HIMALAYAN FORESTS: RESOURCE FOR RURAL LIVELIHOOD, MASSIVE CARBON SINK AND INDICATORS OF CHANGING CLIMATE

[Tshewang Dorji]

[Supervisor: Justin Brookes and José M. Facelli]

Submitted in fulfilment of the requirements for the degree of

[Doctor of Philosophy]

Department of Ecology and Evolutionary Biology

School of Biological Sciences

University of Adelaide

[2020]

Abstract

The Himalayan forests are ecologically important for floral and faunal diversity. The forests provide many invaluable ecosystem services that have significant roles in the socio-economic and cultural lives of people living in these high mountains. However, these forests are facing rapid degradation due to economic growth and climate change. Adapting to these emerging challenges require a better understanding of linkages between human-environment and climate variables to manage these forests sustainably. Various aspects related to forest ecology and restoration, ecosystem services, forest growth and their response to climate are at an early stage of investigation in Bhutan Himalayas. Within this thesis, we have studied the aspects of regeneration and restoration, carbon storage and climate impacts on the high-altitude forest to provide more information relevant for forest management and sustainable utilization of resources.

We investigated the socio-cultural values of high-altitude forests through the lens of ecosystem services to understand the importance of forests in the livelihood of rural communities. By assessing the priorities and values of stakeholders on a suite of ecosystem services, the research identified various gaps in the preferences of ecosystem services that can potentially lead to competing interests in forest management. The study highlighted several challenges that currently undermine sustainable forest management. This includes inadequate information on oak forest ecology and regeneration, forest biomass-carbon storage potentials, and forest’s growth response to changing climate.

A nation-wide study of the old-growth oak forest of Bhutan indicated inadequate forest regeneration, particularly oaks, which is a huge concern for long-term conservation of these forests. Concentrated local experiments using exclosures and canopy gap creation indicated livestock grazing and inadequate canopy gaps as important factors leading to regeneration failure. On the other hand, forests without grazing resulted in undesirable bamboo thickets in the understory. An experiment using tree shelters and planted oak seedlings supported our findings on the adverse effect of grazing on oak regeneration. Tree shelters not only deterred seedling herbivory but also promoted the survival of seedlings and kept bamboo thickets at bay. Based on this study, we recommended tree shelter techniques as a promising avenue

to regenerate oaks in the Himalayas and should be considered in the annual forest plantation programs.

We quantified forest biomass and carbon storage potentials of two dominant forest types of high-altitude Himalayas—broadleaf forest and an adjacent conifer forest—to understand the role of these forest ecosystems in biomass accumulation and, subsequently, the carbon storage. The study estimated a total carbon stock of 366.7 ± 18.3 (Mean ± SE) Mg ha^{-1} (megagram per hectare) for broadleaf forest and 303 ± 9.3 Mg ha^{-1} for conifer forest indicating the importance to conserve these forests for climate change mitigation.

The major challenge for the sustainability of these forests, however, lies in trees response to changing climate conditions, particularly to the ever-changing temperature and precipitation patterns in the Himalayas. We investigated this objective through a dendrochronological approach correlating tree growth of the high-altitude Himalayan larch (*Larix griffithii*) with climate (temperature and precipitation). The growth of trees was positively correlated with summer monsoon rainfall (May-August) and negatively correlated with the summer temperature indicating that summer precipitation and temperature were two crucial factors limiting tree growth. The study resulted in a reconstruction of 639-year (1379-2018) monsoon rainfall patterns over Bhutan Himalayas. The global weather patterns of El Nino-Southern Oscillation and Indian Ocean Dipole demonstrated a strong influence on the rainfall variability and consequently tree growth over Bhutan. With global climate projections predicting high rainfall variability in the future, the response of trees to these variations may significantly impact tree growth and forest distribution, which require further investigation spanning both temporal and spatial scales.

The thesis is a comprehensive dive into the ecology of temperate old-growth forests of Bhutan Himalaya and is a significant contribution to the knowledge of these forests, in how they respond to human interference, and, potentially changing climate.

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Statement of Original Authorship

I certify that this work contains no material which has been accepted for the award of any other degree or diploma in my name in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text. In addition, I certify that no part of this work will, in the future, be used in a submission in my name for any other degree or diploma in any university or other tertiary institution without the prior approval of the University of Adelaide and where applicable, any partner institution responsible for the joint award of this degree.

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I acknowledge the support I have received for my research through the provision of an Australian Government Research Training Program Scholarship.

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Date: 10 November 2020

Acknowledgements

I would like to thank my supervisor **Justin Brookes** for his exceptional guidance, friendship and encouragements, which made my PhD journey a memorable one. My co-supervisor **José M. Facelli** has been equally helpful with his technical advice, guidance, and motivations throughout the PhD study. I take this opportunity to thank **Steven Delean** for his statistical support and R lessons. I want to thank **Himlal Baral** (Center for International Forestry Research), **Prof. Markus Stoffel** (University of Geneva) and **Evelina Facelli** (The University of Adelaide) for their support and guidance in my research.

The production of this PhD thesis has been an arduous task involving years of fieldwork in challenging weather conditions, sacrifices and commitments that cannot be completed by any single person. The list will be endless if I mention anybody by name. Hence, I would like to express my gratitude to all my colleagues directly or indirectly involved in this journey that made this thesis possible. I am thankful to the officials at The University of Adelaide, South Australia and Ugyen Wangchuck Institute for Conservation and Environmental Research (UWICER), Bhutan for facilitating my work and making life more comfortable.

The PhD study was carried out with support from Research Training Program Scholarships (RTP) of Adelaide University funded by the Australian Commonwealth Government, for which I am profoundly grateful and acknowledge the unique opportunity given to me. My fieldworks were supported by several organizations namely, the School of Biological Sciences (University of Adelaide), Forest Ecosystem Restoration Initiatives (Korea Forest Research and Convention on Biological Diversity), Sloping land in transition project (Royal Government of Bhutan and CIFOR) for which I am highly grateful.

Special thanks go out to the two anonymous examiners for their valuable suggestions which have improved this thesis significantly.

I dedicate this thesis to all my lovely parents and family in Bhutan whose love and encouragements have always been an inspiration to me. My wife, son and daughter, has been very compassionate and even agreed to accompany me to Australia on this long journey- to see this finally happen. Thank you.

Chapter 1: Introduction

1.1 BACKGROUND

The Himalayas are often referred to as “the roof of the world” (Ives 1981), “the third pole” (Wang et al. 2014; Bahadur 1993), “the water tower of Asia” (Bandyopadhyay 2013), and so on. The different names indicate the significance of these high mountain ranges in sustaining the life-supporting functions at both the local and global level. The geologically productive mountains with high altitudinal variations ranging from tropical plains to the snow-covered mountain peaks coupled and wide climatic conditions (hot and humid to cold and dry) make the Himalayas the centre of biodiversity. The eastern Himalayas fall within the Indo-Burma biodiversity hotspot having 13500 plants species and 2185 vertebrate species with a high degree of endemism (Myers et al. 2000). In addition to being a biological hotspot, the Himalayas are a water source to seven of the world’s major rivers and countless tributaries. The water from rain, snowmelt and glaciers gradually drain through the thick vegetation into the rivers downstream that feed the world’s most populated region (Chatterjee et al. 2010).

In recent decades, the Himalayas had undergone unprecedented changes in the backdrop of increasing population pressure and changing climatic conditions. Natural forests and resources that were once thought to last forever have depleted at an alarming rate due to deforestation and over-exploitation of natural resources (Ives 1987; Singh 1998). Facilitated by the anthropogenic pressures, forest composition has been changing from native broadleaf forest to invasive pine-dominated forests (Singh et al. 1984; Bruggeman et al. 2016; Gilani et al. 2015). Many important broadleaf forests have failed to reproduce under its canopy (Andersson 1991; Callaway 1992; Johnson 2004; Kelly 2002; Larsen and Johnson 1998; Linhart and Whelan 1980; Lorimer et al. 1994; Palmer et al. 2004; Perrin et al. 2006; Watt 1919; Shrestha and Paudel 1996; Singh and Rawat 2010; Singh et al. 1997), which raises the question of their sustainability. Such situations will have severe social and ecological impacts as the change in vegetation or land-cover leads to change in the supply of ecosystem services (Naudiyal and Schmerbeck 2017). For instance, deterioration and degradation of oak forests were linked with the drying of natural springs and reduction in freshwater discharges (Singh and Pande 1989) and decline in wildlife abundance (Mcshea et al. 2007).

Climatically, the warming of the Himalayas is predicted to be above the global average temperature in the 21st century (Singh and Bengtsson 2005). Perennial snow cover and glaciers that feed the major rivers are receding at a rate higher than before (Bajracharya et al. 2006; Bolch et al., 2012; Ding et al., 2006). The precipitation pattern is becoming more erratic and unequal in distribution (Basistha et al. 2009). Some areas receive heavier than normal rains while others receive lesser rains. The precipitation during the rainy season itself has become fewer but more intense during the periods causing droughts and flash floods. Loss of lives and properties, crop failures, disease outbreaks and poverty are on the rise that has the potential to disrupt the social harmony and global economy.

The forestry sector is seen as a crucial sector that has the potential to contribute to the social and millennium development goals, reduce global carbon emissions and mitigate climate change (Candell and Raupach 2008). However, the understanding of the natural environment and processes that can sustainably contribute to the above goals are limited, mainly due to less science-based information on these ecosystems, often leading to conflicts between development and conservation (Ives and Messerli 1989). This is particularly true in the eastern Himalayas where complex ecosystem-human interactions are dominant, and only limited studies have been conducted.

Bhutan Himalayas is located in between China in the North and India in the south and forms an essential component of the Eastern Himalayas. The country has a wide range of elevations (150 m to 7570 m) and vast climatic conditions that result in diverse vegetation types and rich flora and fauna (Sargent et al. 2009) (Figure 1). Broadly, it can be divided into the alpine zone (above 4000 m), temperate zone (2000-4000 m), and subtropical-tropical zone (150-2000 m). The temperate zone consisting of broadleaf and conifer forests dominates the total forest cover. The tropical forest consists of tropical warm vegetation types while the alpine region is devoid of forest cover. The abundant and intact natural resources are reserved in the virgin forests, which is currently estimated at 71% of the total land area (DoFPS 2016). It has been reported that the forests of Bhutan have the potential to serve as a refuge to the endemic flora and fauna of the Himalayas and can be regarded as “a vital resource for the Himalayas” (Sargent et al. 2009). As forest cover decreases in most of the countries, Bhutan is a unique nation where forest cover has been increasing for the last few decades (Bruggeman et al. 2016). This success in forest conservation can be attributed to the strong affinity to Buddhist religion backed by regulatory frameworks and enabling government policies. Nature conservation in Bhutan is driven by the fundamental Buddhist principle of living in harmony with nature. At the same time, realizing the importance of forestry sector in socio-economic wellbeing and combating global warming and climate change, more impetus was given to forest conservation in the form of National Forest Policies and National Environment Strategy (Sears et al. 2018).

Bhutan adopted the development philosophy of Gross National Happiness (GNH) where economic development of the country is carefully balanced with environmental conservation and spiritual wellbeing of the people. Similarly, a requirement to maintain a minimum of 60% forest cover at all times was mandated in the constitution of Bhutan (RGOB 2008). The country is currently regarded as a carbon-neutral country and has further plans to reduce its carbon emissions, reforest its barren lands, and plans to become a carbon-negative country. More than 50% of the country is declared as protected areas, which are connected to each other through a network of biological corridors. The biological corridors provide a flat form for the rare and endangered plant and animal species to grow and thrive. The country, as a result, has succeeded in its nature conservation.

Figure 1. Bhutan’s high-altitude temperate forest is a source of rural livelihood, a global carbon sink and an indicator of changing climate.

The necessity for maintaining a good forest cover also rises from the point of socio-economic development of the country. Most of the population are dependent on forest and its resources for their livelihood, socio-cultural activities, and a suite of other ecosystem services. Nature-based eco-tourism, hydropower and agricultural farming are the key economic sectors of the country. Bhutan is by large an Agrarian society with more than 57% of the total population entirely dependent on subsistence agricultural farming. Each rural household practices small-scale agricultural and livestock farming.

Bhutan also faces the challenge of being a small developing country with a fragile mountain ecosystem and a dual objective of economically developing the country while preserving its natural and cultural heritage, and value systems. Managing the highly diverse and complex mountain ecosystem with limited scientific knowledge not only hinders sustainable forest management but also makes it uncertain. The poor availability of appropriate long-term forest management information and ambiguity in how ecosystems may respond to changing climatic conditions further add to a number of unanswered questions that undermine the sustainable management of forest and its natural resources. Bhutan's least disturbed and most extensive forest cover make it a heavenly place for both ecological and climatic research. However, despite the potentials, not many studies are currently undertaken focusing on forest ecosystem services and valuation, regeneration and restoration ecology, carbon sequestration and tree-growth climate relationships. Understanding these processes have relevance to forest conservation, sustainable management, and the wise use of natural resources in the face of changing climatic conditions.

Bhutan's forestry sector provides the highest ecosystem services (ICIMOD 2017), which is estimated at USD 15.5 billion per year which is significantly higher than even the Gross Domestic Product of USD 3.5 billion per year (Kubiszewski et al. 2013). Ecosystem services and valuation of the forest has become necessary to protect the existing forest and garner plough-back funding mechanisms, to meet further conservation costs, mainly through ecosystem services and carbon credits. Valuation should consider not only monetary assessments but also socio-cultural values to include the perception of local communities, their priorities, and values. This is because forests and farming communities are strongly intermingled. The subsistence farming on the hills is entirely dependent on the forests for water, fodder and grazing grounds for livestock, leaf litter for manure, timber for house construction and firewood for cooking and warming. The people have a strong affection towards their natural environment, and these forests serve as an important incubator for spiritual wellbeing and many social and cultural values. Thus, ecosystem valuation should encompass wide-ranging ecosystem services that are physical as well as less tangible socio-cultural and spiritual values that people place on these forests. Understanding these components will help to determine the true value and importance of forests by the local communities and policymakers. Such a process will help educate the general public on the benefits of forest conservation and human wellbeing.

Valuation of forest and its ecosystem services relies heavily on a better understanding of forest ecology and status of forest structure and recruitment and how these ecosystems are responding to anthropogenic and climate pressures. Forest regeneration is an essential feature that will ensure forest re-growth, survival, and sustenance of ecosystem services. Bhutan being an agrarian dominant society, forest grazing by livestock (Figure 2) is an integral part of the farming system (Norbu 2000; Roder et al. 2002). While the vast forest cover is an advantage for livestock farming, overgrazing by large herds of migratory and sedentary livestock often lead to forest regeneration failures, particularly in the broadleaf forests. Overgrazing and oak regeneration failure are thus prevalent issues throughout the Himalayan forest extending from India (Singh and Pande 1989; Singh et al. 1997; Upreti et al. 1985), Nepal (Metz 1997; Shrestha 2003; Shrestha and Paudel 1996; Vetaas 1997) and Bhutan (Covey et al. 2015; Dorji 2012; Tashi 2004). This is a significant concern for the long-term sustainability of old-growth forests. Studies on forest structure and regeneration dynamics will enable us to determine species recruitment and future forest composition that are necessary for forest conservation and sustained provision of ecosystem services. Identifying important forest types with regeneration problems/failures and designing techniques that will assist in restoration programs are therefore important. Many native forest types in the Himalayas are facing regeneration problems, particularly in the broadleaf forests. Artificial regeneration, which involves raising regeneration-problematic seedlings in the nurseries and restocking in the degraded forests using techniques that deter herbivory such as fences and tree shelters have relevance for nature conservation and saving the specific forest type. Such measures should be given special consideration in the Himalayas.

Figure 2. Cattle grazing is an integral part of the farming system in Bhutan. Large herds of cattle graze freely in the forest.

A better understanding of how a forest or ecosystem responded to the past climate and comparing to how they are responding to the present climate will enable us to predict their responses to the future climate. As meteorological recordings in the eastern Himalayas only began recently (after the 1980s in Bhutan), tree-growth climate comparisons are limited (Figure 3). The approach to deal with this setback is addressed by using high-resolution climate proxies such as tree-rings (Singh et al. 2009; Krusic et al. 2015; Sano et al. 2013), ice and glacier cores (Dahe et al. 2000; Banerjee and Azam 2016), lake sediment cores (Lone et al. 2019; Sanwal et al. 2019) and laminated stalagmite records (Zhang et al. 2018). Reconstructions from diatoms (Fayó et al. 2018), fish otoliths (Darnaude et al. 2014) and permafrost (Saito et al. 2013) are also gaining popularity. Such information will enable us to look into the future based on long term past climate variabilities. Our prediction and forecasting abilities will be improved, which will help to adapt to the changing climate conditions appropriately.

Changing climate and habitat environments would mean that some species may gradually shift their distribution ranges. This is particularly true for the species growing near the tree line whose distribution would be severely impacted by the upward shifting forests. Some species may permanently vanish while some others may expand their distribution ranges. Growth of some species may be favoured by gradual warming or hampered by erratic precipitation, both of which have implications for forest carbon sequestration and carbon accounting. High-altitude temperate forests of Bhutan are generally least disturbed, and most have attained old-growth state that is of high conservation significance. The forests are floristically very rich, with more than 35 tree species (Dorji 2012). The upper canopy is dominated by very old-growth oak trees with DBH (diameter at breast height, cm) of 85 to above 223 and height reaching up to 39 m mixed with conifer species like *Abies densa*, *Picea spinulosa and Tsuga dumosa*. The middle storey is dominated by broadleaf species like *Acer campbelli*, *Rhododendron spp*, *Gambelli ciliata*, *Ilex dipyrena*. *Prunus* and *Corylus ferox*. The understorey is mostly dominated by *Daphne bholua*, *Peris formusa*, *Berberis* and a bamboo species *Yushania microphylla* (Dorji 2012; Tashi 2004). The rich forest, along with different stratification, is home to various wild animals, birds and other mammals and a source of fresh springs (Singh and Pande 1989).

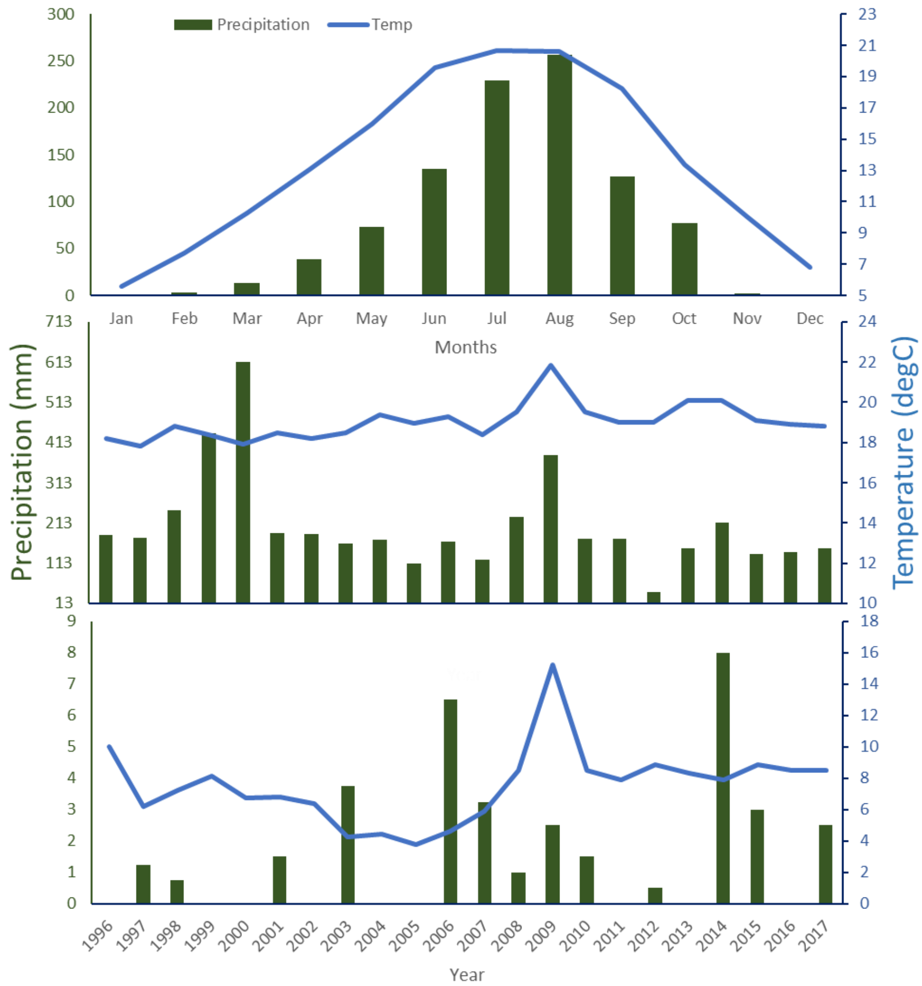

Figure 3. Climatic records of a local met station (Drukgyel Dzong) located close to the study site. Figure shows the precipitation and temperature fluctuations over the months (top), across the years in summers (middle) and winters (bottom).

The old-growth forests are consistently being recognized worldwide for their roles in carbon sequestration potentials (Carey et al. 2001; Luyssaert et al. 2008), although very little is understood in the Bhutan Himalayas. Few studies on the carbon storage capacity of forests in Nepal (Adhikari et al. 1995; Garkoti 2007; Verma et al.

2012) indicated the huge carbon storage potentials of forest in the central Himalayas. Forest biomass and carbon assessments will be invaluable for precise accounting and reporting of the carbon storage capacity of the forests and are crucial for mitigating regional and global carbon emissions. Thus, establishing baseline information on carbon storage of important forest types and forests with high carbon density should be given special conservation consideration. These investigations and assessments are relevant for forest conservation and sustainable management plans.

In this PhD thesis, we have attempted to address the above points by focusing on the most widely distributed temperate forests of Bhutan Himalayas. By using ecological methods and biophysical measurements, we aimed to assess the human-environment-climate interactions and how this interaction shapes the mountain ecosystems. Most specifically, we show how high-altitude forests can serve as a source of rural livelihood, massive carbon sink and indicators of changing climate. Our investigations on ecosystem services, forest regeneration and restoration, forest carbon storage and forest growth-climate relationships form crucial components of sustainable forest management that have policy implications. Our study is expected to help decipher both temporal and spatial climate-sensitive forest management options across Bhutan Himalayas in the face of climate change.

1.2 STUDY AREA

The kingdom of Bhutan is situated in the Eastern Himalayas, sandwiched between China in the North and India to the South (Figure 4). The temperature and rainfall vary significantly due to great altitudinal variations. Temperature ranges from over 20 °C to freezing points while the average annual precipitation varies from 800-3000 mm (Roder 1992). The climate is strongly influenced by the summer monsoon winds which bring the majority of rain in the summers (June-September). There are four distinct seasons consisting of Spring (March-May), Summer (June-August), Autumn (September-November), Winter (December-February).

Our study site ranges from 2500-3500 m in altitude and falls within the temperate forests and adjoining ecosystems of Bhutan (Figure 4). The forests are dominated by mixed conifer species and cool temperate broadleaf species (Figure 5). The forests occupy the hills of the inner valleys which are fertile and where major Bhutanese population live and practice agricultural farming. Forests play an important role in maintaining soil fertility by transferring nutrients to agricultural fields (Roder et al. 2003). Forests are also a source of timber, non-wood forest products, fodder, grazing and other ecosystem services, including the cultural and spiritual wellbeing of local communities (Dorji et al. 2019).

Study 1 (Chapter 2) was conducted in seven high-altitude old-growth oak forests adjacent to villages of Western Bhutan where socio-cultural values and importance of forest and its ecosystem services were investigated. Study 2 (Chapter 3) was conducted in the old-growth oak forests of Bhutan. The forest structure, composition and regeneration dynamics of oak with respect to human disturbances were investigated. The study also used permanent fenced and unfenced plots established since 2000 in western Bhutan to understand the effects of forest grazing on tree species regeneration. Based on the results of study 2, study 3 (Chapter 4) focused on assisted artificial regeneration of planted oak seedlings using tree shelters and grazing to determine the best possible technique to restore oak forests. Study 4 (Chapter 5) was concentrated in two major forest types of temperate Himalayas; Mixed conifer forest (referred as "conifer forests" henceforth) and oak-dominated broadleaf forest (referred as "broadleaf forest" henceforth). Detailed above ground (AGB) and belowground (BGB) tree biomass carbon and soil organic carbon (SOC) was estimated to understand the carbon storage potentials of high-altitude forests.

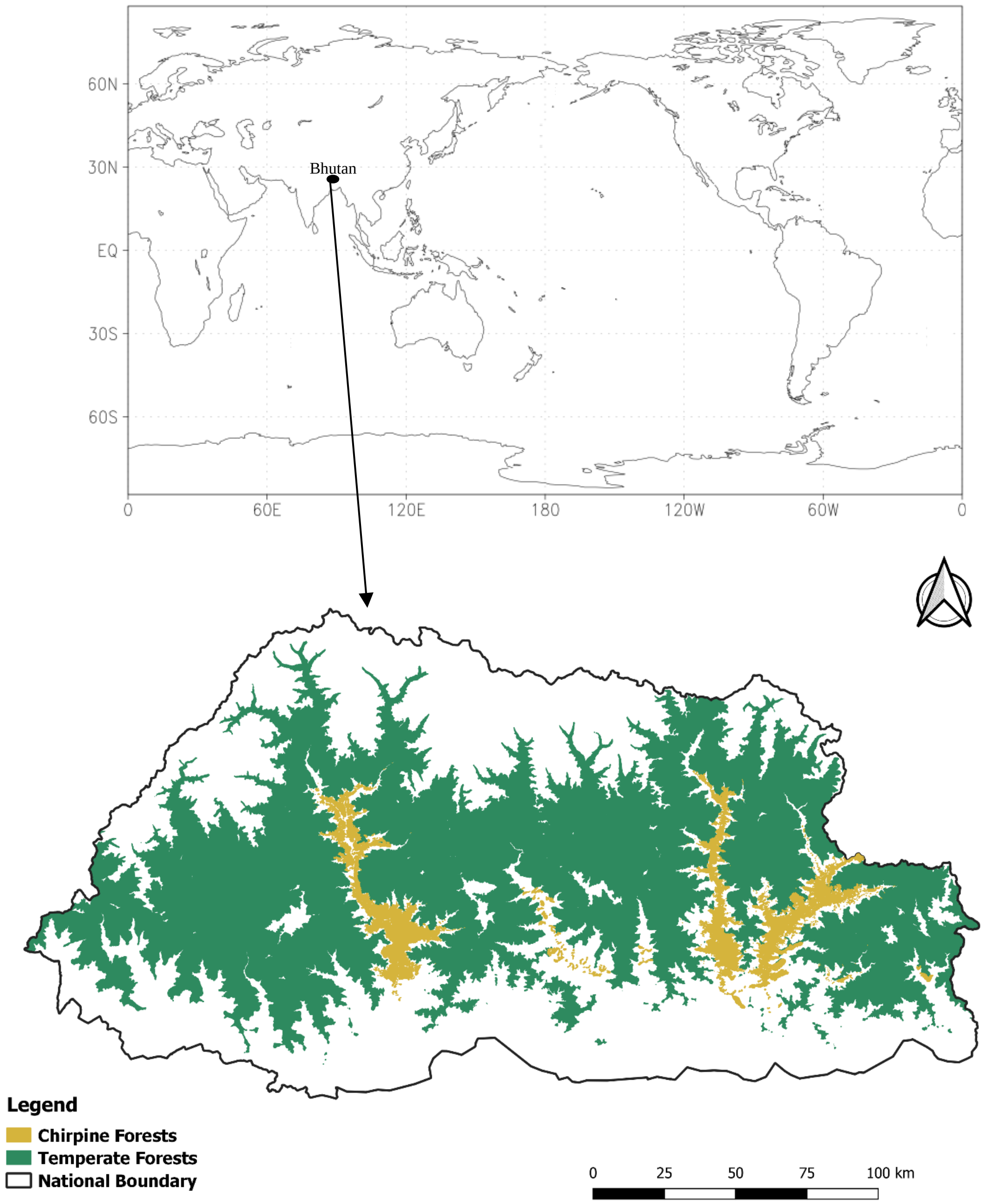

Figure 4. Map of the world showing the location of Bhutan (Above) and map of Bhutan showing the temperate forest distribution.

Study 5 (Chapter 6) was conducted at a high-altitude *Larix griffithii* forest adjacent to the tree line. The climate-tree growth relationships were investigated to reconstruct past climate based on annual tree-rings.

Figure 5. The mixed conifer forest (left) and cool broadleaf forest (right) dominates the temperate forest of Himalayas.

1.3 PURPOSES AND OUTLINE OF THESIS

The primary objective of my thesis is to generate information on sustainable forest management of temperate Himalayas of Bhutan and attempt to interpret the research findings into the forest management plans and policies, which would subsequently lead towards recognition of their inherent value and further protection. By focusing on two major temperate forest types namely a brown oak-dominated forest and mixed conifer forests of Bhutan, I was able to concentrate on specific research questions that are invaluable for social wellbeing, ecological balance, and climate-change mitigation. Basically, they all relate to the complex nature of human-environment interaction and their outcomes and pursue a middle path to sustainable development through the presented work.

Chapter 2 aims to investigate the dependence of rural people on the suite of ecosystem services and how they value their forests from the socio-cultural perspective. Special emphasis was given to the identification of ecosystem services, their priorities on ecosystem services and their viewpoints on how these services can be enhanced. Following questions were taken into consideration.

a) Which ecosystem services are viewed to be scarce and how to protect them?
b) Was there a mismatch in the priorities of ecosystem services between local communities and forest managers?

Chapter 3 focused on the forest structure and regeneration of brown oak forests throughout the country and aimed to understand the forest composition and recruitment status. Research questions for this study were:

a) Is the oak forest regenerating adequately? What are the factors that promote or hamper oak regeneration?
b) Does fencing and protection from herbivory lead to better recruitment of oak seedlings?
c) What changes in current forest management practices are required to protect these important forest type?

Chapter 4 is essentially an extension of study 2 and aims to explore the restoration techniques of oak seedlings using different tree shelters. Specific questions addressed were:

a) Is grazing a major contributor to oak regeneration failure?
b) What are the most appropriate types of tree shelters that can deter herbivory and restore the oak forests?

Chapter 5 aimed to estimate the forest carbon storage potentials of the two temperate forest. Following points were focused:

a) Can temperate forest serve as an important carbon storehouse of the Himalayas?
b) How temperate broadleaf forest differ from conifer forest in terms of their carbon storage potentials?
c) What are the major carbon pools in the forest that contribute to the carbon absorption?

Chapter 6 studies the growth response of high-altitude *Larix griffithii* to climate, and the study aims to:

a) Which climate variables are most sensitive to the growth of high-elevation *Larix griffithii?*
b) Does *Larix* have the dendroclimatic potentials for reconstructing past climate?
c) What are the global climatic factors that determine the growth of trees in the Himalayas?

The diagrammatic representation of the thesis is presented in Figure 6. All the studies conducted in this thesis aims to generate information on the Himalayan temperate forests that will lead to a better understanding of forest ecology-human interactions and aid in the conservation and sustainable utilisation of forest resources.

