and yearly mean precipitation above 1000 mm, as described by Josse *et al.* (2009a, 2009b). An altitudinal threshold of 1500 m a.s.l. was chosen as this is the altitude at which the species composition typically changes, as lowland or lower montane tree species are displaced by a floristically different assemblage of upper montane species (Gentry 1993; Josse *et al.* 2009a). However, it is recognised that this threshold varies geographically. On large equatorial inland mountains this transition usually occurs at an altitude of 1200-1500 m a.s.l. but it may occur at much lower elevations on small outlying island mountains and away from the equator (Bruijnzeel 2002). On large inland mountain systems cloud forests may typically occur between 2000 and 3500 m a.s.l., whereas in coastal and insular mountains this zone may descend to 1000 m a.s.l. Under exceptionally humid conditions a cloud forest zone may develop on steep, tropical island or coastal mountains at elevations as low as 500 m a.s.l. (Bubb *et al.* 2004). The threshold of 1500 m was therefore selected as a compromise, in discussion with experts familiar with the region. The intention here is to ensure that all tree species associated with upper montane forest are included in the assessment, and through a process of expert consultation, it was adjudged that a lower altitudinal limit of 1500 m would achieve this.

The focus here is on tree species associated with moist, upper montane or cloud forests. However, some species that are also associated with other types of vegetation such as seasonal (moist) forests have been included in the assessment, because there are areas where species occur in more than one vegetation type, as in the overlap between xerophitic and seasonal vegetation, and between seasonal and cloud forest vegetation. Therefore some cloud forest species are shared with seasonal forest, as for example in Argentina and Bolivia, where species that might be considered to be threatened with extinction would be excluded if only cloud forest species were included. Trees are defined here as upright woody plants with a dominant above-ground stem that reaches a height of at least 3 m (Körner 1998), including palms and woody ferns. Species such as *Chusquea* sp., although their main stem reaches above the height threshold were excluded from the analysis, as they are considered as tall grasses.

As national RL assessments have been undertaken in individual countries, e.g. Bolivia (Meneses and Beck 2005), Colombia (Calderón *et al.* 2002), Ecuador (León-Yánez *et al.* 2011), Peru (León *et al.* 2006) and Venezuela (Llamozas *et al.* 2003), the objective here is to focus primarily on those species that are shared by more than one country so as not to duplicate national efforts.

4.3 Methods

As a first step, a workshop was held in Ecuador involving regional specialists (see **Appendix I**), with representation from each country in the region (Argentina, Bolivia, Colombia, Ecuador, Peru and Venezuela). Based on this expert knowledge, a consolidated list was produced of the candidate tree species that are known to occur in the Tropical Andes. The development of this list was supported by accessing data from a range of sources, including the Missouri Botanical Gardens database (www.Tropicos.org), regional herbaria (Colombian National Herbarium (COL), Venezuelan National Herbarium (VEN), Bolivian National Herbarium (LPB), Herbarium of the Universidad Pontificia Católica in Ecuador (QCA) and San Marcos Herbarium of the Universidad. Nacional Mayor de San Marcos, Peru (USM)), regional floras and personal databases. The nomenclature of taxa on the list was checked using the Plant List (www.theplantlist.org, accessed March 2011), to identify synonyms and those species unresolved taxonomically. The Angiosperm Phylogeny Group III system (APG III 2009) was followed to provide consistency on the names of the species families.

Geographical distribution data for all the tree species was then compiled using pre digitized data from species' specimens and previously surveyed locations from the different sources of information. The sources of this information included: personal records of the network of regional specialists involved in this assessment, the Missouri Botanical Gardens database (www.Tropicos.org), regional herbaria, and the Global Biodiversity Information Facility (GBIF: www.gbif.org), accessed during November 2010. A spatial database incorporating these distribution data was created in ArcGIS v.10, and then critically examined in order to exclude those points that were incorrectly georeferenced. The database was used to identify those species occurring exclusively at $\geq$ 1500 m a.s.l. by overlaying data on a Digital Elevation Model (DEM) obtained from www.worldclim.org (**Figure 4.1**), with a grid space of 30 arc seconds (0.0083^{o} or approximately 1 km). If at least one of the records was below this altitude threshold, the species was excluded from further analysis. These records were also used to identify those species shared by more than one country (see **Appendix II**). Distribution maps of each taxon that met these selection criteria were then checked by the regional network of specialists, and revised further where necessary.

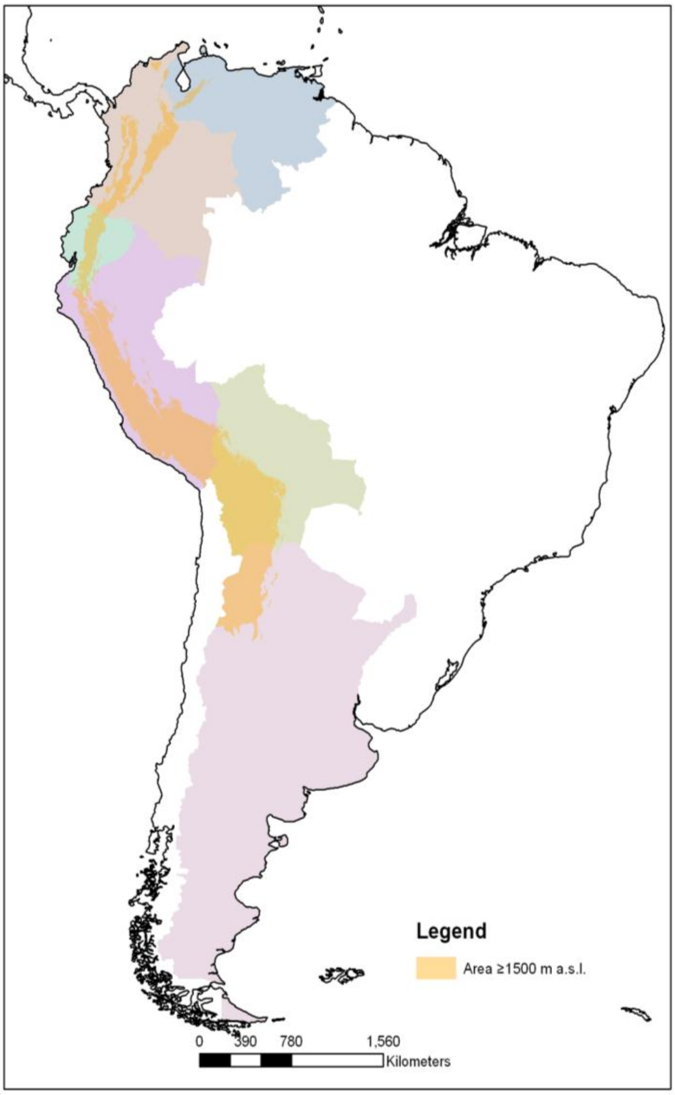


Figure 4.1 Map of the distribution of the study area, which exceeds an attitude of 1500 m in the tropical Andes. Based on the Digital Elevation Model (DEM), obtained from www.worldclim.org.

The IUCN RL criteria were then applied to each taxon, with reference to the distribution maps created, and a preliminary classification was provided. These assessments were conducted by a lead assessor (Tejedor), which were then reviewed in consultation with the network of specialists. In each case, the RL criteria were applied following the IUCN Red List guidelines (IUCN Standards and Petitions Subcommittee 2011). The RL process requires that all the species are evaluated using all of the criteria and that the final category allocated to the species is the one associated with the highest category of threat (IUCN 2001).

Two key elements used in the RL process, at least for those species poorly known (as is the case of many Andean tree species), are the Extent of occurrence (EOO) and the Area of Occupancy (AOO), which are measures of the geographical range of a species. EOO is used to evaluate the species under criteria A and B, and AOO is used to evaluate the species under criteria A, B and D.

EOO is defined as the smallest polygon in which no internal angle exceeds 180° and contains all sites of occurrence, and is commonly calculated using the Minimum Convex Polygon (MCP) around georeferenced data for species distributions. These calculations were performed using a series of packages in R 2.14 (R Development Core Team 2011) such as 'dismo', 'rgdal','rpart','mgcv', kernlab,' among others and ArcGIS v.10. The scripts used in R were modified from those provided by both, the R packages and by Duncan Golicher. The MCP requires at least three points in order to be calculated. For those species that had fewer than three points EOO was not calculated, namely: *Citharexylum rimbachii*, *Crossothamnus gentryi*, *Diplostephium cinerascens, Dunalia trianaei*, *Joosia aequatoria*, *Polylepis microphylla*, *Ribes canescens* and *Tournefortia loxensis.* This result was then filtered further to calculate an EOO that excluded non suitable habitats, by using a classified global land cover map for 2009 (referred to henceforth as "GlobCover") produced by Arino *et al.* (2010), which was obtained from the MERIS imaging spectrometer, at a resolution of 300 m. This was achieved by excluding the following land cover classes: Rainfed croplands, Mosaic cropland (50-70%)/vegetation (grassland/shrubland/forest) (20-50%), Mosaic vegetation (grassland/shrubland/forest) (50-70%)/cropland (20-50%), Artificial surfaces and associated areas (Urban areas >50%), Closed to open (>15%) herbaceous vegetation (grassland, savannas or lichens/mosses), Bare areas, Water bodies, Permanent snow and ice. Also, this map was masked using the DEM, to only include areas ≥1500m a.s.l. Once the EOO map was clipped using GlobCover, it was then projected using the Mollweide (sphere) projection, in order to calculate the area for each of the species.

The AOO is defined as the area occupied by a taxon, excluding cases of vagrancy, which is calculated using gridded (raster) data at a scale appropriate to the taxon (IUCN 2001).

The IUCN guidelines (IUCN Standards and Petitions Subcommittee 2011) suggest a resolution of 4 km^2, as this allows the taxa to be listed as Critically Endangered, as the criteria requires a taxon to have an AOO of < 10 km^2 to qualify. Therefore AOO was calculated using two resolutions: using a 4 km^2 and a 100 km^2 grid cell (IUCN 2011). These two grid sizes were chosen to evaluate the variability of the range size for a given species. These calculations were performed using a series of packages in R 2.14 (R Development Core Team 2011). AOO was not calculated for the 14 species with ≤ 3 records.

Assessment under Criterion A addresses a decline in population size (past, present and/or projected), based on (a) direct observation (A1, A2 and A4 only), (b) an index of abundance appropriate to the taxon, (c) a decline in area of occupancy, extent of occurrence and/or quality of habitat, (d) actual or potential levels of exploitation, and/or (e) the effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites (IUCN Standards and Petitions Subcommittee 2011). Under Criterion A1, a reduction in the population is defined as a decline in the number of mature individuals of at least a 50% over a time period (years) in the past, although the decline need not be continuing, which means that the sources of decline are clearly reversible and understood and have ceased. This criterion was not applicable for any of the species, as the causes of decline for the species have not stopped, as is the case of forest fragmentation and conversion for agriculture or livestock, as detailed in **Chapter 3**. Even in the case of species such as *Cinchona* sp. where the main cause of population reduction has ceased (i.e. exploitation for the extraction of quinine, owing to its replacement by synthetic production), other pressures on the habitat such as forest degradation and loss have not ceased throughout the region where the species is distributed.

Under A2, A3 and A4, the Criteria stipulate that in order to include a species under the threat category there should be a reduction in the population, representing a decline in the number of mature individuals, of at least 30% over the time period (years) specified, which relates to a decline over 10 years or 3 generations, whichever is the longer. A2 refers to the past, A3 to the future and A4 to past and future. For these Criteria, the causes of reduction may not have ceased or may not be understood or may not be reversible. Information on species population declines and information regarding the species' specific characteristics such as generation length, species age at maturity and longevity are scarce. However, the IUCN guidelines (IUCN Standards and Petitions Subcommittee 2011) acknowledge that this lack of information is a common issue for many species, so it is possible to infer or project values based on a decline in the AOO, EOO and/or quality of habitat, by making assumptions about the relationship between habitat loss and population reduction, as long as details of all the factors taken into consideration to avoid over or under estimations are included in the documentation. Generation length for tree species can be longer than 10 years; therefore it is possible to use a time scale of 100 years, as the most significant population declines could have occur during the last 100 years and relevant information is available (IUCN Standards and Petitions Subcommittee 2011). In this assessment 50 and 100 years were used as time lengths to identify the population decline in the past and future with reference to forest loss.

Inferences from deforestation rates and the area of current forest cover were used to estimate the percentage of forest loss over a period of 50 and 100 years in the past (A2) or that might occur in the future (A3). Three scenarios were set with the different data available and different estimation of potential forest loss were explored. For the first scenario (S1), deforestation rates and the total forest area per country for 2010 were obtained from data provided by the FAO (2010). The deforestation rates from the FAO (2010) are percentage annual forest change rate that occurred between 2005 and 2010. For the second scenario (S2), the deforestation rates were obtained from regional averages from different sources from the literature summarised in **Table 3.4** in **Chapter 3** and the total area of current forest cover for the Andean montane forests, calculated using the GlobCover map for 2009 excluding lowland areas <1500 m. a.s.l. and the following land use classifications: Rainfed croplands, Mosaic cropland (50-70%)/vegetation (grassland/shrubland/forest) (20-50%), Mosaic vegetation (grassland/shrubland/forest) (50-70%)/cropland (20-50%), Artificial surfaces and associated areas (Urban areas >50%), Closed to open (>15%) herbaceous vegetation (grassland, savannas or lichens/mosses), Bare areas, Water bodies, Permanent snow and ice. The third scenario (S3) used the deforestation rates provided by the FAO (2010) and total area of current forest cover for the Andean montane forests estimated above. For all scenarios, the area of forest loss per annum was calculated, and then multiplied by 50 and 100 to estimate the area of forest lost during the period. These values were then added to or subtracted from the current forest area in order to calculate the change in forest area over the last or in the next 50 and 100 years. The current forest area was then compared to the estimated forest areas to estimate percentage forest loss. These percentages were used to estimate whether the forest area has been reduced by more than 30% in the past, or is projected to do so in the next 50 to100 years, assuming that the percentage of annual forest change will remain constant over time. The area of forest estimated in the three scenarios described above, for the last and next 50 years, was also used to calculate the percentage of forest loss in a window of 100 years including past and future required for the Criterion A4.

Furthermore, if there was information available for the species indicating that it had been exploited (or overexploited), this was noted in support of sub-criterion (d) of Criteria A1-4:, 'actual or potential levels of exploitation'. In addition, information on the decline of populations as required by sub-criterion (c) was elicited from experts supported by inspection of the distribution records, taking the collection dates into account.

Criterion B addresses the geographical range of the species, which is based on the on the EOO and AOO, calculated as explained above, to identify populations with restricted and declining distribution. Calculation of EOO and AOO allowed the identification of species for which the estimated area was within the thresholds of the threatened category, as shown in **Table 4.1**, to produce a preliminary categorisation. Then, species were evaluated further using the following sub-criteria, at least two of which need to be met, for a species to qualify as threatened under this Criterion: a) if the species is severely fragmented or with few number of locations, b) in continuing decline and/or c) if there are extreme fluctuations in the areas of occupancy or number of mature individuals (in the present or near future). These sub-criteria were estimated using expert knowledge and literature if this was available, supported by reference to the distribution maps and maps of potential habitat (forest cover).

Criterion C addresses small population sizes and their fragmentation, decline, or fluctuations. This Criterion identifies taxa with small populations that are currently declining or may decline in the near future. This Criterion was only used for those species for which an estimate of the total population size was available or could be estimated, based on population density figures given by the experts, in turn based on their collections, field data and knowledge of the species. The information for most species is scarce and although some species appear to be rare, estimations of the total population size was difficult. In every case where data were available, values of total population size exceeded the thresholds of the Criterion. Therefore, none of the species qualified as threatened under this Criterion.

Criterion D is based on an accurate estimate of populations smaller than 1000 individuals (D1). This criterion is also applied to qualify a taxon under the Vulnerable category when the species has an AOO <20 km^2 or is found in ≤5 locations and for which a plausible future threat exists that could drive the taxon to CR or EX in a very short time (D2) (IUCN Standards and Petitions Subcommittee 2011). To apply this Criterion the experts were consulted and where possible they identified those species with small number of individuals per hectare. However, none of the species were identified to have fewer than 1000 individuals in total, although many were identified to be found at low densities or as rare. 11 species had AOO <20 km^2, but in each of these cases, expert knowledge indicated that these were likely cases of under-recording, as each of the species was relatively widely distributed. Therefore, no species was found to meet this Criterion. It should be noted, however, that accurate field inventories of very threatened species throughout their ranges are generally not available in this region, particularly for those species present in more than one country, as examined here.

Criterion E is based on information on quantitative analysis of the species; this was applied by consulting the scientific literature and the expert network. From this, it was identified that for none of the species there has been a population viability analysis been carried out to date. Therefore, Criterion E did not apply to any species.

Uncertainty is an issue that underlies the Red List, as many of the decisions are inferred, estimated and projected. Natural variability, vagueness in the terms and definitions used in the Criteria (semantic uncertainty), and measurement error are further sources of uncertainty (IUCN Standards and Petitions Subcommittee 2011). In this assessment the level of uncertainty was measured in several different ways. Uncertainty was scored for each subcriterion on a scale, represented as high, medium or low uncertainty. For the information provided by the experts, uncertainty was assessed by scoring the degree of uncertainty associated with applying each sub-criterion. For those sub-criteria based on map analysis, uncertainty was scored according to the number of records that were available to carry out the evaluation.

4.4 Results

3750 tree species were initially identified as occurring in the upper montane forests of the Andes. From this total, 917 species were excluded as no georeferenced records were found at the time of the search. Another further 1287 species were excluded as all of their records fell entirely within the boundaries of a single country of South America. 1400 species had at least one record that occurred below the 1500 m threshold and were therefore excluded from subsequent analyses. Of the 146 species remaining, a further 17 were excluded during checking, as a result of taxonomic revision. As a result, 129 taxa were evaluated according to the RL categories and criteria (see **Appendix II** for the full list of species). Ecuador was the country with most species and Argentina the fewest (**Figure 4.2**). The most speciose family was Melastomataceae (11) (**Figure 4.3**). A total of 1663 distribution records were obtained for these species. The number of records per species varied among species (**Figure 4.4**), with 79 species having ≤10 unique records and four having > 50 unique records, and with an overall mean (+/-SD) of 12.9 (+/-11.4) records per species.

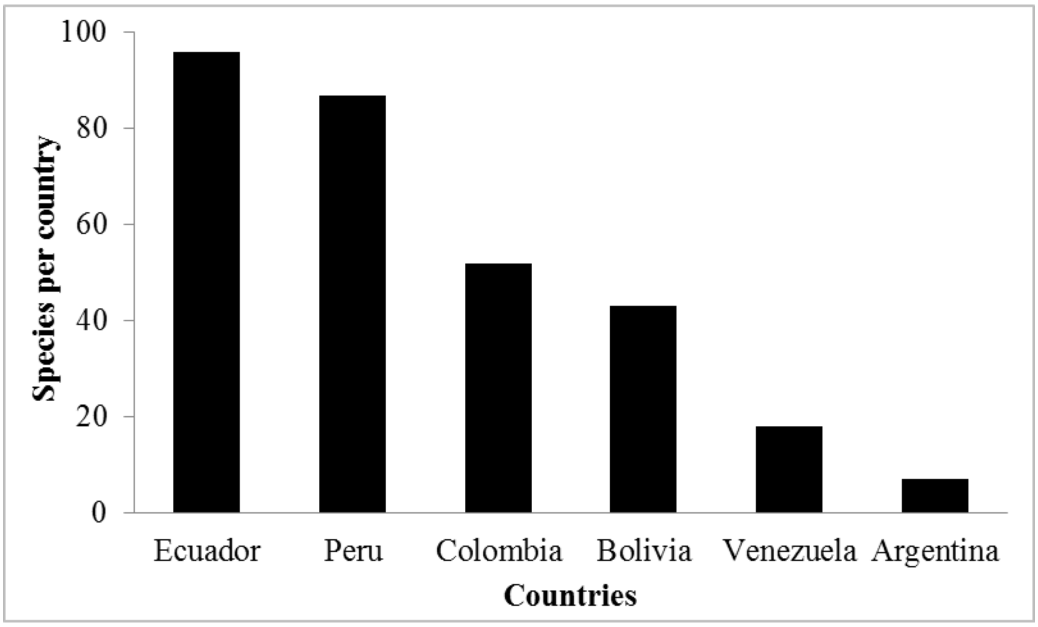


Figure 4.2 Number of species per country that were evaluated using the RL categories and criteria.

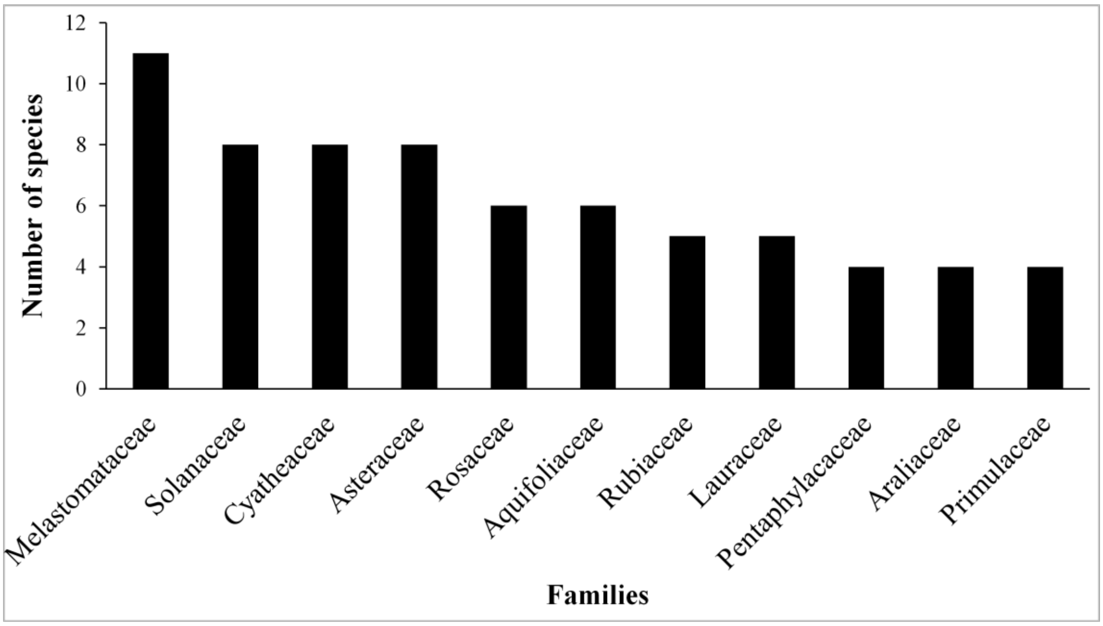


Figure 4.3 The ten families with the largest number of species included in the assessment.

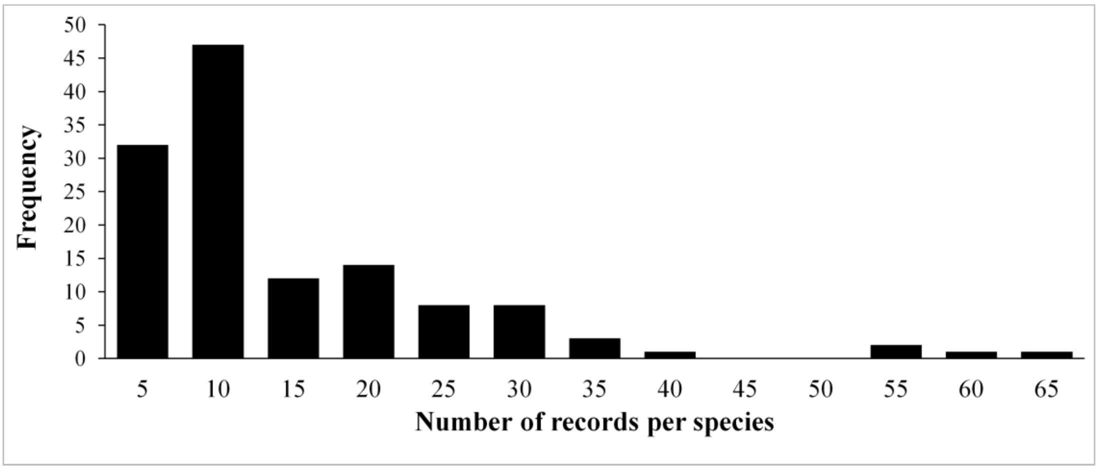
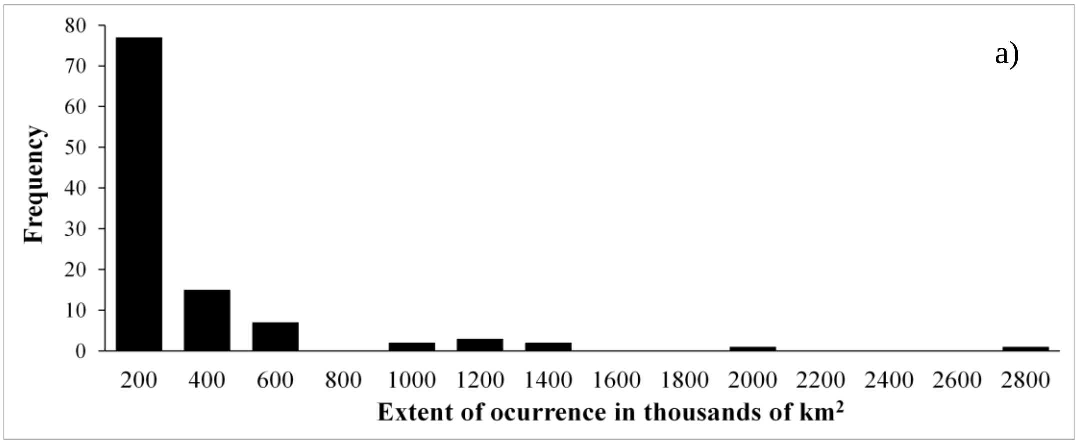


Figure 4.4 Frequency distribution of the abundance of species records for the 129 species evaluated under the RL categories and criteria.