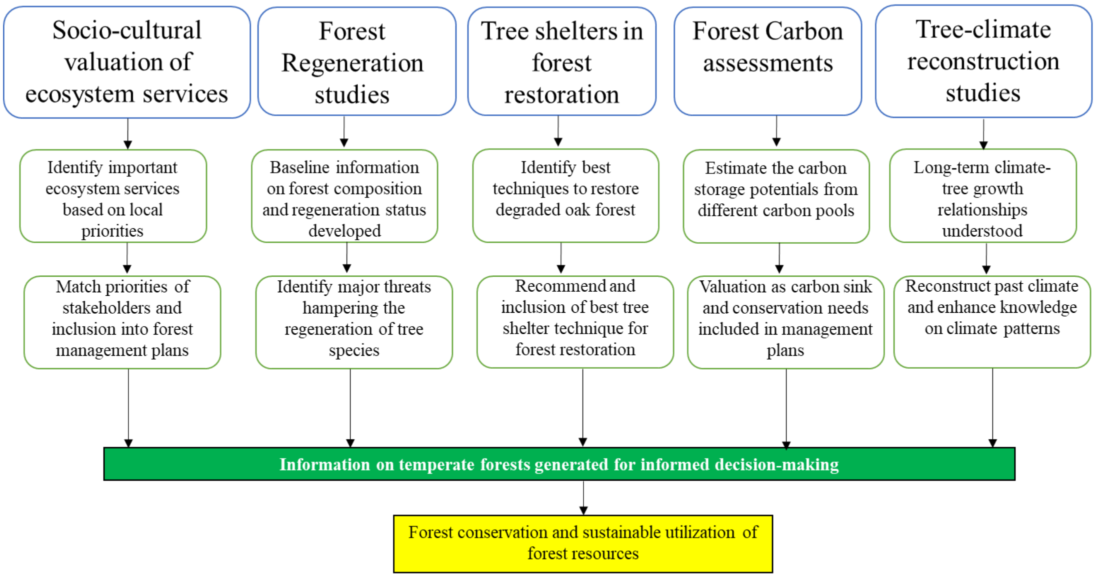

Figure 6 The diagrammatic representation of the objectives and flow of studies towards the goal of sustainable forest management.

1.4 ORGANIZATION OF THE THESIS

This thesis is divided into six sections or chapters comprising of an introductory chapter which introduces five studies or chapters. The first chapter introduces the five individual studies/chapters. The seventh chapter is a general discussion and conclusion section which brings together the results of the five studies and their relevance to forest management in Bhutan. For convenience, the references used in each study are included at the end of every manuscript, while the references for the introductory and general discussion chapters are placed at the end of the thesis as Bibliography. The supplementary materials and appendices are placed at the end of the thesis after the Bibliography section. The summary of the core chapters of the thesis are as follows:

Chapter 2. Socio-Cultural Values of Ecosystem Services from Oak Forests in the Eastern Himalaya (Authors: Tshewang Dorji, Justin D. Brookes, José M. Facelli, Robin R. Sears, Tshewang Norbu, Kuenzang Dorji, Yog Raj Chhetri and Himlal Baral. Published in *Sustainability* **2019**, *11*(8), 2250; **https://doi.org/10.3390/su11082250**)

Summary: In this specific paper, we show how local forest users identify and prioritize ecosystem services from their perception and how they relate to their daily lives and socio-cultural values. Such information is vital for designing an inclusive, sustainable forest management plans that consider the wide-ranging of interests of forest users. The forest, no doubt, provides a huge number of ecosystem services to the local communities. Most of the ecosystem services that have a direct relationship with people's lives were easily identified and recognized, while several regulating and supporting services that are difficult to perceive physically were less appreciated. There also exist distinct areas of ecosystem services priority differences between local forest users and forest managers that need to be streamlined through awareness programs and consultative discussions to protect the forest and invaluable ecosystem services further. Key sustainable forest management practices that will contribute to the overall goal of judicious use of natural goods and services were highlighted in this paper.

Chapter 3. Regeneration dynamics of brown oak (*Quercus semecarpifolia*) forests with special reference to grazing-fencing and canopy gaps in Bhutan, Eastern Himalaya (Authors: Tshewang Dorji, Kesang Wangchuk, Steven Delean, Tshewang Norbu, José M. Facelli, Rinchen Dorji, Kuenzang Dorji and Justin D. Brookes. Submitted for publication in *Forest Ecology and Management*)

Summary: In this manuscript, we present the status of forest composition and natural recruitment of temperate broadleaf forest dominated by brown oak (*Quercus semecarpifolia*), which has failed to regenerate adequately. Based on nationwide forest survey and vegetation data analysis, we confirm the chronic absence of oak saplings and failure in recruitment in the forest. Our results from long-term exclosures and gap-creation experiments show that the combined effects of overgrazing and smaller forest canopy gaps currently hamper forest recruitment. We highlighted that complete exclusion of grazers from the forest is not desirable as it resulted in over dominance by shrubs and bamboo thickets. The study suggested the need to restore the forest through artificial regeneration and use of modern planting techniques that reduce herbivory to assist forest recruitment and conservation of brown oaks in the Himalayas.

Chapter 4. Tree shelters facilitate brown oak seedling survival and establishment in a grazing dominant forest of Bhutan, Eastern Himalaya (Authors: Tshewang Dorji, José M. Facelli, Tshewang Norbu, Steven Delean, Justin D. Brookes, Published in *Restoration Ecology 2020,28(5), 1145-1157;* **http://dx.doi.org/10.1111/rec.13176**)

Summary: In line with the recommendations from study 2, study 3 presents the four years result of assisted forest restoration using artificial regeneration and tree-shelters to regenerate the brown oak forests. Cost-effective restoration techniques using three different types of tree shelters were assessed in combination with grazing on the field performance of planted oak seedlings in the natural forest. We confirmed that forest grazing is the primary factor that leads to regeneration failure. Plant survival and growth were significantly enhanced when protected from grazing through tree shelters. Based on the direct advantages of tree shelters in plant growth and survival and perceived benefits in supporting the local people's rights to free forest grazing, we recommend incorporating tree shelters in forest management plans to restore brown oak forests in Bhutan Himalayas.

Chapter 5. Carbon storage potentials of two major forest types of Bhutan, Eastern Himalayas (Authors: Tshewang Dorji, José M. Facelli, Tshewang Norbu, Madan Kumar Lama, Yog Raj Chhetri and Justin D. Brookes. Submitted for publication in *Journal of Forestry Research*)

Summary: In this manuscript, we present the forest carbon storage potentials of two major forest types of the high-altitude temperate region of Bhutan Himalaya. Such assessments are imperative for proper estimation, reliable accounting and reporting of forest carbon from Bhutan, a country which has pledged to remain a global carbon sink for all times. Based on our field survey and measurement of carbon in all the five carbon pools recommended by IPCC'S good practice guidance, we show that temperate forests store huge quantities of carbon ranging from 303-367 Mg ha^{-1}

(megagram per hectare) in their woody biomass and soils. The study recommended that the special protection of these forests must be considered from the perspective of mitigating carbon emissions.

Chapter 6. 639-year monsoon (May-August) precipitation reconstructions indicate monsoon (May-August) precipitation as the limiting factor for the growth of high altitude *Larix griffithii* in western Bhutan Himalayas (Tshewang Dorji, Karma Tenzin, Dorji Dukpa, Markus Stoffel, Tshewang Norbu, José M. Facelli and Justin D. Brookes. In preparation for submission to Geophysical letters).

Summary: Understanding long-term tree growth-climate dynamics are essential to untangle sustainable forest management practices and adapt to an ever-changing climate. In this manuscript, we use tree-rings of a high-altitude conifer *Larix griffithii* to determine its growth in relation to past climate. Based on the species sensitive response to summer precipitation, we reconstructed high resolution, annually resolved 639 years of summer precipitation of Bhutan Himalaya that indicated high annual to decadal variability patterns. The precipitation patterns over Bhutan were found to be strongly correlated to El Niño Southern Oscillation and Indian Ocean Dipole (IOD). Our study recommended more studies in the Himalayan range to provide decision-makers with information on likely impacts of climate change on monsoon precipitation and design mitigation and adaptation strategies.

Chapter 2: Socio-Cultural Values of Ecosystem Services from forests in the Eastern Himalaya

2.1 STATEMENT OF AUTHORSHIP

Statement of Authorship

Title of Paper	Socio-Cultural Values of Ecosystem Services from Oak Forests in the Eastern Himalaya
Publication Status	☑ Published ☐ Accepted for Publication ☐ Submitted for Publication ☐ Unpublished and Unsubmitted work written in manuscript style
Publication Details	Paper published in Sustainability Journal Received: 27 February 2019; Accepted: 9 April 2019; Published: 15 April 2019

Principal Author

Name of Principal Author (Candidate)	Tshewang Dorji		
Contribution to the Paper	Conceptualization, methodology, data collection and analysis, writing—review and editing		
Overall percentage (%)	70 %		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	15.04.2020

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);
ii. permission is granted for the candidate in include the publication in the thesis; and
iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Himlal Baral		
Contribution to the Paper	Conceptualization, Methodology design, funding acquisition, editing and supervision.		
Signature		Date	19.04.2020

Name of Co-Author	Robin R. Sears		
Contribution to the Paper	Designing and developing research protocol with the lead author, investigation, writing and editing		
Signature		Date	20.04.2020

Please cut and paste additional co-author panels here as required.

Name of Co-Author	Justin Brookes		
Contribution to the Paper	Conceptualization, Methodology design, fieldwork, data analysis, funding acquisition, supervision and editing.		
Signature		Date	21 July 2020

Name of Co-Author	José M. Facelli		
Contribution to the Paper	Supervision, fieldwork, data collection, methodology and data analysis, writing and editing		
Signature		Date	4 August 2020

Please cut and paste additional co-author panels here as required

Name of Co-Author	TshewangNorbu		
Contribution to the Paper	Developing research protocol and data collection, investigation, writing and editing in association with the lead author		
Signature		Date	21.4.2020

Name of Co-Author	Yog Raj Chhetri		
Contribution to the Paper	Developing research protocol and data collection, investigation, writing and editing in association with the lead author		
Signature		Date	21.4.2020

Name of Co-Author	Kuenzang Dorji		
Contribution to the Paper	Methodology, map preparation, fieldwork, data collection, data analysis, writing and editing		
Signature		Date	

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Article
Socio-Cultural Values of Ecosystem Services from Oak Forests in the Eastern Himalaya

Tshewang Dorji [1,2,*], Justin D. Brookes [1], José M. Facelli [1], Robin R. Sears [3,4], Tshewang Norbu [2], Kuenzang Dorji [2], Yog Raj Chhetri [2] and Himlal Baral [3]

[1] The School of Biological Sciences, University of Adelaide, South Australia 5005, Australia; justin.brookes@adelaide.edu.au (J.D.B.); jose.facelli@adelaide.edu.au (J.M.F.)

[2] Ugyen Wangchuck Institute for Conservation and Environmental Research, Thimphu, Bhutan; tshewang.dorji@uwice.gov.bt (T.D.); tnorbu@uwice.gov.bt (T.N.); kdorji@uwice.gov.bt (K.D.); chhetriy@gmail.com (Y.R.C.)

[3] Center for International Forestry Research (CIFOR), Bogor 16115, Indonesia; robin.sears@aya.yale.edu (R.R.S.); h.baral@cgiar.org (H.B.)

[4] Harvard Forest, 324 N Main St, Petersham, MA 01366, USA

* Correspondence: tshewang.dorji@adelaide.edu.au; Tel.: +61-449224611

Received: 27 February 2019; Accepted: 9 April 2019; Published: 15 April 2019

2.2 ABSTRACT

Identification and assessment of socio-cultural values of ecosystem services are increasingly important for the planning and management of forest resources. Key information necessary is how different forest user groups perceive and prioritize different ecosystem services based on their local setting. We assessed the socio-cultural values of ecosystem services of high-altitude oak forests in Western Bhutan using participatory approaches with two important forest users: local communities and forest experts. We found that these forests serve as a pool of 22 ecosystem services under four MEA categories of provisioning (9), regulating (8), supporting (2), and cultural (3) services. Freshwater was unanimously identified as the most valuable service, as well as the most vulnerable, by both the groups. The priorities of local communities inclined towards provisioning and cultural services due to their dependence on these services for their livelihood and wellbeing. Forest experts' priorities were more evenly spread over three categories of services: provisioning, regulating, and supporting services, reflecting their broader interest in resource management, biodiversity conservation, and climate change mitigation. Several regulating and supporting services were not easily identified by many villagers, suggesting that bridging the priorities of local interests with broader national forestry goals may require public partnerships and integrated decision-making about the entire suite of ecosystem services. Several management interventions proposed by the groups were presented for consideration by local users, scientists, and policymakers. For all ongoing and future ecosystem service assessments, we recommend the integration of socio-cultural values with biophysical and monetary assessments to fully value the benefits from the high-altitude oak forests.

Keywords: Bhutan Himalayas; socio-cultural values; mountain ecosystem services; *Quercus semecarpifolia*; oak forest; integrated decision-making

2.3 INTRODUCTION

Socio-cultural valuation approaches to assessing ecosystem services are increasingly recognized as an important tool for understanding the contribution of ecosystems to human well-being (Scholte et al. 2015; Iniesta-Arandia et al. 2014). Ecosystem service assessments that focus on biophysical quantification and monetary valuations help to identify and quantify the functional values of ecosystems (Baral et al. 2014; Chiabai et al. 2011; Kubiszewski et al. 2013); whereas assessments that focus on the importance that people place on ecosystem services for both material (e.g., food, water, timber) and non-material benefits (e.g., spiritual inspiration, sense of place, aesthetic values), provide information on the priorities and needs of local residents and other stakeholders (Viirret et al. 2019; Smith and Sullivan 2014; Raymond et al. 2009; McIntyre-Tamwoy 2004; van Oort et al. 2015). The socio-cultural values of ecosystems are associated with all categories of ecosystem services: provisioning, regulating, supporting, and cultural services (Scholte et al. 2015), and are thus also distinct from assessments of the cultural services of ecosystems, or the strictly non-material well-being provided by natural ecosystems, such as spiritual, aesthetic, and recreational values (Plieninger et al. 2013; Schnegg et al. 2014). While ecological and economic valuation can help to guide technically accurate ecosystem management decisions, understanding the importance that people place on ecosystem services, or the social value, is essential for making socially equitable management decisions (Iniesta-Arandia et al. 2014).

Recent advances in ecosystem service assessments using the social values perspective have been made in diverse ecosystems such as the sea (Viirret et al. 2019) and rivers (du Bray et al. 2019), and in management schemes such as community-based forestry (Paudyal et al. 2018b); agriculture (Smith and Sullivan 2014); and a combination of forests, shrublands, and agricultural ecosystems (Cáceres et al. 2015). These studies have integrated the perspectives, needs, and values of local stakeholders in meaningful ways to inform management decisions about those ecosystems.

Although human-ecosystem interactions are complex, there is evidence of a declining trend in ecosystem services worldwide (Millennium Ecosystem Assessment 2005; Shepherd et al. 2016). As quantitative information on many degraded ecosystems is limited or fails to address social values, there is a need for rapid qualitative assessments to help set priorities for conservation actions and to prevent further deterioration (Grantham et al. 2009; Baral et al. 2013). Social valuation through qualitative approaches enables researchers to gain a sense of the range and importance of the ecosystem services while also identifying threats that require immediate attention (Baral et al. 2013; Ananda and Herath 2009). Qualitative approaches to social valuation such as focus group discussions (O.Nyumba et al. 2018), participatory mapping, and stakeholder interviews involve the collection of information directly from stakeholders and attempt to link ecosystem services directly to human well-being, as defined by the stakeholders themselves (Dunford et al. 2018; Seppelt et al. 2011). The choice of method differs based on the objectives of the assessment, range of ecosystem services, and different stakeholder groups (Dunford et al. 2018; Harrison et al. 2018). Using a combination of social, ecological, and ecosystem models are most effective for environmental planning and management (Lin et al. 2017).

The decline in forest ecosystem services can be addressed through sustainable forest management, which itself has gradually evolved over time from being driven by purely economic goals—seeing the forest for timber production—to a broader perspective of the forest as a socio-ecological system, one that is inclusive of ecological, social, and cultural values (Sharma et al. 2010). Multidisciplinary approaches to forest management through the use of scientific and technical information in combination with historical information (Machar et al. 2016) or traditional and local knowledge (Boafo et al. 2015) are found to be highly effective. Furthermore, the involvement of local stakeholders, with attention to community values and local ecological knowledge, in local ecosystem management is increasingly being recognized at all levels of the decision-making process (Boafo et al. 2015; Vignola et al. 2009), and they are linked with positive conservation outcomes (Young et al. 2013).

The resilience of rural communities to adverse impacts of climate change and poverty can be enhanced through the formulation of conservation policies that are based on assessments of ecosystem services and that directly link to rural livelihood (Vignola et al. 2009). Recent research suggests that local communities' knowledge and perceptions of ecosystem services provide important insights into opportunities and challenges in ecosystem management (Zhang et al. 2015; Plieninger et al. 2014). Involving local communities helps to identify how various stakeholders value, perceive, and prioritize ecosystem services differently (Cáceres et al. 2015). It also enforces linkages between ecosystem managers and local users in mainstreaming ecosystem services and adaptation needs into policies (Vignola et al. 2009). The process of integrating priorities of local communities into management decisions is important for developing countries where the economy largely depends on ecosystem services and where these services particularly benefit poor people (Paudyal et al. 2018b). Assessment of ecosystem services and the valuation of benefits can bring local and international partnerships in conservation efforts through the payment for ecosystem service mechanisms (Paudyal et al. 2018a).

Assessing the biophysical character of ecosystem services in mountain ecosystems can be especially challenging due to the scarcity of data in some regions (Paudyal et al. 2015) and limited understanding of trade-offs and synergies among ecosystem services and uncertainties induced by climate change (Scholes 2016). The complexity of defining and classifying ecosystem services among disparate populations in mountain regions makes comparisons difficult. Several tools and approaches for a comprehensive assessment of mountain ecosystem services were highlighted for the Himalayan region (Baral et al. 2017). Recent studies conducted in Nepal have shown that information collected on social values and people's perception can be an important tool for ecosystem service assessment of mountain forests (Garrard et al. 2012; Paudyal et al. 2015; Paudyal et al. 2018b).

In the eastern Himalaya, natural forest ecosystems form an important component of rural livelihood, wellbeing, and economy. Bhutan's forests, covering 71 percent of the total geographical area, are home to rich biodiversity and comprise the largest contributor of ecosystem services (Kubiszewski et al. 2013). An initial assessment using the benefit transfer method has valued Bhutan's ecosystem services to be worth USD 15.5 billion year^{-1}, with many of the services providing benefits at the global scale (Kubiszewski et al. 2013). The country has a constitutional mandate to maintain at least 60 percent of its land under forest cover for all time and has pledged to remain carbon neutral. This strong commitment to nature conservation is deeply rooted within Bhutanese culture and the Buddhist value of coexistence with nature (Rinzin et al. 2009). The natural goods and services from Bhutan's forests are the source of material prosperity, health, and happiness, and they are strongly linked with the country's development philosophy of Gross National Happiness (Sears et al. 2017). In this richly forested nation, ecosystem service assessments are an increasingly important tool to value the natural capital and integrate ecosystem services in decision-making processes. This is due to the strong dependence on the forest ecosystems and their services by the country's vital economic sectors like hydropower, agriculture, and tourism (Norbu 2012).

In May 2017, Bhutan held the regional symposium on natural capital where key stakeholders, researchers, and policymakers met together to understand the value of natural capital and valuation methods. The country launched the first assessment report on ecosystem services from a single river basin and secured funds for a nationwide assessment of ecosystem services. Those assessments are focused on biophysical quantification and monetary valuation of ecosystem services (Kubiszewski et al. 2013; WWF 2017), giving limited attention to the social values of ecosystem services. Ours is part of a broad study in Bhutan on the social values associated with ecosystem services from local forests (Sears et al. 2018). Such assessments are important for Bhutan to balance forest utilization with the national goal of forest conservation to meet the constitutional mandate to maintain 60 percent of land under forest cover and in fulfilling the pledge to remain carbon neutral at all times.

Focusing on Himalayan oak forests in Western Bhutan (Singh et al. 1997), our study aims to identify the priorities and perceptions of ecosystem services, or the socio-cultural values, by two stakeholder groups: local communities and forest experts. Historically, local communities have been marginalized from national forest planning processes, while the forest experts—professional civil servants—have been tasked with implementing these plans. We suggest that each group should be carefully consulted to ensure that the opinions and priorities of user groups are taken on board for future forest management policies. Furthermore, we suggest that assessing social values of local ecosystems from both local community and manager standpoints will provide a basis to identify more equitable and representative options for the sustainable management of forests and other natural resources. To address this objective, we assess the socio-cultural values ascribed to the provisioning, regulating, supporting, and cultural services by two important forest user groups and compare their preferences and variations. Priority ecosystem services that are recognized by both stakeholder groups to be highly valuable and vulnerable were identified, and conservation measures proposed. This kind of study on how perceptions and priorities of ecosystem services differ among different user groups can help to inform decision-making for truly sustainable forest management that harmonizes forest utilization and ecological function.

2.4 MATERIALS AND METHODS

2.4.1 Study Area

Our study was conducted in western Bhutan, where oak forests dominate the elevational belt between 2400 m and 3100 m above sea level. We worked in seven villages adjacent to the oak-dominant forest, and where social conditions, household occupation, and forest use were similar (Figure 7). Villages were identified in consultation with local range offices and extension agents. Due to the small size of the villages, each village was considered as a study site for the purpose of this study. The livelihoods of local people in the study area are directly dependent on subsistence agriculture, forest products, and livestock farming (Table 1).

The mean annual temperature recorded by a weather station at one of the study sites was 8.5 °C, with a mean maximum temperature of 17.1 °C and mean minimum of −3 °C. The mean annual rainfall is 750 mm. Most of the rain falls in the months of June to September, with two to three snowfalls every year. The soils are rich in organic matter content, slightly acidic, and well-drained, varying from silty clay loam to sandy loam.

Though no formal assessment has been made to categorize these forests in western

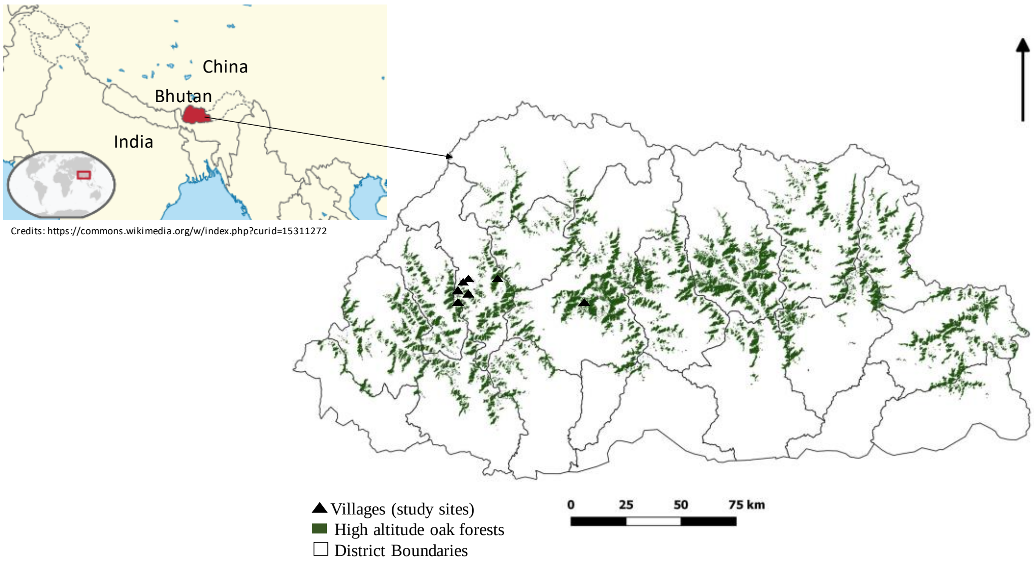

Figure 7. Map indicating the location of Bhutan in Asia (top left), and the location of study sites within Bhutan (shaded triangles). Black lines show district boundaries. Green shaded area indicates the distribution of oak forests.

Bhutan as old-growth, they are commonly referred to as such by Himalayan ecologists, foresters, and conservationists due to the large stature of trees, the structural complexity, and the high diversity (Tashi and Thinley 2008). The overstory of the forests at all sites is dominated by the evergreen oak *Quercus semecarpifolia*, mixed with conifers such as *Tsuga dumosa*, *Picea spinulosa*, *Abies densa* at higher elevations, and *Pinus wallichiana* at lower elevations. The middle layer is occupied by rhododendrons, maple, Himalayan birch, and Alnus trees. The understory is very diverse, with shrubs and herbaceous plants. These forests, along with the adjoining alpine meadows, have been historically grazed by herds of migratory and sedentary cattle.

The forests are part of the Bhutan government's reserve forest system, which provides access to residents for the utilization of forest resources, though under stringent rules. The forests are utilized under forest management plans which aim to minimize the significant change in species diversity while providing forest goods and services to local people (Moktan 2010). During the late 1990s, increased harvesting of the oak trees to meet the urban demand for fuelwood in cold winters led to a nationwide ban on both commercial and subsistence felling of oak trees in 2000 (Moktan 2014). However, the harvest of conifer and other broadleaf trees as timber and fuelwood from these forests continued as a single tree selection system that follows close-to-nature silviculture to minimize forest disturbance, retain old-growth characteristics, and promote tree diversity (Raymond et al. 2018). Most parts of the forest that are distant from the human settlements are intact. In Bhutan and in the study areas, in particular, conifers are preferred for timber, for the ease of working with it, while oak is preferred for fuelwood due to its high calorific value (Orwa et al. 2009).

The study site forms a part of the Himalayan oak forest belt, where the regeneration of oak species is reported to be inadequate (Shrestha 2003; Shrestha and Paudel 1996; Singh et al. 1997; Tashi and Thinley 2008). Because successful forest regeneration is key to sustainable forest management (Gottesman and Keeton 2017), restoration and conservation of these forests have become a top priority for ecologists, conservationists, and foresters. In the context of Bhutan, studies have linked the poor regeneration of oak to overgrazing by large herds of migratory and domestic animals (Tashi 2004; Dorji 2012).

2.4.2 Data Collection

To assess the socio-cultural values that are placed on these oak forests, we used participatory rural appraisal techniques to identify the ecosystem services from the forests and the values and priorities that people place on them. The methods consisted of a combination of household and expert interviews, focus group discussions (FGD), and preference point ratings (Figure 8). The stakeholders were split into two groups: 1. villagers consisting of farmers and village heads (hereafter referred to as the “local community”) who utilize the forest on a regular basis, and 2. forest experts, consisting of, local foresters and senior forestry officials of the forest department who deal with conservation ecology and the management of old-growth oak forests (hereafter referred to as “experts”).

From April to June 2017, a total of 84 households were interviewed using structured perception questionnaires to collect information on household dependence and individual perceptions of the availability trend in ecosystem services derived from the forests over the last 10 years. Sixty-one percent of the respondents were female. The average age of the participants was 53 (± 15 SD) and 43 (± 12 SD) for men and women, respectively. Prior to the start of the survey, our team organized group meetings and awareness programs with the communities to acquaint the residents with the terminology associated with ecosystem services. The participants were chosen based on village records of households. Through the household interview, we also identified six to ten key informants per village who were included in the focus group discussion (FGD).

Table 1. Description of study sites indicating the district, elevation, and coordinates of their location, and the main source of occupation of farmers.

Study Village	District	Altitude (m)	Coordinates	Main Source of Income
Shelling	Wangdue	2539	27°23′24.19″ N 89°58′41.26″ E	Agriculture (Potato), forests, livestock
Jadingkha	Thimphu	3036	27°27′02.61″ N 89°30′18.69″ E	Forests, agriculture, livestock
Shari	Thimphu	2681	27°27′28.98″ N 89°31′32.87″ E	Agriculture, forests, livestock
Nubri	Thimphu	2776	27°27′37.41″ N 89°31′15.38″ E	Agriculture, forests, livestock
Chimithankha	Thimphu	2618	27°26′36.30″ N 89°31′37.51″ E	Agriculture, forests, livestock
Gemina	Thimphu	2459	27°25′36.41″ N 89°32′58.31″ E	Business, livestock, agriculture, forests
Yusipang	Thimphu	2830	27°28′09.15″ N 89°42′04.12″ E	Agriculture, forests, livestock

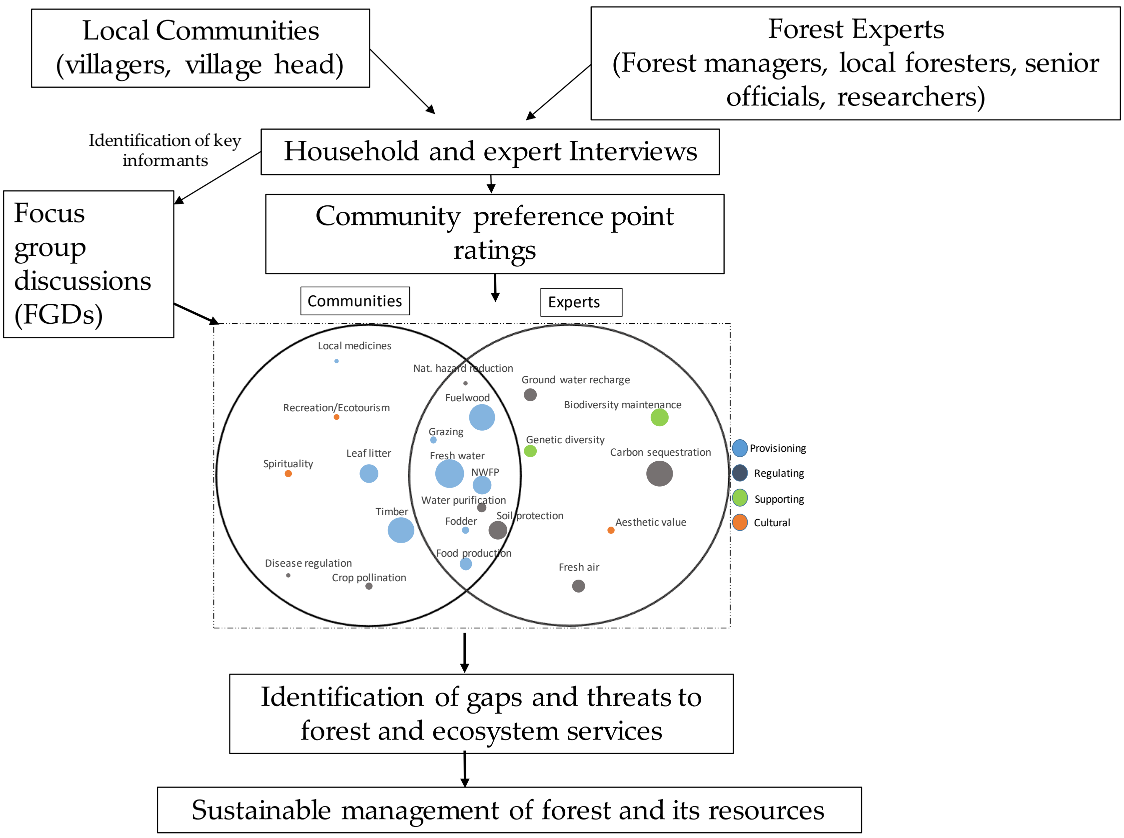

Figure 8. Methodological framework for the assessment of socio-cultural values of ecosystem services from the oak forests and pathway to forest management policies.

FGDs were conducted in each village to collect information on their priorities of ecosystem services and on their perceptions of the availability of and threats to them. We also solicited their ideas for management interventions based on their traditional knowledge. Due to the large number of FGD participants in the two villages, we split these groups into further smaller groups. Thus, a total of nine FGDs were carried out in the seven villages. Group members were from middle-aged to elderly men and women with a good knowledge of past and present interventions in oak forests. Prior to the conduct of FGDs, and based on a narrative analysis of interviews combined with a study of the four pillars of Gross National Happiness, we defined a set of shared social values that represent community aspirations, which we call "community values". These are socio-economic development, general well-being, environmental conservation, coexistence, spiritual sustenance, and cultural vitality. We then used the FGD to identify links between the local forest ecosystem services and these community values (Table 2).

Table 2**.** Community values associated with ecosystem services in the study area.