The calculation of the EOO varied markedly among species **Figure 4.5**. The EOO values without excluding unsuitable areas showed that 102 species had an EOO ≤500,000 km^2, and 9 species were widely distributed with an EOO ≥1,000,000 km^2, e.g. *Weinmannia auriculata* with an estimated EOO of 2,632,149 km^2 (**Figure 4.5a**). 26 species had an EOO <20,000 km^2, 14 of which can be preliminarily classified as meeting Criterion B as Vulnerable (VU), 10 as Endangered (EN) and 2 as Critically Endangered (CR), according to the IUCN thresholds.

By excluding areas of unsuitable areas for the species (**Figure 4.5b**), the EOO was reduced for all the species. The mean (± SD) EOO reduction was 208,687± 35,185 km^2. For example *Weinmannia auriculata*, which originally had a large EOO, the area was reduced to 158,665 km^2. Another example of the considerable reduction in EOO area for the species is shown in **Figure 4.6** where the EOO for *Ceroxylon parvifrons* is displayed. The initial estimate of EOO was 1,927,465 km^2 but when excluding unsuitable areas this was reduced to 196,676 km^2. When unsuitable areas were excluded, 49 species had an EOO <20,000 km^2, 30 of which can be preliminarily classified as meeting Criterion B as Vulnerable (VU), 17 as Endangered (EN) and 2 as Critically Endangered (CR), according to the IUCN thresholds.



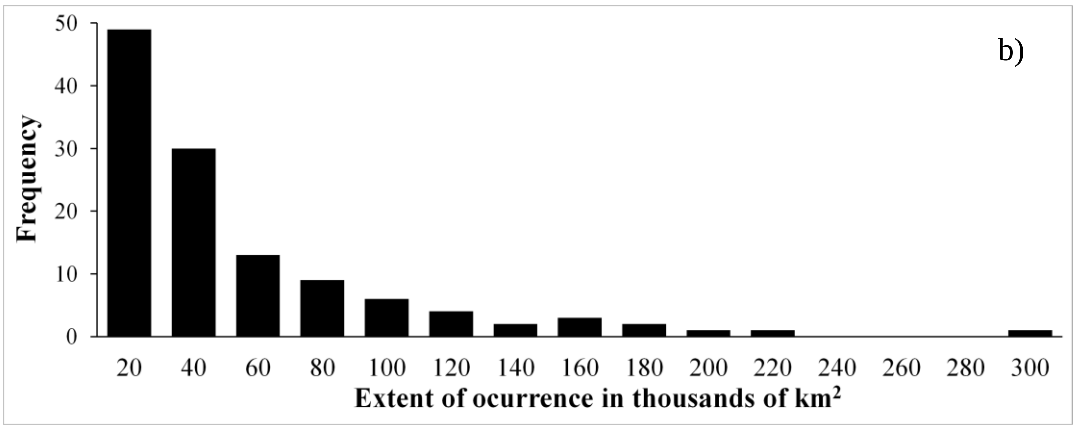


Figure 4.5 Frequency distribution of EOO values for the species based on: a) MCP including the full extent of the distribution. b) Excluding unsuitable areas (see text for further details).

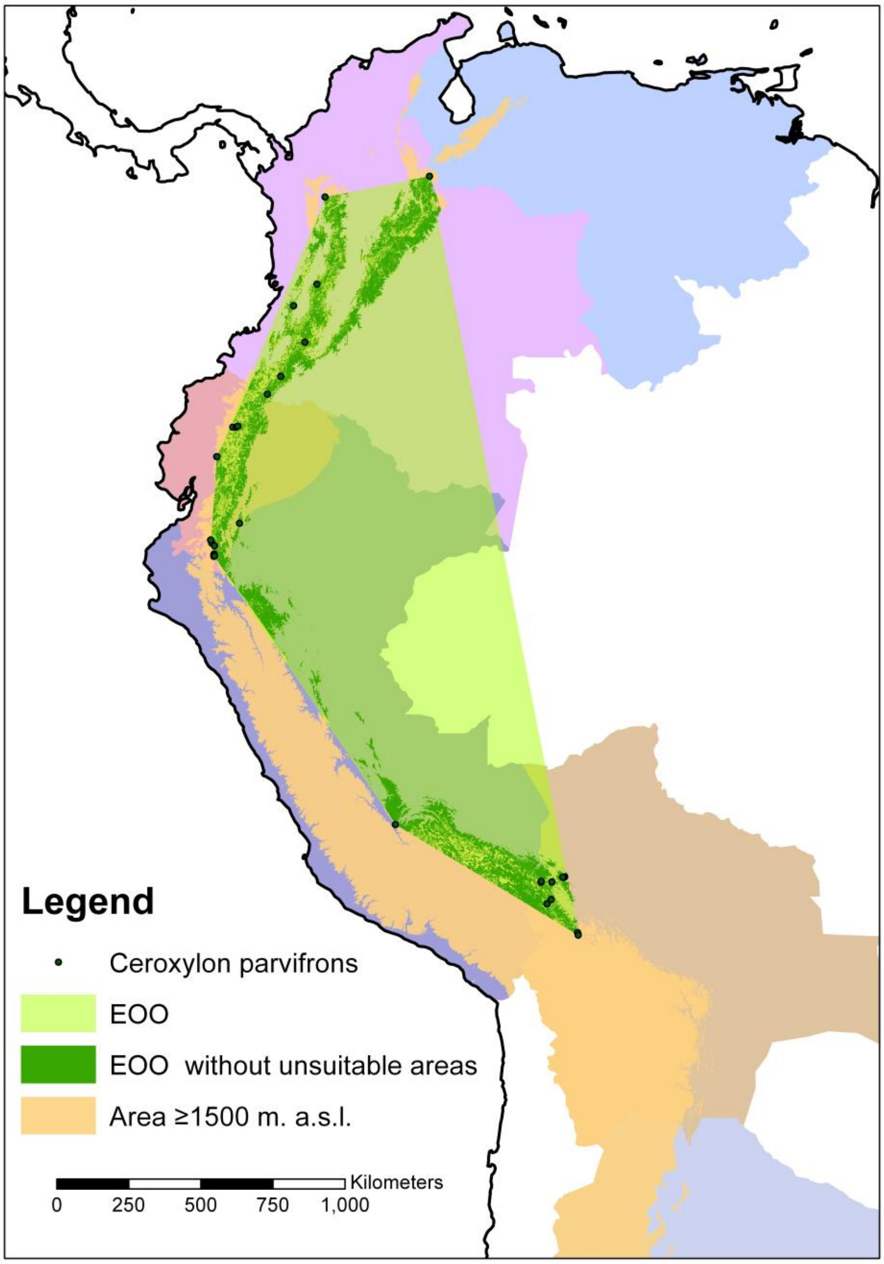
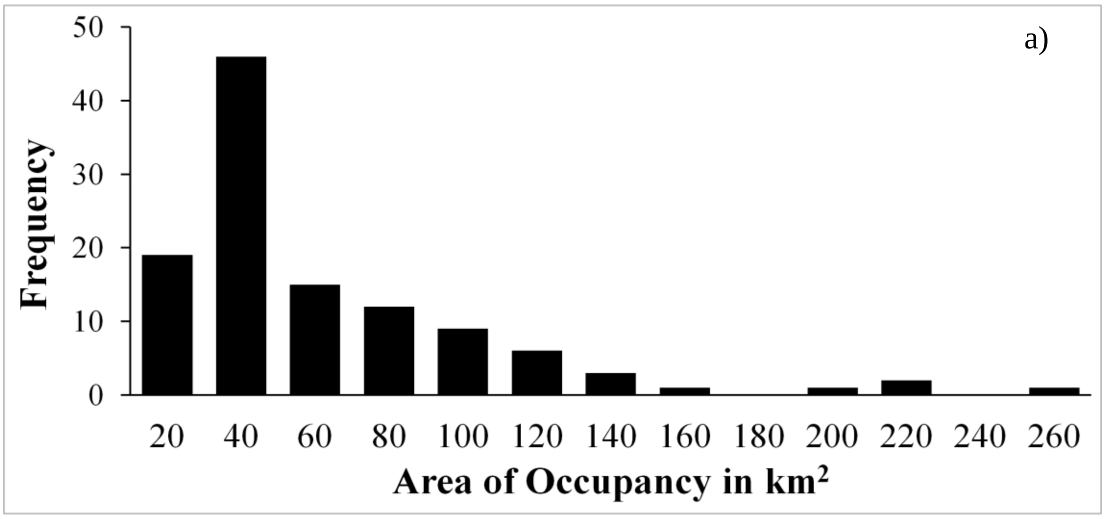


Figure 4.6 EOO for the species *Ceroxylon parvifrons*. Light green polygon shows the MCP which includes the full extent of the species' distribution. Dark green polygon shows the EOO without unsuitable areas (see text for further details).

The AOO was calculated using two resolutions, with respective grid cell sizes of 4 km^2 and 100 km^2 (**figure 4.7**). With an AOO of 4 km^2 resolution, all of the species were associated with an AOO of less than 300 km^2, which in terms of the RL would classify all species under a threatened category of at least Endangered (EN), as the threshold for their inclusion in this category is <500 km^2. However, as the RL guidelines state, it is necessary to explore the different scales to identify a relevant value where the biological aspects of the taxon are incorporated. An additional AOO was therefore calculated for the species at a resolution of 100 km^2, in order to explore the implications of the choice of grid cell size on the classification. 96 species had an AOO <2,000 km^2, 81 of which can be preliminarily classified as Vulnerable (VU), 15 as Endangered (EN) and none of them were Critically Endangered (CR) according to the IUCN thresholds, and without taking into consideration those species with ≤3 records. Therefore, 19 species had AOO ≥2,000 km^2 using a resolution of 100 km^2.



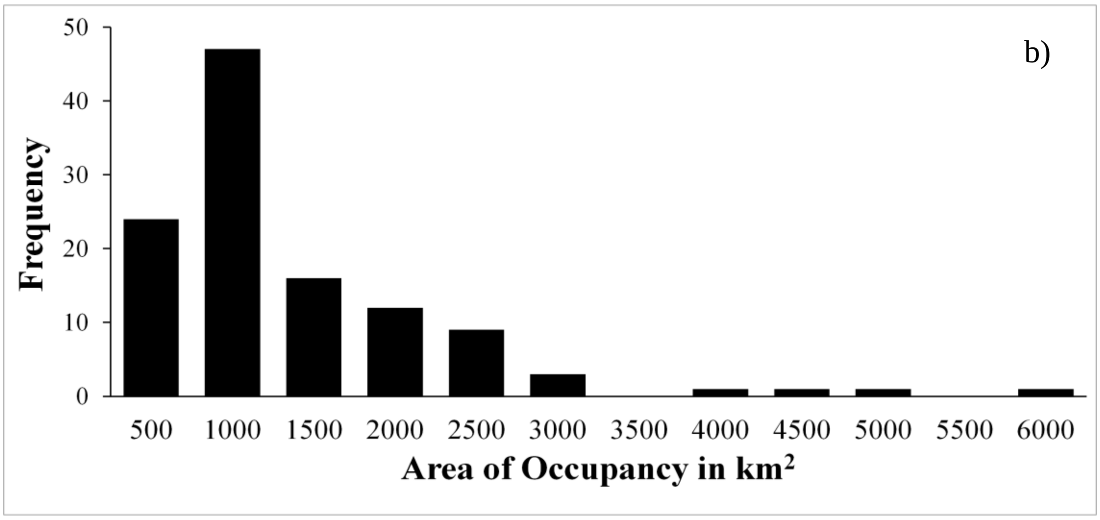


Figure 4.7 Frequency distribution of the AOO with a grid cell size of a) 4 km^2 and b) 100 km^2 for the species evaluated.

Under Criterion A, further information on the deforestation rates and the area of current forest cover was used to estimate the percentage of forest loss that has occurred over a period of 50 and 100 years in the past (A2) or might occur in the future (A3) (**Tables 4.2** and **4.3**). Deforestation rates vary across the Andean countries and sources, as do the percentages of forest loss thought the region**.** Estimates of total forest loss in the past 50 years did not exceed the 30% threshold of the IUCN guidelines; however this value was exceeded over a timescale of 100 years by a small margin (**Table 4.2**). The estimate of projected forest loss in the next 50 years, as shown in **Table 4.3**, reached the threshold at 30.8% forest loss. The estimated forest loss for the next 100 years, in all three methods of calculation, was at least 30% and up to 61.63% of forest loss, based on the estimation method (**Tables 4.2** and **4.3**). Analysis showed that Ecuador and Venezuela showed that if forests continue to be lost at the current rate, these forest areas may be lost in their entirety over the next 100 years.

To estimate population declines under A4 criteria, considerations must relate to both the past and the future. Therefore, to infer population decline from forest loss a combination of percentage forest loss since 50 years ago (**Table 4.2**), for the next 50 years (**Table 4.3**) and a combination of both past and future (**Table 4.4**) was used. The results show that with current deforestation rates, the estimated percentage forest loss will be at least 30% and for S2 this could be at least 47% (**Table 4.4**). Therefore, the estimated forest loss in a time frame of 100 years, including the past and the future, could exceed the 30% threshold, which in terms of the RL would qualify all species as at least Vulnerable (VU). In the case of future forest loss, all species would be classified as Endangered (EN) according to this criterion.

1) FAO past (S1)								
Country	**Current forest area in km^2**	**Annual change rate (%)**	**Forest area 50 years ago in km^2**	**% remaining from 50 years ago**	**% loss from 50 years ago**	**Forest area 100 years ago in km^2**	**% remaining from 100 years ago**	**% loss from 100 years ago**
Argentina	294000	-0.8	411600	71.43	28.57	529200	55.56	44.44
Bolivia	571960	-0.53	723529	79.05	20.95	875099	65.36	34.64
Colombia	604990	-0.17	656414	92.17	7.83	707838	85.47	14.53
Ecuador	98650	-1.89	191874	51.41	48.59	285099	34.60	65.40
Peru	679920	-0.22	754711	90.09	9.91	829502	81.97	18.03
Venezuela	462750	-0.61	603889	76.63	23.37	745028	62.11	37.89
Total	**2712270**	**-0.46**	**3342018**	**81.16**	**18.84**	**3971766**	**68.29**	**31.71**

2) Andes Area and literature rates (S2)								
Country	**Current forest area in km^2**	**Annual change rate (%)**	**Forest area 50 years ago in km^2**	**% remaining from 50 years ago**	**% loss from 50 years ago**	**Forest area 100 years ago in km^2**	**% remaining from 100 years ago**	**% loss from 100 years ago**
Argentina	105232	-0.32	122069	86.21	13.79	138906	75.76	24.24
Bolivia	202601	-0.49	252239	80.32	19.68	301876	67.11	32.89
Colombia	100905	-0.64	133195	75.76	24.24	165485	60.98	39.02
Ecuador	46800	-0.75	64350	72.73	27.27	81900	57.14	42.86
Peru	314113	-0.75	431905	72.73	27.27	549698	57.14	42.86
Venezuela	13315	-1.08	20471	65.04	34.96	27628	48.19	51.81
Total	**782966**	**-0.62**	**1024229**	**76.44**	**23.56**	**1265492**	**61.87**	**38.13**

3)Andes Area and FAO rates (S3)								
Country	**Current forest area in km^2**	**Annual change rate (%)**	**Forest area 50 years ago in km^2**	**% remaining from 50 years ago**	**% loss from 50 years ago**	**Forest area 100 years ago in km^2**	**% remaining from 100 years ago**	**% loss from 100 years ago**
Argentina	105232	-0.8	147324	71.43	28.57	189417	55.56	44.44
Bolivia	202601	-0.53	256291	79.05	20.95	309980	65.36	34.64
Colombia	100905	-0.17	109482	92.17	7.83	118059	85.47	14.53
Ecuador	46800	-1.89	91026	51.41	48.59	135252	34.60	65.40
Peru	314113	-0.22	348665	90.09	9.91	383218	81.97	18.03
Venezuela	13315	-0.61	17375	76.63	23.37	21436	62.11	37.89
Total	**782966**	**-0.48**	**970164**	**80.70**	**19.30**	**1157363**	**67.65**	**32.35**

Table 4.2 Estimation of forest loss in the past 50 and 100 years using: 1) Total forest area per country and deforestation rates provided by FAO (2010). 2) Andean forest area derived from GlobCover in 2009 and mean deforestation rates from the literature available in Table 3.4, Chapter 3. 3) Andean forest area derived from GlobCover in 2009 and mean deforestation rates provided by the FAO (2010).

1) FAO future (S1)								
Country	**Current forest area in km^2**	**Annual change rate (%)**	**Forest area in 50 years in km^2**	**% remaining in 50 years**	**% loss in 50 years**	**Forest area in 100 years in km^2**	**% remaining in 100 years**	**% loss in 100 years**
Argentina	294000	-0.80	176400	60.00	40.00	58800	20.00	80.00
Bolivia	571960	-0.53	420391	73.50	26.50	268821	47.00	53.00
Colombia	604990	-0.17	553566	91.50	8.50	502142	83.00	17.00
Ecuador	98650	-1.89	5426	5.50	94.50	0	0.00	100.00
Peru	679920	-0.22	605129	89.00	11.00	530338	78.00	22.00
Venezuela	462750	-0.61	321611	69.50	30.50	180473	39.00	61.00
Total	**2712270**	**-0.46**	**2082522**	**76.78**	**23.2**	**1452775**	**53.56**	**46.44**

2) Andes area and literature rates for the future (S2)								
Country	**Current forest area from in km^2**	**Annual change rate (%)**	**Forest area in 50 years in km^2**	**% remaining in 50 years**	**% loss in 50 years**	**Forest area in 100 years in km^2**	**% remaining in 100 years**	**% loss in 100 years**
Argentina	105232	-0.32	88395	84.00	16.00	71558	68.00	32.00
Bolivia	202601	-0.49	152964	75.50	24.50	103327	51.00	49.00
Colombia	100905	-0.64	68616	68.00	32.00	36326	36.00	64.00
Ecuador	46800	-0.75	29250	62.50	37.50	11700	25.00	75.00
Peru	314113	-0.75	196321	62.50	37.50	78528	25.00	75.00
Venezuela	13315	-1.08	6158	46.25	53.75	0	0.00	100.00
Total	**782966**	**-0.62**	**541703**	**69.19**	**30.81**	**300440**	**38.37**	**61.63**

3) Andes area and FAO rates future (S3)								
Country	**Current forest area from in km^2**	**Annual change rate (%)**	**Forest area in 50 years in km^2**	**% remaining in 50 years**	**% loss in 50 years**	**Forest area in 100 years in km^2**	**% remaining in 100 years**	**% loss in 100 years**
Argentina	105232	-0.80	63139	60.00	40.00	21046	20.00	80.00
Bolivia	202601	-0.53	148912	73.50	26.50	95223	47.00	53.00
Colombia	100905	-0.17	92328	91.50	8.50	83751	83.00	17.00
Ecuador	46800	-1.89	2574	5.50	94.50	0	0.00	100.00
Peru	314113	-0.22	279560	89.00	11.00	245008	78.00	22.00
Venezuela	13315	-0.61	9254	69.50	30.50	5193	39.00	61.00
Total	**782966**	**-0.48**	**595767**	**76.09**	**23.91**	**408569**	**52.18**	**47.82**

Table 4.3 Estimation of projected forest loss in the next 50 and 100 years using: 1) Total forest area per country and annual forest change rates provided by FAO (2010). 2) Andean forest area derived from GlobCover in 2009 and mean deforestation rates from the literature available in T**able 3.4**, **Chapter 3**. 3) Andean forest area from GlobCover in 2009 and mean deforestation rates provided by the FAO (2010). S1-3: three scenarios using data from different sources (see Methods for details).

1) FAO past and future (S1)				
Country	**Forest area in 1960 in km^2**	**Forest area in 2060 in km^2**	**% remaining in 2060**	**%loss in 2060**
Argentina	411600	176400	42.86	57.14
Bolivia	723529	420391	58.10	41.90
Colombia	656414	553566	84.33	15.67
Ecuador	191874	5426	2.83	97.17
Peru	754711	605129	80.18	19.82
Venezuela	603889	321611	53.26	46.74
Total	**3342018**	**2082522**	**62.31**	**37.69**

2) Forest area in the Andes and literature rates (S2)				
Country	**Forest area in 1959 in km^2**	**Forest area in 2059 in km^2**	**% remaining in 2059**	**%loss in 2059**
Argentina	122069	88395	72.41	27.59
Bolivia	252239	152964	60.64	39.36
Colombia	133195	68616	51.52	48.48
Ecuador	64350	29250	45.45	54.55
Peru	431905	196321	45.45	54.55
Venezuela	20471	6158	30.08	69.92
Total	**1024229**	**541703**	**52.89**	**47.11**

3)Forest area in the Andes and FAO rates (S3)				
Country	**Forest area in 1959 in km^2**	**Forest area in 2059 in km^2**	**% remaining in 2059**	**%loss in 2059**
Argentina	147324	63139	42.86	57.14
Bolivia	256291	148912	58.10	41.90
Colombia	109482	92328	84.33	15.67
Ecuador	91026	2574	2.83	97.17
Peru	348665	279560	80.18	19.82
Venezuela	17375	9254	53.26	46.74
Total	**970164**	**595767**	**61.41**	**38.59**

Table 4.4 Estimation of forest loss in the past 50 years and projected for the next 50 years as calculated in **Tables 4.2** and **4.3**. 1) Total forest area per country and deforestation rates provided by FAO (2010). 2) Andean forest area from GlobCover in 2009 and mean deforestation rates from the literature available in **Table 3.4**, **Chapter 3**. 3) Andean forest area from GlobCover in 2009 and mean deforestation rates provided by the FAO (2010).

The RL assessment carried out here identified one species as Critically Endangered (CR), namely *Polylepis microphylla*, which also in 2011 was evaluated by the Ecuadorian authorities as CR within that country. Its status in Peru has not been verified and therefore upgraded from the preliminary category EN to CR. In the Endangered (EN) category there were 47 species, Vulnerable (VU) 28, Near Threatened (NT) 19, Least Concern (LC) 29 and Data Deficient (DD) 5 (**Figure 4.8**). The 5 species that were classified as DD were *Ilex maasiana*, for which there were two records obtained from Missouri Botanical Garden that were recorded as affiliated to this species, and no other records were available to review. This is a small tree shrub and was not known by the reviewing experts. *Allophylus coriaceus* could be possibly listed as LC, as its EOO is large; however, more information is needed to evaluate this species. *Phenax laxiflorus* was believed to be a small shrub and the records available for the species did not provide further information to perform an evaluation. *Cyathea catacampta* and *Prunus muris* were taxonomically unresolved and further information about them was not available to conduct an assessment.