Community Values	**Link to Ecosystem Services**
Socio-economic	Use of forest to obtain timber, fuelwood, food, NWFPs for subsistence and income
Well-being	Services derived from forests important for health, life sustenance and happiness such as fresh air, disease regulation, aesthetic values and spirituality
Environmental conservation	Living in a healthy environment, balancing between conservation and economic development, e.g., forest protection for biodiversity, carbon sequestration, genetic diversity, homestead plantations and agroforestry.
Coexistence	Forest provides an avenue to appreciate Buddhist values of human beings living in harmony with nature, through awareness, kindness and wisdom, e.g., habitat for wild plants and animals.
Spiritual sustenance	Forest acts as an incubator of spiritual well-being and contentment by harbouring local deities and important religious sites.
Cultural vitality	Forests offer opportunities to display rich culture which can be promoted and passed on to future generations, e.g., cultural values, ecotourism, spiritual sustenance.

For the expert group, we interviewed eleven forestry professionals in total (two female, nine male), representing local forest office and senior forest officials from the forest department (5), researchers (5), and an experienced forest ecologist who had previously worked with the forest department (1). Because of our focused interest on the perception of ecosystem services from the oak forests, and to avoid confusion over other forest types, we limited our interviews to only those experts who were educated and experienced with this forest type. Each had high qualifications (ranging from a bachelor's degree in forestry to a PhD) and at least 10–15 years of work experience. Open-ended interviews spanning their opinions on priority ecosystem services, availability trends, drivers of change, and management recommendations were conducted. The average age of the experts was 42 (± 6 SD).

These surveys were carried out by a multidisciplinary team comprising foresters, researchers, and extension agents as a part of CIFOR's Sloping Lands in Transition (SLANT) research program.

Prioritization of ecosystem services was assessed through questions based on which service respondents valued the most. From the list of ecosystem services identified, each FGD listed the five most important ecosystem services. The ecosystem services were then ranked (from one to five in decreasing order of preference) according to their importance to the community (Cáceres et al. 2015). Participatory resource mapping using charts was employed to encourage equal participation by all the members within the group. The group facilitator also ensured that all participants within the group contributed equally in the discussion, and the information collected represented the views of the whole group rather than only one or two spokespeople (van Oort et al. 2015). All ecosystem services were grouped into the four categories of provisioning, regulating, supporting, and cultural services following the Millennium Ecosystem Assessment framework (Millennium Ecosystem Assessment 2005).

In a subsequent activity, 49 participants from local communities and the 11 experts were asked to distribute 100 preference points across different ecosystem services based on their priority. A 'community priority index' was calculated for each ecosystem service based on weighted scores (Paudyal et al. 2018b; Salihu et al. 2015). Average priority ratings were calculated for each ecosystem service within each stakeholder group. Similarities and variations in the preferences of ecosystem services were determined. Finally, based on their priorities and resource availability trend, both the groups identified potential threats and key conservation areas anticipated to be relevant to forest managers, local stakeholders, and policy makers.

For a comparison of priority ratings between the two groups, we used the lmer function in the lme4 package (Bates et al. 2015) to conduct mixed effects model analysis (Duursma and Powell 2016) of the relationship between stakeholder groups and their priority values ascribed to ecosystem services. We entered the group (local communities and experts) and gender of respondents as fixed factors in the model. Respondents nested within location (villages and offices) and their age were the random factors in the model. Visual observation of residual plots did not show any obvious deviations from homoscedasticity or normality. The *p*-values and test of main effects and interactions were computed by Wald Chi-Squared tests using the Anova function in the car package (Fox and Weisberg 2011; Duursma and Powell 2016). Pairwise group comparisons were done using Tukey's HSD test. All analyses were conducted using R software (CoreTeam 2018; RStudio Team 2016).

2.5 RESULTS

2.5.1 Perception of Ecosystem Services

Twenty-two ecosystem services were identified in this study and categorized based on the MEA categories of provisioning (9), regulating (8), supporting (2), and cultural (3) services (Table 3). Local communities easily identified ecosystem services related to provisioning and cultural services. The services, such as timber, water, and spiritual sites, are directly seen and highly related to local culture and rural livelihood. Mushrooms were the most extensively utilized resource from the forest. Overall, local communities listed 32 different types of mushrooms collected for household consumption and sale. Regulating services were less known to many of the local communities, as these services provided indirect benefits that were difficult to visualize. For example, only a few farmers who had some formal education knew about ground water recharge, water purification, carbon sequestration, disease regulation, crop pollination, maintenance of genetic diversity, and soil protection.

After the group trainings employed to promote an understanding of the technical definitions of ecosystem services and an awareness of the linkages and indirect benefits, many villagers were better able to identify specific services and their local importance. For example, in a village primarily dependent on commercial vegetable farming, participants recognized the importance of adjoining forests for harboring pollinators in increasing their crop production. Similarly, farmers living close to a forest management unit associated the drying of perennial springs in the area to forest logging activities and disturbances of ground water recharge. Villagers pointed out that ecosystem services were less recognized and mostly taken for granted because of the assumption that these services are provided free and forever by nature. They recognized these rich resources as an outcome of strong conservation efforts of the past, and thought that their duty was to do the same for future generations. Overall, both stakeholder groups favored provisioning services over the other three categories, primarily due to their direct link to the sustenance of rural livelihood.

Table 3. Ecosystem services identified in the study, with description, indicators, and associated community values. Ecosystem services based on the MEA ecosystem services group.

ES Category	ES Identified Locally	Description Based on Perceived Importance	Indicator of ES	Associated Community Values
Provisioning services	Freshwater	Water for domestic, agriculture, prayer wheels and hydropower	Number and volume of water bodies	Wellbeing, Socio-economic
	Timber	Timber stock at harvestable age	Harvestable trees ha^{-1}	Socio-economic, Wellbeing
	Fuelwood	Fuelwood obtained from forests	Number and volume of fuel wood	Socio-economic, Wellbeing
	Food	Provision berries, wild fruits, mushrooms, wild vegetables	Amount of food materials	Socio-economic
	Leaf litter	Leaf litter for cattle bedding and farm manure	Amount of leaf litter collected	Socio-economic
	Grazing	Area available in the forests for grazing	Number of grazing animals	Socio-economic
	Fodder	Forage production potentials of forests for livestock	Number of fodder species	Socio-economic
	Local medicines	Variety of plant and fungal species with biomedical value	No. of species and harvestable amount	Wellbeing
	High-value NWFP	Plants and animals with high bioprospecting potential	No. of species and production potential	Socio-economic
Regulating services	Fresh air regulation	Trees provide fresh oxygen and absorb dust particles from atmosphere	Total leaf area; amount of pollutant in air	Wellbeing
	Carbon sequestration	Atmospheric carbon captured by forests and stored as carbon in their biomass	Forest cover, wood biomass per ha	Environmental conservation
	Groundwater recharge	Good forest with vegetative cover regulates runoff and retains water in the soil	Ground water recharge rate (water availability throughout the year)	Environmental conservation

	Natural hazard regulation	Forests act as a natural buffer, protection from winds, landslides and other disasters	No. of hazard incidences per year	Wellbeing
	Water purification	Pure water running in streams from the base of forests	Quality and quantity of clean water	Wellbeing, Environmental conservation
	Disease regulation	Reduced diseases by regulating fresh air and water purification	Number of people affected by water and air borne diseases	Wellbeing
	Crop pollination	Increased production of crops from population of bees and other pollinators	Number of pollinator species	Socio-economic, Environmental conservation
	Soil protection	Forest vegetative cover prevents soil erosion and improve soil fertility through nutrient cycling.	Incidences of landslides, soil erosion or degradation	Environmental conservation
Habitat/supporting services	Biodiversity/habitat	Home to a diverse plants and wild animals	Increasing/decreasing wild flora and fauna	Coexistence, environmental conservation
	Genetic diversity	Forests conserve the genetic diversity for future generations	Appearance of new plants and animals; genetic diversity of populations	Coexistence, environmental conservation
Cultural services	Spiritual and religious values	Forest harbours religious and spiritual sites (temples and caves, sacred sites for local deities) important for wellbeing	No. of locations; no. of people visiting these locations	Spiritual sustenance, Wellbeing,
	Aesthetic values	Forests offers scenic beauty to the landscape for enjoyment by local residents and outsiders	No. of visitors appreciating the visual quality of the landscape	Wellbeing, Socio-economic
	Recreation and ecotourism	Forests used for recreation and ecotourism purposes	No. of recreation sites; no. of visitors	Socio-economic, Wellbeing

2.5.2 Prioritization of Ecosystem Services

Interviews indicated that fresh water from the forest was the most highly valued ecosystem service by both local communities and experts. FGDs further confirmed this by ranking water as the top priority ecosystem service in all group discussions (Table 4).

Water in the study areas was mainly used for household consumption and agricultural purposes. Communities also highlighted the importance of water for rotating the prayer wheels (prayer mills) set over streams and water’s strong association with spiritual wellbeing and contentment of the villagers. In the past, flowing water was used to run traditional water mills for grinding grains, but these have been replaced by electric mills. The high priority of water indicated by experts could be driven by increasing water scarcity in the Himalayas and due to economic values of rivers.

Table 4**.** Village wise ranking of ecosystem services (top five) based on which natural asset the focus group valued the most in their community. For the expert group, preference points were used for prioritization. Letter in brackets indicates the MEA ecosystem service categories (P provisioning, R regulating, S supporting, C cultural).

Location	**Ecosystem Services Ranking Based on Focus Group Discussions**				
	First	**Second**	**Third**	**Fourth**	**Fifth**
Sheling	Freshwater (P)	Fuelwood (P)	Spiritual (C)	Litter (P)	Natural hazard regulation (R)
Jedingkha	Freshwater (P)	Fuelwood (P)	Timber (P)	NWFP (P)	Grazing (P)
Shari	Freshwater (P)	Groundwater recharge (R)	Fuelwood (P)	NWFP (P)	Timber (P)
Nubri	Freshwater (P)	Timber (P)	Fuelwood (P)	Fodder (P)	Fresh Air regulation (R)
Chimithankha	Freshwater (P)	Fuelwood (P)	Timber (P)	Grazing (P)	NWFP (P)
Gemina	Freshwater (P)	Fresh air regulation (R)	Fuelwood (P)	Timber (P)	Spiritual (C)
Yusipang	Freshwater (P)	Timber (P)	Fuelwood (P)	Biodiversit y (S)	NWFP (P)
Experts	Fresh water (P)	Carbon sequestration (R)	Biodiversity (S)	Fuelwood (P)	Groundwater recharge (R)

Our results showed key areas of agreement between the two groups of stakeholders, as well as variations in the preferences of priority ecosystem services. Common preferences for provisioning ecosystem services were water, fuelwood, NWFP, food, and fodder production; regulating services were fresh air regulation, soil protection, water purification, and natural hazard reduction (Figure 9). In general, local communities' preferences were more centered towards provisioning and cultural services that were directly obtained from the forests and linked to their daily lives. On the other hand, expert's priority ecosystem services were more or less spread over the provisioning, regulating, and supporting ecosystem services, with a strong emphasis on carbon sequestration and biodiversity/habitat maintenance services.

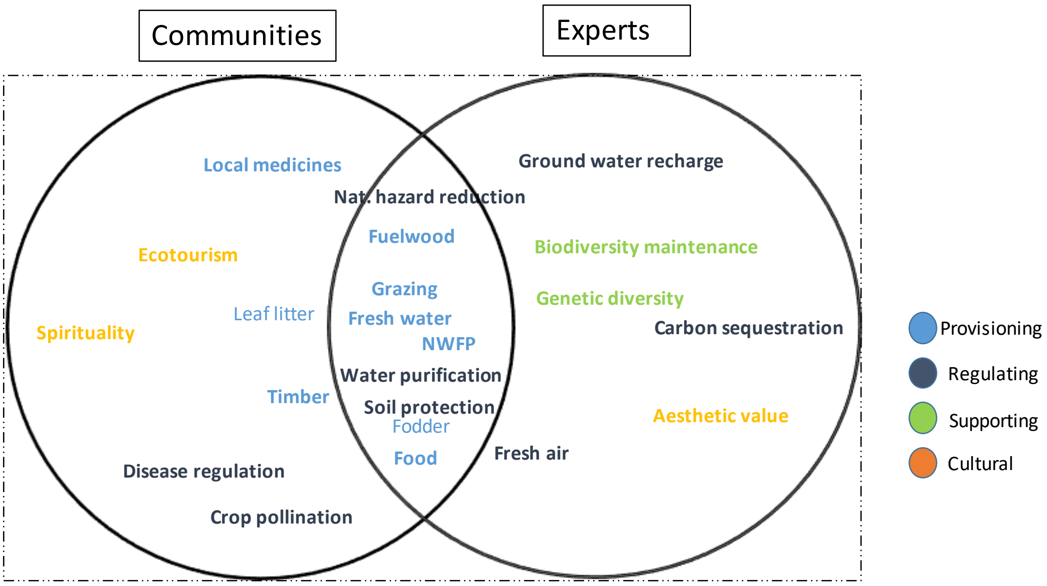

Figure 9**.** Priority ecosystem services assigned by the two forest user groups; local communities and experts indicating common preferences and variations. Colour of the text represents the ecosystem service category.

Results from the community priority index showed strong variations in the scores attributed to various ecosystem services by the stakeholder groups (Figure 10). Although both the groups prioritized fresh water as the most important ecosystem service, the relative importance of water perceived by local communities (30.4 ± 1.4; mean ± SE) was significantly higher ($p < 0.05$) compared to the experts (21.7 ± 3.9; mean ± SE). Similar trends were observed for timber and spiritual services from the forests. In general, local communities' priorities for provisioning services (73 ± 2.4) were significantly higher ($\chi^2 = 9.7$, df = 1, $p < 0.01$) than experts (43.9 ± 8.2). On the other hand, the regulating ($\chi^2 = 5.2$, df = 1, $p < 0.05$) and supporting services ($\chi^2 = 27.4$, df = 1, $p < 0.001$) from the forests were significantly higher for the expert's group (36.8 ± 5.1 & 14.2 ± 2.4) compared to local communities (21.4 ± 2.2 & 1.3 ± 1.0). There was not enough evidence of significant difference ($\chi^2 = 0.44$, df = 1, $p > 0.05$) in the scores attributed to cultural services between the groups. No significant differences were found for the main effects of age, gender, and gender/group interactions in the scores assigned to the four ecosystem categories ($p > 0.05$).

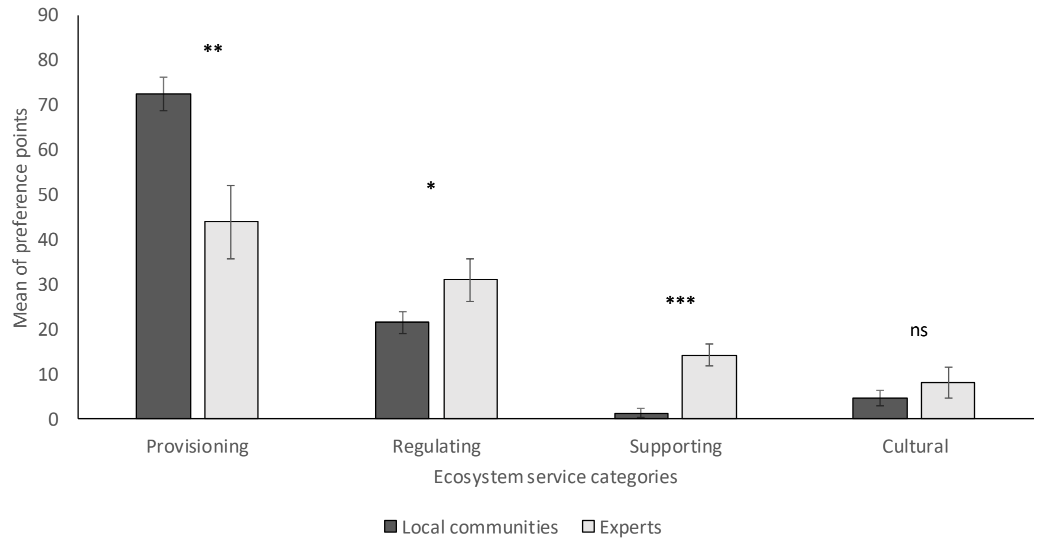

Figure 10**.** Priorities of rural community vs. experts across the four broad categories of ecosystem services based on preference points (total of 100 points) assigned by the participants of two groups. Labels indicate results of the mixed effects model and Tukey's HSD test, *** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$, ns = not significant. Bar represents standard error of the mean.

2.5.3 Perception of Trends in the Availability of Ecosystem Services, Threats, and Management Interventions

Both local communities and experts perceived a declining trend in several ecosystem services, especially freshwater, timber, firewood, and high-value NWFPs, over the last ten years (Figure 11). The concerns for declining trends were greater for experts than for community members, with an additional emphasis on biodiversity, groundwater, and carbon. Both groups believed that these trends were due to high pressure resulting in the overexploitation of those forest resources. In the focus group discussions, many farmers pointed out that in the past, fuelwood, fodder, and timber were readily available in the nearby forests, but that now they had to travel 20 km to 30 km inside the forests to fetch these goods. Forest grazing and food production services from the forests were believed to have remained constant; however, they indicated that utilization of these services has declined over the years due to changing lifestyle patterns as a result of socio-economic development. For example, traditional livestock farming, which relies on open forest grazing, is being replaced by improved breed farming, which requires stall feeding at home. Similarly, modern agricultural farming and readily available food options in the market have reduced local people's dependence on forest fruits and berries.

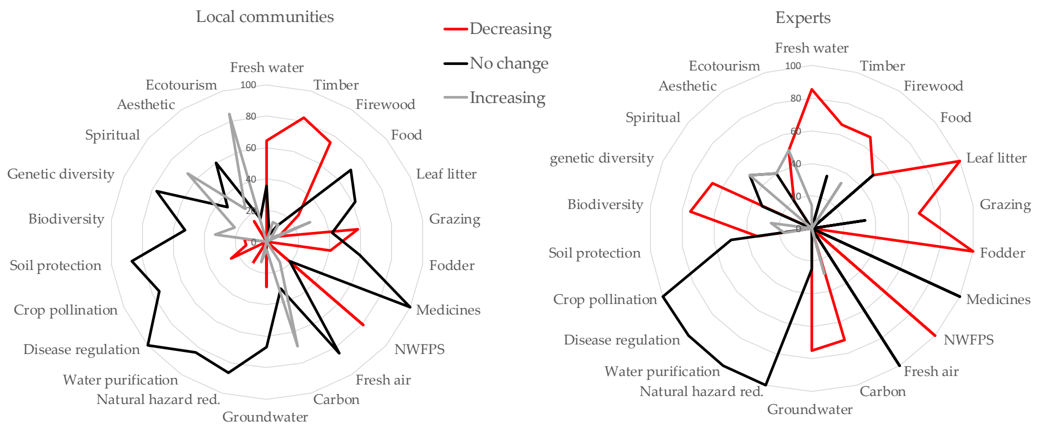

Figure 11**.** Perceptions of rural communities and experts on the change in availability of ecosystem services over the past ten years. Scale represents the number of respondents expressed as a percentage.

Local communities' perceptions of regulating, supporting, and cultural services from the forests were more positive than for provisioning services, and they felt the availability of these has remained constant or improved over time. Local communities think that these values will only grow as forest cover increases as a result of strong conservation commitments and gradual colonization of open pastures by vegetation with decreasing grazing pressure. They indirectly related the increasing forest cover to more tree growth and increased carbon sequestration capacity of old-growth forests. Experts had mixed opinions about carbon sequestration services. Only 29 percent of expert views were in line with the perception of local communities; they attributed the increase in forest cover to the decreasing use of oak as a result of the nationwide ban of oak tree felling to prevent overharvesting in the early 2000 s. However, 71 percent of the experts felt that carbon sequestration potentials were declining due to the extraction of other timber and fuelwood species (e.g., Conifers), fueled by road access and an increasing population. They also highlighted the failing reforestation programs and poor natural regeneration problems in these forests as other reasons for the decline.

There were synergies among the groups in terms of their perception of increasing trends in the access to forests for their spiritual and aesthetic values. Participants attributed the increase to economic progress like road connectivity, more disposable income, and the increasing number of people looking for tranquility and spiritual wellbeing. The focus group discussions strongly supported this view. Eight out of nine focus groups suggested community-based ecotourism as a future enterprise with high potential in their communities. They associated this to the increasing number of local and international tourists seeking natural beauty and tranquility. This is further complemented by the intact forest cover in their locality with rich cultural and spiritual values.

Local communities perceived that the threats to the sustainability of provisioning, regulating, and supporting services from the old-growth oak forests were the result of the overexploitation of resources driven by population growth and economic development, as well as changing conditions associated with climate change (Table 5). Similarly, the expert group emphasized high pressure on forest resources due to the growth in populations of both human and cattle over the years. Overharvesting of resources and overgrazing by cattle were identified as the main threats for the rapid decline in the natural resources and failure of regeneration by oak. While local communities agreed on the overharvesting of resources, they were less aware of the regeneration problem in their local forests. Both groups stressed the importance of reforestation programs to protect their water and other resources and increase carbon sequestration services.

The local communities raised the need for training and education on appropriate harvesting guidelines in relation to the provisioning services from the forests to alleviate overexploitation by both villagers and outsiders in these open-access forest areas. While strict rules are in place for the harvest of forest resources in these forests, they are not always followed or enforced. Some felt that the rules themselves needed reviewing since allowable harvest levels might be too high for the ecosystem to support the practice. In addition, the groups felt that improving the local governance of these local forests through the empowerment of local communities and establishment of custodianship could be a potential management intervention to promote sustainable use and safeguard the resources.

Table 5. Summary of key findings from FGD on priority ecosystem services, perceived threats, and potential management options indicated by local communities and experts.

Ecosystem Service	Perceived Threats/Opportunities	Potential Management Interventions	
		Communities	Experts
Freshwater	Forest disturbance; high pressure on resource due to increasing population and climate change	Protection of water source through revegetation; Prohibition of disturbance at the water source.	Water budgeting; identification of critical watersheds and proper management; Prohibition of forest harvesting in upper catchment forests.
Timber/Fuel wood	Over-exploitation both local and outside residents; longer tree rotation period	Empowering local communities to safeguard their resources through community forestry	Transfer of forest ownership over to local communities; Assist regeneration of forests through plantation programs.
High-value NWFP	Overexploitation of resources by local and outside residents	Education on proper harvesting techniques, ownership to community	Ownership should be given to local communities; Domestication of high-value medicinal species in the farmers field.
Carbon sequestration	Reforestation failures; over-extraction of wood resources	Community-based plantation programs	Detail study on regeneration ecology of oak forests; Replenish felled trees with plantations.
Maintenance of genetic diversity	Keystone species are not regenerating in the forests; Prey-predator balance	Addressing human-wildlife conflicts as the population of prey species; boar, deer have increased.	Detailed study on the ecology of oak forests and wildlife population dynamics. Restoration of oak forest through artificial plantations need to be adopted to address regeneration failures.
Spiritual /Ecotourism	Increased connectivity, Better income; More people seeking tranquillity	Creation of awareness campaigns and advertisements	Protecting spiritual/religious sites from forest logging; providing adequate amenities (e.g., toilets, clean water, guest house) to encourage tourism.

2.5.4 Community Values and Their Linkage to Ecosystem Services

The community values most strongly associated with these oak forests by community members were socio-economic, well-being, and environmental conservation. Socio-economic and environmental conservation values were each associated with seven ecosystem services from these forests, followed by well-being values with six (Figure 12). Cultural values, such as coexistence and spiritual and cultural vitality, were associated with fewer ecosystem services. Participants of the

FGDs felt that their community and the old-growth forests could be a special focus for environmental- and cultural-based ecotourism. They attributed this to their location close to the alpine peaks used by tourists for trekking and the presence of several cultural and religious spots of national significance. They expressed the need for an awareness of such services by the general public through newspaper advertisements and TV broadcasts. Both groups in our study strongly felt that local governance and community-based management of these old-growth forests through community forestry can ensure sustainability. Both groups expressed a need for research on forest ecology and restoration and community-based plantation programs. The stakeholder groups identified several potential areas related to forest management and conservation that required active local and government partnerships. Public consultations, awareness campaigns, and integrated decision-making on all aspects of natural goods and services were proposed during the focus group discussions (Figure 12).

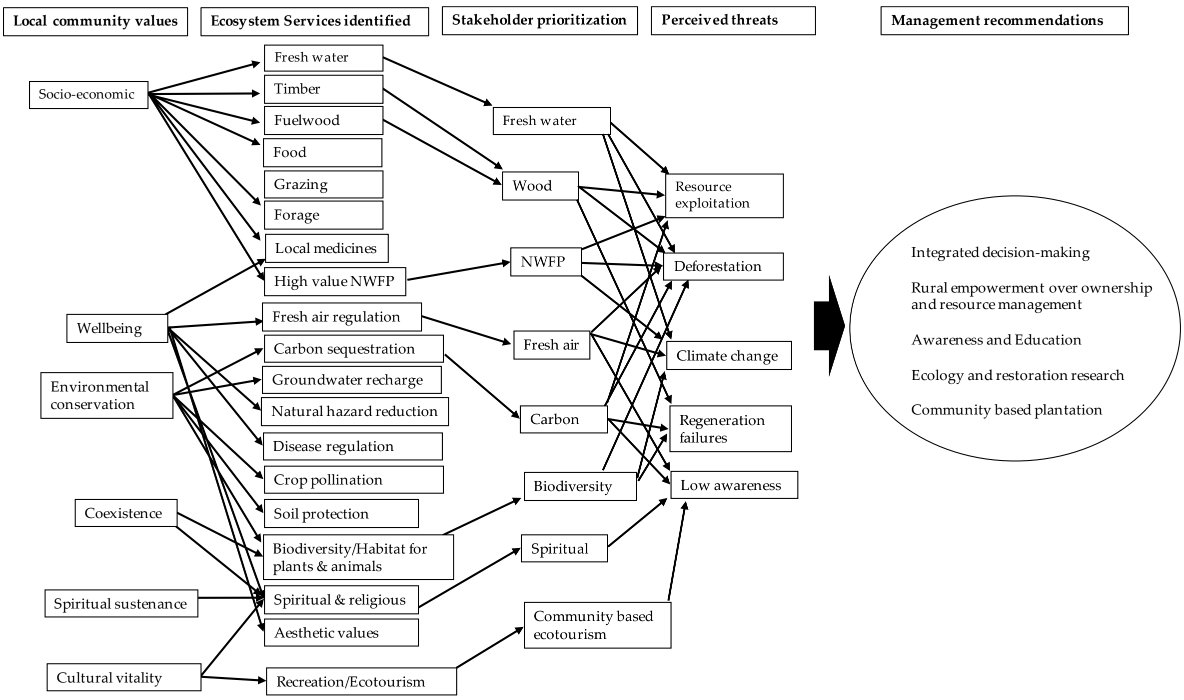

Figure 12**.** Framework on bridging local community priorities with national interests through the incorporation of community values and perceptions of ecosystem services.

2.6 DISCUSSION

Our social valuation revealed a diverse range of forest ecosystem services provided by oak forests in western Bhutan and their relative importance to two distinct forest users groups: local communities and forest experts. By pairing the concept of community values to the perceptions of the full suite of ecosystem services, our study provides a social valuation of these forests centered on the perceptions and needs of two stakeholder groups. In-depth understanding of how different forest users perceive and value ecosystem services is important for landscape-level decision-making (Cáceres et al. 2015). This approach further nurtures partnerships and trust among stakeholders and develops a sense of local ownership over local resources, which is important for wider sustainable forest management goals.

In line with other research in the region (Paudyal et al. 2018b; Sears et al. 2018), our study confirms that community members understand and prioritize provisioning and cultural ecosystem services over regulating and supporting services. This is probably due to their historic and cultural interactions with these services for livelihood sustenance (Cáceres et al. 2015). Cultural services identified in this study represented an important component of rural culture, spiritual contentment, and happiness consistent with findings from other studies (Plieninger et al. 2013). Less tangible regulating and supporting ecosystem service categories, which are difficult to see or measure, were not emphasized by local communities. They were, however, recognized for their importance to local wellbeing. All of these ecosystem services were considered by residents and managers alike to be an important reward of a well-maintained forest ecosystem, a perception that is consistent with positive attitudes by local communities on the regulating services from oak forests of Nepal (Naudiyal and Schmerbeck 2017). The most important reward is the conservation of soil and water, also highlighted by other studies (Sheikh and Kumar 2010; Singh and Pande 1989).

The expert group was aware of the full suite of ecosystem services, and their preferences were distributed more equally across the four service categories. This could be due to their broader interest in forest resource management, biodiversity conservation, and climate change mitigation. Members of this group were educated in all aspects of forest ecology and regularly participated in national and international seminars and environmental awareness programs. Such forums help make it easier to understand the less tangible benefits of regulating and supporting ecosystem services. For example, it was reported elsewhere that regular environmental awareness programs were even more influential than the formal education levels in valuing the perceived importance of regulating services to the communities (Zoderer et al. 2016). Given that residents of rural communities in the Himalayas are not highly educated, special consideration should be given to creating regular environmental awareness sessions focusing on all services, with emphasis on regulating and supporting services, so that communities can fully value the importance of their local forests (Sears et al. 2018).

Priorities for certain ecosystem services among community members were strongly influenced by the individual’s environment where they live and their socio-economic background, which was found in similar studies from Nepal (Garrard et al. 2012; Paudyal et al. 2015). For example, all the FGDs prioritized fuelwood in the top five ecosystem services, which, according to the participants, was due to their high-altitude location and strong dependence on fuelwood for housewarming during most of the year (Moktan 2014). Similarly, villagers from Gemina and Nubri included fresh air regulation services in the top five, reflecting their location close to an industrial estate and realizing the value of forests in fresh air regulation to their wellbeing. In general, all the local communities’ priorities for water, timber, fuelwood, leaf litter, NWFP, and forest grazing indicated their strong dependence on forest resources for their rural livelihood.

The influence of demographic factors such as cultural identity, gender, and income levels of participants on the variations in their socio-cultural values are described in studies conducted outside Bhutan (Zoderer et al. 2016; Paudyal et al. 2015). While we did not explicitly explore these areas, our results suggest that gender and age did not influence the socio-cultural values assigned to ecosystem services.

The level of socio-economic development in the village does seem to affect the dependence on forest resources, if not the values people put on them. We found a lower dependence on several ecosystem services by the local communities as a result of socio-economic development and lifestyle changes. For instance, villager dependence on provisioning services, such as local medicines, wild vegetables, wild fruits, and berries, was reported to have significantly declined over the years, even though their availability in the forest was perceived to have remained constant. Local medicinal plants, which were widely used in the past, are now hardly used due to improved access to modern medicines. Similarly, food and fruits are rarely collected due to the availability of various food options in farms and markets. It is thus predicted that with economic growth and the availability of alternative service options, the dependence on and traditional ecological knowledge about these products will decline over time. Documenting indigenous knowledge and practices should be given priority for transmission to future generations.

Stakeholder priorities of ecosystem services were strongly governed by their perceived trend in the availability of ecosystem services. Ecosystem services that were highly important and perceived to be vulnerable, such as water, fuelwood, and NWFP, received high priority ratings from both the communities and experts. These critical, and vulnerable, ecosystem services are highly important for the wellbeing of local communities and should be a priority conservation area for inclusion in forest management. Experts also gave high scores to carbon sequestration and biodiversity values because they believed that old-growth forests are deteriorating due to human disturbances and poor regeneration. This perception differed from that of local communities, where residents perceived that carbon sequestration services of forest had increased over the years as a result of increasing forest cover brought about by strong conservation programs and the colonization of open pastures by new forests. Local perceptions about increasing forest cover in the region are consistent with remote sensing studies that related forest cover increase to land abandonment and gradual replacement by pine forests (Bruggeman et al. 2016; Gilani et al. 2015). These results highlight a need for studies on the carbon storage dynamics related to the forest colonization of abandoned pastures and fields in this landscape.