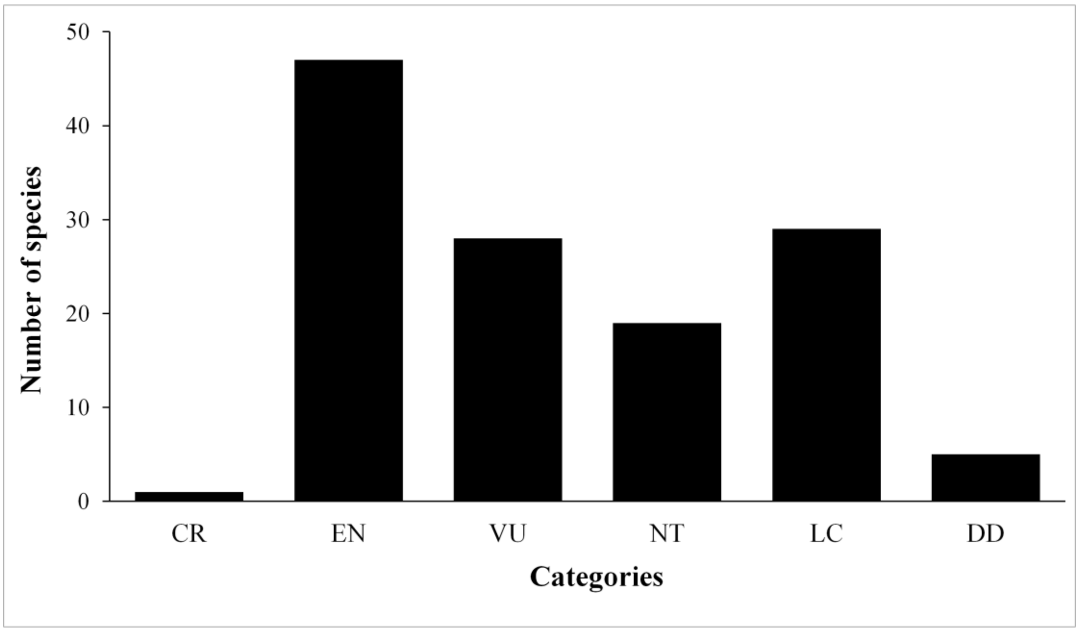


Figure 4.8 Number of species within each of the RL Categories. CR: Critically Endangered, EN: Endangered, VU: Vulnerable, NT: Near Threatened, LC: Least Concern and DD: Data Deficient.

The IUCN requires that the all of the Criteria are considered when carrying out the RL assessment and that the highest level of threat is given. **Tables 4.5** and **Appendix III** show the preliminary category assigned to the species according to the individual Criteria. The classification was made according to the information available to assess the species and taking into consideration any applicable sub-criteria. Additional information collected from the experts assisted in the final classification of the species, which is also presented on these tables.

Criteria	Time duration (years)	Method	Past (P) Future (F) Both (B)	Preliminary Category CR	EN	VU	LC	NT	DD	n.a.
A1 reduction in population size										129
A2 reduction in population size	50	S1	P				129			
A2 reduction in population size	50	S2	P				129			
A2 reduction in population size	50	S3	P				129			
A2 reduction in population size	100	S1	P			129				
A2 reduction in population size	100	S2	P			129				
A2 reduction in population size	100	S3	P			129				
A2 reduction in population size	100	EK	P		3	46			80	
A3 reduction in population size	50	S1	F				129			
A3 reduction in population size	50	S2	F			129				
A3 reduction in population size	50	S3	F				129			
A3 reduction in population size	100	S1	F			129				
A3 reduction in population size	100	S2	F		129					
A3 reduction in population size	100	S3	F			129				
A3 reduction in population size	100	EK	F		5	42			82	
A4 reduction in population size	100	S1	B			129				
A4 reduction in population size	100	S2	B			129				
A4 reduction in population size	100	S3	B			129				
A4 reduction in population size	100	EK	B			6	1		122	
B1 small range (EOO)[1]		EK/Data			10	15	95		9	
B1 small range (EOO)[2]		EK/Data			17	31	72		9	
B2 small range (AOO)[3]		EK/Data			115				14	
B2 small range (AOO)[4]		EK/Data			15	81	19		14	
C1 small and declining population[5]						1			128	
C2 small and declining population[5]									129	
D1 very small population[5]									129	
D2 very small range locations[5]						1			128	
E quantitative analysis[5]									129	

Table 4.5 Preliminary classification category using the IUCN criteria for each of the species. The classification was made according to the information available to assess the species without taking into consideration any applicable sub-criteria. S1: based on the calculation of forest loss using deforestation rates and national forest area by FAO (2010). S2: based on the calculation of forest loss using the regional averages of deforestation rates from the literature and Andean forest area from the GlobCover map. S3: based on the calculation of forest loss using deforestation rates from the FAO (2010) and Andean forest area from the GlobCover map. EK: based on expert knowledge. [1]: based on the MCP area for each species. [2]: based on the GlobCover map area for each species. [3]: based on the AOO at 4 km^2. [4]: based on the AOO at 100 km^2. [5]: based on information available for the species.

After considering all the criteria in the RL assessment for which information was available (**Appendix II**), the criteria which gave the highest category of threat and relatively low uncertainty were considered for the final classification of the species (IUCN Standards and Petitions Subcommittee 2011). Therefore, for each species, the sub-criteria on which classification depended were identified, to specify under which conditions the species were assigned the final category (**Table 4.6**). Results indicated that sub-criterion B2ab(iii) was the most widely met by species throughout the assessment, both in isolation and in conjunction with other sub-criteria.

	Category			
Sub-criteria	CR	EN	VU	Total
A2acd; B2ab(iii,v)		1		1
A2c; A3c; B1ab(iii); B2ab(iii)		1	3	4
A2c; A3c; B1ab(iii,iv); B2ab(iii,iv)			1	1
A2c; A3c; B2ab(ii,iii)			1	1
A2c; A3c; B2ab(iii)			3	3
A2c; B1ab(iii); B2ab(iii)			1	1
A2cd; A3cd; B1ab(ii,iii,iv,v); B2ab(ii,iii,iv,v)		1		1
A2cd; A3cd; B1ab(iii); B2ab(iii)			1	1
A2cd; A3cd; B2ab(iii)			1	1
A2cd; A3cd; B2ab(iii,iv,v)			3	3
A2cd; A3cd; B2ab(iii,v)			1	1
A3c; B1ab(iii); B2ab(iii)		1		1
A3c; B1ab(iii,v); B2ab(iii,v)		1		1
A3c; B2ab(iii)		1		1
B1ab(ii,iii); B2ab(ii,iii)		1		1
B1ab(iii); B2ab(iii)		6	6	12
B1ab(iii,v); B2ab(iii,v)		4		4
B2ab(i,ii,iii)		1		1
B2ab(ii,iii)		1		1
B2ab(iii)	1	18	5	24
B2ab(iii,iv)		1		1
B2ab(iii,iv,v)		2	1	3
B2ab(iii,v)		7	1	8
Total	**1**	**47**	**28**	**76**

Table 4.6 Summary of the different sub-criteria assigned to the species under the threatened categories, CR (Critically Endangered), EN (Endangered) and VU (Vulnerable).

The level of uncertainty measured in this assessment under each of the criteria of the RL varied significantly among criteria (**Table 4.7**). Greater uncertainty was identified in the Criteria for which information was scarce. Criterion B had the largest number of species with low uncertainty as expert knowledge assisted in the process, with Criterion B1 having the largest number of species assessed with lowest level of uncertainty.

Criteria	Uncertainty level High	Medium	Low
A1 Reduction in population size	128	1	
A2: reduction in population size[1]	98	20	11
A2: reduction in population size[2]	129		
A3: reduction in population size[3]	129		
A4: reduction in population size[4]	129		
B1: small range (EOO)[5]	30	6	94
B1: small range (EOO)[6]	16	113	
B2: small range (AOO)[7]	47	58	24
B2 : small range (AOO)[8]	45	44	40
B(a) Severely fragmented[9]	44	59	26
B(a) Locations less than 10[9]	32	57	40
B(b) Continuing decline[9]	31	63	35
B(c) Extreme fluctuation	129		
C1: Number of mature individuals less than 10000	128	1	
C1: Continuing decline in the future more than 10%	128	1	
C2: small and declining population	128	1	
C2: ai) Number of mature individuals >1000	128	1	
C2: aii) % individuals in subpopulation is 90-100%	128	1	
C2: b) Extreme fluctuation of number of individuals	128	1	
D: Individuals less than 1000 and <20 km^2 AOO or less than 5 locations	129		
D2: Restricted area of occupancy and with a threat that could change taxon to CR or EX	128	1	
E: Quantitative analysis	129		

Table 4.7 Uncertainty for all the species under each of the RL criteria and the information used to assign the classification. [1]) A2: based on expert knowledge. [2]) A2: based on the three scenarios detailed in the methodology to estimate the percentage of forest loss over a period of 50 to 100 years in the past. [3]) A3: based on the three scenarios detailed in the methodology to estimate the percentage of forest loss over a period of 50 to 100 years in the future. [4]) A4: based on the three scenarios detailed in the methodology to estimate the percentage of forest loss over a period of 100 years including past and future. [5]) B1: based on the MCP and expert knowledge. [6]) B1: based on the GlobCover map area for each species. [7]) B2: based on the AOO at 4 km^2 and expert knowledge. [8]) B2: based on the AOO at 100 km^2 and expert knowledge. [9]) Based on expert knowledge.

4.5 Discussion

In this assessment, of the 129 tree species evaluated using the IUCN RL criteria, 76 species were classified within a threatened category. 467 candidate species were excluded from the assessment as they are endemic to or only had georeferenced records available from one country of the 6 countries studied. 64 species were known to be in more than one country but were not associated with georeferenced records, and were therefore excluded. Of the national endemic species, 199 have been evaluated previously at the scale of individual countries using the IUCN Categories and Criteria (**Figure 4.8**). 87 of these were listed in the IUCN website: (www.iucnredlist.org, consulted May 2010), of which 84 are species from Ecuador. Taking into consideration these evaluations and the results of the current research, a total of 241 tree species have been identified as being under threat in the tropical Andes (**Figure 4.8**). 32 species were Near Threatened, as these fulfilled most but not all the Criteria and sub-criteria, or were close to the Vulnerable Category.

Country	CR	EN	VU	NT	LC	DD	NE	Total
Ecuador	2	36	52	9	5	1	61	**166**
Peru	9	31	15	2	3	10	50	**120**
Colombia	4	5	5	2	1	0	60	**77**
Bolivia		5	1			1	94	**101**
Argentina							3	**3**
Venezuela								**0**
Total endemics	**15**	**77**	**73**	**13**	**9**	**12**	**268**	**467**
This evaluation	**1**	**47**	**28**	**19**	**29**	**5**	**-**	**129**
Total Andes	**16**	**124**	**101**	**32**	**38**	**17**	**268**	**596**

Table 4.8 Red List classification of national endemic species evaluated previously in national-scale assessments (Calderón *et al.* 2002; IUCN 2010; León-Yánez *et al.* 2011; León 2006; Llamozas *et al.* 2003; Meneses and Beck 2005). In addition, the results of this assessment are presented, focusing on those species present in more than one country, together with the total number of species in each threat category.

The tropical Andes is a centre of plant endemism (Kier *et al.* 2005; Morawetz and Raedig 2007; Myers *et al.* 2000). Brummitt and Lughadha (2003) have further identified the tropical Andes as one of the hottest biodiversity hotspots for conservation, in terms of the species richness, endemism and the relation with the extent of the remaining vegetation available in the hotspot. Previous regional assessments of biodiversity such as those carried out by Brooks *et al.* (2002) and Myers *et al.* (2000) in the tropical Andes biodiversity hotspot have identified that out of an estimated 45,000 plant species present in the hotspot, 20,000 are endemic to the Andes, of which 78 species were identified as threatened in the IUCN RL. Furthermore, these authors identified 3,389 species of mammals, birds, reptiles and amphibians present in the region. 1,567 of these species were identified as endemic, 124 endemic species of which were considered threatened, with two bird species having gone extinct. More recently, Orme *et al.* (2005) identified 2,139 bird species present in this hotspot, of which 483 are estimated to be endemic within the hotspot and 114 are threatened. A further study carried out by Young *et al.* (2001) identified that the tropical Andes above 1000 m. a.s.l. in Venezuela, Colombia, Ecuador and Peru have the highest population decline of amphibians in Latin America, with several species locally extinct. In the case of fish in the tropical Andes, which have been little studied, it has been estimated that there are around 400 to 600 species present and at least 40% of them are endemic, some of which are threatened (Anderson and Maldonado-Ocampo 2011).

Other RL assessments at the national scale have identified the number of endemic species and the number of these that are threatened in different species groups (Vié *et al.* 2009; **Table 4.9**). These values provide a basis for comparison with the total of 241 tree species that are endemic to the region and are also threatened, based on the results of the current assessment and the national-scale assessments undertaken previously. These values suggest that the number of threatened tree species that are endemic to the region is substantially higher than the equivalent number of mammals and birds, but less than that of amphibians, as this assessment only included a fraction of the total tree species that are endemic to the region that didn't fit the criteria to be assessed in this evaluation. When comparing these figures, however, it should be noted that the current assessment was limited to altitudes >1500 m, whereas these figures for other species groups refer to the entire tropical Andean region, including lowland areas.

	Mammals		Birds		Amphibians		Conifers		Cycads	
Country	Total endemic	Threatened endemic	Total endemic	Threatened endemic	Total endemic	Threatened endemic	Total endemic	Threatened endemic	Total endemic	Threatened endemic
Argentina	82	13	12	0	37	21	0	0	0	0
Bolivia	22	4	15	5	63	32	2	0	1	0
Colombia	37	9	65	40	333	158	0	0	6	6
Ecuador	29	11	32	17	155	100	0	0	1	1
Peru	55	19	106	36	217	69	0	0	2	2
Venezuela	19	6	38	14	139	62	5	1	0	0
Total	**244**	**62**	**268**	**112**	**944**	**442**	**7**	**1**	**10**	**9**

Table 4.9 Endemic species from the tropical Andean countries and the number of endemic species that are threatened. Adapted from: Vié *et al.* (2009).

Comparison of the results obtained in this RL assessment with previous assessments of tree species in other locations, such as those described in Newton and Oldfield (2008), shows that the percentage of regionally endemic species threatened in this assessment (59%) was somewhat higher than the mean value (45%) recorded in previous targeted assessments (**Table 4.10**). In comparison with other assessments of tree species undertaken in the Latin American region, the percentage of threatened species is higher than in Guatemala and in Central American dry forest, but slightly lower than that of Mexican cloud forest (González-Espinosa *et al.* 2011). The Mexican assessment included many local endemics, which were excluded from the current assessment, suggesting that the degree of threat to montane tree species in the tropical Andes is at least comparable to that of Mexico. The principal threatening processes are similar in the two regions, namely deforestation and forest fragmentation (González-Espinosa *et al.* 2011). Together, these results provide evidence of the fact that montane biodiversity is particularly threatened (Cayuela *et al.* 2006; Cincotta *et al.* 2000; Jarvis *et al.* 2010).

Assessment focus	No. of tree taxa included	% threatened	No. of taxa included in different Red List categories						
			CR	EN	VU	LC	NT	DD	NE
Andes (**this assessment**)	129	59	1	47	28	29	19	5	0
Acer spp. (maples)	179	26	6	15	25	80	8	20	25
Quercus spp. (oaks)	508	11	13	16	27	97	22	33	ca. 300
Caucasus region	150	21	7	10	15	65	14	39	0
Ethiopia and Eritrea	135	78	42	35	28	19	9	2	0
Magnoliaceae (magnolias)	245	46	31	58	23	20	9	10	94
Guatemala	48	52	10	13	2	0	0	23	–
Cuba	125	90	73	23	17	8	2	2	–
Dry forest of Mesoamerica	544	8	1	17	24	457	39	6	–
Central Asia	97	47	25	12	9	25	8	18	–
Mexican cloud forest	762	61	83	206	175	215	78	2	0

Table 4.10 Comparison of the results in this assessment with recent Red List assessments of different groups of tree species, coordinated by the Global Tree Specialist Group, described in Newton and Oldfield (2008) and González-Espinosa *et al.* (2011) for Mexican cloud forest. CR: Critically Endangered; EN: Endangered; VU: Vulnerable; LC: Least Concern; NT: Near Threatened; DD: Data Deficient; NE: Not Evaluated.

Although deforestation is one of the principal threats to many montane tree species (**Table 3.5** in **Chapter 3**), other threatening processes may also be influential. Threats such as invasion of exotic species, forest degradation and overexploitation (Asner *et al.*

2009) may also be affecting tree populations over the long term (Gibson *et al.* 2011). Species such as *Polylepis* spp. have been heavily exploited in the Andes (Bellis *et al.* 2009; Gareca *et al.* 2010; Jameson and Ramsay 2007), and many species of this genus are now restricted to small forest fragments in the landscape. In countries such as Bolivia, only 11% of its potential distribution area remains covered with *Polylepis* woodland (Gareca *et al.* 2010). Also, species of high commercial value such as *Cinchona* spp., *Podocarpus* spp., *Zanthoxylum* spp. and *Ilex* spp., have also been subjected to overexploitation at different times in the past, which is likely to have reduced population size. *Cinchona* spp., for example, were particularly sought after for their medicinal properties up to the 1950s (Cuvi 2011), until a synthetic substitute for quinine was created. Today these species are still exploited by local communities and the forests continue to be degraded (Ayma-Romay and Padilla-Barroso 2009), despite restrictions established by some countries to halt these activities. The populations of these species are further jeopardised by the fact that some regenerate with difficulty in transformed landscapes, as is the case of *Podocarpus* spp. (Ayma-Romay and Padilla-Barroso 2009).

Many uncertainties arise with the application of the RL, many of which are described in the RL guidelines (IUCN Standards and Petitions Subcommittee 2011). Furthermore, applying the RL to plants and particularly to tree species has been recognized to have particular challenges, relating to the lack of accurate information on their status and distribution (Newton and Oldfield 2008; Nic Lughadha *et al.* 2005). The different types and sources of uncertainty in relation to the RL have been identified and discussed by various authors e.g. Akçakaya *et al.* (2000), Mace *el al.* (2008) and Newton (2010). Regan *et al.* (2002) provide a useful overview of the different kinds of uncertainty that are commonly encountered in conservation ecology. Epistemic uncertainty, which characterizes many of the input variables for RL, is classified by these authors into six types: measurement error; systematic error; natural variation; inherent randomness; model uncertainty; and subjective judgment. These authors also recognise linguistic uncertainty, classified into five types: vagueness, context dependence, ambiguity, indeterminacy of theoretical terms, and underspecificity. Akçakaya *et al.* (2000) describe the main sources of uncertainty in the RL process as 1) measurement error, owing to the lack of information for some or most of the variables used in the rules; 2) semantic uncertainty, which is mostly attributable to the inexact definitions used in the IUCN guidelines, and 3) natural variability, including temporal and spatial variation in the population size and distribution.

In terms of the RL assessment presented here, the largest area of uncertainty is measurement error; this can be viewed from different aspects. Firstly, this may be considered in relation to the so- called “Linnean” and “Wallacean” shortfalls (Whittaker *et al.* 2005), which refer respectively to the inadequacy of taxonomic knowledge and distribution data available to assess the species. This has been identified more generally as a major constraint to the conservation planning process in tropical regions (Cayuela *et al.* 2009), and the identification of species distributions for further conservation actions (Lavoie 2013). Although data are increasingly being made available through digitized biological portals and databases such as GBIF and the Missouri Botanical Gardens database, which provide quantitative georeferenced species distribution data (Bachman *et al.* 2011; Beck *et al.* 2013), such data does not always provide an accurate indication of the full distribution of a species. The georeferenced data for the species used in this research, shown in **Figure 4.9**, illustrates the distribution of the species collections evaluated along the Andes. The first aspect that is apparent is that there seem to be gaps in the collection efforts, many of which are explicable in terms of the limited access to different locations (Feeley and Silman 2009). This lack of species data is likely to have resulted in an underestimation of the species’ ranges (Feeley and Silman 2009; Knapp 2002). However, that lack of data in some species was related to their rarity and degree of habitat specialism, implying a restricted distribution (Feeley and Silman 2009). Using expert’s knowledge, to evaluate the accuracy of the records and the estimated species’ distribution, played an important role in deciding an appropriate category of threat for each of the assessed species, reducing the geo-referencing error encountered in the research. This was reduced even further as each of the species was evaluated at least twice due to the fact that the assessed species were to be found in at least two countries.

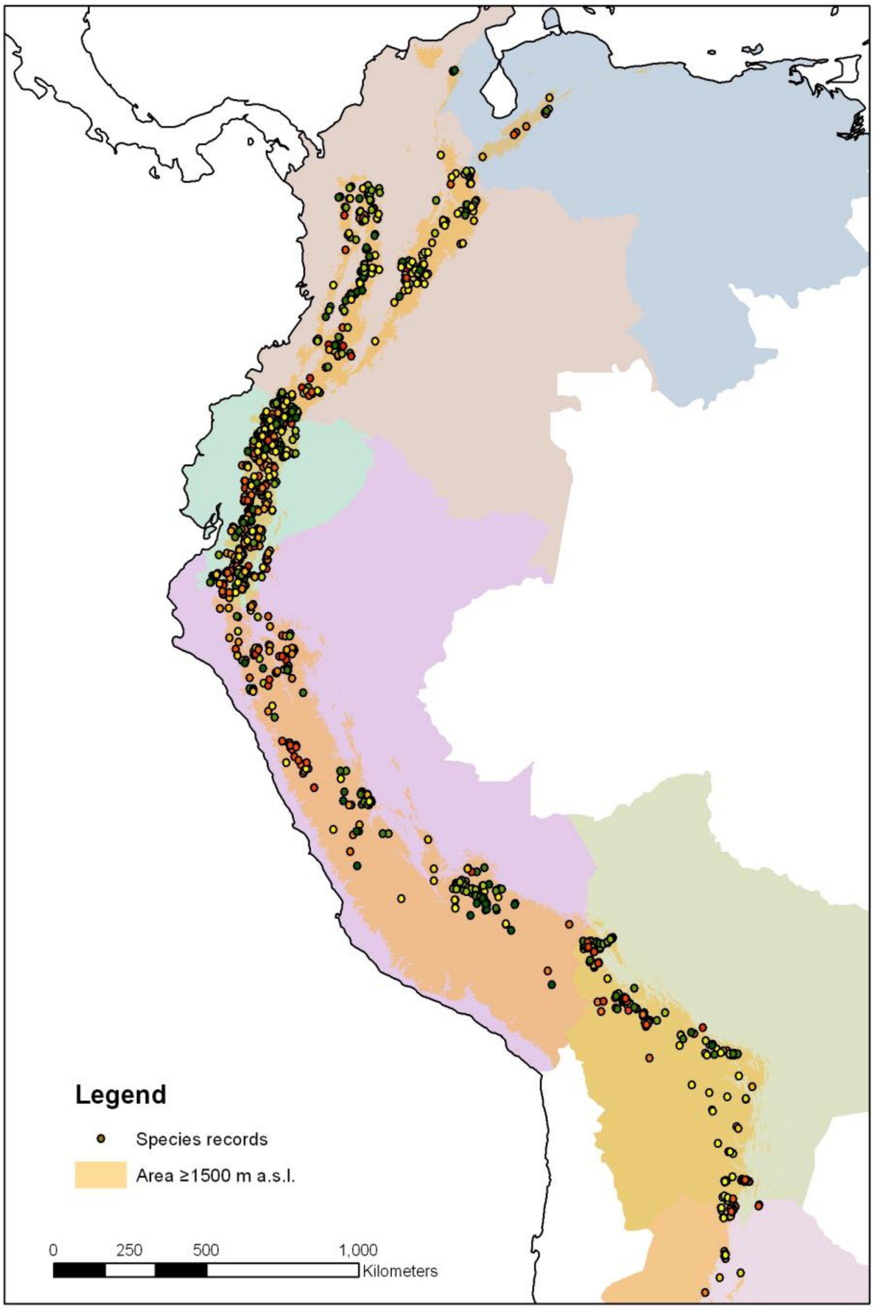


Figure 4.9 Georeferenced distribution of the species collections evaluated along the Andes. Different dot colours are different species.

A second area of uncertainty is semantic uncertainty. Although the latest IUCN RL Categories and Criteria (IUCN 2001), with their corresponding guidelines (IUCN Standards and Petitions Subcommittee 2011), address much of the possible ambiguity of definitions, there is still a significant area of uncertainty in terms of their applicability to tree species, especially when the available data are scarce. For example, the establishment of an appropriate scale to use for AOO estimation of different species has been discussed in the literature (Hartley and Kunin 2003; Willis *et al.* 2003), as the resolution used for this estimation influences the category given to the species (Rivers *et al.* 2011). As illustration, if this value is too coarse, the species may not be able to be listed as CR and vice versa. A further issue in relation to estimation of EOO in relation to the B criteria as indicated by Newton and Oldfield (2008) is that in previous RL assessments of tree species, there was a tendency to use criteria B1 and B2 to classify the species under one of the RL categories (**Table 4.11**). This reflects the strong reliance on herbarium accession data for EOO estimations in tree species, and in plant species more generally (Nic Lughadha *et al.* 2005). While it can be argued that conservation assessments based on range estimates are repeatable and objective (Rivers *et al.* 2011), they only form part of what is required by a comprehensive RL assessment. Perhaps the most significant area of uncertainty in the current assessment was the estimation of actual population size, as inventory data for the majority of tree species in the region is entirely lacking.

	Red List Criterion								
Assessment focus	A1	A2	A3	A4	B1	B2	C1	C2	D
Andes (**this assessment**)	0	16	14	0	12	73	0	0	0
Acer spp. (maples)	1	22	1	0	11	10	1	3	7
Quercus spp. (oaks)	4	12	0	0	17	17	0	1	19
Caucasus region	0	3	0	0	21	11	0	0	0
Ethiopia and Eritrea	0	24	0	0	65	77	0	0	0
Magnoliaceae (magnolias)	0	3	0	3	83	30	4	8	8
Guatemala	0	4	0	0	8	20	0	2	0
Cuba	2	2	1	7	39	96	1	12	17
Dry forest of Mesoamerica	0	0	0	0	40	2	0	0	0
Central Asia	0	3	0	0	12	40	1	0	1
Mexican cloud forest	7	18	17	98	71	53	2	10	4

Table 4.11 Comparison of the frequencies of criteria cited in Red List assessments of tree species cited in Newton and Oldfield (2008). Multiple criteria may be cited for a single taxon.

Additional uncertainty arises from the methodology used for identifying the areas of potential distribution. The method applied here focused on areas that are >1500 m a.s.l., through exclusion of species recorded below this altitudinal threshold. It is possible that owing to the species being under-recorded in certain areas, their distributional range could have been underestimated. Some of the species included in this analysis may therefore actually occur below the altitudinal threshold adopted. Further uncertainty is associated with the use of maps to exclude unsuitable areas. The GlobCover map used in this assessment may have overestimated the species' EOO values, as unsuitable areas were directly related to human activities such as rainfed areas and cropland. Overestimation may have also occurred in areas with land cover types that are not suitable for the species or where the species no longer occur. In addition, some of the transformed vegetation that may not be the original forests but an alternative vegetation type may still support the species and allow their survival. The use of other global forest maps, such as the one produced by Schmitt *et al.* (2009) that defines "forest" using a minimum threshold of 10% tree cover, could also potentially be used to estimate EOO. However, these may exclude isolated patches of shrubs/forest or mixed grasslands/shrubs, where montane tree species in the Andes occur, potentially leading to EOO underestimation.

Expert knowledge played an important role in this assessment. For example, it allowed the identification of the level of rarity of the species and the threats that they face, as well as validation of species distributions where georeferenced data were lacking. Expert judgment has been identified to be important in the RL process but is also an area of uncertainty (IUCN Standards and Petitions Subcommittee 2011). Previously, it has been related to biases in the listing process, where personal interests of the experts resulted in inflated lists of species at risk (Possingham *et al.* 2002). Recently, the RL process has become more robust, reducing reliance on expert knowledge to categorize the species subjectively, but this remains as a fundamental part of the RL assessment (Newton and Oldfield 2008; Rodrigues *et al.* 2006). Expert knowledge has been used widely in previous RL evaluations of tree species (Newton and Oldfield 2008), for example for montane tree species in Mexico (González-Espinosa *et al.* 2011) and Magnoliaceae (Cicuzza *et al.* 2007), but most conservation assessments rely on the use of expert knowledge only when objective observations are not available (Burgman *et al.* 2001; Drew and Collazo 2012; Drew and Perera 2011; Kangas and Leskinen 2005; Orsi *et al.* 2011; Perera *et al.* 2012).

Measuring the level of uncertainty that expert knowledge adds to any conservation assessment is valuable (IUCN Standards and Petitions Subcommittee 2011). In this RL the uncertainty was measured as the level of confidence that experts had for each of the species, in terms of their knowledge of the species and evaluation of the data provided (**Table 4.7**). Expert knowledge relating to Criteria B1 and B2 had the lowest level of uncertainty, as expertise was validated by empirical data provided by the available herbarium records and preliminary estimations of EOO and AOO provided by GIS. Distribution data are typically the most abundant resources available to experts undertaking RL assessments (Bachman *et al.* 2011; Newton and Oldfield 2008). The areas in which the experts had greater uncertainty were related to estimation of the AOO and EOO of those species that had relatively few distributional records. The current research supports the finding of Rivers *et al.* (2011), who suggested that there is a need for at least five good valid records in order to carry out a robust RL assessment.

In conclusion, this assessment identified that the number of threatened trees in the tropical Andean region is high relative to other groups of organisms such as mammals, birds and fish, and provides further evidence of the congruence that occurs in this region of species richness, endemism and threat. Therefore tree species should be factored into conservation plans in the region and be a priority for conservation action, as they are currently not taken into consideration as much as other more “charismatic” species such as birds or large mammals.

Recent research by Giam *et al.* (2010), which used the number of vascular plants that occur globally, described in Kier *et al.* (2005), and the ecoregions where these species occur, described by Olson *et al.* (2001), established a nonlinear mixed-effects species-area relationship, to identify the species richness in each ecoregion and the countries in which they occur. These authors have also added historical land use cover and projected human population density to identify current and future endangerment of vascular plants, together with the average per capita gross national income adjusted for purchasing power parity, to identify the financial resources of individual countries and estimate how they overlap with those threatened ecoregions. Although this is a different approach to the one in this research, it has also concluded that plant species in many countries, such as those in the tropical Andes, are threatened and in need of conservation actions, owing to the large number of endangered species and the poor governance and underfunding, despite efforts of non-governmental advocacy groups.

Other recent research by Waldron *et al.* (2013), analysed mammal biodiversity and country-level conservation funding from different sources including government, donors, trust funds, and self-funding, to establish where there is underfunding at the global scale. These authors concluded that Colombia and Venezuela are in the top 40 countries that are underfunded and have high levels of threatened biodiversity (mammals). This coincides with current research, identifying the Andes as an area with important biodiversity at the global scale, which is currently highly threatened. Despite the funding available for conservation actions, there is still a need for coherent conservation planning and targeted investment to reduce biodiversity loss.

Since the Tropical Andes are an important area of the world for biodiversity conservation and much of it is considered under threat, a range of conservation measures have been proposed in the region in order to respond to these pressures (Dinerstein *et al.* 1995). Different institutions have invested heavily on developing, strengthening and protecting key biodiversity areas (KBAs). Some of these institutions are: Conservation International (CI), BirdLife International, Wildlife Conservation Society, World Wildlife Fund, The Nature Conservancy, as well as the national governments of the states within the region. For example, BirdLife in collaboration with CI have identified important areas for the conservation of birds (BirdLife and Conservation International 2005), based on the number of endangered bird species occurring in the Andes. However, as this assessment itself recognized, it does not integrate information on other endangered fauna and flora, but is a step towards the creation of KBAs in the region. Another example of the efforts of these institution is the work carried out by CI in order to increase the number of biodiversity corridors in the region (Conservation International 2013; Critical Ecosystem Partnership Fund 2006), which has been identified as an area with several endemic species that are threatened and in need of conservation actions.

These examples illustrate the general state of the biodiversity in montane forest in the tropical Andes, the lack of information regarding the species that need to be prioritised for conservation action, and imperfect information regarding the areas where these efforts need to be directed. Threatened trees species do not always form a fundamental role in the conservation actions. The role of protected areas in the conservation of endangered trees species will be explored in **chapter 5**, as identifying the areas where the distribution of multiple threatened species coincide, which could provide valuable information for conservation actions.

4.6 References

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5 Modelling potential species distribution under climate change scenarios for the tropical Andes

5.1 Introduction

Historically, climate change has had a great influence on shaping the distribution of species (Cárdenas *et al.* 2011; Davis and Shaw 2001; Luterbacher *et al.* 2004), and today it is recognised as one of most influential causes of ecosystem change (Parmesan 2006; Parmesan *et al.* 2013; Rosenzweig *et al.* 2008; Walther *et al.* 2002). The rate at which changes will occur during the next century will determine whether species are able to adjust, for example by migrating to suitable habitats, or will face extinction (Saxe *et al.* 2001; Thomas *et al.* 2004). Observed migration rates required for tree species have been estimated in different parts of the world, ranging from 20 to 40 km per century, whereas the projected rate for the 21st Century is expected to be 300 to 500 km per century (Davis and Shaw 2001). Past climate changes in montane forests, as in the Tropical Andes, have shown that an increase of ~6°C has led to an upward migration of lowland species which in turn reduced and in many cases replaced highland taxa (Bush *et al.* 2004; Cárdenas *et al.* 2011; Feeley *et al.* 2012; Urrego *et al.* 2010). These estimations show that previous migration rates are far below the rates required to track projected climate changes in the future, and some species are already showing a contraction in their distribution ranges (Lindner *et al.* 2010; Zhu *et al.* 2012). This highlights the potential impact of climate change on the extinction risk of species (Root *et al.* 2003), especially for endemic species (Malcolm *et al.* 2006; Thomas *et al.* 2004; Thuiller *et al.* 2011) and those in areas where biodiversity is already threatened by habitat loss and degradation (Giam *et al.* 2010; Ramírez-Barahona *et al.* 2011; Travis 2003).

Assessing the effects of climate change on forest communities requires an understanding and quantification of current mechanisms controlling present geographic distributions, and how such distributions may change in future in response to climate change (Sykes and Prentice 1996). Development of such projections requires some kind of modelling approach. Modelling can be performed using ‘mechanistic’ models based on ecophysiology, or ‘bioclimate envelope’ models based on correlations between species distribution and climate. Species distributions correlate with climate at a range of spatial scales (Way and Oren 2010). At a small scale, topography plays a role in mountain ranges, modifying the macroclimate and producing an altitudinal distribution of the species according to the climatic gradient (Trivedi *et al.* 2008). Climatic limitations that restrict species distributions at a continental or altitudinal scale can be physiological constraints on survival or reproduction, or biotic interactions such as competition (Attorre *et al.* 2011; Ruiz-Benito *et al.* 2013). The direct and indirect effects of these constrains may dominate at opposing range margins, such that the cold (poleward or upper altitudinal) margins are driven by intolerance to a stressful climate while warm margins are delineated by biotic interactions (Loehle and LeBlanc 1996; Trivedi *et al.* 2008).

There are several methods that can be used for modelling species distributions (**Appendix IV**). For example, MaxEnt (Phillips *et al.* 2004) has recently become a popular method to model species' potential distribution (e.g. Busch *et al.* 2012; Jiménez-Alfaro *et al.* 2012; Rinnhofer *et al.* 2012; Warren *et al.* 2013), as it can be used effectively when distribution data are scarce (Hernandez *et al.* 2008; Pearson *et al.* 2007). This is a characteristic of many of the tree species within the tropical Andes. Also, this method has recently been used to investigate the vulnerability of plant species to climate change (Crossman *et al.* 2012; Elith *et al.* 2006a). However, this capacity to work with small number of records has been attributed to the fact that the program assumes that the data has been systematically collected or randomly sampled (Phillips *et al.* 2009; Royle *et al.* 2012), which is not the case for the species studied here. This in turn, results in the autocorrelation of the model residuals and inflates model accuracy (Kramer-Schadt *et al.* 2013). Recent research by Golicher *et al.* (2011) identified that the results from MaxEnt are very similar to Generalized Additive Models (GAM). Furthermore, more recently Renner and Warton (2013) Other methods that can potentially be used for modelling species distributions include Support Vector Machines (KSVM), and regression trees. These have been used to determine species potential distribution of tree species in other regions e.g. Golicher *et al.* (2011). GAMs are a method for detecting non-linearity of predictors and response function and then building a parametric model.

This technique applies smoothers independently to each predictor and additively calculates the component response (Franklin 2009; Guisan and Zimmermann 2000; Yee and Mitchell 1991). This is often used in multiple regression analysis. This method has been demonstrated to have high predictive accuracy when used for spatial prediction and species distribution modelling (Franklin 2009). Classification trees or recursive partitioning are used to predict membership of cases of a categorical dependent variable from their measurements on one or more predictor variables (De'ath and Fabricius 2000). Classification trees are developed using different measures that recursively split data sets into increasingly homogeneous subsets representing class membership, based on ranges of values of predictor variables (De'ath and Fabricius 2000). This takes place in three stages: tree building, tree stopping and tree pruning. All classification tree approaches employ hierarchical, recursive partitioning of the data, resulting in decision rules that relate values or thresholds in the predictor variables with pixel classes (Rogan *et al.* 2008). KSVM are perhaps the least widely used of these three methods, but carry out classification and regressions. The approach uses a "kernel trick", which is a function that returns the inner product between two points in a suitable feature space, thus defining a notion of similarity, with little computational cost even in very high-dimensional spaces (Karatzoglou *et al.* 2006).

Identifying the potential effects of climate change on species potential distribution does not only give an insight into individual species, which can contribute to improved conservation status assessments and future conservation measures, but can also inform conservation planning and management at a regional scale (Busch *et al.* 2012). Outcomes could have implications for the choice of selecting areas and networks for conservation priorities, as currently these are typically fixed in space and time (Araújo *et al.* 2004). Furthermore, as climate change occurs, ranges of some species may well shrink, but for some other species ranges may shift, and therefore the composition of communities will be transformed. Therefore, existing protected areas may not include areas where species may be located in the future (Hole *et al.* 2011a; Shaw *et al.* 2012).

Climate change is becoming a focal issue for conservation planning and policy making, as extensive research provides evidence on the potental impacts of climate change on species, increasing their extinction risk (Araújo *et al.* 2005; Butchart *et al.* 2010; Feeley *et al.* 2011; Keith *et al.* 2008; Midgley *et al.* 2002; Schwartz *et al.* 2006; Thomas *et al.* 2004) and more recently in cloud forests (Ponce-Reyes *et al.* 2013). However, only a few studies e.g. (Bomhard *et al.* 2005; Thuiller *et al.* 2005) have assessed species extinction risk under projected climate change using the Red List assessement and none of these have been carried out in the Tropical Andes.

The hypothesis tested in this chapter is that projected changes in distributional range resulting from climate change will increase the extinction risk of many tree species, particularly those associated with high elevations and those with restricted geographical ranges. Therefore, the main objective of this research is to investigate the potential impact of climate change on the extinction risk of selected tree species using different species distribution modelling techniques, using the Red List assessment process described in **Chapter 4**.

5.2 Methodology

Study area

The tropical Andes is a unique region with multiple ecosystem types, reflecting high species and ecosystem diversity as a result of large altitudinal and latitudinal gradients (Josse *et al.* 2003; Young *et al.* 2002; Young 2007). Montane forests of the tropical Andes contain the largest concentration of species with restricted distribution in South America (Dinerstein *et al.* 1995; Latta *et al.* 2011), which is manifested in the high number of endemic species of fauna and flora (Brehm *et al.* 2005; Brooks *et al.* 2002; Churchill 1996, 2009; Fjeldså and Irestedt 2009; Grenyer *et al.* 2006; Latta *et al.* 2011; Myers *et al.* 2000; Pennington *et al.* 2010). Latin America has suffered a high loss of forest habitat (DeFries *et al.* 2005) and the tropical Andes is an area currently subjected to processes of exploitation, colonization, deforestation, fragmentation and resource extraction (**Chapter 3**), to which climate change could further contribute to the long term biodiversity (Bellard *et al.* 2012). The tropical Andes region in this assessment is described as montane areas above 1500 m a.s.l., occurring in Argentina, Bolivia, Colombia, Ecuador, Peru and Venezuela. Montane forests in the tropical Andes are identified as those occurring from Venezuela, starting in the depression of Barquisimeto in Lara state, until they reach their southernmost limit in the Northwest of Argentina (29° S) in the border between Catamarca and La Rioja provinces. Including this sub-tropical point in Argentina as part of the study allows the identification of potential areas to which species may be able to migrate, seeking cooler southern environments (Feeley 2012). To investigate the species distribution in the region predictions were made to an area covering: Longitude -81.3^{o} to -55.3 ^{o} W and Latitude 12.3 ^{o} N to -29.3^{o}S.

Species Data

Presence-only data was obtained for 129 tree species as described in **Chapter 4**. Records were filtered to identify unique records for each location. The total number of records for all the species was 1666, with *Cestrum peruvianum* having the largest number of records (65). 25 species had fewer than 5 records and were therefore excluded from this analysis and classified as Data Deficient (DD). Modelling methods do not perform well with fewer than 5 records (Hernandez *et al.* 2006) and more recently it has been identified that in order to estimate a realistic species' conservation status, at least 15 georeferenced records are preferable (Rivers *et al.* 2011). Having few localities increases the problems associated with the generation of independent data to test the models and the statistical tests to validate them (Pearson *et al.* 2007). Therefore, the analysis undertaken here needs to be interpreted with caution, especially for those species with few records.

A robust modelling process would include absence data (Lobo *et al.* 2010; Raes and ter Steege 2007). However, these data are more limited than presence-data and were not available in this study. Therefore 1000 randomly selected background data points were obtained to characterize the environment in the study region (Golicher *et al.* 2012).

Environmental data

Current climate data were derived from the WorldClim data set (Hijmans *et al.* 2005), which is available at 1 km resolution. Making decisions about which environmental variables to use and their relative contribution remains a challenge for species distribution modelling, as selecting variables that are physiologically relevant is very important (Synes and Osborne 2011; Williams *et al.* 2012). As the species data are sparsely sampled throughout the region, less information is likely to be revealed about the relationship between response and explanatory variables than in more densely sampled regions. Therefore, in this situation, fewer variables may be needed to adequately predict areas with similar environments, although they might inadequately define the range limits (Williams *et al.* 2012). Many environmental variables are typically highly correlated (Braunisch *et al.* 2013; Golicher *et al.* 2012) and the selection of those variables that are the most influential on the species distribution is therefore a challenge (Williams *et al.* 2012). A principal component analysis (PCA) analysis provides a view of the multiple colianearity among the different variables, allowing the visualization of the weights that each variable has on the species' presence points. Hijmans and Graham (2006) found that the more variables were used from BIOCLIM data the more under-prediction is made. Therefore, for the purpose of investigating the effect of variables in the models, five combinations of variables with high component load were tested with the different models, which were: a) 2, 5, 12 and 14; b) 2, 3, 13 and 14; c) 5, 9, 16 and 18 and d) 2, 5, 13 and 14, e) 2, 5, 12 and 13 (see **Table 5.1.** for the variable details).

Based on the PCA weightings, biological interpretability (Golicher *et al.* 2011) and because most of the variability was explained by the first four axes from which each of the variables were taken, the following four variables were selected: 1) Mean annual temperature, 2) Mean diurnal range (mean of monthly (max temp. - min temp.)), 3) Precipitation of the wettest month and 4) Precipitation of the driest month. Some of these variables have been used previously to investigate the potential impact of climate change on the ecoregions in the region (Tovar *et al.* 2013), potential species distribution in some Andean countries (Loiselle *et al.* 2008) and climate change effects in other tropical montane cloud forests in the neotropics (Ponce-Reyes *et al.* 2013).

N	Variables	Comp.1	Comp.2	Comp.3	Comp.4	Comp.5
1	Elevation		0.400			0.133
2	BIO1 = Annual Mean Temperature		-0.396			
3	BIO2 = Mean Diurnal Range (Mean of monthly (max temp - min temp))					0.628
4	BIO3 = Isothermality (BIO2/BIO7) (* 100)			0.576		
5	BIO4 = Temperature Seasonality (standard deviation *100)			-0.575	0.109	0.107
6	BIO5 = Max Temperature of Warmest Month		-0.351			0.208
7	BIO6 = Min Temperature of Coldest Month		-0.334	0.209		-0.237
8	BIO7 = Temperature Annual Range (BIO5-BIO6)			-0.236		0.432
9	BIO8 = Mean Temperature of Wettest Quarter		-0.387	-0.102		
10	BIO9 = Mean Temperature of Driest Quarter		-0.382	0.19		
11	BIO10 = Mean Temperature of Warmest Quarter		-0.385			
12	BIO12 = Annual Precipitation	-0.370			0.208	
13	BIO13 = Precipitation of Wettest Month	-0.536				
14	BIO14 = Precipitation of Driest Month				0.570	
15	BIO15 = Precipitation Seasonality (Coefficient of Variation)	-0.224			-0.409	0.269
16	BIO16 = Precipitation of Wettest Quarter	-0.535				
17	BIO17 = Precipitation of Driest Quarter				0.53	
18	BIO18 = Precipitation of Warmest Quarter	-0.431		-0.354		-0.372
19	BIO19 = Precipitation of Coldest Quarter	-0.210		0.197	0.401	0.266

Table 5.1 PCA output matrix of five components (comp. 1-5) of the environmental variables considered for species modelling. Values presented are weights (regression coefficients). Climatic variables and elevation were obtained from the WorldClim dataset (Hijmans *et al.* 2005).

Climate change scenarios

For analysis of potential distribution under climate change scenarios, data were used from the results of the Hadley Centre Coupled Model for two of the Special Report on Emission Scenarios (SRES) for the year 2080, namely HADCM3 for scenario A2 and B2. These were prepared for the Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Report (IPCC 2007) and were downscaled by Ramirez and Jarvis (2008). These scenarios were designed to consider different trajectories of future economic development, and energy use and perhaps this choice of scenarios is seen as conservative or optimistic as more fossil fuel-intensive scenarios such as A1FI were excluded, but they still have significant implications for biodiversity. Also, recent research suggests that the more extreme warming predictions may be less likely to occur (Huntingford 2013). The A2 scenario represents a very heterogeneous world, where population continues to increase at a higher rate than the B2 scenario. Fragmented and slower economic growth and technological change characterises this scenario when compared to other scenarios (IPCC 2001; Joos *et al.* 2001). This scenario is commonly used for 'business as usual' impact studies, as it projects a 3°C increase in surface air temperature by 2100 on average across the model (Cubasch *et al.* 2001). The B2 scenario depicts a world that emphasises local solutions to social, economic and environmental sustainability, where human population continues to increase with an intermediate level of economic development, and therefore a less energy-intensive scenario featuring a lower emission path, projecting a 2.2°C temperature increase on average across all models (IPCC 2001).

Predictions and interpretation

To determine the species potential distribution the following methods were initially explored, following Golicher *et al.* (2011): generalized additive models (GAM) (Hastie 2008), recursive partitioning (rpart) (Therneau *et al.* 2013), Support Vector Machines (KSVM) (Karatzoglou 2013) and Maximum Entropy (MaxEnt) (Phillips *et al.* 2006). After considering the similarities between MaxEnt and the other methods, only the results of 'GAM',' rpart' and 'KSVM' are presented in this chapter. The scripts used were modified from those provided by both, the R packages and by Duncan Golicher in the R statistical environment (R Development Core Team 2011). KSVM models were used from the 'kernlab' package, regression trees from the package 'rpart' and GAM from the 'mgcv' package, all of which are available in R (R Development Core Team 2011). All the maps and outputs were resampled to a resolution of ~25km^2.

Model validation

Model validation is an important part of modelling species distributions. This is carried out to identify if the model has the potential to discriminate presence data from absence (or background) data better than random. The Akaike information criterion (AIC) is commonly used to identify the most parsimonious models. A similar technique has been found to be the most effective means of optimizing predictions of distributional change for European plants (Thuiller *et al.* 2005). Also, the area under the receiver operating characteristic (ROC) curve, known as the

AUC, is also commonly used in predictive modelling to evaluate the performance of the models used. More specifically, ROC is used to evaluate the ability of models to distinguish between a true presence and a true absence drawn at random (Pearce and Ferrier 2000). ROC results are more accurate when there is truly independent data to test the models as well as the use of real absence data. However, as these were not available for this research, ROC curves were calculated using a subset of the original data, to measure the power of discrimination of the models for a subset of those species with the largest number of records (n>19). This was carried out as follows. First, the data were split into two data sets using the median latitude as a splitting point. One of the subsets was used to build the model and one was used to test the model ability to predict the remaining data set. These predictions were carried for KSVM, Rpart and GAM models, using two different types of background points. One set of background points was collected from throughout the region and the other was collected within the MCP of the species distributions. This provides an overview of the potential discrimination properties of the models taking into consideration the extent of the prediction area of the models. These three models were used to validate the models as it was possible to use R to select a consistent set of background points to be considered as pseudo-abscences and to split the data by the median latitude. The same process was not carried out in MaxEnt as the program generates its own background points and the calculation of AUC is carried out by default, where the program randomly splits the data for model training and model validation. Furthermore, recent literature suggests that MaxEnt predicts very similar to GLMs (Hastie and Fithian 2013) and produces similar results to GAM (Golicher *et al.* 2012), especially when predicting species distribution in montane forests (Golicher *et al.* 2011). MaxEnt has also been identified as to inflate the measures of accuracy as a result especially when the data is not randomly or systematically collected as is the case in the studied species (Renner and Warton 2013).