While local people perceived an overall expansion of forest cover, they were less aware of the regeneration failures by the primary oak species in their forest. This could be due to the presence of very large oak trees in the forest, which masks the lack of young ones to replace them in the future. The regeneration failure is a considerable threat to the future of these oak forests and the important ecosystem services they provide. Their decline could directly impact the wellbeing of temperate farmers (Bisht et al. 2012; Bisht and Kuniyal 2013; Covey et al. 2015; Dorji 2012; Naudiyal and Schmerbeck 2017; Shrestha 2003; Singh et al. 1997). We call for a detailed study on the ecology and restoration of the oak forests in Bhutan and the transmission of ecological knowledge to local communities. As ecosystem restoration is becoming a global priority, and one in which the engagement of experts and resource users will be increasingly involved (Aronson and Alexander 2013), we suggest that the active participation of local stakeholders in all aspects of decision-making and forest management would be beneficial. We propose the organization of environmental awareness programs and community-based, small-scale restoration projects as initial steps to protect and restore these important forests.

Improving local forest governance through empowering local communities in all aspects of management and ownership with strong technical support from Bhutan's forestry department could be a promising way forward for the sustainable management of these forests and resources. Promising experiences show that common-pool resource management mechanisms and institutions, such as community forestry, are effective in regulating and managing valuable natural resources by communities in Bhutan (Brooks 2010; Buffum et al. 2010; Buffum 2012). Similarly, the ecotourism sector represents considerable potential for benefit-sharing from ecosystem services, which can ensure environmental sustainability and socio-economic development in Bhutan (Norbu 2012). For a start, forest managers and extension workers can capitalize on promoting the natural, cultural, and spiritual values through community-based ecotourism. The high biodiversity of these rich forests coupled with their spiritual importance and cultural diversity present a major opportunity for appreciating the secret value of these forests. We support the current government initiative of promoting community forestry in which local communities serve as the primary custodians of the forests. This local empowerment contributes to long-term sustainability of the forests.

2.7 CONCLUSIONS

Our study presented an important social perspective to identify and assess ecosystem services from one Himalayan forest type: the high-altitude oak forest. Contrasting with many valuation studies that focus on biophysical models and monetary aspects of ecosystem services, we carried out an assessment focused on the socio-cultural values of these forests and found strong linkages between these forests and human well-being. We used participatory rural appraisal tools, including household and expert interviews, participatory resource mapping, focus group discussions, and preference point ratings, in seven villages of Western Bhutan, to study preferences and bridge differences in the perspectives ascribed by two stakeholder groups to ecosystem services, local communities and forest experts.

Our results indicated that for both local communities and experts, fresh water, followed by timber and fuelwood, were highly valuable, as well as highly vulnerable, ecosystem services in the study region. We suggest that these ecosystem services should receive the highest priority in forest management and planning. Based on the comparison of four broad categories of ecosystem services, we demonstrated that both stakeholder groups preferred provisioning services over others. However, the preferences and understanding of local communities were remarkably higher than experts from the perspective of provisioning services but lower from regulating and supporting services, indicating the need for investments in awareness programs and environmental education so that local communities fully value the wide range of forest ecosystem services.

Our qualitative approach of identifying the perception of ecosystem services from local people has high potential to identify areas of socio-ecological conflict and bridge the local perspectives within the framework of wider national goals for long-term sustainable management policies in the Himalayan region. Threats to these forests identified in this study included the overharvest of forest resources, overgrazing by cattle, and climate change. These mainly affected the availability of resources to community members and the natural regeneration of oaks. To address these threats, both community members and experts emphasized the need for improved local governance and community-based management of these forests, research on forest ecology, and restoration to support sustainable forest management decisions. They also identified the need for local awareness campaigns about forest ecosystem services and sustainable forest management, as well as integrated decision-making with relation to all ecosystem services. These conditions would lead to more socially equitable and environmentally sustainable management decisions and forest use.

We propose that the social valuation of ecosystem services should be included in all ecosystem service assessments as a critical complement to biophysical quantification and monetary valuations. In this way, a more robust and comprehensive assessment can be achieved, one that reflects both the diverse values of forests and the full needs of forest stakeholder groups.

2.8 AUTHOR CONTRIBUTIONS

Conceptualization, T.D., J.D.B., J.M.F., R.R.S., and H.B.; methodology, T.D., J.D.B., J.M.F., R.R.S., and H.B.; software, T.D.; validation, T.D., J.D.B., and H.B.; formal analysis, T.D.; investigation, T.D., J.D.B., J.M.F., R.R.S., H.B., T.N., K.D., and Y.R.C.; resources, T.D., H.B., J.B., J.F., and T.N.; data curation, T.D., T.N., K.D., and Y.C.; writing—original draft preparation, T.D., J.D.B., J.M.F., R.R.S., and H.B.; writing—review and editing, T.D., J.D.B., J.M.F., R.R.S., T.N., K.D., Y.R.C., and H.B.; visualization, T.D., J.D.B., J.M.F., R.R.S., and H.B.; supervision, J.D.B., J.M.F., and H.B.; project administration, T.D., J.D.B., J.M.F., R.R.S., T.N., K.D., Y.R.C., and H.B.; funding acquisition, H.B., T.D., J.D.B., R.R.S., T.N., K.D., and Y.R.C.

2.9 FUNDING

This research and APC were funded by the Republic of Austria in a grant to CIFOR. The study forms a part of the PhD thesis of the first author. The University of Adelaide, Royal Government of Bhutan, and RTP scholarships of the Australian Commonwealth Government are acknowledged for the funding support provided to the first author.

2.10 ACKNOWLEDGMENTS

The study is part of a collaborative research project between the Ugyen Wangchuck Institute for Conservation and Environmental Research (UWICER), Bhutan and the Center for International Forestry Research (CIFOR) under the framework of the Sloping Lands in Transition (SLANT) research program. The Director, UWICER, Program Director, Conifer Forestry Research Sub-centre (CFRSC), Thimphu and the management are gratefully acknowledged for their strong support during this research. We would like to thank Cheten Thinley, Damber Kr. Ghemiray, and all the colleagues of CFRSC for the excellent support in the field. The authors gratefully acknowledge all the stakeholders involved in the study for providing information and offering special insight into the old-growth oak forests of Bhutan. We would like to express our gratitude to Steven Delean, Adelaide University, for introducing us to mixed models and for his consistent statistical advice. Our paper is improved thanks to the insightful comments of four anonymous reviewers on an earlier version of the manuscript.

2.11 CONFLICTS OF INTEREST

The authors declare no conflict of interest.

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Chapter 3: Regeneration dynamics of brown oak (*Quercus semecarpifolia*) forests with particular reference to grazing and canopy gaps in Bhutan, Eastern Himalaya

3.1 STATEMENT OF AUTHORSHIP

Statement of Authorship

Title of Paper	**Regeneration dynamics of brown oak (*Quercus semecarpifolia*) forests with special reference to grazing-fencing and canopy gaps in Bhutan, Eastern Himalaya**
Publication Status	☐ Published ☐ Accepted for Publication ☑ Submitted for Publication ☐ Unpublished and Unsubmitted w ork w ritten in manuscript style
Publication Details	Journal of Forestry Research

Principal Author

Name of Principal Author (Candidate)	Tshewang Dorji		
Contribution to the Paper	Conceptualization, methodology, data collection and analysis, funding acquisition, writing, and editing reviewer comments		
Overall percentage (%)	70 %		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	20/4/2020

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);
ii. permission is granted for the candidate in include the publication in the thesis; and
iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Justin Brookes		
Contribution to the Paper	Conceptualization, Methodology design, fieldwork, data analysis, funding acquisition, overall supervision and editing and reviewing.		
Signature		Date	21 July 2020

Name of Co-Author	José M. Facelli		
Contribution to the Paper	Supervision, fieldwork, data collection, methodology and data analysis, editing and reviewing		
Signature		Date	4 August 2020

Please cut and paste additional co-author panels here as required

Name of Co-Author	Kesang Wangchuk		
Contribution to the Paper	Methodology development, fieldwork, data analysis, editing and writing		
Signature:		Date	8.7.2020

Name of Co-Author	TshewangNorbu		
Contribution to the Paper	Developing research protocol and data collection, investigation, writing and editing in association with the lead author		
Signature		Date	21.4.2020

Name of Co-Author	Kuenzang Dorji		
Contribution to the Paper	Fieldwork, data acquisition, preparation of maps, writing and editing		
Signature		Date	2.7.2020

Name of Co-Author	Rinchen Dorji		
Contribution to the Paper	Fieldwork, data acquisition, methods, map development and reviewing		
Signature		Date	16/07/2020.

Name of Co-Author	Steven Delean		
Contribution to the Paper	Data sorting, software acquisition, data analysis, writing, editing and reviewing.		
Signature		Date	31-7-2020

Regeneration dynamics of brown oak (*Quercus semecarpifolia*) forests with particular reference to grazing-fencing and canopy gaps in Bhutan, Eastern Himalaya

Tshewang Dorji [1,2,] *, Kesang Wangchuk[2], Steven Delean[1], Tshewang Norbu[2], José M. Facelli [1], Rinchen Dorji[3], Kuenzang Dorji[2] and Justin D. Brookes [1]

[1] The School of Biological Sciences, University of Adelaide, South Australia 5005, Australia; justin.brookes@adelaide.edu.au (J.D.B.); jose.facelli@adelaide.edu.au (J.M.F.)

[2] Ugyen Wangchuck Institute for Conservation and Environmental Research, Thimphu, P.O.Box 2049 Bhutan; tnorbu@uwice.gov.bt (T.N.)

[3] RNR Statistics Division, Directorate Services, Ministry of Agriculture and Forests, Thimphu Bhutan; rinchend@moaf.gov.bt (R.D.)

* Correspondence: tshewang.dorji@adelaide.edu.au or tshewang.dorji@uwice.gov.bt; Tel.: +61-449224611

3.2 ABSTRACT

Brown oak (*Quercus semecarpifolia*) forests are degrading rapidly in the Himalayas due to human pressures coupled with recruitment failures and are widely considered as a vulnerable forest type. No comprehensive assessment has been conducted for the brown oak forests of Bhutan. We carried out a nationwide study to understand forest composition and regeneration status of the species. We used mixed-effects and structural equation models (SEMs) to understand the direct and indirect factors that influence the oak and other tree seedling regeneration. Vegetation analysis showed that the forests have reasonable quantities of oak seedlings, however, suffered from chronic absence of saplings (50 cm < height > 130 cm) and poles (Height >130 cm and DBH < 10 cm) indicating the regeneration failure for the last several decades. SEMs showed that oak seedlings were adversely affected by grazing but promoted by forest disturbances and canopy gaps. All the seedlings recorded showed heavy signs of grazing, with 57 % of seedlings severely browsed to the ground. Combined effects of heavy grazing and inadequate canopy gaps appeared to be two crucial factors responsible for the failure of the seedling transition to upper growth classes, thereby creating a significant gap in the regeneration process. Our results were consistent with the observations from long-term grazing-fencing experiments as well as regeneration from a forest after gap creation that showed higher seedling and sapling recruitment in fenced areas and wider canopy gaps. It is to be noted; however, that complete exclusion of grazing was not desirable as it resulted in massive dominance by a bamboo species (*Yushania microphylla*), which can compete with the recruits and compromise their survival. We suggest that addressing grazing problems or canopy gaps alone may not yield desirable outcomes but should be implemented together with bamboo control, preferably after a seed masting. In the present scenario, more grazing resistant species of conifers, rhododendrons, and *Ilex* were favoured, which are likely to replace the oaks in the future. Considering the substantial gaps in the oak recruitment process spanning over several decades, delaying further or leaving to business as usual situation, can lead to a decline in the forest and invaluable services. There is an urgency for bold actions and collective efforts towards artificial regeneration. The action should entail large-scale restocking complemented by the creation of canopy gaps and adoption of modern planting techniques that reduce herbivory to enhance the sustenance of brown oaks in the Himalayas.

Keyword Brown oak, *Quercus semecarpifolia*, Regeneration, Grazing, Himalaya

3.3 INTRODUCTION

Natural regeneration involves the germination of fallen seeds that gradually restock the forest and relies heavily on the ability of trees to reproduce under a wide range of environments. It forms a critical aspect of forest ecology that guides long-term sustainable forest management. Yet, many essential forest ecosystems globally experience inadequate natural regeneration, which leads to unprecedented loss of important species due to human activities and poor forest management practices (Watt 1919). Although the views on poor regeneration and species loss are varied, overgrazing by large herbivores is commonly regarded as one of the major causative factors. Grazing by large herbivores including domestic livestock and deer herbivores are known to reduce forest diversity through selective browsing (Putman 1996; Didion et al. 2009), soil compaction and soil erosion due to trampling (Mulholland and Fullen 1991), and nutrient loss through removal of biomass (Hatton and Smart 1984). For the regenerating seedlings, grazing can affect not only seedling growth and survival rates but also alter the growing environment within the same landscape through facilitation or removal of the competing vegetation (Popay and Field 2017; Callaway et al. 2005).

Depending on the species and palatability to animals, forest grazing can be used to encourage forest regeneration by controlling the competing vegetation, thereby improving livestock productivity (Didion et al. 2009; Darabant et al. 2007; Buffum et al. 2009). For example, grazing was found useful to control undesirable bamboo growth and facilitate conifer seedling regeneration in temperate forests of Bhutan (Darabant et al. 2007). Similarly, grazing can be deployed to reduce weed and unwanted shrubs that compete with the tree seedlings (Kosco and Bartolome 1983; Popay and Field 2017; Perevolotsky and Seligman 1998). However, more recent studies on forest grazing are less definitive and demonstrate that grazing is likely to contribute to broadleaf forest regeneration failures in combination with inadequate canopy gaps (Tashi and Thinley 2008; Suzuki and Ito 2014).

Grazing effects and responses by the forest vary greatly in conifer and broadleaf forest types (Suzuki et al. 2008) although most studies do not segregate the grazing effects in conifer and broadleaf forest ecosystem. Broadleaf species are generally more palatable and preferred by grazing animals than conifer seedlings. When the species of interest is palatable and grazed, grazing may not be a viable tool to manage the forests. For instance, in the Mediterranean and California’s woodlands, it was evident that grazing reduced the oak seedling survival (Hall et al. 1992), density, height and number of leaves significantly (Lempesi et al. 2017), while in the Atlantic woodlands, grazing resulted in complete failure of oak regeneration (Palmer et al. 2004). Studies conducted in a cool broadleaf community forest in Bhutan found that moderate intensities of forest grazing (0.4 cattle ha^{-1}) can be combined with timber production without adverse impacts on forest regeneration (Buffum et al. 2009). Moderate level of grazing was also found to maintain species diversity (Ram et al. 1989).

The high altitude brown oak (*Quercus semecarpifolia*) is a major forest forming species that occupy the upper elevation range (2000-3500m asl) of the entire east-to-west Himalaya and is believed to be a remnant species of the Himalayas. The forests serve as important grazing ground and a source of fodder for both domestic and wild herbivores (Roder et al. 2002; Norbu 2002). The forests are also associated with diverse socio-cultural values and range of ecosystem services that are imperative to the livelihood of local communities (Dorji et al. 2019). The brown oak forests have suffered rapid degradation across the entire distribution, which is attributed to overgrazing, chronic disturbance through logging and extraction of forest products

(Singh 1998), and climate change (Bisht and Kuniyal 2013). Forest overgrazing, fuelwood extraction and extensive lopping of foliage for fodder resulted in poor regeneration of brown oaks (Singh and Rawat 2012) and gradual replacement by pine forests (Singh et al. 1984). Absence of suitable gaps in the canopy (Metz 1997) and poor forest management (Shrestha and Paudel 1996) were also identified as factors affecting recruitment. In the mountain regions, forests close to human settlements are regularly used and have high disturbance frequency than the forests situated distant from human settlements (Kumar and Ram 2005). Several researchers found that oak regeneration increases with decreasing disturbance (Kumar and Ram 2005; Singh and Rawat 2012), while some researchers adopt an opposing view and tend to believe that large-scale disturbances are in fact required for oak regeneration (Metz 1997; Singh et al. 1997). The forests surrounding monasteries and religious sites are least disturbed and protected by cultural beliefs and should ideally have more regeneration; however, this was not the case (Metz 1997). Thakuri (2010) also found that oak regeneration in Nepal was higher in the disturbed forests than the undisturbed forests and related the trend to disturbances of trees in the canopy. These contrasting results have meant that the factors responsible for successful oak regeneration remain unresolved.

The need for long-term research at wider scales has been strongly advocated in the Himalayan region for a better understanding of forest dynamics (Shrestha et al. 2004; Singh and Thadani 2015). In the absence of crucial long-term information on plant recruitment and demographics, efforts to undertake timely restoration measures remain uncertain. The issue of poor regeneration has remained under the radar for some time primarily because of fund limitation, absence of adequate information and existence of many mature trees presently, that mask the poor regeneration status or the need to regenerate these forests. These forests require preservation as they have rich biodiversity and are the sustenance of subsistence farming on which the majority of the Himalayan population depend. The forests are a source of fresh water, timber, fuelwood, leaf litter, non-wood forest products (NWFPs) and manure and have special significance for spirituality and wellbeing (Dorji et al. 2019). As not many species can grow and adapt at high elevations, the oak trees serve as an important carbon sink in high altitude mountains (Verma et al. 2012).

The forest management is challenged by limited knowledge on Himalayan brown oak forest regeneration and restoration ecology. Our study aims to fill this gap by conducting a nationwide oak study comprising of 68 brown oak forests of Bhutan to assess the current regeneration status and determine the factors responsible for the seedling regeneration. We then support our results with data from permanent regeneration-monitoring plots consisting of grazed and ungrazed (fenced) plots established since 2000. Further, to determine the role of wider forest canopy gaps on oak regeneration, we conducted a regeneration survey in an oak forest that has been cleared for a transmission line and compared the results to the adjacent undisturbed closed forest.

3.4 MATERIALS AND METHODS

3.4.1 Study area

The study was conducted in Bhutan (Figure 13) in sites extending from east to west across the country and at altitudes ranging from 2300 - 3500 m above sea level (temperate to subalpine climate). The geology comprises bedrock of metasedimentary and orthogneiss unit of the high Himalayan crystalline belt (Gansser 1983; Tobgay et al. 2010). Annual precipitation varies from 720 mm to 1500 mm, most of which falls during the monsoons extending from May to September. Mean annual temperature is approximately 4.6 ºC and varies with a mean maximum of 14.6 ºC to a mean minimum of -8 ºC (Darabant et al. 2007; Tashi 2004). The forest stands are dominated by *Quercus semecarpifolia* in the overstorey with conifers such as *Tsuga dumosa*, *Picea spinulosa* and *Pinus wallichiana* scattered as components of canopy dominants in some stands. The middle storey is occupied by *Rhododendron* spp., *Ilex dipyrena*, *Sorbus* and *Symplocos* trees. The understorey is mostly dominated by bamboos (*Yushania microphylla*), *Rosa* spp., *Lyonia ovalifolia*, *Viburnum* spp., *Pieris formusa*, and *Berberis* spp.

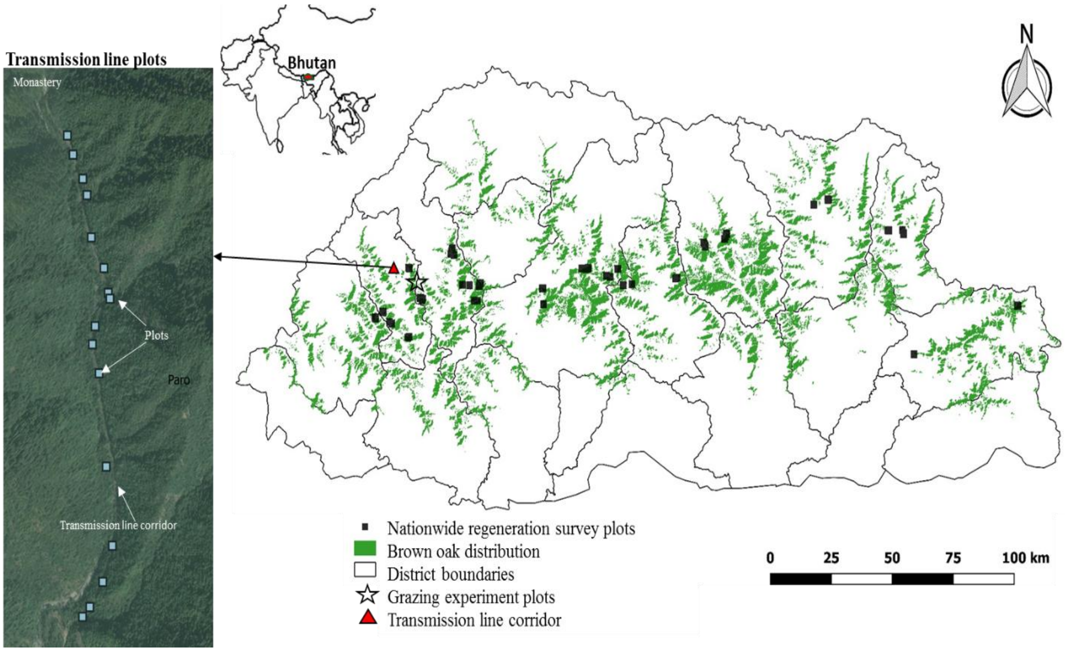

Figure 13. Map of Bhutan showing the probable brown oak forest distribution. The three different study sites are shown in black square boxes for nationwide survey plots, red triangle for transmission line corridor (with a GoogleEarth image of plot layout) and star symbol for the location of the grazing-fencing experiment.

Domestic livestock, including resident and migratory yaks and cattle, graze within the forest. The migratory livestock transit in the forest as they move from higher to lower elevations to avoid cold winters and vice versa in summers. Besides livestock, wild animals such as sambar and barking deer also graze the forests. The density of wild herbivores is considered much lower than the domestic livestock based on the visual investigations of droppings (Darabant et al. 2007).

3.4.2 Forest survey and sampling

a) Nationwide regeneration study

A total of 68 high altitude oak forests covering the east to the west belt of Bhutan were surveyed (Table 6). In each of these forest stands, a plot of 25 m x 25 m was established, where we identified, tagged, and measured all tree individuals greater than 1.3 m high. The height (Ht, m) was measured using a digital hypsometer, and diameter at breast height (DBH, cm) was measured using a diameter tape. A visual assessment of the level grazing intensity in each plot was conducted based on dropping abundance and clipping of foliage. Each plot was classified into heavy, moderate, light, and no grazing accordingly. For fine-scaled assessment of tree regeneration, grazing evidence, vigour, and microsites, subplots of 2 m x 2 m were laid at the centre of the main plots, where regenerating individual of trees were counted and measured for height (ht, cm) and collar diameter (Cd, cm). The microsites on which the seedlings grew, seedling vitality, and grazing evidence on the seedlings were graded as per the scales: Microsite; 1 = leaf litter, 2= moss, 3 = nurse log, 4 = mineral soil. Grazing; 1= no grazing, 2 = low grazing, 3 = moderate, 4 = heavy grazing. Vitality; 1 = dead, 2= weak, 3 = Healthy, 4 =strong. Plots of 5 m x 5 m were sampled at the centre of the plots to survey ground vegetation and shrub layer. The height of the tallest individual, together with their percent cover (%) were recorded for each species. We adopted the location of forests relative to human settlement as a proxy to represent the gradient of forest disturbance which ranged from not disturbed to heavily disturbed (Aryal et al. 2015) (Distance to settlement; within 1 km = heavily disturbed, 1 - 3 km = moderately disturbed, 3-5 km = lightly disturbed and > 5 km = no disturbance). Grazing status within the plots was assessed based on a visual estimate of dungs and foliage clipped by the animals (1 = no grazing, 2 = light grazing, 3= moderate grazing, and 4 = heavy grazing).

Table 6. Description of Nationwide regeneration survey plots

Plot	District	Altitude	Aspect	Slope	BA (m^2/Ha)	Trees/ha
P1	Tashigang	3104	SE	40°	62.7	672
P2	Tashigang	3063	N	45°	70.6	432
P3	Tashigang	3104	SW	5°	101.7	624
P4	Tashigang	2683	S	40°	8.9	432
P5	Tashiyangtse	2535	SW	35°	40.9	640
P6	Tashiyangtse	2586	SW	30°	134.0	640
P7	Tashiyangtse	2786	SW	25°	117.2	672
P8	Tashiyangtse	2590	E	20°	85.1	816
P9	Tashiyangtse	2716	SE	35°	63.4	832
P10	Tashiyangtse	2701	SE	45°	74.2	992
P11	Lhuntse	2356	S	45°	40.7	560
P12	Lhuntse	2672	S	35°	59.8	480
P13	Lhuntse	2704	SW	20°°	92.8	480
P14	Lhuntse	2706	S	15°°	107.6	384
P15	Bumthang	2828	NE	25°	37.8	432
P16	Bumthang	2764	NE	30°	135.6	336
P17	Bumthang	2803	NE	15°	58.5	544
P18	Bumthang	2827	E	15°	66.2	352
P19	Bumthang	2873	W	35°	117.0	464
P20	Bumthang	2812	SW	35°	122.0	272

P21	Bumthang	2770	W	30°	63.8	464
P22	Bumthang	2849	SW	45°	77.8	560
P23	Trongsa	2864	S	40°	77.8	720
P24	Trongsa	2888	SE	35°	180.7	896
P25	Trongsa	2520	S	30°	84.4	112
P26	Trongsa	2527	N	10°	76.7	256
P27	Wangdue	2809	S	60°	112.0	336
P28	Wangdue	2720	E	10°	137.0	192
P29	Wangdue	2902	SE	45°	173.5	240
P30	Wangdue	2912	SW	25°	155.6	352
P31	Wangdue	2876	SW	35°	91.7	240
P32	Wangdue	2869	W	15°	126.5	224
P33	Wangdue	2707	W	20°	200.7	304
P34	Wangdue	2698	N	25°	225.3	416
P35	Wangdue	2774	SW	10°	76.8	576
P36	Wangdue	2776	N	15°	153.3	304
P37	Haa	3417	SW	20°	89.8	176
P38	Haa	3601	S	40°	91.0	400
P39	Haa	3442	S	30°	51.7	336
P40	Thimphu	3021	S	41°	92.8	320
P41	Thimphu	2643	SE	22°	107.3	825
P42	Thimphu	2720	SE	15°	123.3	475
P43	Paro	2650	SW	32°	72.7	500
P44	Paro	3500	SW	22°	144.6	450
P45	Paro	2890	SW	70°	59.9	1575
P46	Paro	2890	NW	82°	72.7	900
P47	Thimphu	2795	SW	35°	31.8	575
P48	Thimphu	2800	SW	40°	83.2	850
P49	Thimphu	2915	S	47°	109.4	2150
P50	Thimphu	2956	SE	35°	76.2	650
P51	Thimphu	3000	SW	38°	58.8	400
P52	Thimphu	3050	NW	37°	111.2	725
P53	Thimphu	3000	SE	60°	90.9	1150
P54	Thimphu	2379	NW	45°	56.2	1450
P55	Thimphu	2391	NW	35°	52.9	1450
P56	Thimphu	3200	S	80°	27.4	2000
P57	Thimphu	3458	SW	35°	80.8	1725
P58	Paro	3415	S	80°	99.8	775
P59	Paro	3455	SW	80°	41.2	975
P60	Thimphu	3105	SE	36°	134.2	1150
P61	Thimphu	3270	SW	50°	64.3	825
P62	Paro	3500	SE	55°	111.3	850
P63	Paro	3040	S	35°	27.3	2150
P64	Paro	3326	SE	80°	78.4	425
P65	Thimphu	2920	NW	40°	56.2	375
P66	Thimphu	2847	SE	47°	66.4	525

P67	Thimphu	2870	SSE	45°	74.0	675
P68	Thimphu	3500	S	38°	104.4	875

b) *Grazing and fencing experiment*

To support our nationwide survey with longitudinal experimental information, we used the oak regeneration data from our long-term grazing and fencing plots in western Bhutan. Twelve 10 m x 10 m plots were established in the year 2000 under mature oak stands. Six plots were fenced using barbed wire to exclude large herbivores, while six others were left unfenced to represent natural control conditions. The plots were established in pairs such that each pair of adjacent plots consisted of a grazed and ungrazed (fenced) plot as a block. Detailed descriptions of the plots can be found in (Dorji 2012) and (Tashi 2004). Each of the plots was divided into 25 subplots of 2 m x 2 m for assessing the tree regeneration and bamboo cover percentage. Hemispherical photographs were taken 50 cm from the ground from the centre of each plot to describe light availability on the forest floor and estimate canopy opening. Shrub layer was assessed in the centre 5 m x 5 m similar to the nationwide survey. The data on oak recruitment overtime used were from five-time points (2000, 2002, 2005, 2011, 2018). All oak individuals below 50 cm height were classified as seedlings and above 50 cm as saplings. Although there were several other tree species regenerating, their density did not differ significantly between the fenced and unfenced plots and was not presented in this study. Further, our fence was designed to exclude large herbivores (both livestock and wild animals) and did not account for smaller herbivores and rodents. Both our observation and published literature in a similar Himalayan setting did not confirm any significant damage by small herbivores or rodents (Darabant et al. 2007; Tashi 2004).

c) *Regeneration survey in transmission line corridor*

Experiments which involve the creation of wide canopy gaps by removing big trees from a forest is expensive, and and an “eye-sore” particularly in old-growth forest where they function as keystone species. To avoid such situation, we purposely overlapped our regeneration study with a national project of electrifying a highly significant remote Buddhist monastery in Western Bhutan. The transmission lines had to pass through an old-growth brown oak forest which was cleared in 2015. The transmission corridor is 6-12 m wide and 10 km long cleared as a Right of Way (RoW) for the 11 KV transmission line. A total of 263 mature oaks and 81 mixed conifers were removed from the transmission line. We laid 17 regeneration plots, 5 m x 5 m each, along the open corridors to assess the regeneration in the open gaps and another nine 5 m x 5 m regeneration plots in the undisturbed forests adjacent to the transmission lines in 2017. The area was relatively far from human settlements, and no grazing by domestic livestock was recorded. There are no historical accounts of migratory livestock grazing in the forest. We did not account for grazing by wild angulates in the study area whose presence was not noticed during the survey. No official subjective assessment of wild herbivore population is also currently available for the area.

3.4.3 Data analysis

To explore the relationship between how human disturbances impact forest structure and regeneration dynamics, we developed mixed-effects model (Duursma and Powell 2016) for each dependent variable (species richness, evenness, and diversity) associated with forest structure and regeneration. Mixed models were built using the lmer function in the lme4 package (Bates et al. 2015) of R software (CoreTeam 2019). In our model, we used species richness, evenness and diversity as dependent variables and distance from the human settlement (an indicator of forest disturbance) as the fixed factor. Plots were entered as the random factor in the model. To further establish the complex direct and indirect effects of interrelated factors that affect tree regeneration, we used the structural equation modelling (SEMs)—a multivariate statistical analysis tool to analyse structural relationships. We used structural equation modelling (SEMs) available with the program piecewiseSEM (Lefcheck 2016) in R (CoreTeam 2019). The method involves the combination of factor and multiple regression analysis and brings multiple variables into a single casual network for testing of various assumptions. They are widely used in ecology due to their flexibility to accommodate a wide range of model structures and hypotheses.

For the grazing and fencing experiment data, we used mixed-effects model (lme4 package; (Bates et al. 2015)) in R (CoreTeam 2019). We entered grazing treatment (fenced and unfenced) and time (years) as fixed factors, while plots nested within blocks were entered as random effects to account for variations as a result of the hierarchical grouping of samples and repeated measures over time. Log transformation of data was done to meet the assumption of normality and homoscedasticity or errors. Homoscedasticity or normality assumptions were checked through the residual diagnostic plot of fitted values and model residuals and q-q plots. Model estimates of the response variables were back-transformed to the measurement scale for reporting. The *p*-values and related tests for the mixed-effect models were obtained by the *F*-tests using the Anova function in the package 'car' (Fox and Weisberg 2011). Group comparisons were made using Tukey's HSD.