Area predictions

The AUC values were used to produce binary maps that assist the calculation of the potential area of distribution for the species. This required selection of an appropriate AUC threshold. Models with values of AUC >0.9 are considered to be highly accurate, those providing values in the range 0.7–0.9 ‘useful’, and those lower than 0.7 ‘poorly accurate’ (Guisan *et al.* 2007). For this research the threshold was set at 0.9, in order select only the grid cells that have the highest probability of occurrence to prevent outliers that may have arisen as a result of coordinate imprecision unduly affecting the distribution maps. Using this threshold discounts specificity and is used to measure sensitivity (Golicher *et al.* 2011). In order to identify the areas where the species are likely to occur, a MCP was used to restrict the species distributions and to compare the results with the Red List carried out in **Chapter 4**. A Minimum Convex Polygon (MCP) is described in the IUCN RL Categories and Criteria as the smallest polygon in which no internal angle exceeds 180° and contains all sites of occurrence (IUCN 2001), which in this case are the georeferenced locations. MCPs were therefore plotted for each projected species distribution, under each climate change scenario, using R (R Development Core Team 2011).

Suitable forest area and protected areas

In order to identify the area that is suitable for each species, a classified global land cover map for 2009 (referred to henceforth as ‘GlobCover’) produced by Arino *et al.* (2010) was used to exclude non suitable habitats. These data were obtained from the MERIS imaging spectrometer, at a resolution of 300 m, which was reclassified to ~5 km. This was achieved by excluding the following land cover classes: Rainfed croplands, Mosaic cropland (50-70%)/vegetation (grassland/shrubland/forest) (20-50%), Mosaic vegetation (grassland/shrubland/forest) (50-70%)/cropland (20-50%), Artificial surfaces and associated areas (Urban areas >50%), Closed to open (>15%) herbaceous vegetation (grassland, savannas or lichens/mosses), Bare areas, Water bodies, Permanent snow and ice, for areas above 1500 m. a.s.l.

Furthermore, the outputs of the modelled suitable habitat for the species’ current distribution was mapped, for those species that were classified as under the threatened Category, i.e. VU, EN and CR, using the RL Criterion A3, and overlaid with the corresponding ecoregions, described by Olson *et al.* (2001), to identify those ecoregions with the largest number of threatened species. Also, the role of current protected areas on the potential species distribution under each projected climate change scenario was explored by filtering the predicted area by the protected areas map produced by IUCN and UNEP-WCMC (2012), for the countries in the region.

Red List assessment including and excluding future threats

In **Chapter 4** the Red List status for each taxon was assessed excluding potential future threats under climate change. For a comparative assessment of the potential impacts of future climate change, Criterion A of the IUCN Red List Categories and Criteria, Version 3.1 (IUCN 2001) was used, as this assesses the population reduction and geographic range parameters (**Table 4.1**). Criteria C, D and E were not applied, as specific required information for most species such as population size parameters and population viability analyses, were not available for the species.

Criterion A3 was applied by subtracting the EOO projected under the climate change scenarios from the current EOO, using model outputs. MCPs were used to calculate the EOO in both cases. In addition, for calculation of both current and projected EOO values, areas of unsuitable habitat were excluded, using the GlobCover map. According to IUCN guidelines, population reduction must be measured over the longer of either 10 years or three generations, up to a total of 100 years (IUCN Standards and Petitions Subcommittee 2011). However, generation length is not available for most of the species evaluated in this assessment, although most tree species have generation lengths spanning several decades. Climate change projections for the years 2080-2100 were used, resulting in a period of potential population reduction of at least 70 years measured from 2009. The IUCN Red List Categories and Criteria stipulate thresholds for using criterion A3, in terms of species population decline, where a decline of ≥ 30% has been established for a species to qualify as threatened in the Vulnerable (VU) Category, with corresponding values of ≥ 50% and ≥ 80% for Endangered (EN) and Critically Endangered (CR) respectively. These values were used to reassess the species as to identify potential changes in the classification carried out in **Chapter 4**.

Calculations of potential current distribution using predictive modelling were used to reassess the species under Criterion B and give a preliminary category, solely based on the specific area thresholds for the species current distribution, as this is the first parameter to be able to list the species under a threatened category (**Table 4.1**). These thresholds are: EOO of <20,000 km^2 for VU, EOO <5,000 km^2 for EN and <10 km^2 for CR. This therefore allows comparison of the preliminary category given to the species in **Chapter 4**, where the estimated EOO was calculated using the 'Globcover' map alone.

Assumptions

Certain assumptions need to be made explicit when carrying out species distribution modelling, so as to identify potential areas of uncertainty in the models. The main assumptions in the approach taken in this research are: 1) current climatic conditions are considered to be in an equilibrium state with current species distributions, 2) climatic projections are potential areas that may be suitable for a species to colonise and become established, but this does not actually take into considerations other biotic interactions that may limit potential migration, such as competition and dispersal barriers; 3) the land use map is a snapshot in time of the year 2009, which does not take into consideration the change in land use that may occur during the period forecast by the models.

5.3 Results

Model validation

The process of splitting the data into two by the median latitude and then using one set to fit the models and the other to test the data, with the five different sets of variables, enabled evaluation of model ability to predict species occurrences. The average AUC values for the three different models fitted for the 13 of the most abundant species, when tested with background data from the whole region, showed that the models had a high mean AUC value: KSMV: 0.83 (±0.04), Rpart: 0.84 (±0.02), GAM: 0.86 (±0.04). None of the models predicted an AUC lower than 0.5 (**Figure 5.1**). However, mean AUC values decreased considerably for many of the species when the background points were selected within the species' MCPs, namely: KSMV: 0.63 (±0.04), Rpart: 0.64 (±0.04), GAM: 0.66 (±0.04) (**Figure 5.2**). This result shows that when values were used from locations distant from where the data were collected, the models provided in most cases high AUC values, whereas when the models were fitted with data from within the MCPs the models ability to discriminate presence against background data was reduced, and for many species the AUC value was below 0.5. Overall, GAM produced the highest AUC value and therefore the results from this model are presented from now on. Also the model outputs for the set of variables that gave the highest AUC values using GAM with the MCP background data will be presented, namely variables 2, 3, 13, and 14 (**Table 5.1**).

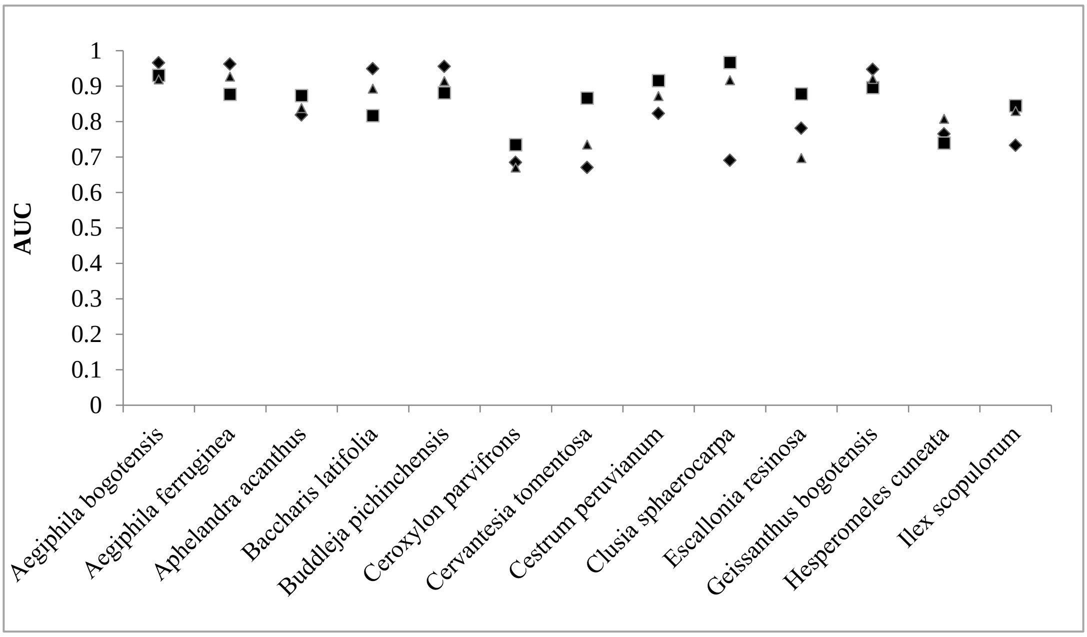


Figure 5.1 Mean AUC values for 13 of the most abundant species, from five combinations of variables, using different climatic variables for three models (♦=KSMV, ▲=Rpart, ■=GAM). Models used background data from the whole region.

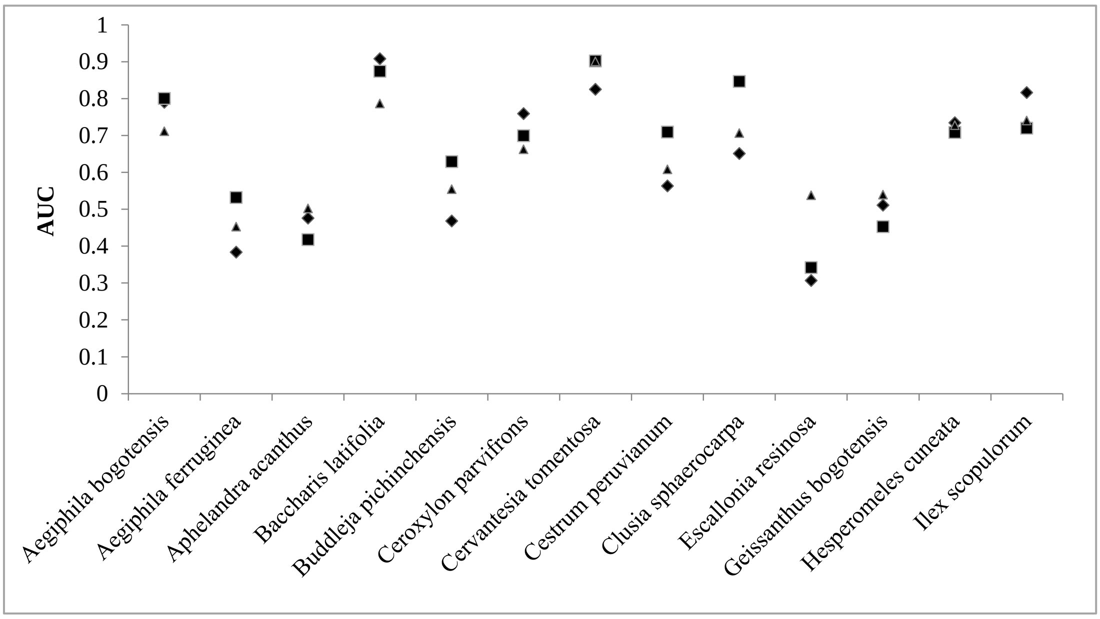


Figure 5.2 Mean AUC value for 13 of the most abundant species, from five combinations of variables, using different climatic variables for three models (♦=KSMV, ▲=Rpart, ■=GAM). Models used background data from within the MCP of each species.

Predictions of current distribution

Using a specific set of climatic variables to predict the species distribution does not encompass all the factors influencing their current distribution, and for some species these variables may not be as important restricting the species occurrence as those not considered in this assessment. Therefore, the potential area of distribution for many species evaluated, using the results for the GAM model, was widely distributed over the region, with a mean (± SE) area of 3,767,049 (±326,364) km^2, a minimum area predicted of 114,525 km^2 for C*alliandra taxifolia* and maximum area of 8,069,350 km^2 for 15 species, namely: *Axinaea grandifolia*, *Azara salicifolia*, *Cinchona pyrifolia*, *Clusia pseudomangle*, *Clusia sphaerocarpa*, *Cyathea catacampta*, *Cybianthus laetus*, *Gynoxys calyculisolvens*, *Gynoxys sancti-antonii*, *Oreopanax seemannianus*, *Schefflera inambarica*, *Solanum goniocaulon*, *Weinmannia auriculata*, *Zanthoxylum brisasanum* (**Figure 5.3**).

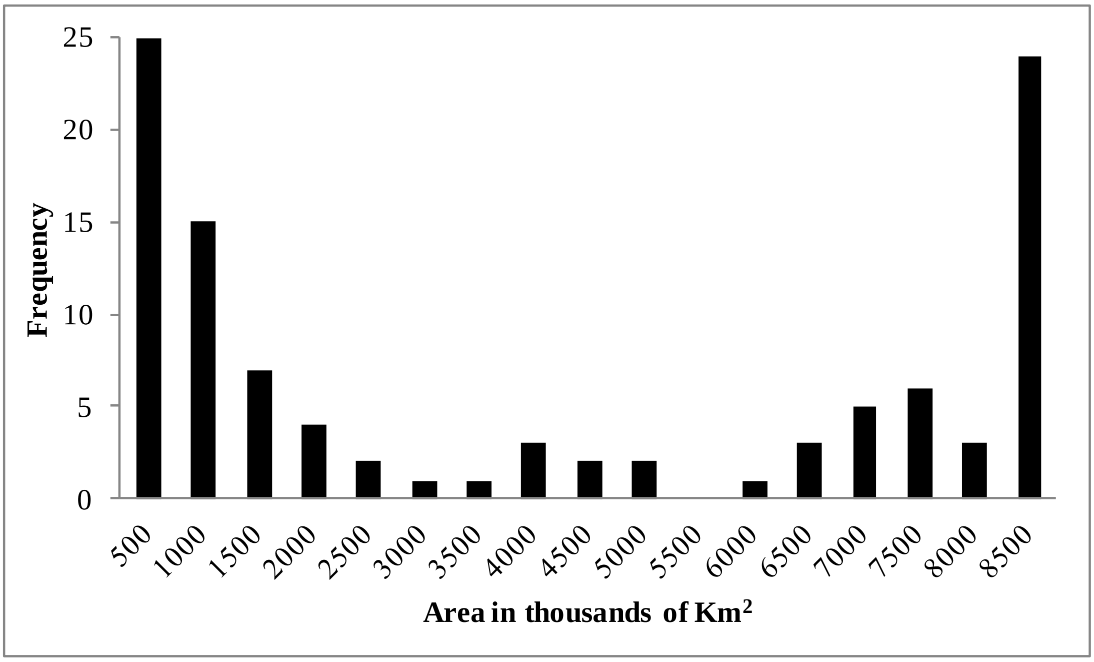


Figure 5.3 Frequency distribution of the full extent of the species' potential distribution from the model (GAM) predictions.

Restricting each of the species' distributions with the use of a suitable habitat map ('GlobCover') of areas above 1500 m a.s.l, reduced the species' distribution areas considerably, with a mean area of 451,234 (±20,710) km^2 with a minimum area of 53,825 km^2 for *Ilex rimbachii*; and a maximum area of 657,375 km^2 for 23 species, which were the same 14 species with the largest distribution with the full distribution area, with the addition of *Daphnopsis espinosae*, *Dendrophorbium balsapampae*, *Ilex scopulorum*, *Ilex uniflora*, *Miconia harlingii*, *Nectandra subbullata*, *Persea brevipes*, *Prunus urotaenia* and *Ribes canescens*.

The estimates of distributional range were further restricted by the MCP around current distribution data and the 'GlobCover' map to enable comparison between model outputs and current patterns of distribution. **Figure 5.4** shows that by using these two parameters to restrict the potential distribution for the species, the mean area predicted was 32,965 (±3,569) km^2. 68 species had an area below the mean and 47 species had an area <20,000 km^2, which is the threshold for the threatened categories under the B1 criteria, in relation to the Extent of Occurrence (EOO) (IUCN 2001). The minimum predicted area was for *Berberis jobii*, for which the model predicted the distribution outside the MCP and therefore the suitable habitat within the MCP was null. The maximum area predicted was for *Senna versicolor* with an area of 170,900 km^2.

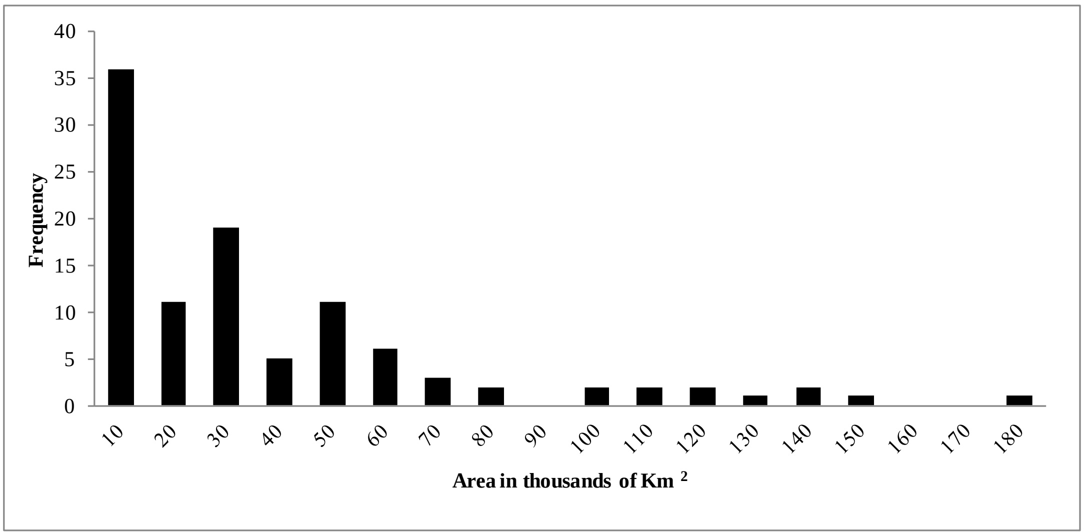


Figure 5.4 Frequency of species' potential distribution from the model (GAM) prediction, restricted to the MCP and 'GlobCover' map of suitable habitats.

Prediction of potential distribution under climate change scenarios

When EOO values were calculated without excluding unsuitable areas, the results for the (GAM) model showed that for 10 species, the predicted area under the A2 climate change scenario was larger than the predicted current distribution. These species were: *Citharexylum joergensenii*, *Clethra rugosa*, *Cyathea frigida*, *Geissanthus argutus*, *Ilex uniflora*, *Prunus pleiantha*, *Schinus pearcei*, *Schoepfia flexuosa*, *Senna versicolor* and *Smallanthus fruticosus*. For the other 94 species the predicted area was less than the current potential distribution. Under the B2 scenario, 12 species were predicted to have a larger potential distribution than predictions for their current distribution, namely: *Berberis lehmannii*, *Dendrophorbium balsapampae*, and the 10 species under A2 scenario. For the other 92 species, the distribution under climate change was less than the current potential distribution.

As expected, when EOO values were calculated by excluding unsuitable areas (i.e. using the 'GlobCover' map) and by clipping with the MCP around current distribution data, the projected species distributions under the climate change scenarios were considerably reduced compared to the predicted current distribution.

Using these lower EOO values, under the A2 scenario the projected distributional area decreased for 73 species and remained unchanged for 27 species. Under the B2 scenario, the projected area decreased for 75 species and remained unchanged for 25 species. The mean projected distributional area for the species under the A2 scenario was 26,198 (±3,193) km^2 and for the B2 scenario was 27,124 (± 3,208) km^2. The lowest values were recorded for *Alchornea anamariae*, for which projected area was reduced from 3,250 km^2 to 0 km^2, and *Berberis jobii*, for which values were 0 km^2 under both scenarios. The maximum projected area for both scenarios recorded was for *Senna versicolor* with areas of 167,600 km2 and 169,575 km2 for the A2 and B2 scenarios, respectively. The mean difference between the area of current potential distribution and the area projected to be suitable for each species under climate change (within the MCP) was 7,177 (±1,088) km^2 (28%) and 5841 (±947) km^2 (23%), for the A2 and B2 scenarios, respectively (**Figure 5.5**).

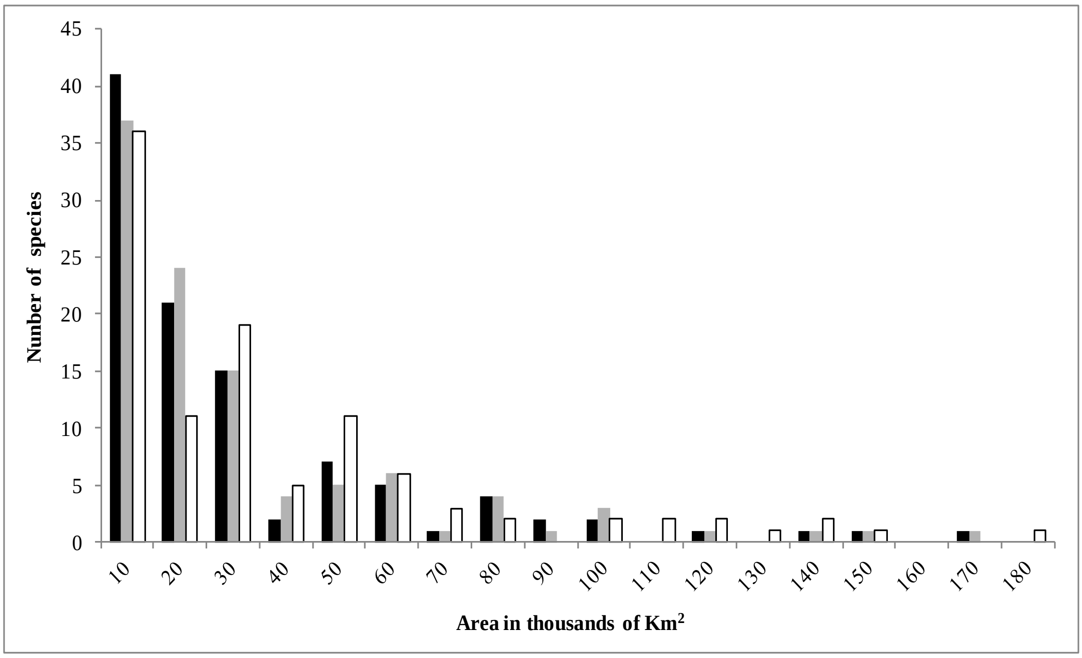
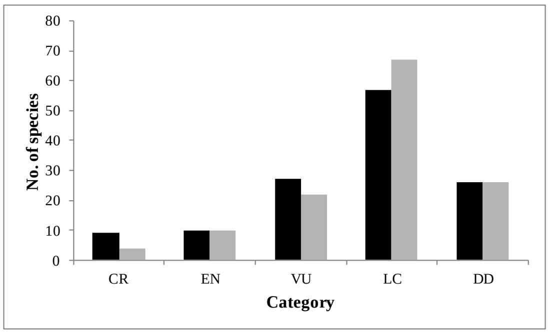


Figure 5.5 Frequency of species' EOO under two climate change scenarios A2 (black) and B2 (grey), and predicted current distribution (white), excluding unsuitable areas using the 'GlobCover' map and clipped by the MCP.

Model predictions and the Red List

Criterion A3 of the Red List was used to predict whether a population reduction was projected to occur for each species individually, as a result of climate change. When the projected distributional area of each species was used, resulting from the GAM, none of the species met any of the EOO thresholds associated with threatened Red List categories. In this case, all species were classified as LC. When currently unsuitable areas were subtracted from the EOO estimate using the "Globcover" map, all species again classified as LC. When estimates of EOO were further reduced by clipping with the MCP associated with current distributional range, 46 species qualified as threatened according to the A3 criterion, under the A2 climate change scenario. These included 9 in the CR Category with a ≥80% reduction of the suitable area, 10 EN with a ≥50% reduction of the suitable area and 27 species VU with a ≥30% reduction of the suitable area. Under the B2 scenario 36 species would preliminarily be in the threatened category, 4 CR with a ≥80% of reduction of the suitable area, 10 EN with a ≥50% reduction of the suitable area and 22 species VU with a ≥30% reduction of the suitable area (**Figure 5.6**).

a)



b)

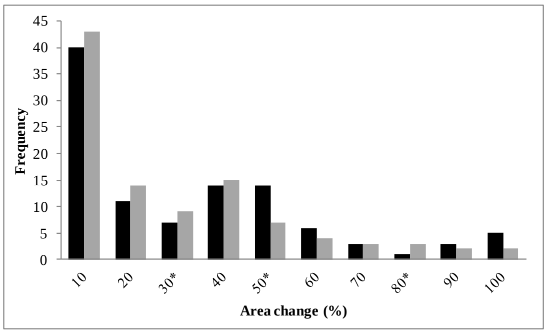


Figure 5.6 Preliminary category assigned to the species, according to the A3 criterion taking into consideration the percentage change from the modelled predicted area of distribution to the A2 (black) and B2 (grey) scenarios, a)

According to the IUCN RL thresholds: Critically Endangered (CR) with a ≥80% reduction of the suitable area, Endangered (EN) with a ≥50% reduction of the suitable area, Vulnerable (VU) with a ≥30% reduction of the suitable area, Least Concern (LC) and Data Deficient (DD). b) Frequency of species per class of percentage change in suitable area asterisks, are the RL thresholds from modelled current distribution.

Taking into consideration the information obtained for the species' current and potential distribution under climate change scenarios, it is possible to provide an overview of the possible changes to the species RL classification. **Table 5.2** illustrates that the preliminary category assigned to the species using the outputs from the modelled present distribution, with the use of B criterion thresholds and modelled potential distribution using the A3 criterion thresholds. The number of species that are threatened under the CR category increased and the EN and VU decreased. However, column (f) in **Table 5.2** shows the highest given category under any of the cases considered (a-e), highlights the fact that potentially 105 species could be considered under a threatened category.

Category	Final (a)	Present (b)	Model present (c)	A2 (d)	B2 (e)	Highest (f)
CR	1	0	1	9	4	8
EN	47	17	20	10	10	63
VU	28	31	26	25	22	34
NT	19	0	0	0	0	5
LC	29	72	57	58	68	16
DD	5	9	25	25	25	3

Table 5.2 Overview showing the species Categories according to the (a) Final RL given considering all the Categories and Criteria (**Chapter 4**); (b) Present distribution based on 'GlobCover' and species' MCP, taking into consideration the thresholds of Criterion B1; (c) Modelled potential present distribution using current climatic variables, 'GlobCover' and the species' MCP taking into consideration the thresholds of Criterion B1; (d) Modelled potential distribution using climatic variables from scenario A2, 'GlobCover' and the species' MCP, taking into consideration the thresholds of Criterion A3; (e) Modelled potential distribution

using climatic variables from scenario B2, ‘GlobCover’ and the species’ MCP, taking into consideration the thresholds of Criterion A3, and (f) Highest category given to the species in any case (a-e).

To further explore the potential changes in the species classification, **Table 5.3** shows the changes in the categories, from the Final RL given to all taxa, considering all the Categories and Criteria (**Chapter 4**) and results from the modelled potential current distribution using current climatic variables, ‘GlobCover’ and the species’ MCP. This shows that 19 species have been uplisted to a higher category of threat, 42 remained in the same category and the remainder were downlisted or were Data deficient (DD).

		Final RL (a)						
	Category	**CR**	**EN**	**VU**	**NT**	**LC**	**DD**	**Total**
Modelled present rating (b)	**CR**				1			1
	EN		9	7	2	1	1	20
	VU		10	9	3	4		26
	LC		15	11	11	20		57
	DD	1	13	1	2	4	4	25
	Total	1	47	28	19	29	5	129

Table 5.3 Matrix of categories given to the species according to all Categories and Criteria for (a) Final RL given considering all the Categories and Criteria (**Chapter 4**) and preliminary categories given to the species according to Criterion B1 thresholds for (b) Modelled potential current distribution using current climatic variables, ‘GlobCover’ and the species’ MCP using the outputs of GAM.

Furthermore, **Table 5.4** shows the matrix of potential changes in the species preliminary classification taking into consideration the Criterion B1 thresholds, using current species distributions based on ‘GlobCover’ and species’ MCP and modelled potential present distribution using current climatic variables, ‘GlobCover’ and the species’ MCP. This shows that, from the current distribution to the modelled distribution, 19 species increase to a higher threatened category, 93 species remained in the same category and 17 were Data Deficient.

Modelled present rating (b)	Present (a)					
	Category	**EN**	**VU**	**LC**	**DD**	**Total**
	CR	1				1
	EN	10	6	3	1	20
	VU		18	8		26
	LC			57		57
	DD	6	7	4	8	25
	Total	17	31	72	9	129

Table 5.4 Matrix of preliminary categories given to the species according to the RL Criterion B1 thresholds for both (a) Present distribution based on 'GlobCover' and species' MCP and (b) Modelled potential current distribution using current climatic variables, 'GlobCover' and the species' MCP.

Criterion A3 of the Red List was used to establish whether a population reduction was projected to occur for each species individually, as a result of climate change. These results illustrate that for the species evaluated with the A2 scenario, 26 species were uplisted in the category of threat, 5 of which moved from VU to CR, 5 from LC to VU, 2 from LC to EN and one from LC to CR, 34 remained in the same category and the remaining 69 species were downlisted from category or were DD (**Table 5.5**). For the B2 scenario 23 species were uplisted in the threat category, with 4 species moving from NT, VU and EN to CR and 16 from NT or LC into a threatened category, 30 species remained in the same category and the other 76 species were downlisted or were DD (**Table 5.6**).

Final RL (a)

	Category	CR	EN	VU	NT	LC	DD	Total
A2 scenario (b)	**CR**		1	5	2	1		9
	EN		4	2	2	2		10
	VU		8	8	6	5		27
	LC		21	12	7	17	1	58
	DD	1	13	1	2	4	4	25
	Total	1	47	28	19	29	5	129

Table 5.5 Matrix of categories given to the species according to all the Categories and Criteria for (a) Present distribution based on 'GlobCover' and species' MCP and using the Criterion A3 thresholds for (b) Modelled potential distribution using climatic variables from scenario A2, 'GlobCover' and the species' MCP.

Final RL

	Category	CR	EN	VU	NT	LC	DD	Total
B2 scenario (b)	**CR**		1	2	1			4
	EN		2	4	2	2		10
	VU		5	6	6	5		22
	LC		26	15	8	18	1	68
	DD	1	13	1	2	4	4	25
	Total	1	47	28	19	29	5	129

Table 5.6 Matrix of categories given to the species according to all the Categories and Criteria for (a) Present distribution based on 'GlobCover' and species' MCP and using the Criterion A3 thresholds for (b) Modelled potential distribution using climatic variables from scenario B2, 'GlobCover' and the species' MCP.

Areas of most threatened species

To establish areas which could potentially be targeted for future conservation actions, having a high density of threatened species, a calculation of the number of species that were classified under a threatened Category using Criterion A3, under climate change scenario A2 was made, by taking into consideration the species' present modelled distribution, based on 'GlobCover' and the species' MCP. The results showed that the areas with the largest number of threatened species are found in Ecuador, the south of Colombia and the north of Peru (**Figure 5.7**). Furthermore, the ecoregions that contain the largest number of endangered species, are 'Eastern Cordillera real montane forests' with a mean of 10 species, followed by 'Iquitos varzea' with a mean of 9 species, "Northern Andean paramo" with a mean of 8 species and 'Northwestern Andean montane forests' with a mean of 7 species occurs throughout this region (**Figure 5.8**).

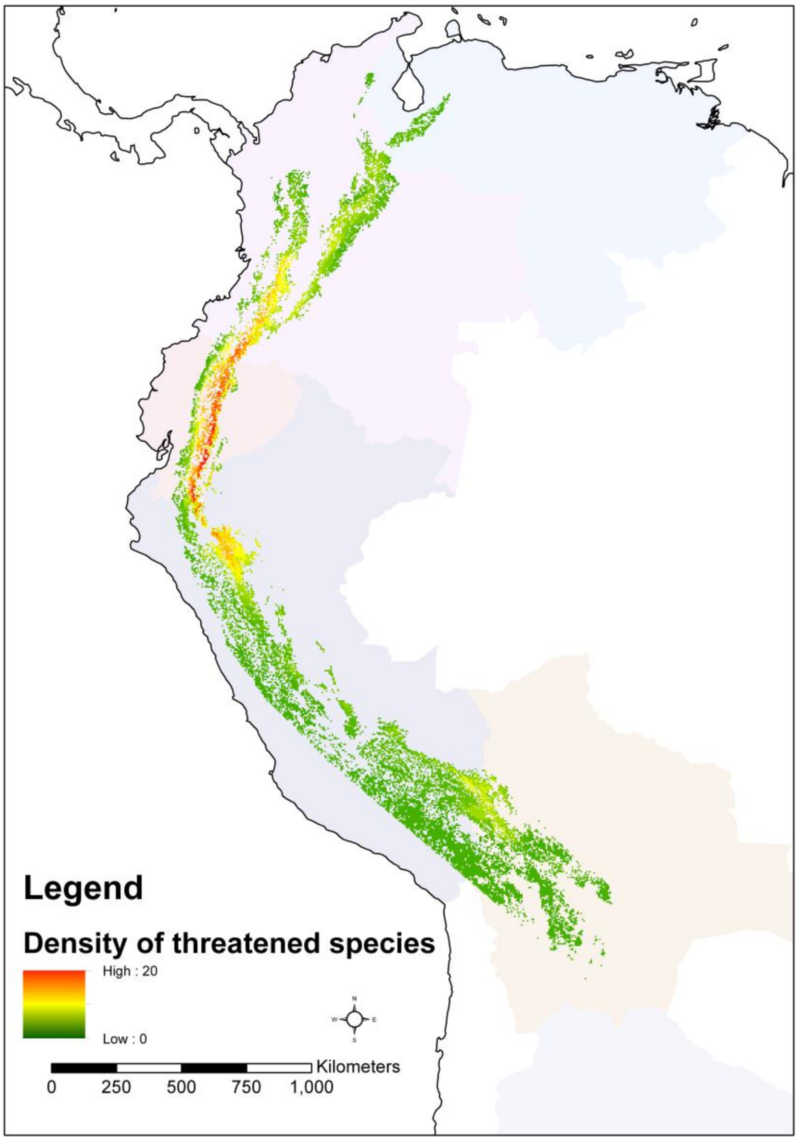


Figure 5.7 Density of threatened species in the study area, based on the present modelled distribution of the species and the category of threat according to the RL Criterion A3 under climate change scenario A2.

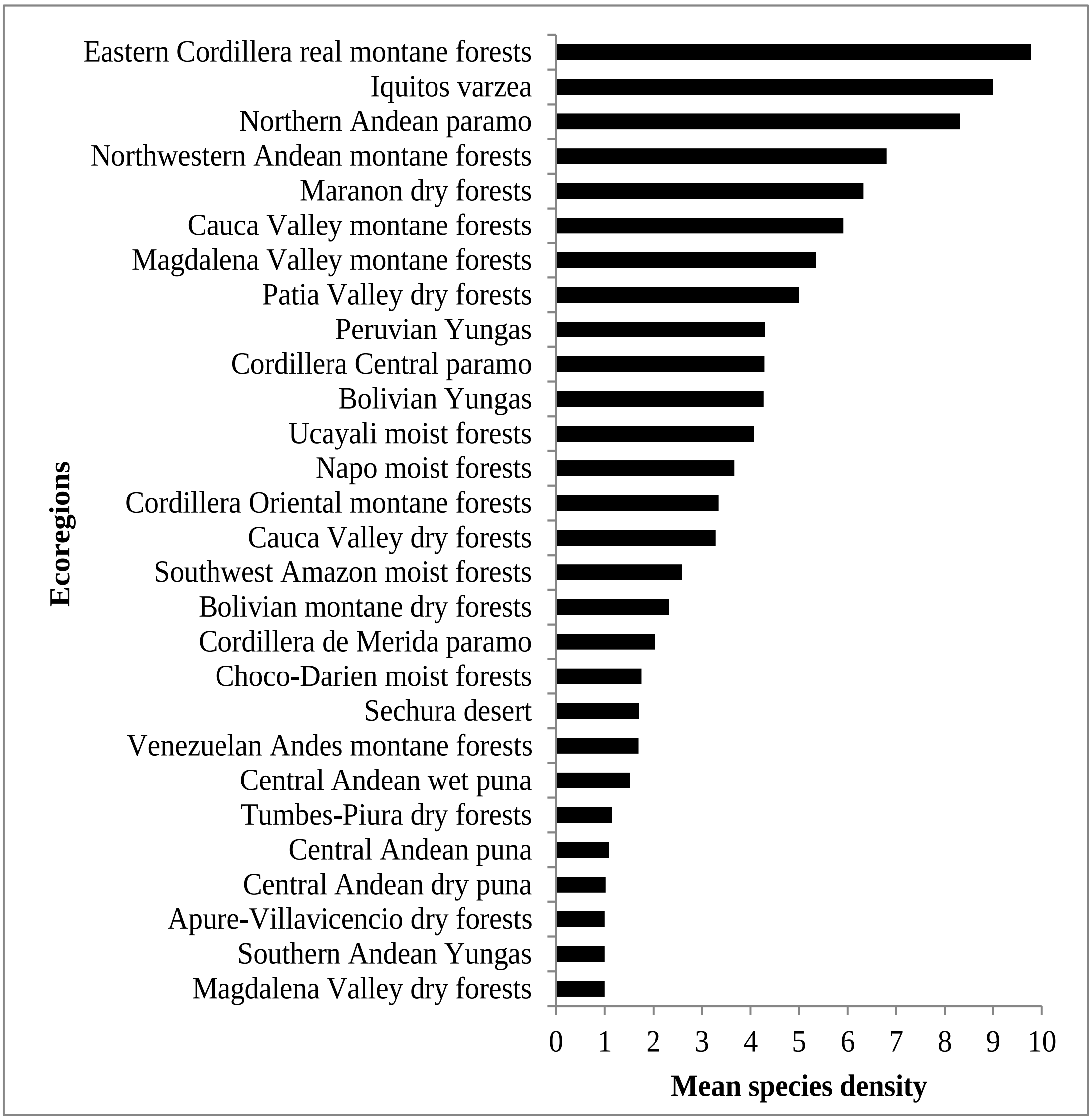


Figure 5.8 Ecoregions in the study area, described by Olson *et al*.(2001), and mean density of species classified in the threatened (VU, EN or CR) categories, according to the RL Criterion A3, based on the model outputs of the climate change scenario A2.

Protected Areas

Protected areas play an important role in the conservation of species, in particular in the Tropical Andes, where as described above, many species are already threatened can potentially be at even further risk as protected areas may not cover the species EOO. The total area with protection status in the study area was of ~110,625 km^2, taking into consideration the suitable areas of the 'GlobCover' map, (**Figure 5.9**).

For the species' present potential distribution, the percentage of area protected within the species EOO was on average 19.6 (± 0.91)%,with two species, *Berberis jobii* and *Cyathea catacampta*, not having protected areas within their predicted current EOO and *Citharexylum joergensenii* on the other hand, having the largest proportion of EOO protected with 50%.

The percentage of areas protected in the projected EOO under climate change A2 and B2 scenarios declined to an average of 18.6 (±0.97)% and 19.1 (±0.96)%, respectively. Under scenario A2 seven species had no protected areas within their EOO, namely: *Alchornea anamariae*, *Berberis jobii*, *Calliandra taxifolia*, *Cyathea catacampta*, *Palicourea candida*, *Prunus pleiantha* and *Symplocos canescens*; and under scenario B2: *Alchornea anamariae*, *Berberis jobii*, *Cyathea catacampta* and *Symplocos canescens.* For both scenarios the species with the maximum area protected was also *Citharexylum joergensenii* with 50% of the area being within protected areas (**Figure 5.10a**). For those species that were considered to be in a threatened category under Criterion A3, taking into consideration the projected distribution under climate change scenarios A2 and B2, the mean protected area was 18.3 (± 0.97)% and 18.7 (± 0.95)%, respectively (**Figure 5.10b**).

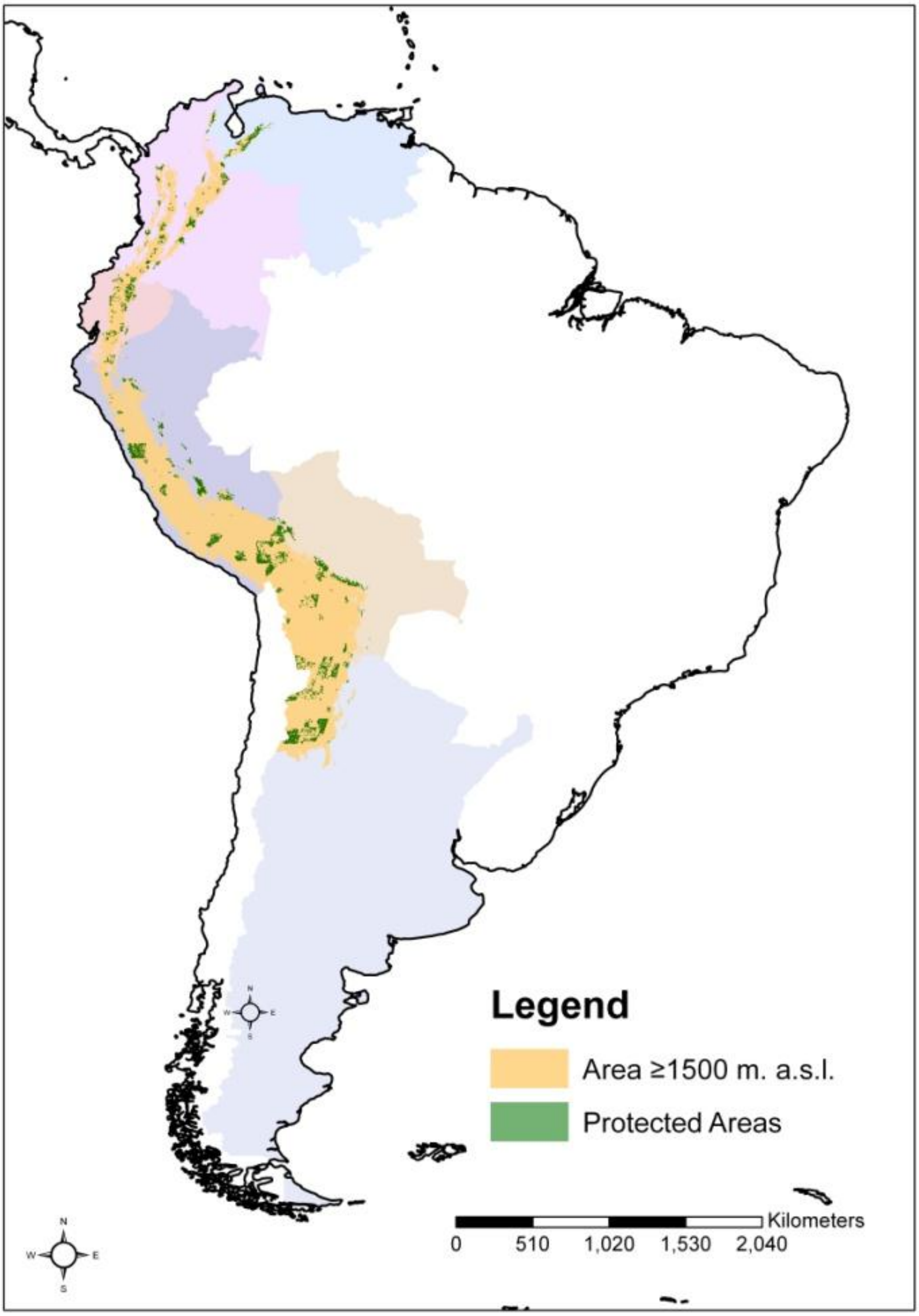


Figure 5.9 Protected areas (green) excluding unsuitable areas using the 'GlobCover' map in the study area.

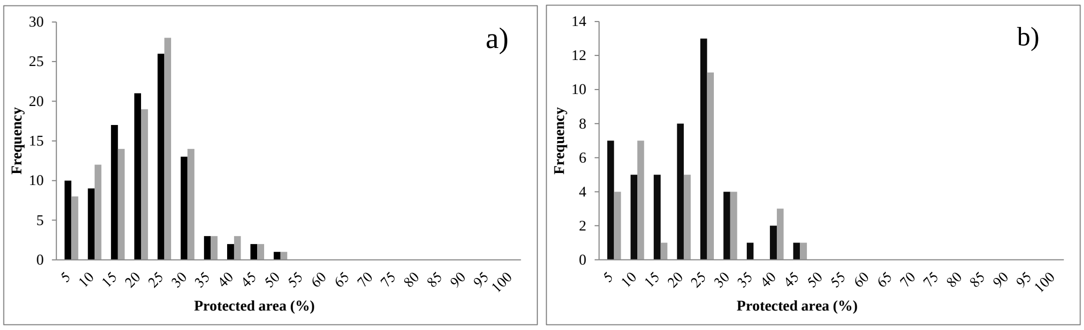


Figure 5.10 Percentage protected area available in the species' EOO. a) For all the species evaluated from the modelled predicted area of distribution under the A2 (black) and B2 (grey) scenarios and b) for those under a threatened Category, taking into consideration the IUCN RL criterion A3 for climate change scenarios A2 (black) and B2 (grey).

5.4 Discussion

Predictive modelling has become an important and cost-effective tool for regional biodiversity assessments, biodiversity management and conservation planning (Elith *et al.* 2006b), especially in poorly surveyed regions, such as the Andes, that are already threatened by other anthropogenic actions (Tejedor Garavito *et al.* 2012; **Chapter 3**). A large proportion of the current research into predictive modelling has been carried out using presence only data from herbaria and museums (Elith and Leathwick 2007) and these have provided an insight into the knowledge gap on the potential distribution and the consequent effects of climate change in tropical tree species. Models have been built with as few as 5 records, with some reliable results (Hernandez *et al.* 2006). However, knowledge of species distributions is poorly developed (Whittaker *et al.* 2005), and much uncertainty still remains about inference from model prediction using presence data only (Hastie and Fithian 2013).

In this evaluation, as for all research carried out in predictive modelling, a large number of caveats should be considered when interpreting the results. Limitations of predictive modelling using presence only data have been recognised in the literature e.g. (Feeley and Silman 2010; Golicher *et al.* 2012; Jiménez-Valverde *et al.* 2008; Lobo *et al.* 2010; Lobo *et al.* 2008; Synes and Osborne 2011; Thuiller *et al.* 2008). Here, the modelling was limited initially by the properties of the data that was available, and those species that are potentially rare with only a few records were not evaluated and were therefore classified as DD. The use of the area under the receiver operating characteristic curve (AUC) to evaluate the choice of variables and models has potential weaknesses, some of which are described by Golicher *et al.* (2012) and Lobo *et al.* (2008), including the fact that it weights omission and commission errors equally and that it does not give information about the spatial distribution of model errors. However, some of the effects of the over predictions in this assessment were reduced by comparing the results from predictions from collecting pseudo-absences from within the MCPs, which reduces the inflated AUC values from data from outside the species' climate envelope (Lobo *et al.* 2008). Furthermore, potential biases in the climatic variables have previously been tested by Loiselle *et al.* (2008) in two of the Andean countries, Bolivia and Ecuador, who concluded that significant parts of the climatic gradient were poorly represented in herbarium collections. However climatic bias in collections did not greatly affect distribution predictions for plant species and were used to estimate species potential distributions. Also, selecting thresholds to identify the realized distributions for each of the species in the output maps involved an arbitrary method that assists interpretability. This threshold was set at a 90% cut off. This could have left some of the species locations out, but reduced further errors in the interpretation.

The uncertainties surrounding modelling species distributions, particularly under climate change scenarios, relate in part to the choice of spatial scale to project the species distribution. The use of a coarse spatial scale hinders the precise forecast of the species potential distribution, as it may hide potential refuges for species and environmental heterogeneity that could increase species survival, especially in mountain areas where estimation of risks of extinctions could be overestimated (Thuiller *et al.* 2005). Also, the choice of scenario, which in this case was HADCM3 scenarios A2 and B2, were conservative, compared to more extreme scenarios available such as the A1FI. However, recent literature has demonstrated that these more extreme conditions may be unlikely to occur within the timeframe that these were projected by the IPCC (Otto *et al.* 2013), as local feedbacks may have contributed to the slower increase in temperatures than originally predicted

(Armour *et al.* 2012), some of which are related to the increased carbon storage capacity of the oceans and land in recent times (Ballantyne *et al.* 2012).

This assessment only evaluates the potential changes in the species distribution under projected climate change scenarios to assess the future population decline related to potential habitat loss. However, these results highlight the fact that as many as 105 species were preliminarily identified as threatened in at least one of the different cases evaluated in this assessment (**Figure 5.2**). It also gives an indication of the extent of protected areas that falls within the species EOO, and identified that for all of the species evaluated here, this area is ≤50%. It is recognised that modelling technique used here is limited, as this could have been improved by including species migration, population dynamics, biotic interactions and community ecology into the models (Guisan and Thuiller 2005), as these could provide a better understanding and further inform the RL assessment (Akçakaya *et al.* 2006). However, these results are in line with the studies carried out in other regions, such as in Africa (Bomhard *et al.* 2005) where the authors, by using climate change scenarios and the IUCN RL assessment, identified that climate change will greatly contribute to the change of species classification, uplisting up to a third of the 227 Proteaceae taxa assessed in a threatened category.