Finally, to understand the forest regeneration due to large forest openings and compare to an adjacent closed-canopy natural forest, we used an Independent-samples t-test on the regeneration counts recorded from the two groups. Log-transformation of the oak counts was done to meet the statistical assumptions of normal distribution. The final results were back-transformed and reported. All statistical analysis was carried out using the R software (CoreTeam 2019).

3.5 RESULTS

3.5.1 Structure and composition of high-altitude oak forests

A total of 59 different tree species were recorded from the oak forest during the nationwide survey, of which 24 were evergreen broadleaf, 27 deciduous broadleaf, seven evergreen conifer, and one deciduous conifer life-forms (Annexure 1). *Quercus semecarpifolia* dominated all the plots as the canopy dominants represented by a higher basal area (Table 7). Some plots were dotted with few conifers such as *Tsuga dumosa*, *Picea spinulosa*, and *Pinus wallichiana*. *Rhododendron arboreum* and *Ilex dipyrena* mostly dominated the middle storey with few *Symplocos* spp. And *Eurya* spp. in some plots. The average basal area of the trees was 90.5 m^2 ha^{-1}, with a mean of 675 trees ha^{-1}. A maximum DBH of 222 cm was recorded for *Q. semecarpifolia* (Table 2). The maximum height of 86 m was also recorded for *Q. semecarpifolia*. We also recorded 33 Shrub species. The shrub layer (in percent cover) is mostly dominated by *Daphne bholua* (21.5 %), *Berberis* spp. (18 %) and *Rosa sericea* (10 %). A total of 125 herb species were recorded on the forest floor, which was mostly dominated (in percent cover) by *Carex* spp. (25 %), *Rubia* spp. (10 %) and ferns (8%).

Table 7. Dominant tree species recorded in the survey, along with the forest structural attributes and regeneration patterns (seedling and saplings).

Lifeform	Species	BA (m^2 ha^{-1})	Trees ha^{-1}	Av. DBH (cm)	Av. Ht (m)	Max DBH (cm)	Max ht (m)	Seedling ha^{-1}	Sapling ha^{-1}
Evergreen broadleaf	*Quercus semecarpifolia*	103.5	409	46.1	19.3	222	86	3261	6
	Quercus glauca	1	7	32.2	18.5	87	39	41	0
	Rhododendron	2.6	149	12.2	7.8	69.4	22	159	135
	Ilex dipyrena	7	32	18.3	10.4	58.7	23	143	53
	Osmanthus suavis	0.1	1	10	6.3	21.5	13	30	6
	Eurya accuminata	0.4	31	10.9	7.7	47	20	44	0
	Myrsine semiserrata	0.1	3	14.9	9.8	19.7	15	110	0
	Persea	1.3	11	32.3	16.7	88.4	30	184	0
Deciduous broadleaf	*Acer campbellii*	0.3	6	25.6	15.1	44.5	23	363	0
	Betula utilis	0	1	20.2	7.5	26.8	10	6	0
	Fraxius paxina	0.2	2	39.9	21.8	69.5	24	110	0
	Gambelea cilliata	0.2	2	29.2	14.2	69.5	23	221	0
	Litsea	0	1	10.9	9.3	12.8	11	6	0
	Prunus cerasoides	0.2	7	19.6	14.4	35	30	110	0
	Viburnum	0.3	5	20.3	9.5	58	15	37	0
Evergreen conifers	*Pinus wallichiana*	1.4	40	16.9	17.7	70	40	778	157
	Picea spinulosa	1.3	8	33.1	21.1	103.8	45	100	12
	Abies densa							88	0
	Juniperus recurva	0.1	1	20.8	10.7	36	15	36	0
	Taxus baccata	0.6	6	29.4	12.9	62.5	23		
	Tsuga dumosa	2.9	23	24.6	13.9	159.5	69	243	24
Deciduous conifer	*Larix griffithii*	0.3	1	51.7	29.7	54	31	6	0

(BA = Basal area; Av. = Average; DBH = diameter at breast height, ht = Tree height)

3.5.2 Regeneration of tree species

A total of 27 tree species were recorded regenerating on the forest floor during the survey. *Quercus semecarpifolia* showed the highest regeneration density with a mean of 3261 seedlings ha^{-1}, followed by conifers with 1255 seedlings ha^{-1} (*Pinus wallichiana* with 778 seedlings ha^{-1}, and *Tsuga dumosa* with 243 seedlings ha^{-1}). Recruits of *Ilex dipyrena* and *Rhododendron arboreum* with a mean density of 196 and 294 ha^{-1} represented the regenerations for the middle-storey tree species. The majority of the regeneration (95.9%) of *Q. semecarpifolia* were less than 30 cm tall, and no sapling above the height of 100 cm was recorded from the entire survey. There was a complete absence of individuals in the sapling and pole stages (Figure 14). More than 57 % of *Q. semecarpifolia* seedlings were heavily grazed, and 42 % were lightly grazed, indicating the strong effect of grazing on recruitment. *Rhododendron* and conifers were least affected by grazing, while *Ilex dipyrena* recruits showed light signs of grazing.

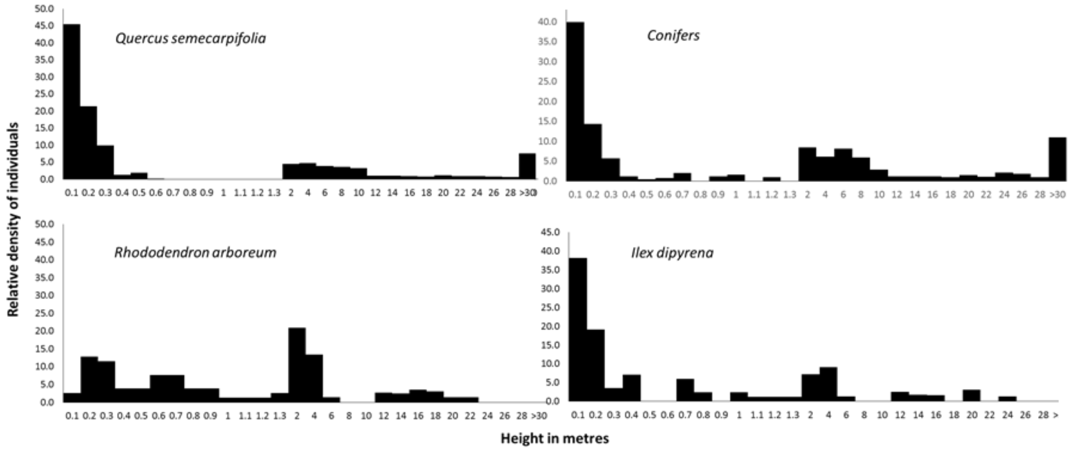

Figure 14. Forest structural features of dominant tree species in the brown oak forests (data pooled) based on height class intervals expressed as relative density within each plot. Note that seedlings and saplings are represented by individuals below the 1.3 m height, while individuals above 1.3 m height represent trees.

3.5.3 Anthropogenic-Regeneration interactions

Forest structural features and regeneration of oak seedlings were strongly influenced by the proximity of the forest to human settlements/pastures (Figure 15). Species richness and evenness were comparatively higher in the forests close to the human settlements, which also resulted in higher tree diversity in these forests. There was a direct relationship between the *Q. semecarpifolia* seedling density and the forest disturbance. Highest oak seedling density was recorded in forests, that were located at about 1-3 km from the villages. The forests with intermediate forest disturbances and lower grazing pressure recorded a higher number of seedlings than either disturbed or undisturbed forests located away from human settlements.

Oak seedlings in the forests near villages (<1 km) showed signs of heavy grazing, which resulted in a significantly lower number of oak seedlings compared to the forests where grazing animal pressure is low (Figure 15 & Figure 16). No oak saplings above the height of 60 cm were recorded during the entire study (Figure 14). Visual investigation of several grazed seedlings in the field showed severe signs of repeated grazing that substantially removed the foliage (Figure 16) Seedling density also significantly declined in the undisturbed forests located further from the settlements.

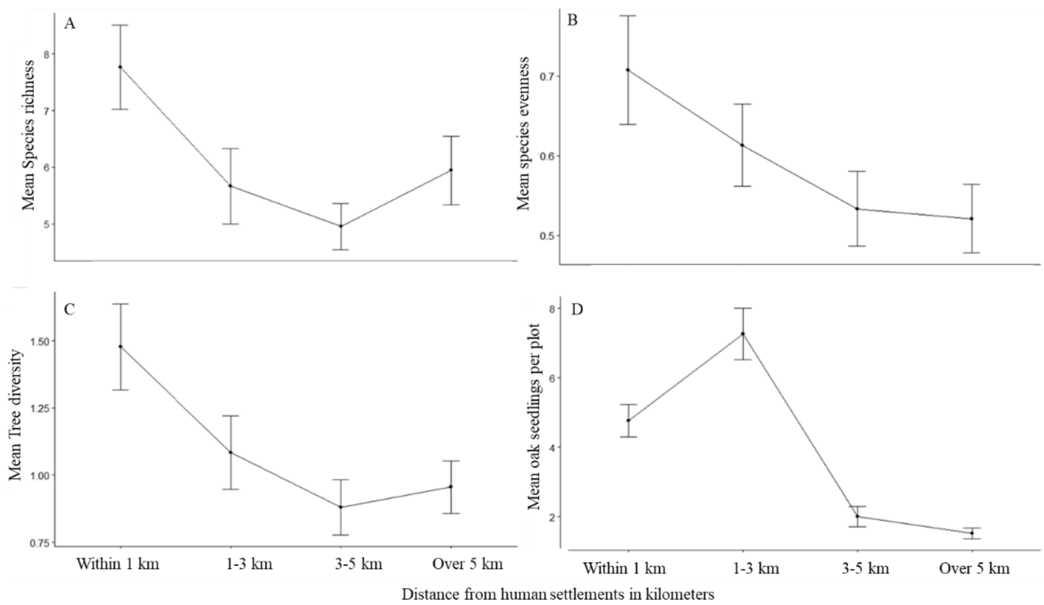

Figure 15. Forest structural features and oak regeneration in relation to the location of forests from disturbance source (distance from settlements). Mean tree species richness (A), mean species evenness (B), mean tree diversity (C) and mean oak seedling count in the plots (D) with respect to plots distance from human settlements.

Figure 16. Overgrazing can result in complete defoliation of the tree seedlings and compromising seedling survival. Shown in the pictures are severely defoliated oak saplings (A-B), top browsed *Tsuga dumosa* sapling (C) and *Osmanthus suavis* (D).

Our SEMs explained 47% of the variations in total tree seedling and oak regeneration and about 22%, and 23% of the regeneration variations for conifers, and Rhododendrons, respectively. The modelling result also confirmed that grazing, canopy opening, and distance from human settlements are important factors that can influence tree regeneration. The SEMs support the hypothesis of grazing, having a significant negative effect on the total count of seedling regeneration with a standardised path coefficient of -0.46 (p <0.001). Similar relationships were observed for oak regeneration with a path coefficient of -0.4 (p <0.001), rhododendrons with -0.19 (p <0.05), and conifer regeneration with -0.28 (p < 0.05). Canopy opening also showed a significant positive effect on total seedling regeneration (path coefficient = 0.40, p <0.001), oak regeneration (path coefficient = 0.45, p < 0.001), and rhododendrons (path coefficient = 0.29, p < 0.05). However, no such relationship of canopy opening with conifer regeneration was detected in this study (path coefficient = 0.23, p > 0.05, Figure 17). The distance of forest from the human settlements was significant and inversely related to tree diversity in all the SEMs (path coefficient = -0.54, p < 0.001); however, the relationship between tree diversity and seedling regeneration was not significant (Figure 17).

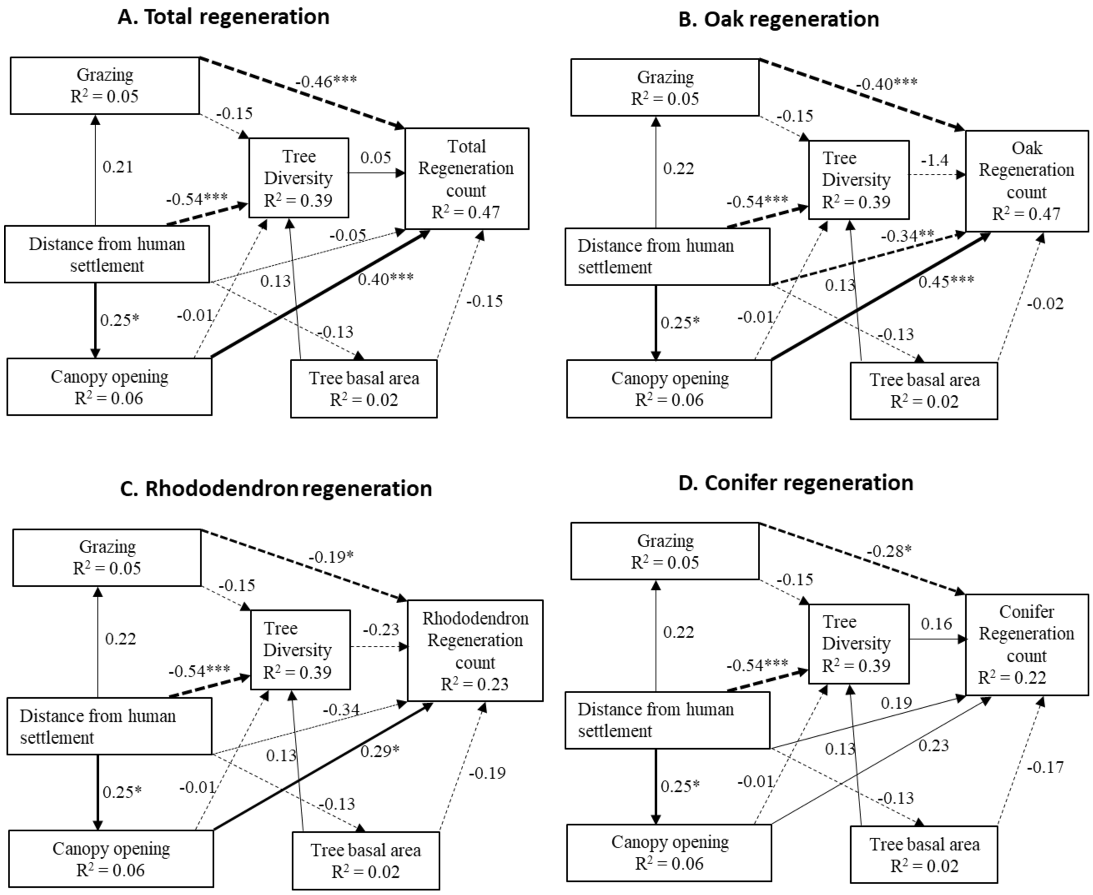

Figure 17. Structural equation models showing the direct and indirect causal effects of various variables on total seedling regeneration (A), oak regeneration (B), Rhododendron regeneration (C), and Conifer regeneration (D). Solid lines indicate positive and dotted lines represent negative relationships shown with the standardised path coefficient values. The thickness of the lines indicates the significance of relationships. Multiple R^2 values are shown in the response variable boxes. Significance level are shown by ***$P < 0.001$, **$P < 0.01$, *$P < 0.05$.

3.5.4 Grazing and fencing experiment

Grazing and fencing experiment observed for 18 years also supported our results of the negative effect of grazing on oak seedlings. The results showed considerable variations in the oak seedling density, indicating large fluctuations in seedling emergence and mortality over the years, particularly in the fenced plots (Figure 18). There is an increase in the oak seedlings observed after the year 2000 (in the fenced plots) and after 2010 (in both fenced and unfenced plots). Higher seedling recruitment was found in the fenced plots compared to the unfenced plots. We also noted significant changes in the forest understorey (Shrub layer) as a result of fencing (Figure 18B). Most notably, a bamboo (*Yushania microphylla*) increased significantly within the fenced plots (Figure 19). Increased numbers of unpalatable shrubs like *Pieris formusa*, *Berberis aristata* and *Daphne bholua* in the grazed plots were also observed.

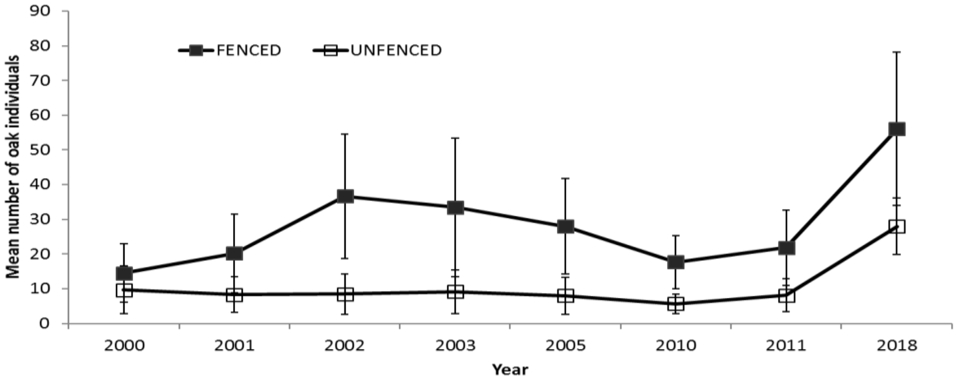

Figure 18. Oak regeneration count per plot over the years since the establishment of the fenced and unfenced plots. The bar shows the standard error of the mean.

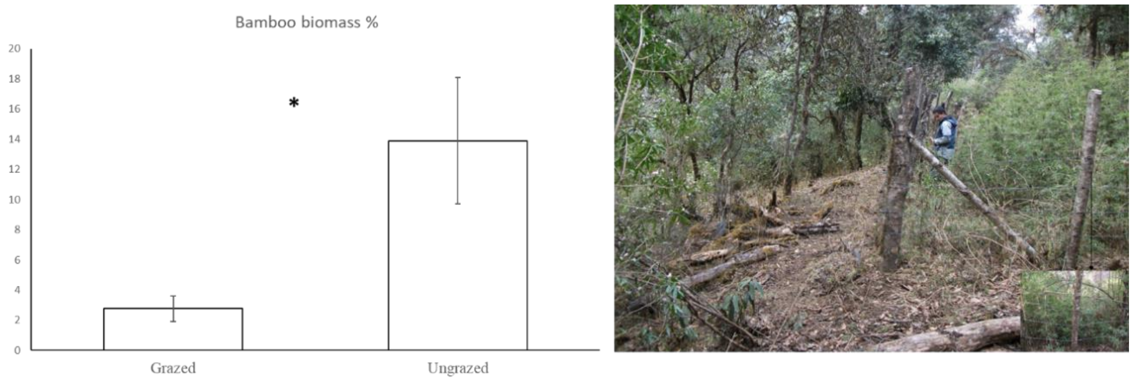

Figure 19. Comparison of the grazed and ungrazed plots showed significant dominance by bamboo thickets in the fenced plots. Bar represents the standard error of the mean.

There was no significant difference in oak seedling density between the grazed and fenced plots after 18 years (p = 0.938). However, sapling density with a mean of 2.5 ± 0.8 (SE) for the grazed plot was significantly different from the fenced plot with a mean of 13.4 ± 4.4(SE) (Table 8) showing the positive effect of fencing (p = 0.02) and fencing over time interactions ($F_{4,40}$ = 3.99, p = 0.008, Table 9).

Table 8. Model estimated mean number of seedlings and saplings in the grazed and ungrazed plots

	Total		Seedlings		Saplings	
	Grazed	**Fenced**	**Grazed**	**Fenced**	**Grazed**	**Fenced**
Year	Mean (SE)	Mean (SE)	Mean (SE)	Mean (SE)	Mean (SE)	Mean (SE)
2000	5.97 (3)	7.14 (3.6)	4.3 (2.5)	5.1 (3)	1.1 (0.4)	1.1 (0.4)
2002	6.8 (3.4)	15.0 (7.5)	4.5 (2.6)	12 (7.0)	1.4 (0.4)	1 (0.3)
2005	7.0 (3.5)	16.4 (8.2)	5.1 (3)	11.3 (6.6)	1.4 (0.4)	4.4 (1.4)
2011	7.8 (3.9)	12.1 (6.0)	5.0 (2.9)	4.8 (2.8)	2.3 (0.7)	6.7 (2.2)
2018	23.3 (11.7)	38.5 (19)	19.3 (11.3)	20.3 (11.9)	2.5 (0.8)	13.4 (4.4)

Table 9. Results of mixed-effects model analysis for oak total, seedling and sapling counts. Bold letters indicate a statistically significant effect.

	Total			Seedlings			Saplings		
Fixed effects	F	df	Pr(>F)	F	df	Pr(>F)	F	df	Pr(>F)
Fencing effects	1.22	1,10	0.295	0.448	1,10	0.51	7.56	1,10	**0.02**
Time effects	4.11	4,40	**0.007**	3.12	4,40	**0.02**	12.1	4,40	**<0.001**
Fencing*Time	0.24	4,40	0.913	0.49	4,40	0.74	3.99	4,40	**0.008**

3.5.5 Oak regeneration in the open areas along the transmission corridor

We compared the regeneration in the open gaps created by tree removal for transmission lines (Figure 20 A & B) with the adjacent natural forest (Figure 20 D). The study found that open gaps cleared under the transmission lines had a significantly higher number of *Q. semecarpifolia* seedlings (mean = 12.58 ± 1.20 SE per plot) compared to the adjacent undisturbed, closed forests (2.00 ± 1.23, t (25) = 6.550, p < 0.001, Independent-samples t-test, Figure 21). All the seedling regeneration were young with height less than 25 cm indicating the recent regeneration following the canopy opening (Figure 20 C). The adjacent natural forests understorey were very dense covered by *Yushania microphylla* bamboo (Figure 20 D). No account of grazing animals or grazing signs on seedlings was observed during the survey in both types of canopy conditions.

Figure 20. Oak regeneration survey conducted in an open transmission corridor (A, B), with a young oak regeneration (C), and the adjacent natural forest (D).

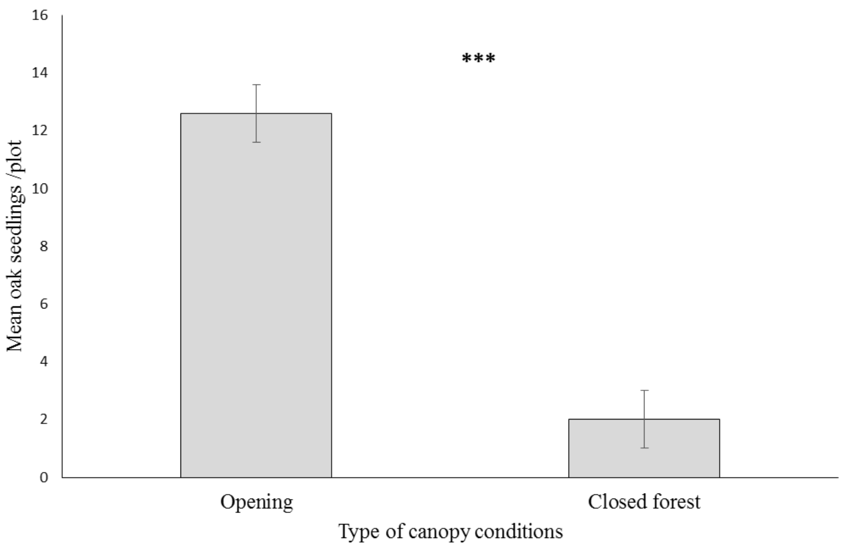

Figure 21. Comparison of brown oak regeneration (mean count in 5 m x 5 m plots) in the open transmission corridor and adjacent undisturbed closed forests. The bar represents standard error of the mean.

3.6 DISCUSSIONS

Our study showed that tree seedling regeneration of high-altitude brown oak forests is dependent upon the structural features of the forest (e.g., canopy gaps) and how local people utilise them (e.g., forest grazing). It was observed that oak trees produced sufficient seeds that germinate into young seedlings but failed to recruit into a sapling or higher growth classes. The analysis of population structure and regeneration patterns confirmed the chronic absence of saplings and small trees in the entire forest, an indication of oak recruitment failure in last several decades, which mirrors studies of the same system in Nepal (Shrestha et al. 2004) and India (Singh et al. 1997). Visual investigation of seedlings in the forests exhibited severe browsing of top shoots and defoliation of seedlings indicating the strong bearings of grazing on oak recruitment. A similar negative effect of grazing on oak seedlings and saplings have been reported from the Himalayas (Singh and Rawat 2012; Dorji et al. 2015) and worldwide (Lempesi et al. 2017; Perrin et al. 2006; Hall et al. 1992). As a result, grazing greatly affected oak recruitment even if there were adequate oak seedlings, in line with the findings in other oak species (Kelly 2002).

Forest grazing is a dominant feature of the alpine and sub-alpine forests in the Himalayas (Namgay et al. 2013; Moktan et al. 2008), including the oak forests, where forests serve as the primary source of fodder and nutrient transfer to agricultural fields (Roder et al. 2002). Large herds of migratory cows and yaks, as well as sedentary cattle and wild herbivores, graze the forest throughout the year without providing sufficient time for the plants to recover from grazing damage. Past studies have found that tree seedlings may recover from one or two seasons of light grazing. However, when plants are repeatedly browsed, it leads to complete defoliation and reduced photosynthetic ability, eventually causing high seedling mortality (Canham et al. 1994; Liu et al. 2016). Similarly, studies from Central Himalaya have found that three months of cattle grazing can result in 75 % of oak seedling mortality (Singh et al. 2011).

Further, the selective feeding behaviour of grazing animals can change the forest structure and composition over time. As indicated by our study, grazing diminished the density of palatable tree seedlings, e.g., oaks, while on the other hand fostered the dominance by non-palatable tree species, e.g., conifers, ilex and rhododendrons. The results from our long-term grazing and fencing experiment showed strong dominance by unpalatable species in the grazed plots while the palatable ones declined significantly. Our finding is, therefore, congruent with studies in the similar forest of the Himalayas that report gradual replacement of Himalayan oak forest by other species (Naudiyal and Schmerbeck 2017; Singh et al. 1984). As oak forests provide a suite of ecosystem services indispensable for wellbeing and socioeconomy of the region, changing forest composition can have significant bearings on the livelihood of rural communities (Dorji et al. 2019).

Forest grazing, however, is not always detrimental as it appears to be. Some studies in the Himalayas also pointed out the positive aspect of forest grazing. As shown from our study, fencing and complete exclusion of grazing led to strong dominance by bamboos and is not a desirable option for regenerating seedling from the competition for space, nutrients, and lights perspective. Darabant et al. (2007) reported the benefits of forest grazing in controlling the competing vegetation (e.g. bamboo) and facilitating the regeneration of conifer seedlings. Conifers are generally grazing resistant species, which could have responded positively to grazing and reduction in the surrounding vegetation. For regenerating palatable species like oaks, grazing exclusion must be complemented by periodic removal of unwanted vegetation or low levels of grazing to control the competing vegetation (Gottesman and Keeton 2017; Darabant et al. 2007). Past studies indicated that controlled grazing could maintain sparser mid and understorey layer, providing conducive environments for seedling survival and establishment (Darabant et al. 2007; Popay and Field 2017). However, when grazing pressure is not managed, over browsing of highly palatable oaks might outweigh the grazing benefits and lead to complete regeneration failure. The findings point out the need to treat different forest types separately, should grazing be used as a forest management tool to control unwanted vegetation growth.

The profuse oak regeneration in the wider forest gaps cleared under transmission corridors and subsequent absence in the adjacent closed forests from our study indicates that forest clearings and canopy openings are also necessary in addition to grazing management. Similar views were also suggested by past studies that emphasized that creation of larger gaps in the forest facilitates regeneration of oak species (Singh et al. 1997; Tashi and Thinley 2008; Thakuri 2010; Lorimer et al. 1994; Gottesman and Keeton 2017). The studies appear to support the hypothesis of large-scale disturbances required for oak regeneration (Metz 1997). In the old-growth forests of Bhutan, where human disturbances are low and large gaps absent, oak regeneration is currently being compromised even if the grazing levels are low. As indicated by our study, oak seedling density was higher in the disturbed forests close to human settlements compared to the least disturbed forests located away from human settlements. Forests surrounding villages undergo regular small-scale disturbances from selection cutting (Buffum et al. 2008; Moktan et al. 2009) and the extraction of timber, fuelwood, and fodder needs—that create smaller canopy gaps for oak regeneration. It is obvious that the existing management practice of strict protection of oak forests is not desirable and does not guarantee oak regeneration. However, it is to be noted that when the fuelwood, fodder demands and grazing pressure are high, it often results in harmful coppicing and deterioration of the regenerative capacity of the oak species in the forest adjacent to the villages (Tshering et al. 2014).

Thus, human pressure, canopy gaps, and grazing synergistically and individually affect the regenerative potential of oak forests as indicated by our SEMs. Even in the presence of large gaps in the forest, severe grazing can lead to oak recruitment failure (Singh et al. 1997) as gaps are known to be preferred feeding spot of herbivores (Kuijper et al. 2009). Without some form of grazing, gaps themselves become prone to invasion by inferior shrubs and tree species (Gottesman and Keeton 2017; Harmer et al. 2005) and bamboos in the Himalayas (Darabant et al. 2007; Dorji 2012). It is therefore evident that gap creation should be accompanied by controlled grazing or weed removal, preferably following a mast seeding—typical for *Quercus semecarpifolia*. For instance, the increase in the mean number of oaks after 2000 (fenced plots) and 2010 (in both fenced and unfenced plots) are likely due to mast seeding observed in those years (Singh et al. 2011). Forest canopy manipulations should target the removal of conifers, tall understorey trees and bamboos to create more gaps and reduce competition for resources (Lorimer et al. 1994) while maintaining mature oak trees as a seed source. Canopy gap creation should, however, be strictly complemented with low grazing levels (Suzuki and Ito 2014; Kirby 2001). We recommend that long-term monitoring of recently created gaps is essential to understand the natural vegetation dynamics. Grazing protection trials involving tree shelters (Stange and Shea 1998; Garcia et al. 2011; Dubois et al. 2000) have been found very effective in regenerating the brown oaks in the Himalayas (Dorji et al. 2020).They should be given special consideration, particularly due to the advantage of free movement of grazers around the tree shelters which control undesirable vegetation.

Reducing grazing pressure from the forest can be another management option which can be achieved through improved livestock breeding practices (Wangchuk et al. 2014). For example, the increase in seedling numbers after 2010 even in unfenced plots could be linked to the decrease in the herders and local cattle population of high altitude Himalayas as a result of changes in socioeconomic conditions and policy decisions (Wangchuk and Wangdi 2015; Aryal et al. 2015). Since 2004, Bhutan made changes in the legal framework on the collection rights of Cordyceps, a highly priced medicinal fungus, to the high-altitude residents, most of which are cattle and yak herders. The improvement in the living standard meant that people gradually adopted and kept fewer numbers of high yielding crossbred cattle which require more specialised management at home (Wangchuk et al. 2014), compared to the traditional system of large numbers roaming freely in the forest and low productive farming. The positive effect on regeneration due to the reduction in grazing was supported by our data as many of the seedlings recorded in the grazed plots were recent recruitments (less than ten years old).

Unless manipulations in forest canopy and grazing are included in sustainable forest management plans, recruitment will continue to fail, which will determine the sustenance of brown oak forest in the Himalayas. Given the long-term failure of oak regeneration spanning over several decades and over the entire forests, there is an urgent need for a concerted effort to explore artificial regeneration. Raising seedlings from seeds in forest nurseries and restocking of the natural habitats has become imperative to save this species. There is future research needs to explore planting techniques, including those that reduce grazing damage and diversionary feeding (Putman 1996). Grazing management strategies like reducing the population and practising rotational grazing may be beneficial to provide “windows of opportunity” for tree regeneration (Didion et al. 2009).