Evidence that tree species in the Andes are migrating to higher altitudes at a rate of +1.1 m elevation year^{-1} has been recently identified (Feeley *et al.* 2011). This highlights the fact that species are likely to be lost under climate change if they are either slow to migrate or associated with previously stable climates (Fjeldså *et al.* 1999; Golicher *et al.* 2011). The climatic stability of forests and forest patches at high altitude in the Andes, which has been maintained through time, even in climatically stressful events in the past, including the most recent Little Ice Age, has been identified of fundamental value for the survival of species as biological refugia (Fjeldså *et al.* 1999; Hole *et al.* 2011b). Recent studies have concluded that due to climate change the upper boundaries of almost all biomes in the tropical Andes are likely to show an upslope displacement (Tovar *et al.* 2013). This potentially could create novel climates (Williams *et al.* 2007), where the species may be able to migrate to and thrive in. However, as many of the species evaluated here have distribution ranges that are already restricted to the highest altitudes, these could readily face extinction as they may not have places to migrate to. Other potential impacts of climate change on forest composition, which have been highlighted by a recent study in temperate areas (Meier *et al.* 2012), were that pioneer or early successional species have been found to respond rapidly to new climates, whereas late successional species respond much slower to these changes, increasing further the extinction risk of these species and encouraging species invasion.

The results of this research indicate that comparing the results of this research with those obtained in the RL assessment in **Chapter 4**, climate change increased the risk of extinction of 18-20% tree species evaluated, depending on the climate scenario. While these results suggest that climate change represents a significant threat to tree species in the tropical Andes, they contradict suggestions that climate change will become the most important cause of biodiversity loss in coming decades (Bellard *et al.* 2012; Dawson *et al.* 2011; Thomas *et al.* 2004). These results are partly attributable to the fact that upper montane species in the Andean region are already being subjected to a number of current threats. Pre-eminent among these is forest loss and degradation, caused by conversion of forest to agricultural land use, and over-exploitation of tree species for products such as timber and fuelwood. Estimates of deforestation rate for the period 2005-2010 based on national statistics vary from 0.17-1.89% per annum, depending on the country (FAO 2010). Deforestation rate estimates based on analyses of satellite remote sensing imagery provide a regional mean value of forest loss of 0.62% per annum in recent decades, with values ranging from 0.32-1.08% for individual countries (**Table 3.4**, **Chapter 3**). With respect to exploitation, species such as *Polylepis* spp. have been intensively harvested in the Andes over the past century (Gareca *et al.* 2010; Jameson and Ramsay 2007), and many species of this genus are now restricted to small forest fragments. Tree species of high commercial value such as *Cinchona* spp., *Podocarpus* spp., *Zanthoxylum* spp. and *Ilex* spp. have also been subjected to overexploitation at different times in the past, which is likely to have reduced population sizes (**Chapter 3**; Tejedor Garavito *et al.* 2012). Other threats affecting tree species in the region include the impacts of fire, browsing animals, urban expansion, infrastructural development and mining (**Chapter 3**;

Tejedor Garavito *et al.* 2012). Climate change therefore represents an additional potential threat to species that are currently being subjected to multiple additional threats. Potentially, climate change could interact with other threats such as spread of pests and diseases, intensity of land use or land cover change, and the frequency and intensity of fires. Such potential interactions are poorly understood (Staudt *et al.* 2013), and could potentially be very significant, but were not explicitly considered here.

Any insight into the potential impacts of climate change on tree species' distribution is an important element to be included in conservation planning. Furthermore, the main outcome of this investigation can be viewed as a preliminary selection of those species that are potentially more at risk in terms of the potential range shifts under climate change scenarios. This highlights priorities for future work, including the need for more detailed information on the life-history traits for threatened species to integrate demographic modelling approaches, as suggested by recent research by Fordham *et al.* (2013). These authors suggest that using more detailed information is key for conservation prioritisation and intervention. But rather than focusing primarily on climate change, it may therefore be more relevant to consider the relationship between current threats and species extinction risk biodiversity loss and is increasing, as pointed out by Maslin and Austin (2012). While it's possible that the criteria for selecting the species for this evaluation had an impact on the number of threatened species and their geographic distribution, **Figures 5.7** and **5.8** show that the areas with the highest densities of endangered species coincide with areas and ecoregions that have been previously identified as conservation priorities (Dinerstein *et al.* 1995), with a critical conservation status (e.g. 'Cauca valley montane forest' and 'Magdalena valley montane forest'), a threatened status ('Northwestern Andes montane forest'), or Vulnerable ('Cordillera Real Oriental montane forest'), as detailed in **Table 3.7**.

Identifying threatened species not only focuses on prioritising conservation of these species; the aim is to enhance the overall conservation status of the forest system where they are found and increase forest resilience to potential climate change impacts. Species that may not be as sensitive to climate change could still be important in terms of their contribution to ecological structure, composition and function (Crossman *et al.* 2012). Also, the approach to identify the current area of species' current EOO that is already protected indicated that for most species this is ≤50%. Therefore this should encourage a regional-scale overview for where habitat re-creation, corridors and ecosystem resilience can be established to increase the percentage areas of protection for at least the most threatened species.

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6 Discussion and conclusion

6.1 Summary of research findings

There is mounting evidence that climate change is largely attributed to anthropogenic activities (IPCC 2013; Parmesan *et al.* 2013). Changes in climate are predicted to impact biodiversity and increase the extinction risk for many taxa (Thomas *et al.* 2004). Many tree species are known to have limited adaptive capacity to respond to rapid environmental change (Lindner *et al.* 2010). However, the lack of data from forest systems has limited the integration of specific species into forest conservation planning (Whittaker *et al.* 2005). The management of tropical montane forest ecosystems is an example of where bridging this knowledge gap would be beneficial, as these are high on the list of the world's most threatened ecosystems (Bruijnzeel *et al.* 2010; Hamilton 1995; Kapos *et al.* 2000; Ledo *et al.* 2009; Stadtmüller 1986).

The IUCN Red List assessment (RL) is considered an authoritative approach for assessing the extinction risk of species (Lamoreux *et al.* 2003; Newton and Oldfield 2008; Rodrigues *et al.* 2006; Zamin *et al.* 2010). It provides a cohesive structure and serves multiple purposes: 1) To inform conservation policies and influence legislation, 2) The identification of priority areas for conservation, 3) To guide conservation investment and 4) To encourage species-based conservation and management, biodiversity evaluation and monitoring (Hoffmann *et al.* 2008). To date, progress has identified the threat category for many tree species (Newton and Oldfield 2008; Oldfield *et al.* 1998); however, many tree species still await assessment. Progress has been limited by a number of factors, including the lack of appropriate data to assess the status and distribution of many species. This is especially apparent in regionally endemic taxa, which are distributed in more than one country and their evaluation is carried out at the national level, not including their full distribution.

This thesis investigates the potential effects of climate change on montane forests and in particular how climate change can increase the extinction risk of tree species in the tropical Andes. In this thesis, I 1) Identified that projected changes in climate, where temperature was considered to be a particularly key parameter that could potentially lead to changes in habitat conditions of vulnerable montane forests 2) Evaluated the extinction risk of 129 endemic tree species in the tropical Andes, using the RL Categories and Criteria, with 76 species classified within a threatened category, with the level of uncertainty associated with the classification process. 3) Evaluated the potential impact of climate change on the species distribution under different climate change scenarios and reassessed the species using the RL, and established the changes in RL Category. Under a more severe climate change scenario (A2), it was estimated that 5 species moved from VU to CR, 5 from LC to VU, 2 from LC to EN and one from LC to CR. I also identified areas and ecoregions where the most threatened species are currently distributed.

6.2 Novel contributions to species extinction risk assessments

Vulnerability of montane forest areas to potential climate change

Understanding the level of vulnerability of specific ecosystems will lead to a better assessment and improve conservation efforts to halt biodiversity loss. Climate change will have major implications for the future of current ecosystems, in particular their structure, function and composition (Gottfried *et al.* 2012; Parmesan and Yohe 2003; Sommer *et al.* 2010; Wu *et al.* 2010). **Chapter 2** tested the hypothesis that montane forests, at the global scale, vary in their vulnerability to projected climate change scenarios defined by the IPCC and showed that this is true with a particular sensitivity to changes in temperature. The results indicate that all montane forests are likely to be highly vulnerable to changes in temperature under scenario A2, particularly montane forest ecorergions in Africa and Asia. Precipitation will affect ecoregions to a lesser extent; South America and Asia will experience the largest changes in precipitation. Increasing global temperatures and changes in precipitation patterns will have a detrimental impact on the water balance of these forests (Bruijnzeel *et al.* 2010), raising the average altitude at the base of the orographic cloud bank (Pounds *et al.* 1999), affecting montane forest's ecosystem integrity and their water availability. Although potential interactions between precipitation and temperature were not evaluated in this investigation, this could lead to further changes within the water balance of montane forests (Fung *et* *al.* 2011; Pounds *et al.* 2006). Sangarun *et al.* (2007) found that increased air temperatures decrease the percent relative humidity of forests in Thailand's montane cloud forests. Prolonged high temperatures may expose species to increased water stress, leading to increased tree mortality (Allen *et al.* 2010) and the reduction of population sizes, threatening those species with few and small populations (Abbott and Le Maitre 2010). A strong relation between the canopy density and site-specific water availability has been established; water deficit can restrict seedling establishment and therefore lead to changes in forest ecosystem structure and functioning (von Arx *et al.* 2013).

The ecoregion classification used in this assessment, as described by Olson and Dinerstein (1998), has been widely used to conduct global and regional conservation assessments. In addition, the corresponding 'Global 200' priority ecoregions presented by Olson and Dinerstein (2002), assisted the identification of those ecoregions that are already of international importance due to their species richness, endemism and current threats, for which climate change can potentially pose a further threat. Of the 211 ecoregions considered in this study, 141 of them are considered a global conservation priority ("Global 200"). Many of these ecoregions scored high in the ranks for vulnerability to climate change, being associated with the some of the largest M values (i.e. >9) under projections of scenario A2 for temperature. For example, the Rwenzori-Virunga montane moorlands and Albertine Rift montane forests are also part of the Eastern Afromontane biodiversity 'hotspot'(Myers *et al.* 2000).

Lack of understanding of all the processes that are involved in natural systems and the feedback processes of climate change, such as the effects that clouds have on temperature, the speed that forests die back (Allen *et al.* 2010), the amount of CO_2 that is absorbed by oceans and the speed that methane in the poles will be released as temperatures rise, give rise to large areas of uncertainty in the potential impacts of climate change worldwide (Maslin and Austin 2012). However by applying a precautionary principle, it is necessary to continue the identification of the potential effects of climate change in areas of high vulnerability with acceptance of uncertainty, as uncertainty should not disguise the risk that extreme changes can have on ecosystems (Thomas *et al.* 2004).

Conservation status of montane forest in the tropical

Montane forest in the tropical Andes are a major conservation priority, both as 'hotspots' of global biodiversity (Myers *et al.* 2000) and as priority ecoregions (Olson and Dinerstein 1997), owing to their biological richness, high level of endemism (Bush *et al.* 2007; Olson and Dinerstein 1997) and for having the largest concentration of species with restricted distribution in South America (Dinerstein *et al.* 1995; Jørgensen 2011; Jørgensen *et al.* 2006; Latta *et al.* 2011). They are considered fragile ecosystems, playing an important hydrological and ecological role and despite this, they are the least known ecosystems in the tropics (Ataroff and Rada 2009; Bubb *et al.* 2004; Gentry 1995; Kessler 2000; Stadtmüller 1986).

Many threats have been identified to have been contributing to the loss and degradation of these forests (**Chapter 3**). A panel of experts from throughout the region, identified that livestock, deforestation for land use change to agriculture, logging and fragmentation are widely considered to be the major threats to these forests (Tejedor Garavito *et al.* 2012), while recent research (Feeley *et al.* 2011; Herzog *et al.* 2011; Román-Cuesta *et al.* 2011; Tovar *et al.* 2013; Urrutia and Vuille 2009; **Table 3.6**) also identified that climate change could potentially have considerable impacts on these montane forests. The identification of threats and threatened species should focus conservation planning and lead to a more efficient way to mitigate biodiversity loss. This can be done by reinforcing the role of existing protected areas, but also promoting the creation of protected areas, forest restoration and sustainable forest management (Tejedor Garavito *et al.* 2012). Also, this could assist land management policies aimed at the conservation of biodiversity in productive rural landscapes, where there is an integration of conservation, productive systems (agriculture and livestock) and the human population; where landowners are encouraged to set aside part of their land for conservation or are encouraged to adopt agroecological methods (Perfecto and Vandermeer 2008, 2010; Perfecto and Vandermeer 2012; Perfecto *et al.* 2010).

Actions are required to encourage countries to reduce biodiversity loss, forest degradation and protect the ecosystem services they provide. International policies have been agreed by Andean countries such as the Convention on Biological Diversity (CBD), the Framework Convention on Climate Change (UNFCCC), Convention on International Trade of Endangered Species (CITES) and the Global Strategy for Plant Conservation (GSPC). These initiatives encourage ecosystem resilience, the increase of protected areas and the conservation of endangered species. These political initiatives have been accommodated to be implemented at the national level in many countries, but individual countries have also created their own policies to encompass the protection and restoration of forests (**Chapter 3**). Although different institutions have invested heavily in developing, strengthening and protecting key biodiversity areas in the tropical Andes, conservation actions in the region still require the inclusion of threated tree species.

Extinction risk of tree species that occur in montane forests in the tropical Andes

The evaluation of the current conservation statutes of montane forest in the tropical Andes (Chapter 3; Tejedor Garavito *et al.* 2012), identified that montane forests are currently under threats from different sources. Although mechanisms are in place to mitigate to some extent the rate of biodiversity loss, the use of the RL Categories and Criteria can provide further valuable information on the decisions for conservation actions that include tree species that are in the endangered category. A RL assessment can be done with the use of information of the species distribution, population and threats (IUCN 2001). However, as information is limited for many of the species occurring in the tropical Andes, the use of a network of regional experts on montane tree species can particularly assist the identification and validation of species data, playing a fundamental role in the RL process (Newton and Oldfield 2008).

In **Chapter 4**, 129 species were assessed using the RL Categories and Criteria. These species were chosen for being endemic to the region, as the georeferenced data available identified them as having a distribution in a least two countries and that the records were unique to areas ≥1500 m. a.s.l. 76 species were classified within a threatened category: 1 species was Critically Endangered (CR), 47 Endangered (EN) and 28 Vulnerable (VU). The rest were: 19 Near Threatened (NT), 29 Least Concern (LC) and 5 Data Deficient (DD). By incorporating results from previous RL assessments of other endemic species to a particular country to this assessment, it is possible to identify that the number of species endangered in the region reaches 241 species out of 328 evaluated as a whole. This means that 52% of the evaluated tree species are threatened and a further 285 species are DD or are still Not Evaluated (NE). Compared to other assessments of tree species in other locations and of other taxa such as mammals, birds and fish in the region (**Table 4.9**), these results show that the number of threatened tree species in the Andes is relatively higher and provides further evidence of the congruence that occurs in this region of species richness, endemism and threat (**Chapter 4**). Forests are habitat to at least half of the global biodiversity (Millenium Ecosystem Assesment 2005) for which, in the long term, their protection will increase their resilience and have a positive impact on dependant taxa. Tree species should take priority as much as other more charismatic species, as the consequences of current and potential threats will increase the strain on key elements of species dependant on forests for their survival (Biringer 2003).

Potential impact of climate change on the extinction risk of selected tree species

Climate change is becoming a focal issue for conservation planning and policy making as species are predicted to go extinct under climate change scenarios (Araújo *et al.* 2005; Butchart *et al.* 2010; Feeley *et al.* 2011; Keith *et al.* 2008; Midgley *et al.* 2002; Schwartz *et al.* 2006; Thomas *et al.* 2004), and this applies to species in cloud forests too (Ponce-Reyes *et al.* 2013). However, only a few studies (Bomhard *et al.* 2005; Thuiller *et al.* 2005) have assessed species extinction risk under projected climate change using the Red List assessement and none of these have been carried out in the tropical Andes.

Modelling species distributions can help drive the direction of future research by identifying species and areas for which to target conservation action, although there are limitations and caveats identified (Golicher *et al.* 2012; Lobo *et al.* 2010; Synes and Osborne 2011; Thuiller *et al.* 2008). At least 18 species have been identified to have a potential change in the RL classification from NT and LC to a category of threat under a more severe climate change scenario (A2). Also, the majority of threatened species occurred in ecoregions that are already priorities for conservation owing to their high endemism, species richness and current threats: Cauca valley montane forest, Magdalena valley montane forest, Northwestern Andes montane forest and Cordillera Real Oriental montane forest (Dinerstein *et al.* 1995).

Uncertainty and their impact on the assessment of RL status

As uncertainty exists within the RL process, in this assessment it was possible to identify the different levels of uncertainty under each of the Criteria applied. The level of uncertainty in this assessment was related to the data availability and its validation, the latter being a key instrument in this assessment; it was possible to use the panel of experts to assess the data and outputs for all the species assessed and reduce the uncertainty for the preliminary values given to the Extent of Occurrence (EOO) and Area of Occupancy (AOO), as well as to identify the current threats to individual species. Limitations on species distribution modelling have been identified extensively; however their usefulness has an overriding value in terms of their contribution to the RL to identify species extinction risk. However, by exploring the level of uncertainty in the RL process and by using the precautionary principle (IUCN Standards and Petitions Subcommittee 2011), this can assist in clarifying the areas where further research can be focused.

6.3 Critical evaluation of current research

As with most of the research carried on climate change or the RL assessment, this research needs to be used with caution. Here is an outline of the methods used and in the assumptions made throughout this thesis.

Climate change models

Models cannot capture all the factors affecting natural systems, their weight in the model and their implications. For example, at a regional scale, it is difficult to predict and conversely model known important parameters like precipitation due to their highly variable nature (Maslin and Austin 2012). Two climate change models were used in this thesis: HADCM3 for scenario A2 and HADCM3 for scenario B2, from the Hadley Centre Coupled Model for two of the Special Report on Emission Scenarios (SRES). However, these two models give a view of the potential implications for many montane forest areas under climate change scenarios, and choosing more extreme climate conditions would lead to a more drastic future position for montane forest ecosystems. The use of the Manhattan Distance (M) with a threshold of 2SD (standard deviation) from the mean, as a measure for extreme climatic conditions, needs to be taken with caution as, although it corresponds of significant ecological impacts, the way particular ecoregions and forests systems respond to climate changer varies (Beaumont *et al.* 2011; Palmer and Raisanen 2002).

Some authors have attempted to address, with varying success, species extinction risk based on the RL assessment. Thomas *et al.* (2004) used an adapted version of the RL assessment and two climate change scenarios for projection to 50 years and 100 years, with increase in temperature of 1.8-2 °C and >2°C, and of CO_2 levels of 500-550 p.p.m.v and >550 p.p.m.v respectively. The two models were used to assess the extinction risk of a range of species with a wide range of generation lengths. Their method incorporated the species-area relationship to identify the species potential change in their Area of Occupancy (AOO), which falls under the RL Category B2, for which the thresholds are set lower that for EOO (see **Table 4.1**). Their conclusion was that 15–37% of species sampled in regions and taxa were going to be 'committed to extinction'.

Bomhard *et al.*(2005) carried out an RL assessment of 227 Proteaceae taxa endemic to South Africa and used liner relationships for abundance and area, with projected climate change and land use scenarios for time periods ranging from 20 to 80 years. Their conclusion was that under the more severe scenarios, up to 7% of the taxa could move into a Critically Endangered (CR) category and 2% could potentially become Extinct. Shoo *et al.* (2005) looked at the altitudinal effect of climate change on population size in birds in Australia, identifying upland birds as the most sensitive, and in some severe climate change scenarios all the species studied would be threatened, based on RL Criterion A thresholds. In addition, Thuiller *et al.* (2005) evaluated the climate change threat to European plants, relating the large proportion of species loss to variations on temperature and moisture conditions, based on four climate change scenarios for the year 2080, namely: HadCM3 A1, A2, B1 and B2. They found that in the scenario with non-migrating species, montane species are those most at risk, with up to 60% of the studied species threatened. From these examples, it is possible to appreciate that most of the RL list assessments so far have been based on the relationship of area and species abundance, to assess the reduction of population size required by the RL assessment. These examples have been criticised by Akçakaya *et al.* (2006) for their lack of accuracy in the use of species generation lengths and for using incorrect timeframes that arbitrarily encompass a range of species that have a varied generation length, and for over/underestimating the real threat to the species by not taking into consideration further aspects that have an implication of the species extinction rates.

Analysis of climate change using modelling techniques at a regional scale poses a range of limitations for their use and interpretability for on the ground conservation actions; most are related to the choice of model, variables, scale and scenarios to predict the potential areas that would be suitable for the species. The methodology used in this thesis has identified and minimised (where possible) the impacts of caveats and limitations of the modelling process. Further improvements would include strengthening the data for the modelling and including additional model scenarios and assemblages of variables, incorporating further data variability and providing ground validation for the predictions of present distributions, and analysing data that has been systematically collected, as most of the data that is available has elements of bias - primarily owing to the way in which data has been collected, mostly by road-sides and where species are already known to exist (Feeley 2012; Feeley and Silman 2011; Feeley *et al.* 2011). If current, available data does not provide an accurate indication of the full distribution of a species, its use to drive modelling data and hence predictions should be addressed.

Red List assessment

The usefulness of the RL assessment to assess species extinction risk, to inform policy makers and to assist the targeting of conservation actions has been widely recognised (Mace *et al.* 2008; Miller *et al.* 2006; Possingham *et al.* 2002). It is used to assist conservation decisions at different levels, particularly at national and regional levels (Miller *et al.* 2007). Despite its usefulness, there are areas of uncertainty in the application of the RL assessment, which must be addressed and minimized where possible, as has been done in this research (**Table 4.7**). Most of the uncertainty relates to the lack of accurate information on the species status and distribution. Calculations of species EOO and AOO had a fundamental role in this assessment; the guidelines suggest that these sites should encompass all the known species locations (IUCN 2001). In this assessment these measurements were based on available georeferenced data and the use of a recent land use map for the EOO, with two resolutions for the calculations of area, $4km^2$ and $100km^2$, for the estimation of AOO. Therefore under or over estimations for some of the species could have occurred. However, by discussing these preliminary estimations with the panel of experts and consulting literature, a final decision based on current knowledge of the species, provided a means to validate the data. Species analysed in this thesis are regionally distributed (i.e. in more than one country) and therefore their EOO would not place most of them into a threatened category. Up to 72 species were classified using the values for AOO, but the use of the EOO was more valuable to identify the level of threat to these species in the full extent of their distribution. The importance of having these two values calculated has also been useful for the RL assessment of other taxa considered by Rivers *et al.* (2011), where the authors investigated the effect of the number of records needed to evaluate a species using the RL assessment.

Another area of uncertainty is the estimation of population reduction required for Criterion A3, as it uses climate change models to estimate the loss in species due to reductions in habitat availability. Although this method has been criticised in the literature (Akçakaya *et al.* 2006), it provides an outlook of the potential species range shifts and potential uplisting of species to a higher category of threat. The use of predictive modeling demonstrates that, although climate change could be a major factor that will increase species extinction risk, many species are already highly threatened by deforestation, forest degradation and overexploitation processes that are already occurring in many montane forests in the tropical Andes (**Chapter 3**). Conservation actions must prioritise the minimisation of current threats and then mitigate the plausible consequences that climate change could have on species.

The data available for this assessment was collected from different sources, including online databases, herbarium vouchers and personal data sets, making up more than 1600 records for the 129 species evaluated. Despite the extensive data collection, there was still insufficient data to carry out the assessment with high accuracy, or at all, for some of the species (**Figure 4.9**). The lack of data for some species may be due to species rarity, collection efforts and data access processes that vary between areas and countries in the region. Access to data differs between countries with Ecuador having large amounts of data online (Feeley and Silman 2011). Species collections tend to be selective and not systematic (Feeley 2012). Some of the areas are under-collected, being remote areas or due to political conflicts (e.g. southern Colombia) or used to grow illicit crops - a threat to the forests in the region (Dávalos *et al.* 2011). Data limitation was reduced by data validation by the panel of experts. However, validation of the AOO and EOO of those species that had relatively few distribution records could be strengthened. In this research at least two records, each one in a different country, were needed to carry out an assessment with the assistance of the experts, but not enough to create an MCP for which 3 records were required. Furthermore to create a relatively adequate potential distribution model a minimum of 5 unique records were needed. This supports the recent research finding of Rivers *et al.* (2011), who suggest that there is a need for at least five good valid records in order to carry out a robust RL assessment. Even though, it is recommended to use the largest number of unique records when possible.

6.4 Conclusions and further research

Recent research has focused on the potential impact of climate change on the distribution of species or ecosystems, mainly based on the areas where these could migrate to or where they would be more at risk (Beaumont *et al.* 2011; Tovar *et al.* 2013). The research presented here looks further into the potential impacts that climate change can have on montane forests and species that are already threatened by multiple factors (e.g. forest degradation, deforestation and land use change). Montane forests have been identified to be globally threatened, and the tree species that are distributed regionally throughout the tropical Andes need to be taken into consideration in the designation and management of areas for conservation. At least 74 species of the 129 evaluated here have been identified in a threatened category, with several potentially being uplisted into a threatened category due to climate change. These results, in relation to some other endemic species in the region, have shown that the number of threatened species is higher than, for example, the equivalent number of mammals (Brooks *et al.* 2002; Myers *et al.* 2000) and birds (Orme *et al.* 2005) assessed in the region. More important is the fact that many species remain to be evaluated (Joppa *et al.* 2013) and there is a large data gap, especially for rare species and in remote locations (Feeley and Silman 2010, 2011).

Incorporating climate change into conservation efforts needs to be taken with caution, as changing current conservation priorities could jeopardise efforts to mitigate current biodiversity threats such as deforestation and degradation, which are urgent, or could neglect certain areas that may not be affected by climate change as much as may be the case for montane forests (Tingley *et al.* 2013). However, as for many montane forests and specific ecoregions already threatened, considering further potential threats is valuable, especially when this could potentially increase their extinction risk to unprecedented levels.

Strengthening and creating conservation areas is necessary to increase viable populations of threatened species and to provide incentives for private landowners to assist with the inclusion of management techniques that pursue conservation values. These can be used to create conservation corridors and buffer zones to existing protected areas. By doing so, it would maintain ecosystem resilience by addressing the gradual changes that affect ecosystem stability, such as land use, nutrient stocks, soil properties and biomass of long-lived organisms (Scheffer *et al.* 2001).

The outcomes of this RL assessment can inform and encourage both *in situ* (in their natural habitat) and *ex situ* (outside their natural habitat) conservation, to increase knowledge of the actual species' distribution and safeguard the species' survival, maintaining viable populations in their environment. It would also provide baseline knowledge, to assist local botanic gardens to expand species' protection and provide information to local communities on the species' role in forest ecosystem services, which will increase socio-economic values and well-being in the long term (Oldfield and Newton 2012).

Uncertainty surrounds all aspects of forest conservation; especially in areas such as the Andes where many environmental aspects are still to be understood and taken into consideration for better management practices. However, as demonstrated in this research, collaborative research and contributions from regional experts have largely reduced the knowledge gap for some of the aspects considered in this research, and created links for further collaboration to continue the assessment of threatened tree species in the region.

Species distribution modelling provides an overview of the potential change in the suitability of the current forest areas, in which species may not be able to thrive in the future. This preliminary assessment is a precedent to encourage the search of further information as to how these particular species would be able to adapt and interact with their ecosystems to thrive in the face of a changing climate and current threatening factors.

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7 Appendices

Appendix I: Network of experts in the region

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Appendix II: Species data (CD)

Files inside the CD:

File name	Content
Species data	• Species list.xlsx : all species considered • Final classification.xlsx: species RL assessment • Climate change species classification.xlsx: species RL classification under climate change scenarios
CD	Outputs for the species models for current distribution for GAM, Rpart, KSVM, and MaxEnt.
A2	Outputs of predictions for climate change scenario A2 for GAM, Rpart, KSVM, and MaxEnt.
B2	Predictions for climate change scenario B2 for GAM, Rpart, KSVM, and MaxEnt.
Analysis	Overlay of climate change maps and current climate

Appendix III: Preliminary category for the species under each RL criterion

Species	Criteria																												
	A1	A2 (1)	A2 (2)	A2 (3)	A2 (4)	A2 (5)	A2 (6)	A3 (7)	A3 (8)	A3 (9)	A3 (10)	A3 (11)	A3 (12)	A4 (13)	A4 (14)	A4 (15)	A2 (EK)	A3 (EK)	A4 (EK)	B1 hull	B1 Glob Cover	B2 4km^2	B2 10km^2	C1	C2	D1	D2	E	**Final**
Acca macrostema	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	VU	LC	LC	EN	EN	DD	DD	DD	DD	DD	**EN**
Aegiphila bogotensis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Aegiphila cuatrecasasii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	EN	DD	DD	DD	DD	DD	**LC**
Aegiphila ferruginea	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	LC	DD	DD	DD	DD	DD	**LC**
Alchornea anamariae	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**NT**
Allophylus coriaceus	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	DD	DD	DD	DD	DD	DD	DD	**DD**
Aphelandra acanthus	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Axinaea glandulosa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Axinaea grandifolia	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	VU	VU	EN	VU	DD	DD	DD	DD	DD	**VU**
Axinaea lanceolata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	VU	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Axinaea oblongifolia	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	VU	VU	EN	EN	VU	DD	DD	DD	DD	DD	**EN**
Azara salicifolia	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Baccharis latifolia	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Bejaria mathewsii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Berberis grandiflora	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Berberis jobii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	EN	EN	EN	VU	DD	DD	DD	DD	DD	**NT**

Berberis lehmannii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Brunellia brunnea	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	DD	DD	DD	DD	DD	DD	DD	**EN**
Brunellia inermis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	VU	LC	LC	EN	VU	VU	DD	DD	DD	DD	**EN**
Buddleja coriacea	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Buddleja pichinchensis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	LC	LC	VU	EN	LC	DD	DD	DD	DD	DD	**VU**
Calliandra taxifolia	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**VU**
Cecropia bullata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	VU	EN	EN	VU	DD	DD	DD	DD	DD	**VU**
Cecropia telenitida	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Ceroxylon parvifrons	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	VU	LC	LC	EN	LC	DD	DD	DD	DD	DD	**EN**
Cervantesia tomentosa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**NT**
Cestrum peruvianum	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Chionanthus pubescens	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Cinchona pitayensis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Cinchona pyrifolia	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**VU**
Citharexylum joergensenii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	VU	VU	EN	EN	DD	DD	DD	DD	DD	**NT**
Citharexylum rimbachii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	EN	EN	VU	DD	DD	DD	DD	DD	DD	DD	DD	DD	**EN**
Clethra rugosa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Clusia pseudomangle	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Clusia sphaerocarpa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Clusia volubilis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**

Crossothamnus gentryi	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	**EN**
Cyathea arnecornelii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	DD	DD	DD	DD	DD	DD	DD	**EN**
Cyathea austropallescens	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Cyathea carolihenrici	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Cyathea catacampta	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	EN	EN	EN	VU	DD	DD	DD	DD	DD	**DD**
Cyathea corallifera	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	EN	EN	EN	EN	DD	DD	DD	DD	DD	**NT**
Cyathea frigida	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Cyathea halonata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	EN	EN	EN	EN	DD	DD	DD	DD	DD	**VU**
Cyathea parvifolia	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Cybianthus laetus	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Daphnopsis espinosae	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Dendrophorbium balsapampae	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	EN	DD	LC	VU	EN	VU	DD	DD	DD	VU	DD	**EN**
Diplostephium cinerascens	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	**LC**
Dunalia trianaei	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	DD	DD	DD	DD	DD	DD	DD	VU	DD	**EN**
Duranta armata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	EN	EN	EN	EN	DD	DD	DD	DD	DD	**NT**
Escallonia corymbosa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Escallonia resinosa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**NT**
Freziera dudleyi	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Freziera microphylla	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	EN	DD	VU	VU	DD	DD	DD	DD	DD	DD	DD	**EN**
Freziera suberosa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	EN	EN	DD	LC	VU	EN	EN	DD	DD	DD	DD	DD	**EN**

Geissanthus argutus	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	DD	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**VU**
Geissanthus bogotensis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Graffenrieda calyptrelloides	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**EN**
Gynoxys calyculisolvens	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Gynoxys sancti-antonii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	VU	VU	EN	VU	DD	DD	DD	DD	DD	**NT**
Hesperomeles cuneata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Ilex colombiana	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	VU	EN	EN	EN	DD	DD	DD	DD	DD	**EN**
Ilex maasiana	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	DD	DD	EN	EN	DD	DD	DD	DD	DD	**DD**
Ilex rimbachii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Ilex scopulorum	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Ilex sessiliflora	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Ilex uniflora	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Iochroma lehmannii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	DD	DD	EN	EN	EN	VU	DD	DD	DD	DD	DD	**EN**
Joosia aequatoria	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	**EN**
Magnolia yarumalensis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	EN	VU	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**EN**
Meliosma bogotana	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Meriania radula	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Miconia beneolens	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	VU	VU	EN	EN	DD	DD	DD	DD	DD	**EN**
Miconia bipatrialis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**VU**
Miconia calophylla	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	DD	DD	DD	DD	DD	DD	DD	**EN**
Miconia harlingii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**

Miconia velutina	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Monnina pseudosalicifolia	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	VU	VU	EN	VU	DD	DD	DD	DD	DD	**EN**
Myrcianthes discolor	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Myrsine oligophylla	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Nectandra subbullata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	EN	DD	EN	EN	EN	VU	DD	DD	DD	DD	DD	**EN**
Ocotea arnottiana	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Ocotea benthamiana	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Ocotea infrafoveolata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**VU**
Oreopanax bogotensis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Oreopanax ruizii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**EN**
Oreopanax seemannianus	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	LC	DD	DD	DD	DD	DD	**LC**
Palicourea candida	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**VU**
Persea brevipes	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	VU	EN	EN	VU	DD	DD	DD	DD	DD	**EN**
Perymenium jelskii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Phenax laxiflorus	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	EN	DD	DD	DD	DD	DD	**DD**
Piper andreanum	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Piper bogotense	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Piper laguna-cochanum	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**EN**
Podocarpus glomeratus	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Polylepis crista-galli	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**EN**

Polylepis microphylla	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	**CR**
Prunus muris	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	EN	DD	DD	DD	DD	DD	**DD**
Prunus pleiantha	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Prunus urotaenia	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Randia micracantha	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	VU	EN	EN	EN	DD	DD	DD	DD	DD	**LC**
Ribes canescens	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	**EN**
Ruagea microphylla	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	EN	EN	EN	VU	DD	DD	DD	DD	DD	**EN**
Ruagea ovalis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Saurauia bullosa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Saurauia lehmannii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Schefflera inambarica	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**VU**
Schinus meyeri	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	EN	EN	EN	VU	DD	DD	DD	DD	DD	**VU**
Schinus pearcei	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**EN**
Schoepfia flexuosa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Senna versicolor	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Sessea crassivenosa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Sessea dependens	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Smallanthus fruticosus	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**LC**
Solanum cajanumense	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**NT**
Solanum goniocaulon	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	VU	VU	EN	VU	DD	DD	DD	DD	DD	**VU**
Solanum stenophyllum	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**

Symplocos canescens	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	VU	EN	EN	VU	DD	DD	DD	DD	DD	**VU**
Symplocos coriacea	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	VU	VU	EN	VU	DD	DD	DD	DD	DD	**VU**
Symplocos reflexa	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**
Ternstroemia lehmannii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	VU	EN	EN	EN	DD	DD	DD	DD	DD	**EN**
Tournefortia loxensis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	DD	**LC**
Tournefortia undulata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	EN	DD	DD	DD	DD	DD	**LC**
Weinmannia auriculata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	LC	EN	LC	DD	DD	DD	DD	DD	**LC**
Weinmannia cundinamarcensis	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**LC**
Weinmannia jelskii	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	VU	EN	VU	DD	DD	DD	DD	DD	**EN**
Xylosma cordata	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	DD	DD	DD	EN	EN	DD	DD	DD	DD	DD	DD	DD	**EN**
Zanthoxylum brisasanum	n.a.	LC	LC	LC	VU	VU	VU	LC	VU	LC	VU	EN	VU	VU	VU	VU	VU	VU	DD	LC	LC	EN	VU	DD	DD	DD	DD	DD	**EN**

Table 1: Preliminary and final classification of each species. (1) A2 past 50 years FAO deforestation rates and FAO forest area; (2) A2 past 50 years Andes area and deforestation rates from literature; (3) A2 50 past years Andes area and FAO deforestation rates; (4) A2 past 100 years FAO rates and FAO forest area; (5) A2 100 past years Andes area and deforestation rates from literature; (6) A2 100 past years Andes area and FAO rates; (7) A3 next 50 years FAO rates and FAO forest area; (8) A3 next 50 years Andes area and deforestation rates from literature; (9) A3 next 50 years Andes area and FAO deforestation rates; (10) A3 next 100 years FAO rates and FAO forest area; (11) A3 next 100 years Andes area and deforestation rates from literature; (12) A3 next 100 years Andes area and FAO deforestation rates; (13) A4 past/future FAO rates and FAO forest area; (14) A4 past/future Andes area and deforestation rates from literature; (15) A4 past/future Andes area and FAO deforestation rates; (EK) Expert Knowledge. FAO rates: average deforestation rates from all the countries that occurred between 2005 and 2010, available from FAO (2010). Andes area: calculated in this research for areas ≥1,500 m. a.s.l using ArcGIS 10. Deforestation rates from literature: averages from countries available in **Table 3.4 in Chapter 3.**

Appendix IV: Methods for modelling species distribution

Species distribution models attempt to provide detailed predictions of distributions by relating presence or abundance of species to environmental predictors (Elith *et al.* 2006). As Higgins *et al.* (2003) described it: "*All models are caricatures - there is no correct model. But useful models can be parameterized to capture the important features of a process within a restricted domain*". All model approaches are based on some form of empirical data. Model applications require that the fitted functional relationships in the models are general and will hold under changed environmental conditions. Models using a space time substitution are based on rich and detailed spatial information and have a very high level of empiricism. Consequently, this assumption is most critical for this type of model. In order to use empirical spatial models for predictions, it is assumed that the probability of species (or vegetation) occurrence conditional upon environmental conditions is constant in time (Masek 2001).

With the current access to more data and the use of powerful machines, statistical methods can provide a valid and powerful approach to modelling large scale potential distributions under environmental change scenarios (Guisan and Zimmermann 2000). This section provides an overview of the methods available and their application to predict the potential effects of climate change on forests and tree species.

Statistical models

Empirical-statistical vegetation models relate species or community occurrence to site variables that best fit the empirical data. In general, statistical models require less information than process-based models and often better match the spatial and temporal resolution of General Circulation Models (GCM) and present experimental knowledge (Kienast 1998). Thus it is important that statistical models use input parameters that have a physiological - or biological - rationale or are at least highly correlated with physiologically or biologically relevant data (Kienast 1998). The statistical model involves the choice of statistical method, error function and significance tests (Austin 2002).

Generalised Linear Models (GLM): Mathematical extensions of linear models which can handle non-linear relationships and different types of statistical error distributions such as Gaussian, Poisson, Binomial, negative binomial and Gamma, also called the exponential family of distributions (Guisan and Zimmermann 2000; Parviainen and Luoto 2007). Linear multiple regression models predict the response of a variable from a vector of multiple inputs or predictor variables. However, this type of model makes a lot of assumptions about the structure of the data and predictive stable but possibly inaccurate predictions. Also, as ecological data often does not follows the pattern of linear models, GLMs are designed to cope with non-normal distributions of the response variables, transforming them so a linear model can be applied more accurately (Franklin 2009).

Generalised Additive Models (GAM): A non-parametric extension of GLM, GAM is a method for detecting non-linearity of predictors and response functions and then building a parametric model. GAMs allow the data to determine the shape of the response curves, rather than being limited by the shapes available in a parametric class (Yee and Mitchell 1991). This technique applies smoothers independently to each predictor and additively calculates the component response (Franklin 2009; Guisan and Zimmermann 2000). This is often used in multiple regression analysis. They are known to have high predictive accuracy when used for spatial prediction and species distribution modelling (Franklin 2009).

Bayesian approach: Models based on Bayesian statistics combine a *priori* probabilities of observing species or communities with their probabilities of occurrence conditional on the value (or class of values) of each environmental predictor. Conditional probabilities $p(y|xi)$ can be, for instance the relative frequencies of species occurrence within discrete classes of a nominal predictor. A *priori* probabilities can be based on previous results or literature. This results in an a *posteriori* predicted probability of the modelled entity at a given site with known environmental attributes. In vegetation mapping, a *posteriori* probabilities are calculated for each vegetation unit and the unit with the highest probability is predicted at every candidate site (Guisan and Zimmermann 2000).

Environmental envelopes: Bioclimate envelopes can be defined as constituting the climatic component of the fundamental ecological niche, or the 'climatic niche'. Thus, bioclimatic models in their purest form consider only climatic variables and do not include in their processing other environmental factors that influence the distribution of species, such as soil type and land-cover type. The definition of a bioclimate envelope, as with Hutchinson's definition of the fundamental ecological niche, also does not include the influence of biotic effects such as competition for resources (Pearson and Dawson 2003).

Machine learning methods

Machine-learning algorithms refer to induction algorithms that analyze information, recognize patterns, and improve prediction accuracy (Rogan *et al.* 2008).

Classification trees: are used to predict membership of cases of a categorical dependent variable from their measurements on one or more predictor variables (De'ath and Fabricius 2000). Classification trees are developed using different measures that recursively split data sets into increasingly homogeneous subsets representing class membership, based on ranges of values of predictor variables (De'ath and Fabricius 2000). This takes place in three stages: tree building, tree stopping and tree pruning. All classification tree approaches employ hierarchical, recursive partitioning of the data, resulting in decision rules that relate values or thresholds in the predictor variables with pixel classes (Rogan *et al.* 2008). An important advantage of classification trees is that they are structurally explicit, allowing for clear interpretation of the links between the dependent variable of class membership and the independent variables (Rogan *et al.* 2008). They are useful for devising prediction rules that can be rapidly applied and repeatedly evaluated, to assess the adequacy of linear models and to summarise large multivariate data sets (Insightful Corporation 2001). Probably the most important advantage of tree-based models is that they can identify and express in relatively simple form non-linear and non-additive relationships. A tree-based model can capture this kind of interaction by splitting the data into subsets based on the first predictor and then identifying entirely different relationships with other predictors in the two resulting subsets (Michaelsen *et al.* 1994).

The key difference between classification and regression trees is the nature of the response variable. Classification trees predict class membership probabilities for categorical response variables, while regression trees predict average values for interval or ratio scale response variables (Michaelsen *et al.* 1994). The idea is to finally prune the tree to a size or final groups that is likely to provide robust predictions for new data. The best tree size is the smallest one that produces an estimated error rate (Franklin 2009).

<u>Random Forests:</u> Random forests are a combination of tree predictors such that each tree depends on the values of a random vector sampled independently and with the same distribution for all trees in the forest. The generalization error for forests converges as to a limit as the number of trees in the forest becomes large. The generalization error of a forest of tree classifiers depends on the strength of the individual trees in the forest and the correlation between them (Breiman 2001). Studies have found that random forest is an accurate algorithm for predicting habitats (Garzón *et al.* 2006).

The variables predicted to be important in the model help us to understand what variables are driving the distribution of vegetation types. Some distributions are strongly driven by climate, whereas others are driven primarily by edaphic or land-use variables (Prasad *et al.* 2006). It is important to ask how reasonable the maps of future climate scenarios are because one of the main goals in predictive vegetation mapping is to assess the performance of models when predicting under changed climatic conditions. For example, Keenan *et al.* (2011) compare the accuracy of different methods to predict species distributions under different climate scenarios, i.e. RF, random forest; CTA, classification tree analysis; GBM, generalized boosting model; MARS, multivariate adaptive regression splines; GAM, generalized additive model; MDA, mixture discriminant analysis; GLM, generalized linear model; ANN, artificial neural networks; SRE, surface range envelope. They conclude that RF is one of the most appropriate methods to predict the potential distribution of tree species under projected climatic conditions.

<u>Maximum entropy (MaxEnt):</u> estimates the likelihood of a species being present by finding the distribution of maximum entropy (i.e. that is closest to uniform) subject

to the constraint that the expected value of each environmental variable under this estimated distribution matches its empirical average (Phillips *et al.* 2006). MaxEnt uses the 'background' data of the environmental layers in the modeling process. The output of MaxEnt are values between 0 (low) and 1 (high). MaxEnt uses presence and absence (or random background) data. This likely makes it able to correctly identify as suitable, at least some of the 'new' environmental space, if the conditions are closer to the conditions under which the species is currently present than to the conditions under which it is absent. MaxEnt a seems to be able to predict species distributions under novel combinations of climate space (Hijmans and Graham 2006).

The MaxEnt program takes as input a set of point locality data and a set of measurements of environmental (e.g., bioclimatic) variables. It produces as output a map of values over the area covered by the point locality data; these values indicate the relative likelihood that each pixel in the study area contains environmental conditions corresponding to those at the point locations. By selecting an appropriate cut-off value, this continuous map can be converted to a binary map of predicted species presence or absence. One important aspect of this method is that it requires only locations at which the species is known to be present; locations at which the species is inferred to be absent are not used by MaxEnt (Andersen 2010). The output of MaxEnt is an exponential function that assigns a probability to each site (Franklin 2009). The program uses receiver operating characteristic (ROC) curves to diagnose model performance. These curves plot sensitivity (a measure of the rate of correct prediction of known locations) against 1 - specificity (a measure of the rate of correct prediction of absences). The area under the ROC curve, or AUC, is then an index of model accuracy (Andersen 2010). This method has been used by Crossman *et al.* (2012) to evaluate the potential impact of climate change on the distribution of plant species, with adequate results which could, accurately, identify the potential effects of species distribution under potential climate change.

<u>Artificial neural networks (ANN):</u> Masek (2001) uses ANN to model a most suitable forest type based on the conditional probability of vegetation in the environmental space. ANN's are used to parameterise vegetation–environment relationships for the region. The output of the ANN model is an environmental

suitability index for the forest types. The type with the highest environmental suitability is the most likely outcome for any location. Hence, if supplied with slightly changed maps of input variables (i.e. environmental conditions after climate change), the best-suited forest type under future conditions can be estimated (Ostendorf *et al.* 2001)

Process-based models

Process-based vegetation models attempt to describe the components of a system in a mechanistic way and avoid relating vegetation data with parameters that are often uninterpretable in terms of physiology (Kienast 1998). In terms of extinction-risk evaluation, niche-based models (environmental envelopes) tend to predict a stronger level of extinction and a greater proportion of colonization than the process-based model, because niche-based models do not take phenotypic plasticity and local adaptation into account (Morin and Thuiller 2009).

BIOCLIM: a fitted, species-specific, *p*-dimensional environmental envelope (Boxcar) –to model plant species distributions, using one-by-one degree latitude-longitude grid cells. This approach is based on calculating a minimal rectilinear envelope in a multi-dimensional climatic space (Guisan and Zimmermann 2000).

HABITAT: is a similar approach to BIOCLIM, as it gives very similar results although they differ in their classification procedure (Guisan and Zimmermann 2000). HABITAT it uses convex polytope envelopes (minimum convex polygon). A rectilinear envelope can be defined from a very simple classification tree as well (only one dichotomy per predictor), whereas the more complex polytope envelope would need a more detailed tree including more terminal nodes (Guisan and Zimmermann 2000).

DISTRIB (www.nrs.fs.fed.us/atlas): This model predicts the potential habitat for individual tree species, using random forest to predict suitability. Model reliability is based on agreement among various individually modelled regression trees. Predictor importance is based on prediction stability under permutations of data. Built-in bootstrap/predictor-subsetting among 1000 regression-trees to get honest out-of-bag evaluations of prediction strength (USDA Forest Service 2008).

In terms of forest ecosystems, a number of modelling studies have been conducted using forest gap models to assess the impacts of climatic change on forest biomass and species composition in mountainous regions. VEMAP, a continental-scale vegetation response study of the United States, considered how three biogeographical models (BIOME2, DOLY, MAPSS) respond to a double CO_2 scenario. Simulated alpine and subalpine regions in the Western U.S. migrate to higher elevations, and thus decrease in area, while subalpine montane forest boundaries also move upward (Beniston 2003). Using gap model simulations applied to British Columbian mountains, certain upward-moving forest ecosystems could actually disappear from their potential habitats because of the lack of winter cooling, vital for regeneration and the robustness of trees, and a greater sensitivity to droughts and frosts. In all forest impact studies, both in latitudinal and altitudinal terms, projected climatic change will be more rapid than the migration capacity of forests. The faster the rate of environmental change, the greater the probability of species extinction and the disruption of ecosystems (Beniston 2003).

Conclusions

The overview of the available methods to predict species potential distribution and the responses of their environment to potential climate change clearly shows that a wide variety of methods exists and that choosing the most accurate method depends on the type of data used; however, some methods are more easily implemented where the data available is not as abundant as it is the case of the tropical Andes. Nowadays, it is common to find the use of a variety of methods in order to validate the accuracy of the different methods with a given set of data e.g. (Crossman *et al.* 2012; Golicher *et al.* 2011). Therefore it can be concluded that a combination of methods needs to be used to provide reliable result from the research.

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