3.7 CONCLUSION

The nationwide regeneration study of brown oak forests in Bhutan showed long-lasting recruitment failure of brown oaks, an urgent message to forest managers, conservationists, and policymakers to save this vital forest type. The study also confirms that the brown oak forests of Bhutan are no different from the neighbouring Himalayan countries like India and Nepal in terms of regeneration failure problems and needs regional partnerships in ecological research and restoration measures. Our results revealed that strict protection of oak forest might not enable or guarantee brown oak regeneration. Forest management should aim at opening gaps in the forests through felling of mature conifers and associated trees species while maintaining mature oak trees as a seed source. Most importantly, grazing manipulations need to be considered to protect oak seedlings and control unwanted weed growth. The use of tree shelters appears promising as an immediate measure to protect oak recruits from grazing and unwanted bush growth. The wide-scale chronic absence of individuals between young seedling and mature trees urgently calls for the need to restock oak forests through assisted or artificial regeneration and protection from grazing.

3.8 ACKNOWLEDGEMENTS

The study was conducted as a part of the PhD thesis of the first author for which the Australian Commonwealth Government's Research Training Program and the University of Adelaide are kindly acknowledged. The authors would like to thank the partners in Bhutan consisting of Director-UWICER, Dr Kaka Tshering, Dorji Gyaltshen, Karma, Damber Kr. Ghemiray, Cheten Thinley, Madan Kr. Lama, Pema Thinley and all colleagues at Conifer Forestry Research Sub-centre, Yusipang for their support in the fieldwork. The nationwide oak survey team comprised of Tsithup Tshering (UWICER), Kezang Dorji, Pema Rinzin & Ugyen Wangchuk (SWS), Tenzin Wangdi (Tashigang division), Dorji Lhamo (Mongar division), Sither Dorji & Karma Yeshi (WCNP), Sonam Tenzin & Sangay Drukpa (BWS), Nagphel (Haa Range) and D.P. Dhahal (JSWNP) are duly acknowledged for their support during the fieldwork. The funding support for the study was granted by the Royal Government of Bhutan, and the Forest Ecosystem Restoration initiative of the Korea Forest Service and Convention on Biological Diversity via grant SSFA/2017/86543.

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Chapter 4: Tree shelters facilitate brown oak seedling survival and establishment in a grazing dominant forest of Bhutan, Eastern Himalaya

4.1 STATEMENT OF AUTHORSHIP

Statement of Authorship

Title of Paper	**Tree shelters facilitate brown oak seedling survival and establishment in a grazing dominant forest of Bhutan, Eastern Himalaya**
Publication Status	☑ Published ☐ Accepted for Publication ☐ Submitted for Publication ☐ Unpublished and Unsubmitted w ork w ritten in manuscript style
Publication Details	Restoration Ecology Journal, Published online 23 April 2020, **DOI:10.1111/rec.13176**

Principal Author

Name of Principal Author (Candidate)	Tshewang Dorji		
Contribution to the Paper	Conceptualization, methodology, data collection and analysis, funding acquisition, writing, and editing reviewer comments		
Overall percentage (%)	70 %		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	20/4/2020

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);

ii. permission is granted for the candidate in include the publication in the thesis; and

iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Justin Brookes		
Contribution to the Paper	Conceptualization, Methodology design, fieldwork, data analysis, funding acquisition, overall supervision and editing and reviewing.		
Signature		Date	21 July 2020

Name of Co-Author	José M. Facelli		
Contribution to the Paper	Supervision, fieldwork, data collection, methodology and data analysis, editing and reviewing		
Signature		Date	4 August 2020

Please cut and paste additional co-author panels here as required

Name of Co-Author	Steven Delean		
Contribution to the Paper	Data sorting and analysis, editing and writing		
Signature		Date	31-7-2020

Name of Co-Author	TshewangNorbu		
Contribution to the Paper	Developing research protocol and data collection, investigation, writing and editing in association with the lead author		
Signature		Date	21.4.2020

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RESEARCH ARTICLE

Tree shelters facilitate brown oak seedling survival and establishment in a grazing-dominant forest of Bhutan, Eastern Himalaya

Tshewang Dorji[1,2,3], José M. Facelli[1], Tshewang Norbu[2], Steven Delean[1], Justin D. Brookes[1]

Brown oak (*Quercus semecarpifolia*) forest is essential for ecological and socioeconomic functions, mainly grazing in the Himalayas. The tree has failed to regenerate naturally and is a threatened species. Restoration of brown oaks is crucial to ensure sustainability while maintaining livestock grazing in these habitats. Achieving this requires cost-effective restoration techniques that are practicable and sympathetic to the multiple uses of the forest. We assessed the combined effect of grazing (control) and three tree shelters (Protex tubes, mesh wires, and wooden frames) on the field performance of oak seedlings in a forest with heavy grazing pressure. Seedling survival and morphological indicators, including seedling height, collar diameter, sturdiness quotient (SQ), and leaf mass per area (LMA) indices, were measured. More than 90% of control seedlings without protective shelters suffered severe browsing and demonstrated significantly lower survival rates compared to tree shelter seedlings, indicating that grazing was the primary factor governing regeneration success. Seedling survival in tree shelters was three times higher, while the height increase was two times higher than the control. Additionally, locally made mesh wire and wooden tree shelters were more effective than Protex and control in producing quality seedlings reflected by the SQ and LMA values. We suggest that tree shelter is a promising option to restore brown oaks due to its efficacy to defend grazing and support the local community's rights to forest grazing. Our finding is expected to support Bhutan's forest policy of incorporating grazing and tree regeneration into forest management.

Key words: brown oak forests, forest grazing, Himalayas, *Quercus semecarpifolia*, tree shelters

Implications for practice

- Traditional large-scale fencing in forest plantations needs to be re-evaluated as they are expensive, prone to weed invasion, and are against the sentiments of herders.
- Tree shelters offer the middle path to forest restoration and should be given special consideration to restore keystone species like brown oaks in the Himalayan countries.
- Tree shelter supports Bhutan's overall forest policy of incorporating cattle grazing into forest management to support rural livelihood with minimal bearing on forest regeneration.

Introduction

Bhutan has committed to remain carbon neutral for all time during the 2015 Paris climate summit and has pledged to reforest its barren lands and degraded forests (RGOB 2009). Sustained maintenance of present forest cover, as well as restoration of degraded ecosystems, are crucial to meet this pledge. Recently, realizing the importance of forest and associated benefits, forest restoration in developing countries has garnered substantial donor support from global forest restoration partnerships such as the Reducing Emissions from Deforestation and Forest Degradation (REDD+) (Alexander et al. 2011; Pandey et al. 2013). Research and exploration of "ecologically sound designs" (Alexander et al. 2011: p 683) and techniques that are practicable, cost-effective, and ecologically sound will form a core restoration strategy in the Himalayan countries.

Brown oak (*Quercus semecarpifolia* Sm.) is a major forest-forming species at higher mountain elevation (2,400–3,500 m asl) in the Himalayas (Singh et al. 1997; Saran et al. 2010). Because of their remote distribution and less human influence coupled with biological and structural complexity and unique ecosystem functions, they are often regarded as old-growth forests by researchers in the Himalayas (Tashi 2004; Dorji et al. 2018). The species falls within the evergreen oak forest classification of Bhutan and occupies about 31,464 ha of forest area,

Author contributions: TD conceived and designed the research; TD, JDB secured project funds; TD, TN performed the experiments; TD, SD, JDB, JMF analyzed the data; SD, JDB, JMF contributed analysis tools and software; all authors wrote and edited the manuscript.

[1]School of Biological Sciences, The University of Adelaide, South Australia 5005, Australia
[2]Ugyen Wangchuck Institute for Conservation and Environmental Research, Conifer Forest Research Sub-centre, P.O.Box 2049, Thimphu, Bhutan
[3]Address correspondence to Tshewang Dorji, email tshewangdorji1@gmail.com

doi: 10.1111/rec.13176
Supporting information at:
http://onlinelibrary.wiley.com/doi/10.1111/rec.13176/suppinfo

4.2 INTRODUCTION

constituting about 1 % of the total forest cover. These forests and their associated ecosystems are, however, degrading at an alarming rate caused by rapid population growth and economic development (Ives & Messerli 1989; Shrestha & Paudel 1996). Inadequate natural regeneration and recruitment by this species were reported throughout the Himalayas (Singh & Singh 1987; Shrestha 2003; Tashi 2004; Dorji 2012; Singh & Rawat 2012; Covey et al. 2015). Many of these forests are gradually being replaced by conifer species (Singh et al. 1984; Naudiyal & Schmerbeck 2017), which is a concern for ecologists and forest managers. Some explanations suggested for the poor regeneration of this species are unreliable seed production and seed masting (Singh et al. 2011), short-lived seeds, acorn predation (Singh & Singh 1987; Shrestha 2003) and seed vivipary (Singh 2014). Laboratory experiments found high germination and seedling establishment under favorable moisture and light conditions. The best germination occurred at a temperature of about 15°C (Bisht et al. 2012). In the natural environment, the species prefers a mean annual rainfall in the range 1000 – 2500 mm, mean annual temperature of 5 – 17 °C, and dry season spanning not more than 4 - 6 months (Tashi 2004). The trees are known to produce abundant seeds every year with seed masting every 8 – 10 years (Singh et al. 2011). The seeds are recalcitrant and are short-lived. Germination of seeds takes place profusely immediately after falling on the ground during the early rainy season (June – July). Germination of seeds before falling (partial vivipary) is also observed. The young seedlings prefer moist sites with well-drained loamy soil and partial shading. At the initial phases, the seedlings grow at a slow rate averaging 5 - 10 cm in height per year with records of repeated dieback of top shoots for the few years until new and more vigorous shoots are produced (Larsen & Johnson 1998; Tashi 2004). Despite the abundant seed production and profuse germination, the seedling establishment under natural conditions has remained poor (Singh et al. 1997; Shrestha et al. 2004; Covey et al. 2015). Slow growth, excessive lopping of trees (Shrestha & Paudel 1996; Singh et al. 1997), and climate change (Bisht & Kuniyal 2013) are some other factors believed to affect seed production and germination directly. Studies carried out in different parts of the Himalayas related the recruitment failure to overgrazing (Singh & Pande 1989; Shrestha & Paudel 1996), insufficient canopy gaps (Shrestha 2003), or combination of both (Singh et al., 1997).

Vetaas (2000) reported that the shaded forest floor provides favorable conditions for regeneration, which contradicted with other studies that suggest wider canopy gaps for recruitment success (Singh et al. 1997). Tashi (2004) proposed a mixture of closed canopy for early seedling establishment and canopy gaps combined with low grazing pressure in later stages of plant growth. Long-term regeneration studies in Bhutan, however, showed poor regeneration and failure of the seedling establishment even after the manipulation of canopy gaps through selective logging (Covey et al. 2015). Nevertheless, the exclusion of large herbivores through exclosures resulted in higher oak sapling density (Dorji 2012). It appears that seedling recruitment to adult tree size is prevented by repeated grazing and browsing.

Although attempts to exclude livestock in the plantation areas using barbed wire fences are frequently utilized in reforestation projects, the practicality of fencing large areas is limited by both funding and montane terrain. Further, when grazing is excluded in forests, it is often accompanied by rapid colonization of impenetrable bamboo thickets, increasingly subjecting seedlings to habitat modification and intense competition for light and resources (Darabant et al. 2007; Dorji 2012). Consequently, natural regeneration and the establishment of brown oak have remained poor. The situation requires intervention in the form of artificial regeneration involving direct seeding or planting of seedlings raised in nurseries and their protection from grazing (Shrestha & Paudel 1996).

Tree shelters are widely used to guard plants against animal grazing until the plants reach a height safe from herbivory. Tree shelters made of plastic tubes are commercially available. The heavy-duty mesh tree shelters are also increasingly used to avoid undesirable modifications in the growing environment induced by plastic containers. Although human-assisted mass plantations and protection from herbivory using tree shelters have been applied to restore oak woodlands in developed countries (McCreary 2011; Jacobs 2001; Kittredge 1992), it has never been implemented or studied in Himalayan countries. Further, most of the studies focussed on low altitude oaks (Davies 1985; Dubois et al. 2000; Devine & Harrington 2008; Garcia et al. 2011), which differ significantly from the high-elevation Himalayan oaks. Generating information on the suitability of tree shelters to restore brown oak within its habitat is crucial for the conservation of this species. In the Himalayas and particularly in Bhutan, the technique can provide several advantages. Firstly, it will promote the growth of the desired species. Secondly, the method accommodates grazing animals around the tree shelters, thereby improving the rural livelihood of herders and farmers. Finally, grazing animals are known to control the overgrowth of unwanted vegetation in Himalayan forests (Darabant et al. 2007), which will help in weed management and improve pasture quality in restored forests (Ian & Roger 1996). Tree shelters, therefore, can provide a middle path to restoration strategy. The conventional method of fencing and planting large tracts of reforested areas are expensive and is against the sentiments of herders and local communities who use these forest as grazing grounds. This study aimed to evaluate the effect of different tree shelters on the growth and survival of oak seedlings while letting grazing practices to continue as usual. The information generated will inform decision-making processes in terms of adopting ecologically and socially viable techniques to restore the Himalayan old-growth brown oak forests. The study was conducted from 2014 to 2019.

4.3 METHODS

4.3.1 Study area descriptions

The study was conducted in an old-growth oak forest of Chimithankha, Western Bhutan (27°26'13.1"N; 89°30'43.2"E) at an altitude of 2952 m asl (Figure 22). The forest comprises of mature brown oak trees (*Q. semecarpifolia* - an evergreen oak) mixed with few conifers: hemlock (*Tsuga dumosa)*, fir (*Abies densa)* and spruce (*Picea spinulosa)* in the upper canopy (25-30 m). The middle storey (10-20 m) is composed of rhododendrons (*Rhododendron arboreum*, *R. barbatum*), maple (*Acer* spp.), ilex (*Ilex depryena*), *Corylus ferox* and birch trees (*Betula utilis)*. The understorey was dominated by bamboos (*Yushania microphylla*), *Pieris formusa, Rosa spp., Berberis asiatica, and Daphne bholua.* The mean annual temperature measured from 2015 to 2018 was 8.5℃, with a mean yearly rainfall of 800 mm occurring mostly in the summer monsoon season (June - September). The soils in the study site were well-drained, very deep, dark greyish brown silty clay loam to sandy clay loam in texture. The topsoils were very stable, with no visible signs of erosion, due to the presence of good vegetation cover, thick leaf litter, and high organic matter, which acts as a binding agent. Grazing is predominantly practiced in the area by migratory yaks and cattle in the winter (December to February) and summer (May-August), respectively. The local cattle and wild herbivores graze throughout the year (Fig. S1).

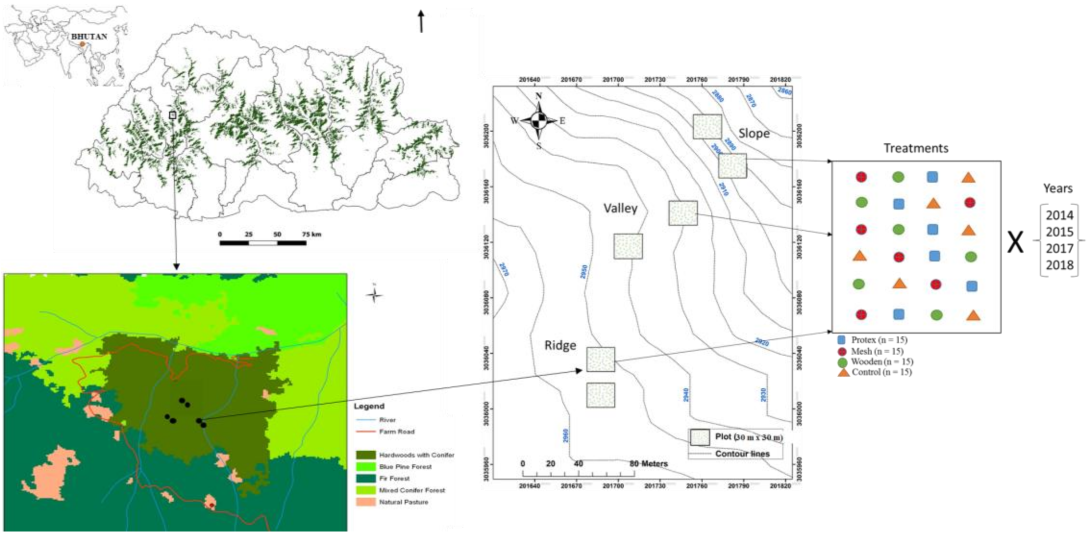

Figure 22. Map of Bhutan showing the location of the study area along with different forest types. The diagrammatic representation presents the hierarchical organization of experimental plots nested within locations (ridge, valley, and slope) and treatments nested within the plots along with the repeated data measurement years.

4.3.2 Experimental design and seedling measurements

In 2014, we planted 530 three-year-old brown oak seedlings, which were raised in poly-pots in an open-air nursery of the Conifer Forest Research Centre, Department of Forests and Park Services, Bhutan. The nursery was located at an elevation of 2600 m above sea level with a mean minimum temperature ranging from −0.7°C in winter (December – February) to 15.4°C in summer (May-July) while the mean maximum temperature ranges from 4.5 °C in winter to 19.1°C in summer. The mean annual rainfall recorded from 2007 to 2017 was 860 mm. In the nurseries, seedlings were raised under similar climatic and growing conditions, from the same seed source and in a similar potting mixture of sand (1 part), soil (1 part), and manure (2 parts) and watered regularly. The seedlings were planted manually in the forest with the earth surrounding the root ball (after removal of poly-pots) in pits of the size of 45 cm x 45 cm at a depth of 30-35 cm and spacing of 2 m x 2 m. The four treatments used were control (i.e., no shelter) and three tree shelter types made from different materials (Fig. S2). The three types of tree shelters were commercially available Protex tubes (18" Protex Pro/Gro Solid Tube Tree Protectors, Forestry Suppliers) (referred hence as 'Protex'), and two cost-effective structures constructed from locally available materials; iron-mesh wires (referred as 'mesh'), and wooden frames (regarded as "wooden").

We planted the oak seedlings in two randomly positioned plots (30 m x 30 m) on each of the ridge, valley, and slope to account for the effects of local topography on seedling survival and growth (Figure 22). The four treatments described above were then randomly assigned to the individual seedlings. The trials faced NW and N aspects with moderate slopes ranging from 17 to 33 % (Table 10). Each plot consisted of about 60 seedlings, with 15 replicate seedlings of each treatment distributed across the plots at a location. To locate the seedlings during yearly monitoring and data collection, the seedlings were permanently demarcated and tagged with individual identification numbers. Seedling loss because of tree/branch fall and animal interference meant that for the final data set, each location consisted of 54-63 seedlings, and these were distributed unevenly, with plots containing 16-44 seedlings each. A total of 177 seedlings were therefore used for this study. All treatments were replicated within each plot over all years, with the exception that there were no control treatment seedlings in one of the plots in the ridge location in the final year of sampling. Data on survival and growth parameters were collected at four-time points (2014, 2015, 2017, and 2018).

Table 10. Site description of the blocks and measurement of environmental variables.

Block Names	**Altitude (m)**	**Aspect**	**Slope (%)**	**Canopy opening (%)**	**Soil Moisture Content (%)**	**Soil Hardness (mm)**
Valley	2950	NW	17	34.1	28.56	19.8
	2953	NW	18	48.6	17.95	21.2
Ridge	2957	N	21	28.1	18.44	19.2
	2974	N	22	10.8	9.86	19.17
Slope	2976	NW	23	22.2	10.86	17
	3010	NW	33	25.5	16.26	14.4

Data on survival and growth parameters were collected at four times (2014, 2015, 2017, and 2018). Seedling height (Ht, cm) and collar diameter (0.5 cm above the cotyledon scar, Cd, mm) were measured and recorded with measuring tape and digital caliper at an accuracy of 0.1cm and 0.1 mm respectively (Fig. S4). The total number of leaves on each seedling were counted manually and recorded. During the 2018 fieldwork, we also randomly sampled leaves from seedlings growing under different treatments. A total of 187 leaves were sampled from different growing positions of the seedlings and were scanned using a flatbed scanner set at a dpi of 300.

The leaf area of individual leaves was measured using software LIA32 available free online (https://www.agr.nagoya-u.ac.jp/~shinkan/LIA32/). The individual leaves were measured for their fresh weight (g) and oven-dried weight (g) after drying the leaves in an oven set at 65 °C for 48 hours or until a constant weight was obtained.

4.3.3 Measurement of Environmental variables

The air temperature and relative humidity inside the Protex tubes were measured and compared to outside conditions by two HOBO Onset data loggers (Onset Computer Co. MA, USA). As the data loggers outside were housed in protective HOBO shields (similar to placing a logger inside wooden and mesh shelters), we did not install loggers inside the mesh and wooden shelters. We, instead, assumed that environmental conditions inside wooden and mesh shelters would be comparable to the outside logger due to similar ventilation and flow of gaseous exchange between the two setups. The loggers were set at 1-h measurement intervals and downloaded every six months using BoxCar Pro for Windows, Version 4.3 (Onset Computer Co.). Soil moisture content was measured using hydrosense (Campbell Scientific Inc., Logan, Utah) and soil hardness by a push cone (Yananaka's soil hardness tester, Kiya Seisakusho, Tokyo). Canopy opening (percent cover) was measured by hemispherical photographs taken by a digital camera (Canon 60d) fitted with a fisheye lens and estimated using LIA32, available online at http://hp.vector.co.jp/authors/VA008416.

4.3.4 Data Analysis

Survival analysis of the seedlings was conducted using the non-parametric Kaplan-Meier estimate (Kaplan & Meier 1958), a powerful tool to study "Time-To-Event (TTE)" data as it accommodates both censored (case in which seedlings survive until the final recording) and event observations (where hazard or a dead event occurred) (Scherm & Ojiambo 2004). Kaplan-Meier survival analysis compared the cumulative survival proportion against time (years) between the treatments. For seedling quality, we used two indices; sturdiness quotient (SQ) to assess the seedling's growth performance in the field (Haase 2008; Tsakaldimi et al. 2012) and leaf mass per area (LMA) as the ability to withstand stress and survive (Puglielli et al. 2015). Seedling sturdiness quotient (SQ) is measured by dividing the seedling height (cm) by the root-collar diameter (mm), which provides a quick and reliable estimate of seedling quality in the restoration programs (Equation 1). SQ assesses the stocky or spindly nature of growing stock and is a handy tool to predict higher survival in the field after planting. Spindly seedlings will have a higher height to collar diameter ratio, while stocky seedlings will have a lower height to root ratios. In general, a ratio of less than six is considered desirable to achieve higher plantation success (Roller 1977). Therefore, SQ can be useful to assess quality seedling and to determine the plant's survival and ability to withstand physical damage (Thompson 1986). The LMA is the ratio of leaf dry weight (g) to leaf area (cm^2, equation 2) and is an essential indicator of seedling functioning such as photosynthesis, respiration, chemical composition, and resistance to herbivory (de la Riva et al. 2016).

$$SQ = \frac{\text{Seedling height (cm)}}{\text{collar diameter (mm)}} \dots \dots \dots \dots (1)$$

$$LMA = \frac{\text{Leaf dry weight, (g)}}{\text{Leaf surface area } (mm^2)} \dots \dots (2)$$

4.3.5 Statistical analysis

Kaplan-Meier survival analysis was performed using the R software (CoreTeam 2019). As the survival responses to treatments in all the locations (Ridge, Valley, and Slope) showed similar trends, data were pooled together, and seedling survival was compared among treatments. A log-rank test (Mantel-Cox) was used to determine significant differences in the survival distribution due to different types of treatments at $p < 0.05$. Tukey's range test was deployed to conduct pairwise comparisons.

For analysing the treatment effects on longitudinal growth data (height, collar diameter, and sturdiness quotient) recorded over the years, we used linear mixed-effects models (lme4 package; (Bates et al. 2015)) in R (CoreTeam 2019). The location (α_i), shelter treatment (γ_k), and year (time of sampling; δ_m) were entered as fixed effects in the model. Plots nested within locations ($b_{1,j[i]}$) and seedlings nested within the plots ($b_{2,l[j[i]k]}$) were entered as random effects to account for variation in the hierarchical groupings of the locations of samples and the repeated measures on individual seedlings through time. We also included random effects for the interactions of the plot level variation with the treatment and year effects. The linear mixed-effects model that was fitted separately for each of the dependent variables (y_{ijklm}) can be represented as follows:

$$y_{ijklm} = \mu + \alpha_i + b_{1,j[i]} + \gamma_k + \alpha\gamma_{ik} + \gamma b_{1,j[i]k} + b_{2,l[j[i]k]} + \delta_m + \delta\alpha_{im} + \delta b_{1,j[i]m} + \gamma\delta_{km} + \alpha\gamma\delta_{ikm} + \gamma\delta b_{1,j[i]km} + \epsilon_{j[i]klm},$$

where μ is an intercept term, and $\epsilon_{j[i]klm}$ is the residual error term, and the subscripts (i,j,k,l,m) represent the number of levels of each categorical factor in the model.
Log transformation of the data was carried out when required to meet the statistical assumptions of normality and homoscedasticity or errors. The normality of residuals was checked using the constant variance of residuals (examined with the residual diagnostic plot of model fitted values against model residuals) and normal quantile-quantile (q-q) plots. Satterthwaite and Kenward-Roger approximations were used to determine denominator degrees of freedom for the F-tests of mixed-effect model terms. We reported results from both tests because the imbalance in the number of replicate seedlings per treatment across plots resulted in slightly different inferences for the highest-order interaction term (treatment*location*year) for some response variables, with the Kenward-Roger approximation providing a more conservative test. Model estimates of the response variables were back-transformed to their raw scale for plotting and reporting.

For the leaf morphology data, which was collected for only one field season (2018), we used a two-way ANOVA to estimate mean differences among treatments on different locations. All post-hoc comparisons were made using Tukey's honestly significant difference (HSD).

4.4 RESULTS

4.4.1 Seedling survival

A log-rank test (Mantel-Cox) conducted to examine the differences in the survival distribution showed statistically significant results for the different interventions, X^2 (3, N = 138) = 68.8, p < 0.001. All tree shelter types demonstrated significantly higher survival rates than control (Figure 23). The oak individuals in the mesh tree shelters (N = 36, N of events = 1, survival = 97.3 %) displayed significantly higher survival rates than wooden (N = 41, N of events = 11, survival = 73.2 %), Protex (N = 76, N of events = 29, survival = 61.8 %) and control groups (n = 52, n of events = 40, survival = 26.3 %). The wooden and Protex tree shelters also exhibited higher survival rates after mesh. The wooden and Protex tree shelters did not differ significantly in their survival rates (X^2 = 0.967, p > 0.05). The survival curve for the control group was the lowest reflecting the greater browsing damage and significantly lower survival rates of individuals in this group (Figure 23).

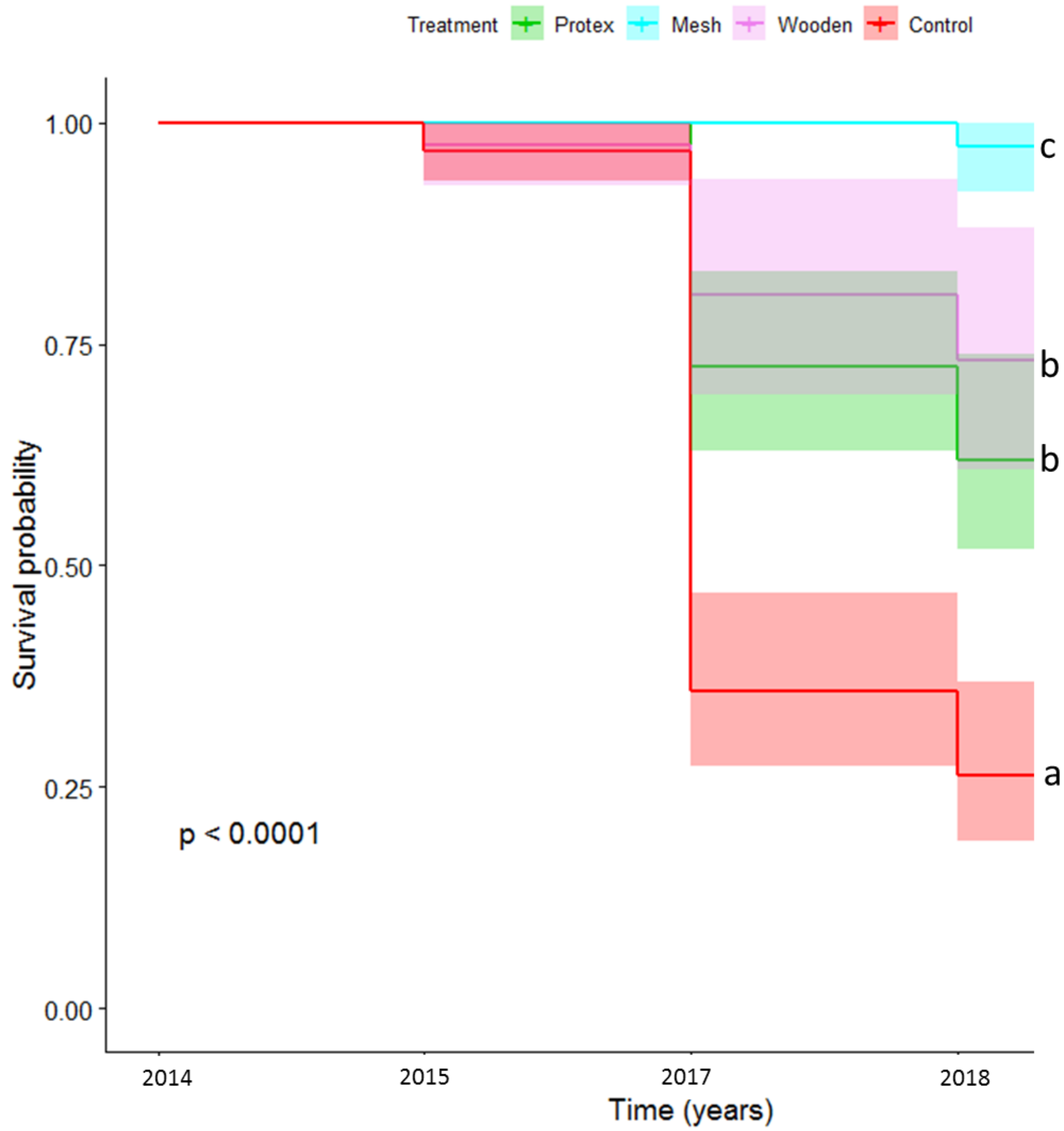

Figure 23. The Kaplan-Meier survival curves and the 95 % confidence intervals (shaded area) shows the survival probability of different treatment groups against time (in years). Different letters indicate significant differences at $p<0.001$ (log-rank test) for various treatments.

4.4.2 Seedling growth parameters

The mixed-effects model fitted separately for each dependent variable; height, collar diameter, and SQ showed significant main effects as well as treatment-time interactions (TableS 1). There were clear differences between treatments and over time, as well as their interaction for all the three dependent variables (TableS 1), indicating the considerable impacts of tree shelters on seedling growth over the years. Although there were no clear differences in seedling height between locations, there was some evidence for differences at the level of the three-way interaction between location, treatment, and time ($F_{18,342}$ = 2.36, p = 0.002). A similar result was evident for the sturdiness quotient variable ($F_{18,380}$ = 1.75, p = 0.03). However, it is worth noting that the more conservative Kenward-Roger test yields non-significant results for the highest-order interaction term for the height and sturdiness quotient variables, respectively ($F_{18,13}$ = 2.28, p = 0.067 and $F_{18,13}$ = 1.72, p = 0.159), suggesting that the patterns are not clear.

The mean height differed among treatments, and these differences depended on the year of measurement and location (Table 11). However, the effect of treatment, year, and treatment*year interaction were highly significant on the height growth of seedlings (Table 11). Tukey's post hoc test showed that the mean height, collar diameter, and SQ of seedlings at the beginning of the study (2014) was similar in all treatments ($p > 0.05$, Table 11). Four years later, in 2018, the mean height of seedlings in tree shelter groups more than doubled (average height = 53 cm) and were significantly different from the control seedlings (average height = 26.2 cm; $p < 0.001$; Table 11). In general, there was a significant increase in the height growth of the control seedlings and tree shelter seedlings in the later part (3- 4 years) of experiment initiation on all locations of ridge, valley, and slope (Figure 24a).

Collar diameter increased over time ($p < 0.001$) at all locations except for the ridge where growth remained fairly constant after 2015 (p = 0.02, Figure 24b), resulting in a significant effect of location in the model (TableS 1). Top dying of old shoots and rapid emergence of spindly new shoots was observed in the Protex treatment after 2015, resulting in significantly lower collar diameter compared to other treatments (p = 0.02, Figure 24b). Overall, seedling collar diameter differed between tree shelter treatments ($F_{3,122}$ = 4.16, p = 0.007) over the years ($F_{3,6.8}$ = 141.77, p < 0.001) and treatment over time interactions ($F_{9,366}$ = 4, p < 0.001). Collar diameter grew by more than two times in the mesh and wooden tree shelters followed by one and half times and by two times in control seedlings and Protex seedling, respectively, over the four years (Table 11). The mean SQ of seedlings in 2014 ranged from 6.93 to 7.12 and was not significantly different between the treatment groups ($p > 0.05$). There was a small decrease in the SQ value, a year after planting (2015), indicating time taken by the seedlings to establish on the mineral soil (Figure 24c). These values changed dramatically over four years ($F_{9,8.8}$ = 62.6, p < 0.001) and were significantly different between the treatments in 2018 ($F_{3,7.7}$ = 11.4, p = 0.003; Table 11 and Figure 25).

Table 11. Estimated means for the growth parameters (± SE) compared between the treatments at the start of the experiment (2014) and four years after the experiment (2018). No significant differences in growth parameters were observed among the treatments in 2014. Therefore, different letters indicate significant differences in the year 2018. A pairwise comparison was computed using Tukey's HSD.

Growth parameters	**Year**	**Control**	**Protex**	**Mesh**	**Wood**
Height (cm)	2014	19.30 ± 1.18	21.49 ± 1.30	22.89 ± 1.53	19.22 ± 1.13
	2018	26.18 ± 2.52 ^{a}	51.36 ± 3.70 ^{b}	58.83 ± 3.97 ^{b}	51.26 ± 3.80 ^{b}
Collar diameter (mm)	2014	2.80 ± 0.17	3.09 ± 0.18	3.20 ± 0.21	2.76 ± 0.18
	2018	5.54 ± 0.54 ^{ab}	4.91 ± 0.35 ^{a}	7.46 ± 0.50 ^{b}	6.58 ± 0.49 ^{b}
SQ	2014	6.93 ± 0.36	7.00 ± 0.36	7.12± 0.41	6.93 ± 0.39
	2018	4.57 ± 0.46 ^{a}	10.84 ± 0.74 ^{b}	7.84 ± 0.45 ^{c}	7.69 ± 0.53 ^{cd}

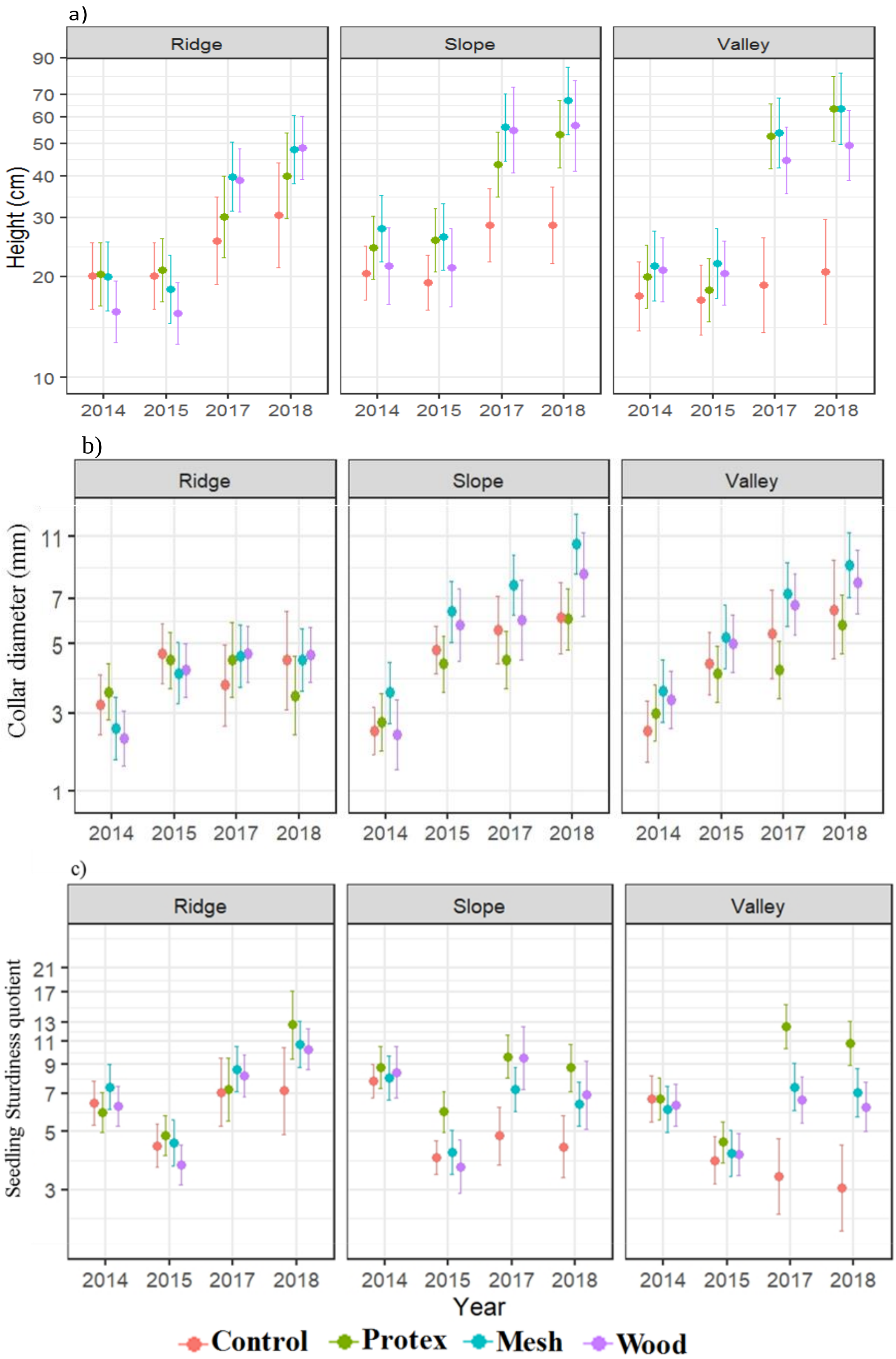

Figure 24. Interaction plots of variations in a) height, b) collar diameter, and c) SQ of oak seedlings by interventions (treatment) at different locations (ridge, valley, and slope) over time (years from 2014 to 2018). Bars represent the 95% confidence intervals.

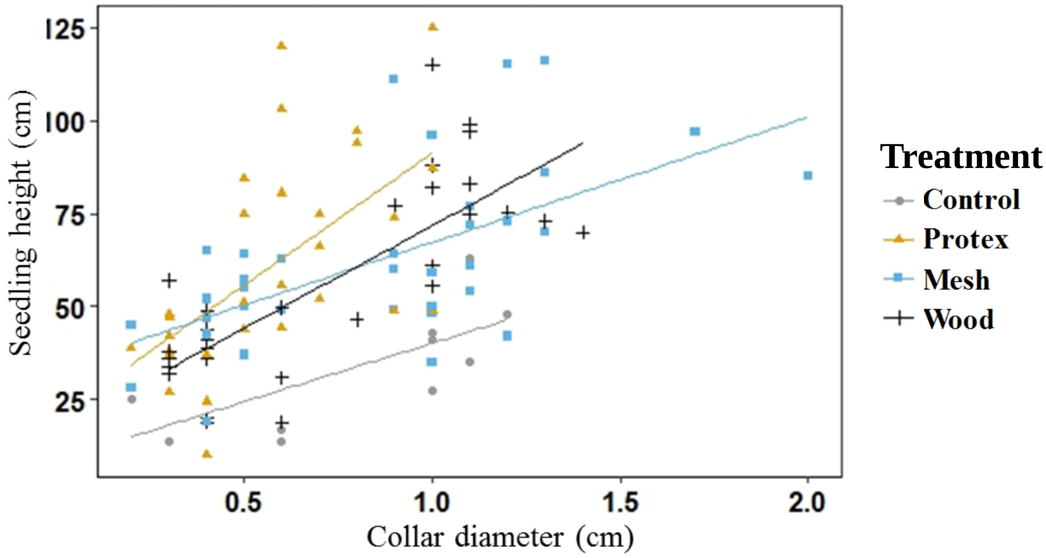

Figure 25. Seedling height to collar diameter ratio of different treatments after four years of plantation in the field.

4.4.3 Leaf morphology and LMA

There was a significant difference in the mean number of leaves per seedling between the treatment groups ($F_{3,191} = 26.7$, $p < 0.001$, two-way ANOVA, Figure 26A). Pairwise comparisons revealed that seedlings in mesh shelters have a significantly higher number of leaves compared to other treatments. Seedlings in wooden tree shelters followed the mesh tree shelter with a significantly higher number of leaves than Protex and control. No significant differences in the number of leaves per seedling were observed between Protex and control types ($p > 0.05$; Figure 26A). The effect of tree shelters or location on oven-dry weights of leaves was also not significant ($p > 0.05$, two-way ANOVA, Figure 26B). On the other hand, the leaf area ranged from 13.4 ± 1.8 cm^2 to 22.1 ± 6.5 cm^2 (mean ± SE), which differed significantly among the treatment groups ($F_{3,28} = 3.13$, $p = 0.04$, two-way ANOVA, Figure 26C), but not with location differences ($p > .05$, two-way ANOVA). The pairwise analysis showed that the leaf area of Protex seedlings (22.1 ± 6.5 cm^2, mean ± SE) was significantly larger than the control seedlings (13.4 ± 1.78 cm^2, $p = 0.005$, Tukey HSD test) and mesh tree shelters ($p = 0.005$, Tukey HSD test, Figure 26C). Variations in leaf area consequently led to significant differences in the LMA ratios between the treatment groups ($F_{3,28} = 7.52$, $p < 0.001$, two-way ANOVA, Figure 26D). No differences in LMA values due to locations were observed ($F_{2,28} = 0.42$, $p > .05$, two-way ANOVA, Figure 26C). The Protex group (93.6 ± 5.5 g/m^2) depicted significantly lower LMA values compared to control (127.6 ± 9.4 g/m^2, $p < 0.001$, Tukey HSD test), mesh (127.3 ± 5.3 g/m^2, $p < 0.001$,) and wooden groups (116.8 ± 3.6 g/m^2, $p = .005$, Figure 26D). No significant difference was observed in the LMA values of control, mesh, and wooden treatments ($p > 0.05$, Tukey HSD test, Figure 26D).

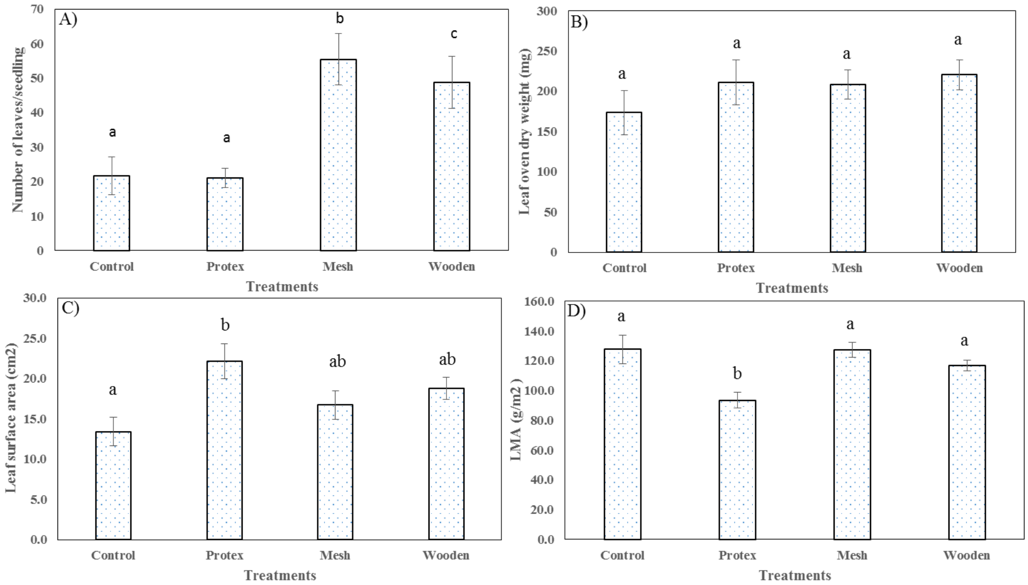

Figure 26. Comparison of leaf morphological features. A. The number of leaves per seedlings in different interventions, B. Dry weight of leaves of seedlings from different treatments, C. Leaf area measurements from different treatments, and D. Leaf mass per area values of seedlings from different treatments. Different letters indicate a significant difference at p<.05, one-way ANOVA. Bars represent the standard error of the mean.

4.4.4 Comparison of the incidences of grazing intensity

During the four years of observation, there was strong evidence of herbivory on the seedlings of *Q. semecarpifolia* in the control treatment without protection (Figure 27a). Of the surviving control individuals, 66.7 % were heavily browsed, 25 % were lightly browsed, and about 8.3 % were not browsed (Table 12). The control individuals that demonstrated light or no browsing damage were the ones protected from herbivory by thorny neighbors (e.g., *Berberis asiatica and Rosa* spp.), which grew and surrounded the seedlings during the experiment period (Figure 27e). The most common form of grazing damage was browsing of the new shoots (from the top), which were tender and less spiky. Grazing incidence was lower in the wooden tree shelters; however, 13.8 % of individuals still suffered heavy browsing and 10.3 %

moderate browsing. No signs of grazing were observed on the seedlings in the Protex and mesh tree shelters.

Table 12. Grazing intensity incidence recorded on surviving seedling for different treatments in 2018 (%); heavily grazed = Top leader shoots and all foliage grazed; Grazed = Leader shoot intact with no or little foliage; no grazing= top shoot and foliage intact with no visible signs of grazing.

	Observation of grazing intensity in 2018		
Treatments	Heavily grazed (%)	Grazed (%)	No grazing (%)
Control	66.7	25	8.3
Protex	0	0	100
Mesh	0	0	100
Wooden	13.8	10.3	75.9

Figure 27. Representative photographs show varying grazing evidence on oak saplings in different treatments. a) Severely grazed oak sapling without a top shoot and foliage, b) No grazing observed on sapling growing in Protex tubes; however, saplings were tall and slender, c) Saplings in iron mesh grew healthy without grazing signs, d) Saplings growing in wooden tree shelters, and e) Seedlings (shown in red circle) protected by thorny neighbours were protected from grazing.

Figure 28. Oak seedlings were grazed to a stick-like appearance when tree shelters were removed.

Nevertheless, heavy grazing occurred on seedlings when the tree shelters were removed (Figure 28). No noticeable signs of insect herbivory and rodent damage (basal girdling) on the seedlings were observed during this study. Because animals actively grazed the bamboos and shrubs around the tree shelters, no overgrowth of unwanted vegetation (bamboos and shrubs) occurred around the tree shelters during this study. No weeding in and around the interventions was, therefore, required during the entire experiment period. In larger and fenced grazing exclusion plots adjacent to our experiment, we did observe strong dominance by bamboo and shrub thickets (Fig. S5).

4.4.5 Comparison of environmental variables

A maximum temperature of 33.2 °C was observed in Protex tubes when the maximum outside temperature was 25.9 °C in one afternoon of June 2018. Significant variations in summer temperature between inside Protex tubes and outside were observed from noon, which continued up to 5 pm, possibly due to heating by the sun and lack of air movement inside the solid-walled Protex. The maximum temperature inside the Protex tubes during this time averaged 27.3 ± 0.8 °C (mean ± SE) and was significantly higher than the outside temperature 24.8 ± 0.5 °C (mean ± SE) ($p < 0.05$, $F= 6.640$, one-way ANOVA, TableS 2).

Conversely, a low temperature of -7.7 °C was observed in the early morning of January 2017 inside the Protex tubes when the outside minimum temperature was -6.7 °C. The minimum temperatures were recorded in winter months of December and January between 5 am to 8 am and ranged from -7.3 to -7.7 °C and -5.4 to -6.7 °C for Protex and outside, respectively. Although there were differences in the daily minimum winter temperatures, statistically, there was no evidence of a significant difference between the two comparisons (TableS 2). Relative humidity (RH), on the other hand, appeared to be influenced by temperature changes. During the summer, when temperatures were high, RH was significantly lower inside the Protex compared to RH outside (TableS 1). In the winter months, when the temperature was lower, RH inside the Protex was significantly higher than the RH outside (TableS 2). Mean soil moisture content in the Protex was substantially higher than outside. In contrast, mean soil hardness was significantly lower inside the Protex than the outside environment (TableS 2).

4.5 DISCUSSION

We have demonstrated that herbivory is a primary factor determining the success of regeneration and the establishment of oak seedlings in the Himalayan old-growth oak forests. About 90 % of the seedlings growing without protection were severely grazed, and seedling survival dropped to about 23 % during our study. We confirmed that the use of tree shelters techniques that deter herbivory could enhance survival and establishment of oak seedlings, indicating their potential to provide a long-term solution between forest managers and herders and should be given due consideration in restoration projects.

Elsewhere, the views on tree shelters are varied, depending on the design and make of tree shelter itself and the local site factors. For instance, studies recommended the height of tree shelters of at least 5 ft (Kelty & Kittredge 1986) to protect seedlings from large herbivores. Similarly, Keeton (2008) observed that seedlings inside shorter tree shelters (60 cm height) were top-browsed by deer and recommended taller shelters for adequate protection. In our study, we used taller tree shelters (about 1.55 m), assuming that the tree growth above or over-topping this height would be safe from grazing animals. During our study, the height of plants was much lower than the tree shelters, and as a result, no top browsing of seedlings was recorded. Nevertheless, modifications in the height of tree shelters may be required based on the severity of grazing on the shoots and branches overtopping the shelters.

All types of tree shelters used were effective in protecting grazing damage. Oak seedling survival, as well as height-growth, increased consistently with protection from grazing. The rate of survival of the seedlings in the three tree shelters was 3-4 times higher than the unprotected control seedlings. Heavy grazing incidences occurred to the control seedlings, which resulted in higher mortality. No record of girdling by rodents was observed during the study, although studies elsewhere have indicated browsing and girdling as an important factor for seedling mortality (Keeton

2008). Wood structures, however, allowed for some side-browsing, presumably by deer, due to wider gaps between the frames used in tree shelter construction, therefore reducing their effectiveness to some extent. Plant survival was much higher in the mesh (97.3 %) compared to the wooden shelters (73.2 %) and Protex tubes (60.4 %). The open gaps and ventilation in the mesh and wooden shelters possibly maintained similar temperatures and gaseous exchange with the growing habitat, thereby attenuating adverse microenvironment effects such as those produced by Protex (Sharew & Hairston-Strang 2005). The drawbacks and benefits of plant microenvironment modifications by Protex tubes remain controversial worldwide. For example, Kelty & Kittredge (1986) observed that the solid plastic material used in tree shelter construction provides more growth advantages to the seedlings by altering the microenvironment in addition to protecting from browsing, a foundation on which tree shelters became widely used in forestry practices in Britain. While on the other hand, other researchers observed the harmful effects of altered growing conditions on plants; for example, the opaque nature of Protex greatly reduced the light transmissivity (Sharew & Hairston-Strang 2005), and the impermeable material elevated CO_2 concentrations (Garcia et al. 2011). Based on our study, the temperature and humidity within the Protex changed widely compared to natural conditions. For instance, we observed high temperatures with low humidity during summers and prolonged extremely low temperatures during winters inside the Protex tubes. We assumed that repeated dieback and resprouting of top shoots of seedlings inside the Protex tubes could be a response to seedling desiccation induced by the unfavourable microclimatic conditions. For seedlings that did not exhibit die-back symptoms, height growth was accelerated relative to the slower collar diameter growth leading to spindly seedlings. The translucent nature of Protex tubes, which allows low light transmission could have promoted the rapid height growth of seedlings at the expense of collar diameter growth (Sharew & Hairston-Strang 2005). In such cases, seedling sturdiness was highly compromised, which is indicated by high SQ values (height over collar diameter) and lower field survival, especially in dry or windy areas. Roller (1977) stressed the importance of lower SQ values, which should parallel diameter for ensuring better field survival and plantation success. The Protex seedlings, as a result, were less sturdy, delicate, and depicted lower survival rates. Most of the seedlings were unable to stand without support and were less desirable in plantation programs as they were prone to wind and physical damage.

From our study, seedlings in mesh and wooden structures were morphologically (and most likely physiologically) much healthier (due to desirable SQ and LMA values) than those in Protex. The more vigorous seedling, together with higher survivorship, indicates that mesh and wooden tree shelters are preferable to Protex. The lower SQ values of control seedlings is a direct interference from repeated browsing, which reduced height growth but had no effect on collar diameter. The height of unprotected oak seedlings was reduced to half, while collar diameter grew without any interference, which is in line with the findings of other herbivory studies (Stange & Shea 1998; Dorji et al. 2015).

Our results are in line with studies by Sweeney & Czapka (2004), which recommend herbivory protection as the top priority in forest restoration and with Ward et al. (1999), who stress the importance of planting higher quality seedlings in combination with tree shelters to achieve good restoration success. The added advantage of using tree shelter and grazing simultaneously is the reduced effort in weed control. The unwanted vegetation like grasses and bamboos growing around the tree shelters were controlled by grazing animals and was of little concern in our study. Subsequently, we did not focus on weed control in the experiment. Our views on weed control partly agree with other researchers like Lantagne et al. (1990) and Dubois et al. (2000), who found no effect of weed control on oak seedlings in a tree shelter experiment. However, where grazing pressure is low to control the unwanted vegetation, the benefits of weed control in tree shelter experiments are documented (Garcia et al. 2011). We observed that livestock grazed almost everything on the ground that is palatable along with unprotected oak seedlings. We also found strong incidences of grazing when tree shelters disappeared from the seedlings due to unknown reasons. For instance, few of the mesh and Protex tree shelters that vanished from the seedlings in the observation plots in 2017 were not replaced. The following year in 2018, heavy grazing of the seedlings occurred, which reduced the seedlings to a sticklike appearance with very little foliage. Canham et al. (1994) reported that seedling death might not happen by heavy grazing for one to two seasons, however, when large herds of animals repeatedly graze throughout the year, it can lead to complete depletion of seedling foliage, decrease the photosynthetic ability of the plants (Liu et al. 2016) and eventually lead to seedling mortality. Based on field observation and in line with the above studies, we assumed that severe browsing of top shoots and foliage leading to total depletion of photosynthetic parts has resulted in higher mortality rates of control seedlings in our study. We believe that repeated grazing and damage by large herds of grazing animals can lead to complete oak regeneration failure (Trumbull et al. 1989).

Nevertheless, we observed that few seedlings in the control treatment (about 8 %) escaped more serious grazing damage when they were growing in association with thorny and unpalatable shrubs (*Berberis asiatica, Rosa sericea*). These unpalatable shrubs mimicked tree shelters in nature and offered the benefits of plant-neighborhood association (Tashi 2004; Baraza et al. 2006; Kim 2017). We suggest that encouraging these thorny bushes on the forest floor—which gets neglected in present forest restoration activities—can protect grazing damage and enhance survival of oak seedlings, while at the same time promoting forest floor biodiversity.

Our study used LMA ratios, a leaf trait of pivotal importance to understand the plant's ability to capture light and gain carbon, which in turn determines the fitness of plants in their environment (Poorter et al. 2009; Puglielli et al. 2015). Higher values of LMA were associated with plants exposed to strong overhead sunlight (Poorter et al. 2009) or a strategy to survive adverse growing conditions like drought (Gratani & Varone 2004) and grazing (Mcintire & Hik 2002; Dorji et al. 2015). We found a significant effect of tree shelters on LMA values of seedlings in general. Protex seedlings had lower LMA ratios as compared to other interventions. The seedlings in Protex tubes developed larger leaves with higher leaf areas compared to other treatments, presumably to maximise the photosynthetic tissue in the light-limited environment, and as a result had lower LMA values and weaker seedlings.

On the other hand, when there were favorable environmental conditions (represented by the seedlings in the wooden and mesh tree shelters), leaves grew healthier in both leaf area and leaf dry weight and demonstrated higher LMA values, which was also associated with better survival and growth in the field. In the case of control seedlings under repeated grazing stress, leaf area was reduced considerably by repeated clipping. The reduction in leaf area gets compensated with thicker and spiny leaves—an ecological strategy to withstand repeated grazing (Dorji et al. 2015), and consequently depicting higher LMA values.

We suggest that using tree shelters is a promising option to regenerate and restore the high altitude brown oaks in the Himalayas. Although, it will be meaningful to consider and reduce the number of grazing stock in the forest and reforested areas (Norbu 2002; Buffum et al. 2009), in practice, the reduction of grazing animals from these forests remains a challenge due to socio-economic reasons. Seasonal migration and rotational grazing for one or two seasons to avoid extreme climate and improve forage productivity, respectively, are widely practised and are an integral part of the farming system (Moktan et al. 2008; Namgay et al. 2013). However, this type of farming system is not adequate to regenerate the slow-growing brown oak seedlings, which require long-term protection of about 15-20 years until they become safe from browsing animals. Rotational grazing and protection of planted areas for a longer-term are not feasible due to limited availability alternate grazing land as well as due to invasion by undesirable shrubs in the absence of grazing. Further, the dependence of rural livelihoods on a suite of ecosystem services, including free grazing in the forests, is a historical right (Dorji et al. 2019) and the primary source of rural household income (Moktan et al. 2008; Namgay et al. 2013).

In such situations, protecting primary species of interest is a viable compromise, particularly following regular mass plantation programs. Fencing large patches of re-forested areas are widely practised in Bhutan but were not always practical due to cost implications, overgrowth of unwanted vegetation, and conflict with local sentiments for grazing rights. Considering the substantial investments in time, money, and human resources in plantation projects—often with unsatisfactory outcomes—tree shelters can be a promising technique. Firstly, it protects the primary species of interest and allows free grazing within the forest. Grazing animals indirectly control the growth of unwanted vegetation and reduce the cost of weeding. It is recommended to use tree shelters that best mimic the natural growing conditions as these shelters were associated with higher survival and better plant growth. The use of cost-effective, locally prepared tree shelters in partnership with local communities is suggested to provide a middle path to reforest the poorly regenerated oak forests. Our study is expected to complement one of the many ongoing efforts Bhutan is undertaking to blend forest grazing with restoration efforts to ensure long term sustainability of the high altitude old-growth oak forests.

Although our study clearly showed that tree shelters are effective and cheaper, the limitation in the extent of tree shelter establishment in restoration works suggests, however, that other measures to protect from grazing should also be investigated. As our study did not explore these measures specifically, we recommend that more studies and trials involving un-palatable, thorny shrubs which mimic tree shelter need to be explored as a cost-effective means to restore oaks in the grazing dominant landscapes of Himalayas. Studies around the globe has shown high seedling mortality even when the seedlings are protected from herbivory or girdling (Keeton 2008), highlighting the importance to include other mortality factors, such as drought stress, belowground competition, and soil nutrients (Hau & Corlett 2003), which vary primarily from site to site (Sweeney & Czapka 2004). The repeated dieback and resprouting of oak seedlings inside the Protex tree shelters in our study is an example of such a situation that warrants longterm monitoring and further research.

4.6 ACKNOWLEDGMENTS

The study was carried out within the framework of the long-term restoration of old-growth oak forests in Bhutan. It is a part of the Ph.D. thesis of the first author at the University of Adelaide, Australia, under the Commonwealth Government's Research Training Program Scholarships (RTP). The Royal Government of Bhutan, the University of Adelaide, and RTP of the Australian Commonwealth Government are gratefully acknowledged for their support. The authors would like to acknowledge the generous support of the funding partners. Rufford grants and Royal Government of Bhutan supported the initial establishment of the experiment in 2014, while Sloping lands in Transition (SLANT) project funded by Austrian Development Agency at Centre for International Forestry Research (CIFOR) and Forest Ecosystem Restoration initiative of the Korea Forest Service supported the maintenance of permanent plots and subsequent fieldworks. The authors would like to thank Dr. Himlal Baral, CIFOR, for his continuous support to this project. The authors would also like to thank Madan Kr. Lama, Yog Raj Chhetri, Damber Ghimirey, and all the colleagues at Conifer Forest Research Sub-Centre, UWICER, Bhutan, for their support during the fieldwork. The lab mates of terrestrial ecology, Adelaide University are duly acknowledged for their suggestions on improving this manuscript. Finally, we would like to thank the three anonymous reviewers for their valuable suggestions, which significantly improved the paper.

4.7 CONFLICT OF INTEREST

The authors declare no conflict of interest.

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Chapter 5: Carbon storage potentials of two major forest types of temperate Bhutan, Eastern Himalaya

5.1 STATEMENT OF AUTHORSHIP

Statement of Authorship

Title of Paper	Carbon storage potentials of two major forest types of Bhutan, Eastern Himalayas
Publication Status	☐ Published ☐ Accepted for Publication ☑ Submitted for Publication ☐ Unpublished and Unsubmitted w ork w ritten in manuscript style
Publication Details	Journal of Forestry Research

Principal Author

Name of Principal Author (Candidate)	Tshewang Dorji		
Contribution to the Paper	Conceptualization, methodology, data collection and analysis, funding acquisition, writing, and editing reviewer comments		
Overall percentage (%)	70 %		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	20/4/2020

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);

ii. permission is granted for the candidate in include the publication in the thesis; and

iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Justin Brookes		
Contribution to the Paper	Conceptualization, Methodology design, fieldwork, data analysis, funding acquisition, supervision and editing.		
Signature		Date	21 July 2020

Name of Co-Author	José M. Facelli		
Contribution to the Paper	Supervision, fieldwork, data collection, methodology and data analysis, writing and editing		
Signature		Date	4 August 2020

Please cut and paste additional co-author panels here as required.

Name of Co-Author	Yog Raj Chhetri
Contribution to the Paper	Developing research protocol and data collection, investigation, writing and editing in association with the lead author
Signature		Date	21.4.2020

Name of Co-Author	Tshewang Norbu
Contribution to the Paper	Developing methodology, field work and logistics, funding acquisition and writing
Signature		Date	1/7/2020

Name of Co-Author	Madan Kr. Lama
Contribution to the Paper	Fieldwork, data acquisition and reviewing
Signature		Date	01/07/2020

Carbon storage potentials of two major forest types of temperate Bhutan, Eastern Himalayas

Tshewang Dorji [1,2,] *, José M. Facelli [1], Tshewang Norbu[2], Madan Kumar Lama[2], Yog Raj Chhetri[2] and Justin D. Brookes [1]

1. The School of Biological Sciences, University of Adelaide, South Australia 5005, Australia
2. Ugyen Wangchuck Institute for Conservation and Environmental Research, Thimphu, P.O.Box 2049 Bhutan

* Corresponding author: tshewangdorji1@gmail.com

5.2 ABSTRACT

Bhutan's high-altitude forests are unique ecosystems that are home to diverse ecosystem services, including carbon absorption from the atmosphere that offset greenhouse gas emissions. Bhutan has made a commitment to remain carbon neutral, and forest management is a key component in this strategy. To comprehend the full value of these forests, the Bhutanese government is evaluating the carbon storage potentials of the forests. In this paper, we aim to study the carbon storage potentials of two dominant high-altitude forest types: a broadleaf forest dominated by oak and another mixed conifer forest. We used biomass equations to quantify carbon stored as above-ground biomass (AGB) and belowground biomass (BGB). We carried out soil sampling to determine soil organic carbon (SOC) content and density to a depth of 100 cm. We estimated a SOC stock of 366.7 ± 18.3 (Mean ± SE) Mg ha^{-1} (megagram per hectare) for broadleaf forests and 303 ± 9.3 Mg ha^{-1} for conifer forest from a total of five carbon pools recommended by IPCC. Carbon stored in the tree biomass dominated the proportion of total carbon stocks in both the forest types with 63.96 % for broadleaf forest and 40.34 % for conifer forest. Soil organic carbon (SOC) constituted a substantial proportion of the total carbon pool. Soils in the conifer forest were richer in SOC density with a mean of 138.84 ± 14.08 (SE) Mg ha^{-1} compared to soils in the broadleaf forest with a mean of 107.18 ± 9.84 Mg ha^{-1}. No significant differences were detected in SOC between grazed and ungrazed forests in this study. Forest floor biomass comprising of herbaceous layer, deadwood and litter on the forest floor was strongly correlated with SOC concentration, indicating the importance of forest floor components in soil carbon accumulation. The forest floor biomass, which is currently less-valued in the forest management, should be reconsidered and a holistic approach

including species diversification and stand stratification (e.g., preservation of tree, shrubs, and woody debris) and soil protection needs to be developed for carbon forestry. The high-altitude forests merit special attention considering the growing need for mitigating global carbon emissions and Bhutan's own commitment to remain carbon neutral for all times.

5.3 INTRODUCTION

Terrestrial ecosystems, particularly those under forest cover contain as high as 80% and 40% of the total above-ground and below-ground terrestrial carbon respectively and have been widely recognized as a global carbon sink (Dixon et al. 1994; Song and Woodcock 2003). Recently, as nations try to mitigate climate change and decentralize forest governance from central to the local level, forestry has shifted from sole objective of timber production to a holistic approach, inclusive of a suite of social, ecological and environmental services including biomass and carbon management (Sandbrook et al. 2010). In such a move, public partnerships in afforestation and reforestation, forest management (Schroeder 1991) and value-addition of wood products are seen to dramatically increase the quantity of sequestered carbon and limit carbon emissions from using other alternatives (Sathre and Gustavsson 2009; Chakravarty et al. 2015). A greater emphasis has also been considered in the global climate policy to support the developing countries in reducing emissions from forest degradation and deforestation (Potvin and Bovarnick 2008). Trees are known to accumulate carbon continuously as they grow and even perform CO_2 absorption and carbon reservoir functions at old age (Stephenson et al. 2014; Carey et al. 2001; Luyssaert et al. 2008). Some researchers, however, maintain that forests cannot continue forever as a carbon sink and may become a source of carbon emission, especially in the face of climate change (Hadden and Grelle 2016). For instance, global deforestation itself is a major source of carbon emission and added about five billion metric tons of carbon dioxide every year to the atmosphere (WRI 2018). While old forests are ageing and can potentially become a carbon source in the absence of proper management (Baccini et al. 2017), slowing deforestation integrated with creation of more forest plantations and management options that increase forest productivity can improve carbon sequestration from the atmosphere (Pugh et al. 2019; Dixon et al. 1994; Potvin and Bovarnick 2008). Therefore, sustainable forest management must include actions to keep the forest young, provide ecosystem services and increase carbon stocks (Nabuurs and Karjalainen 2007).

Himalayan forests dominate the temperate region and are important global carbon pools that are biologically very diverse and productively very highly (Rana et al. 1989; Singh and Thadani 2015). Along with the above-ground biomass carbon, temperate forests are known to store as high as 69% of total soil organic carbon (SOC) in their soil (Ali et al. 2019). Accurate assessment of biomass and carbon in these ecosystems is essential to develop necessary management decisions and improve the carbon sequestration and storage potentials. Despite the potentials, studies on forest biomass and carbon storage are limited, particularly in the high-altitude Himalayas, resulting in poor understanding of the role of the Himalayan ecoregion in the global carbon balance. Bhutan Himalayas is one such region where forest occupies about 71% of the total landmass, and carbon quantification is at an early stage of the investigation. Bhutan has further committed to increase its forest cover through public partnerships in reforestation of the degraded areas. The huge forest cover is estimated to absorb about 6 million tonnes of carbon annually, far above the 1.5 million tonnes of carbon emitted by the country annually—making Bhutan a carbon-neutral society (NEC 2012). Bhutan further plans to become a carbon-negative country by offsetting carbon through the development of policies that enable environment-friendly practices such as hydropower, electric vehicles, and organic farming. Consequently, carbon assessment studies including the carbon captured in different forest types, are crucial for forest management decision-making at the national level and proper accounting of the carbon budget at the global level.

The oak-dominated cool broadleaf forest (hereafter referred as a broadleaf forest) and mixed conifer forest (conifer forest) dominate the temperate and alpine region of the country constituting about 70 % of total forest (DoFPS 2016). The forests are intricately woven into the cultural-fabric and lifestyles of farming communities, thereby having socio-cultural, ecological and economic values (Dorji et al. 2019). Most of the forested area in remote locations are intact and exist as old-growth stands. However, grazing by domestic and wild herbivores are inevitable and persist throughout the forest types.

Forest management that ensures sustainability, continuous supply of ecosystem services and promotion of carbon storage to mitigate global carbon emissions are top priorities for Bhutanese people. Bhutan initiated the nationwide forest inventory from

2012-2015 that gives a comprehensive understanding of the country's forest resources. A national estimate of 709 million tons of carbon has been calculated based on this national level assessment (DoFPS 2018). Although quite similar in approach to the National level assessment, the present study concentrates on two specific local forest types which are not adequately captured in the broader national-level assessments. Local-level forest specific assessments are essential to value the significance of special forest types while complementing the national-level estimates to improve the accuracy of national greenhouse gas emission-absorption reporting. Our study aimed to quantify total carbon stored in two major forest types of temperate Bhutan. We estimated the carbon captured in all of the five-carbon pools suggested by the IPCC Good Practice Guidance for Land Use, Land-Use Change and Forestry (IPPC 2003), that consisted of 1. above-ground biomass (AGB), 2. belowground biomass (BGB), 3. dead wood or coarse wood debris, 4. leaf litter and 5. SOC. The information generated from our study can be used as a baseline for accurate monitoring of carbon stored in these forests for future carbon sequestration studies.

5.4 MATERIALS AND METHOD

5.4.1 Study area

The study was conducted in two forest types in Western Bhutan (Figure 29) consisting of broadleaf forest dominated by brown oak (*Quercus semecarpifolia*) and mixed conifer forest dominated by hemlock (*Tsuga dumosa*), Juniper (*Juniperus recurva*), spruce (*Picea spinulosa*) and Fir (*Abies densa*). The elevation ranged from 2950-3010 m for broadleaf forest and from 3273-3336 m above sea level for conifer forests. Slope percent were similar in both the forests ranging from 27-40 %. The mean annual temperature for the broadleaf forest was recorded at 8.5 ºC and a mean temperature of 4.6 ºC based on a nearby weather station. Annual precipitation varied from 900-1500 mm, mostly occurring during monsoon from May to September. The plots for our study are located in undisturbed forests away large-scale disturbances. Both the study areas fall within Bhutan's forest management units (FMUs) where small group selection cutting was carried out for commercial timber in the past. There is no commercial logging happening presently. Nevertheless, the forests face some degree of human interference due to their proximity to the villages. The intensity of disturbance is increasingly high in the confer forest lately due to the growing demand for rural timber which is allotted by the forest range offices under selection cut system.

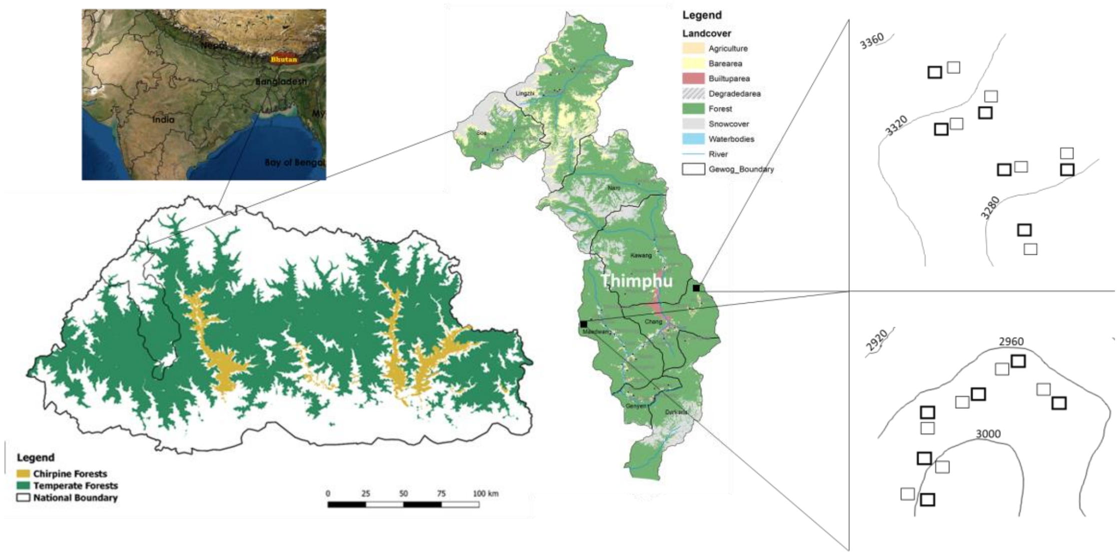

Figure 29. The map shows the location of Bhutan (top left) and a map of Bhutan with the distribution of temperate forests. The studies are located in two forest types in Thimphu district (middle) representing a mixed conifer forest (top right) and a broadleaf forest (Bottom right). Square shapes represent the fenced (thick) and unfenced plots with 40 m contour intervals and elevation (m).

5.4.2 Tree layer inventory

In each forest type, we established 12 vegetation plots of 25 x 25 m and surveyed all woody vegetation inside the plots. All woody plants above the height of 1.3 m (Breast height) were measured including diameter at breast height (DBH, cm) and height (Ht, m). Total above-ground dry biomass (AGB) was determined using the power function equation $Y = ax^b$, where Y is the AGB (Kg), x is the DBH and *a* and *b* are the species-specific parameter estimates of the model (Payandeh 1981; Blujdea et al. 2012). The biomass equations for the species were developed through the destructive sampling of the trees during the recent National Forestry Inventory (DoFPS 2018). For the few species where equations were not available within Bhutan, we used the equations developed for similar forest types by Adhikari et al. (1995). The below-ground biomass (BGB) which is difficult to measure was estimated using the established root to shoot biomass ratio of 0.26 which was reported uniform across latitudes, soil types or forest types (Mokany et al. 2006; Plugge et al. 2016; Cairns et al. 1997). Biomass was converted to carbon by using the conversion factor of 0.47 suggested by IPPC (2006), which indicates 47 % of all tree dry biomass to be carbon (Mohd Zaki et al. 2018).

5.4.3 Shrub, deadwood and leaf litter layer inventory

Inside the vegetation plot, we established three 1 m x 1 m plots to measure shrub layer, deadwood and leaf litter. To further account for changes in understorey biomass due to grazing, we laid out 1 m x 1 m similar plots in six long-term monitoring fenced plots in each forest type. All shrubs inside the plot were harvested, and their fresh weight was recorded in the field. Subsamples of leaves, branches and stems of harvested shrubs were weighed and taken for further analysis. Similarly, we divided the 1 m x 1 m plot into four quadrats of 25 cm x 25 cm and collected dead wood and leaf litter from one of the four quadrats randomly. The samples were taken to the laboratory and oven-dried at 60 °C until a constant weight was obtained. The dry weight of samples and subsamples were used to calculate the corresponding dry weight of each biomass layer. The dried samples were finely grounded and analysed for carbon content (%) using the elemental determinator CHN 628 (LECO, UK). The total C for each layer was obtained by multiplying dry weight with carbon content % (Tashi et al. 2016). All chemical analyses were carried out in the Soil and Plant Analytical (SPAL) laboratory, National Soil Service Centre under the Ministry of Agriculture and Forests, Thimphu, Bhutan.

5.4.4 Soil sampling and SOC determination

A total of 72 soil profiles (three from each plot) were excavated during this study. Before digging of the soil, we measured the soil hardness or soil resistance using a push cone soil hardness meter (Yananaka’s soil hardness tester, Kiya Seisakusho, Tokyo) and soil moisture content using a hydrosense (Campbell Scientific Inc., Logan, Utah). We also measured leaf litter and organic matter thickness of each soil profile using a measuring scale. Soils were dug to a depth of 100 cm and categorized into different depth classes; 0-10 cm, 10-30 cm, 30-60 cm, and 60-100 cm. The bulk density samples of each depth layer were obtained using a metal ring of known volume. The bulk density samples were oven-dried at 105 °C to constant weight. Bulk density (BD) was obtained by dividing the dry weight of soil inside the ring by the ring volume (gcm^{-3}). The BD values of the samples were corrected for coarse fragments (> 2 mm) to obtain BD of the fine soil (Poeplau et al. 2017). For all carbon stock estimation, corrected BD was used. For soil chemical analysis, respective soil samples of about 1 kg were collected from each depth classes and taken to the laboratory for SOC analysis. The samples were gently crushed by hand after air drying for 2-3 days. The samples were filtered through 2 mm mesh sieve. Chemical analysis was performed on the soil that passed through the sieve. The proportion of soil > 2 mm, consisting of pebbles, were weighed and recorded. SOC content (%) in the soil fraction (< 2 mm) was determined by titration using the Walkley-Black method (Walkley and Black 1934). The SOC density at different depths was calculated following formula (Equation 3) Sanderman et al. (2011).

$$SOC\ (Mg\ ha^{-1}) = \%\ C\ \frac{mg\ C}{g{<}2mm}\ x\ GC\ \frac{g{<}2mm}{g\ soil}\ x\ BDfine\ x\ L\ (cm)\ x\ C\ \frac{10^{8}cm^{2}}{ha}\ x\ \frac{Mg}{10^{9}Mg}\ \ldots.(3)$$

GC = Gravel correction; L= Layer thickness; BD fine = Bulk density fine soil; C = Correction for units

5.4.5 Statistical analysis

Comparison between broadleaf and conifer forest for each of the give carbon pools were computed using Welch Two Sample t-test. Mixed-effects models (Fox and Weisberg 2011) were developed to understand how each of the response variables such as litter layer, soil organic matter, soil carbon content and SOC vary with respect to forest type and grazing. In the model, forest type and grazing were entered as fixed factors, while plots nested within the forest type were entered as a random factor to account for variations induced by forest and plot selection. A mixed-effects model was also employed to quantify and understand the variations in soil bulk density, carbon content and SOC along the soil depths. We entered forest type, grazing and soil depth as fixed factors and plots nested within forest types as a random factor. All mixed-effects models were conducted using the *lmer* function of the *lme4* package (Bates et al. 2015). The normality of residuals was examined through a residual diagnostic plot of model fitted values against model residuals and quantile-quantile (q-q) plots. The p-values and test of main effects and interactions were obtained by Wald Chi-Squared tests using *Anova* function in the *car* package (Fox and Weisberg 2011; Duursma and Powell 2016). All pairwise comparisons were obtained using Tukey's HSD test. Linear regression models were developed to study the relationship between the dependent variable (SOC) and independent variables of forest floor biomass (leaf litter, herbaceous biomass and coarse wood). All analysis was performed in the RStudio (RStudio Team 2016) using the R software (CoreTeam 2019).

5.5 RESULTS

5.5.1 Tree layer forest biomass and carbon storage

A total of 27 tree species, eight shrub and 32 herb species were recorded from the broadleaf forest, whereas ten tree, 12 shrub and 40 herb species were recorded from the conifer forest. The above-ground tree biomass was calculated at 415.36 tons ha^{-1} and 221.83 tons ha^{-1} for broadleaf and conifer forest respectively. Mature *Quercus semecarpifolia* trees (60.1 %) and *Q. thomsonii* (26.25 %) dominated the major portion of tree biomass in the broadleaf forest, while in the conifer forest, *Tsuga dumosa* (54.3 %), *Juniperus recurva* (34.41 %) and *Abies densa* (8.7 %) shared the biomass proportion. The total carbon stored in the trees was estimated at 214.32 and 116.16 Mg ha^{-1}in broadleaf and conifer forest, respectively (Table 13).

Table 13 Carbon stored (Mg ha^{-1}) in the five carbon pools of the two forest types. Bold letters represent significantly different *p*-values.

Carbon (Mg ha^{-1})	Location	Broadleaf	Conifer	*p*-value
Trees	Above ground	195.22 ± 50.57	104.26 ± 19.59	0.306
	Below ground	50.76 ± 13.14	27.11 ± 5.09	0.124
Herbaceous layer	Above ground	0.52 ± 0.17	0.15 ± 0.07	0.400
	Below ground	0.27 ± 0.08	0.1 ± 0.03	0.400
Coarse wood	Forest floor	6.96 ± 3.19	18.51 ± 6.14	**0.020**
Leaf litter	Forest floor	5.82 ± 1.01	14.18 ± 1.76	**0.021**
Soils	0-10	22.28 ± 1.74	29.91 ± 1.73	**<0.001**
	10-30	25.37 ± 1.95	31.59 ± 2.01	**0.006**
	30-60	29.98 ± 3.8	37.25 ± 4.6	0.064
	60-100	29.55 ± 2.35	40.09 ± 5.74	0.389

5.5.2 Herbaceous layer forest biomass and carbon storage

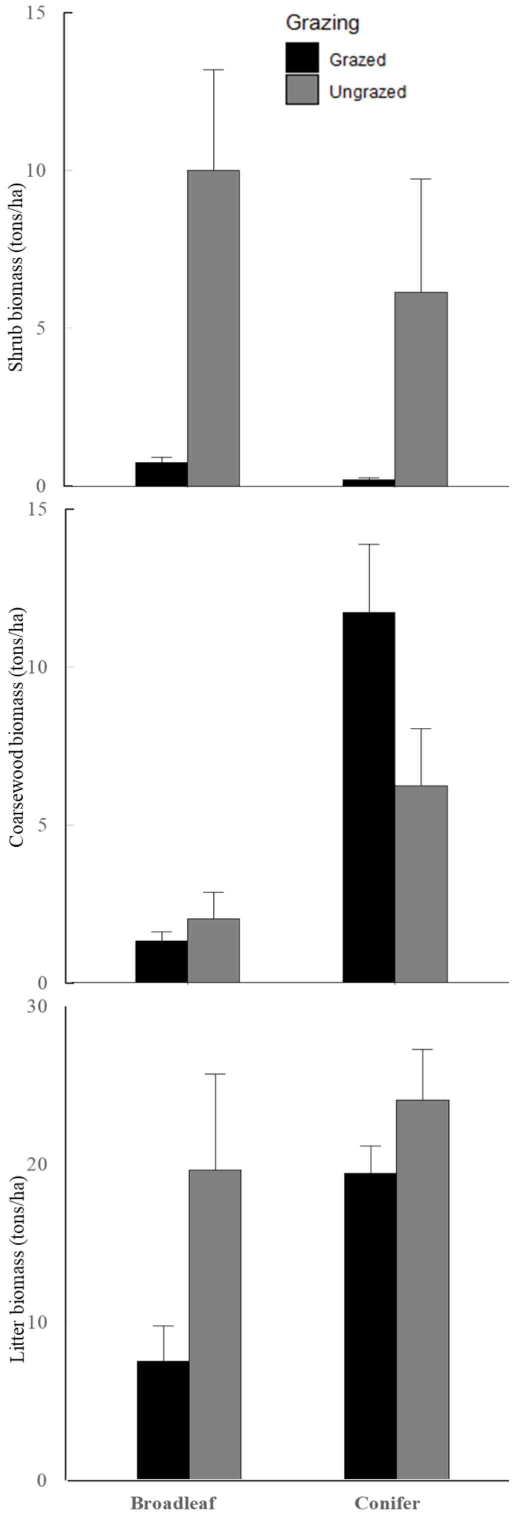

Figure 30. Shrub, Coarse wood and litter biomass (tons ha^{-1}) variations between grazed and ungrazed plots in the two forest types.

A total of eight and 12 species were recorded in the herbaceous layer on the forest floor of broadleaf and conifer forest, respectively. *Yushania microphylla*, a native Himalayan bamboo dominated the forest floor with considerably high coverage in the fenced plots. Most of these bamboos were found dying particularly in the conifer forests during this study, following a mast flowering observed a few years earlier. Shrub layer biomass was not significantly different between the forest types, however, differed dramatically between grazed (0.47 ± 0.24 tons ha^{-1}, mean ± SE) and ungrazed (16.175 ± 6.78 tons ha^{-1}, mean ± SE) plots (p =0.003, Figure 30). The biomass of the shrub layer contributed to 0.24 and 0.1 % of the total carbon stored in broadleaf and conifer forest, respectively (Table 13).

5.5.3 Coarse wood layer

The coarse wood debris did not vary greatly between grazed and ungrazed plots. However, significantly higher accumulation of coarse wood biomass was recorded in the conifer forest compared to the broadleaf forest (t = 3.397, p-value < 0.005, Figure 30). Biomass of woody debris on the forest floor varied from 1.7 ± 0.29 tons ha^{-1} to 9.17 ± 0.57 tons ha^{-1} (mean ± SE) for broadleaf and conifer forest, respectively. Woody debris contributed to about 2.08 % and 6.43 % of the total carbon stocks of broadleaf and conifer forest, respectively (Table 13).

5.5.4 Leaf litter layer

Huge variations in the litter thickness on the forest floor were observed between the two forest types. Average litter depth of broadleaf forest was recorded at 4.11 cm for grazed and 6.58 cm for ungrazed plots, which were significantly lower than conifer forest measured at 6.77 cm for grazed and 9.66 cm for ungrazed plots (Table 13). Similarly, leaf litter biomass on the broadleaf forest floor was 7.58 ± 2.19 tons ha^{-1} for grazed and 19.63 ± 6.01 tons ha^{-1} (mean ± SE) for ungrazed plots while for conifer forest it was 19.42 ± 1.71 tons ha^{-1} for grazed and 24.04 ± 3.16 tons ha^{-1} for ungrazed plots. The proportion of carbon contributed by leaf litter was 1.82 % and 5.41 % for broadleaf and conifer forest. The carbon stored in the leaf litter also varied significantly between the two forest types (Table 13).

5.5.5 Soil variables and SOC density

No significant difference was observed in the thickness of the organic matter between grazed and ungrazed plots ($F_{1,20}$ =2.91, p>0.05, Figure 31 and TableS 3). However, the mean organic matter thickness of 5.42 cm for the broadleaf forest was significantly lower than 29.56 cm recorded for conifer forest ($F_{1,20}$ =34.49, p<0.001, Figure 31 and TableS 3). Soil hardness was significantly higher in broadleaf forests compared to conifer forests ($F_{1,20}$ =23.87, p<0.001, TableS 3). Grazed plots in the broadleaf forest recorded significantly higher soil hardness than ungrazed plots (Figure 31). No such variations were observed in the conifer forests. Soil moisture content recorded to a depth of 25 cm was also significantly higher in conifer forests with 17.84 % than broadleaf forests at 12.47 % ($F_{1,20}$ =2.91, p<0.05, TableS 3).

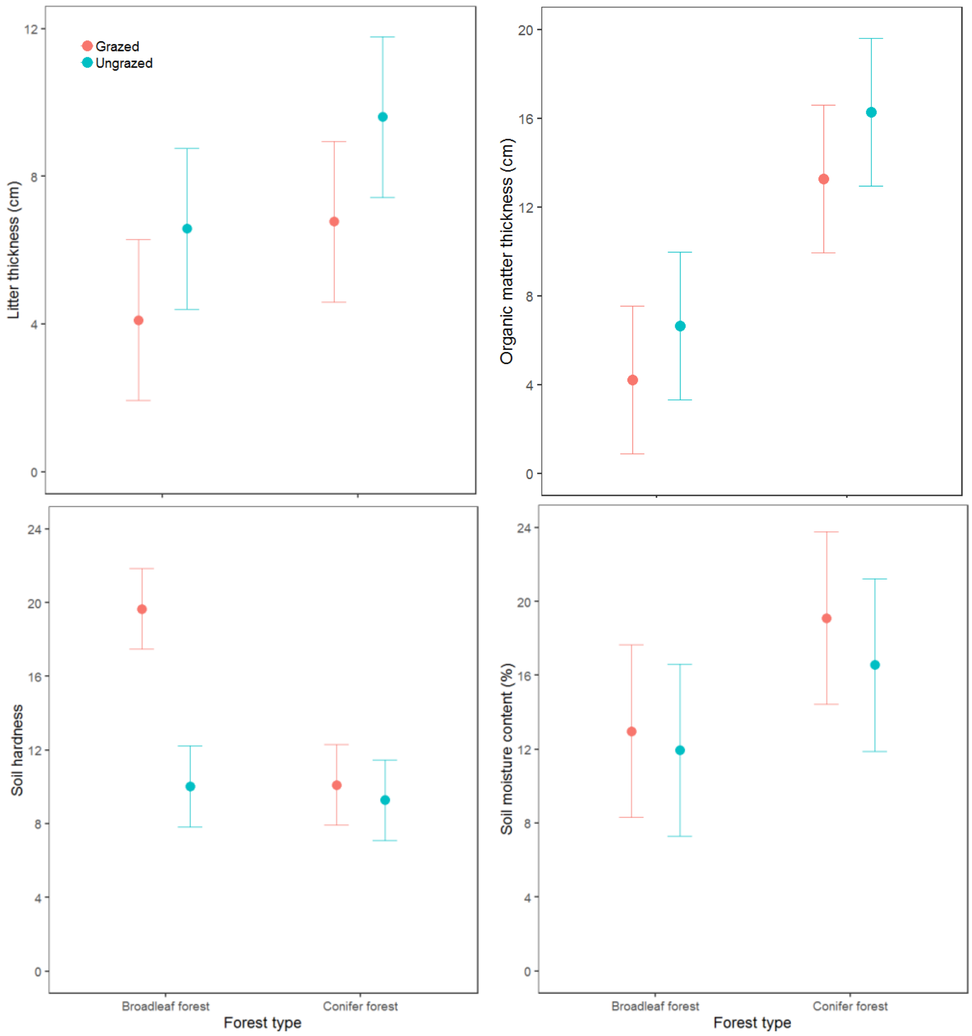

Figure 31. Comparison of different forest floor components (litter layer, organic matter layer) and soil physical properties (soil hardness, soil moisture content) between grazed and ungrazed plots grouped within the forest types. The error bar represents the 95 percent confidence interval.

Although there were variations in the surface soil hardness due to forest types and grazing, soil bulk density didn't show variations due to forest type ($F_{1,22}$ =0.43, p>0.05, Figure 32 and TableS 4), or grazing ($F_{1,22}$ =1.54, p>0.05, Figure 32 and Table S2). Nevertheless, bulk density significantly increased with soil depth in both the forest types (Figure 32 and TableS 4). The mean SOC content at 0-10 cm, 10-30 cm, 30-60 cm and 60-100 soil depths were 5.51%, 3.96%, 2.54% and 1.55% for broadleaf and 9.73%, 5.41%, 3.48% and 2.59% for the conifer forest. The total SOC density was estimated at a mean of 107.18 ± 9.84 (± SE) Mg ha^{-1} for broadleaf forest and 138.84 ± 14.05 Mg ha^{-1}.

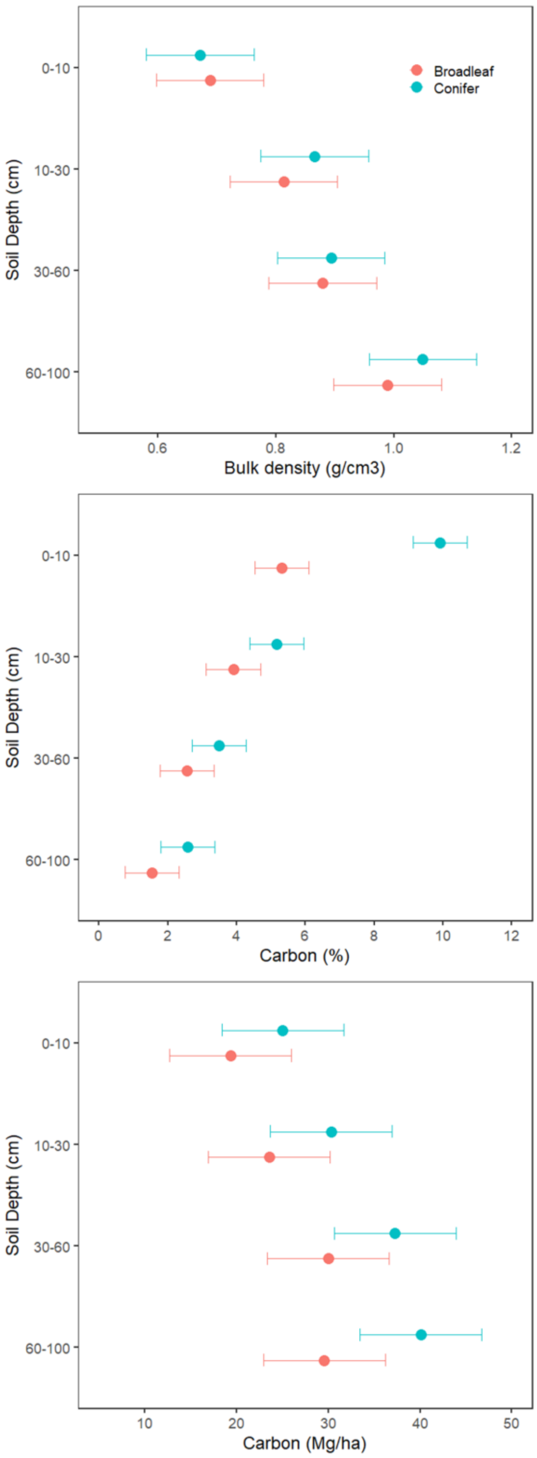

Figure 32. Variations in soil bulk density, SOC content (%) and SOC density (Mg ha^{-1}) with soil depth in the broadleaf and conifer forest. The error bar represents the 95 percent conifidence interval.

The mean SOC densities for broadleaf forest at different soil depths were 22.28 ± 1.74 Mg ha^{-1} (0-10 cm), 25.37 ± 1.95 Mg ha^{-1} (10-30 cm), 29.98 ± 3.8 Mg ha^{-1} (30-60 cm) and 29.55± 3.8 Mg ha^{-1} (60-100 cm). Similarly, mean SOC densities for conifer forest at different soil depths were 29.91 ± 1.73 Mg ha^{-1} (0-10 cm), 31.59 ± 2.01 Mg ha^{-1} (10-30 cm), 37.25 ± 4.6 Mg ha^{-1} (30-60 cm) and 40.09 ± 5.74 Mg ha^{-1} (60-100 cm). The differences in mean values of SOC density at varying soil depths is due to differences in the thickness of depth interval, SOC content and bulk density between the depths. As a result, SOC density was higher in the deeper soils despite a significant drop in SOC content (%) (Figure 32). Soils of broadleaf and conifer forests constituted about 32% and 48% share respectively to the total carbon pool. Conifer forests recorded higher SOC density than the broadleaf forest ($F_{1,22}$ =5.39, p<0.05, Figure 32 and TableS 4) at all soil depths ($F_{3,66}$ =8.13, p<0.001, Figure 32 and TableS 4). No significant differences were observed in the SOC content ($F_{1,22}$ =2.09, p>0.05) or SOC density ($F_{1,22}$ =0.03, p>0.05, Figure 32 and TableS 4) between grazed and ungrazed plots.

5.5.6 Forest floor biomass and its relationship with soil carbon

Soil organic carbon stock for the upper soil layers (0-30 cm), was positively correlated with the forest floor biomass, particularly leaf litter and deadwood biomass (Figure 33). Although herbaceous layer biomass did not show a significant relationship with SOC, their accounting in the total forest floor biomass significantly improved the correlation (Figure 33).

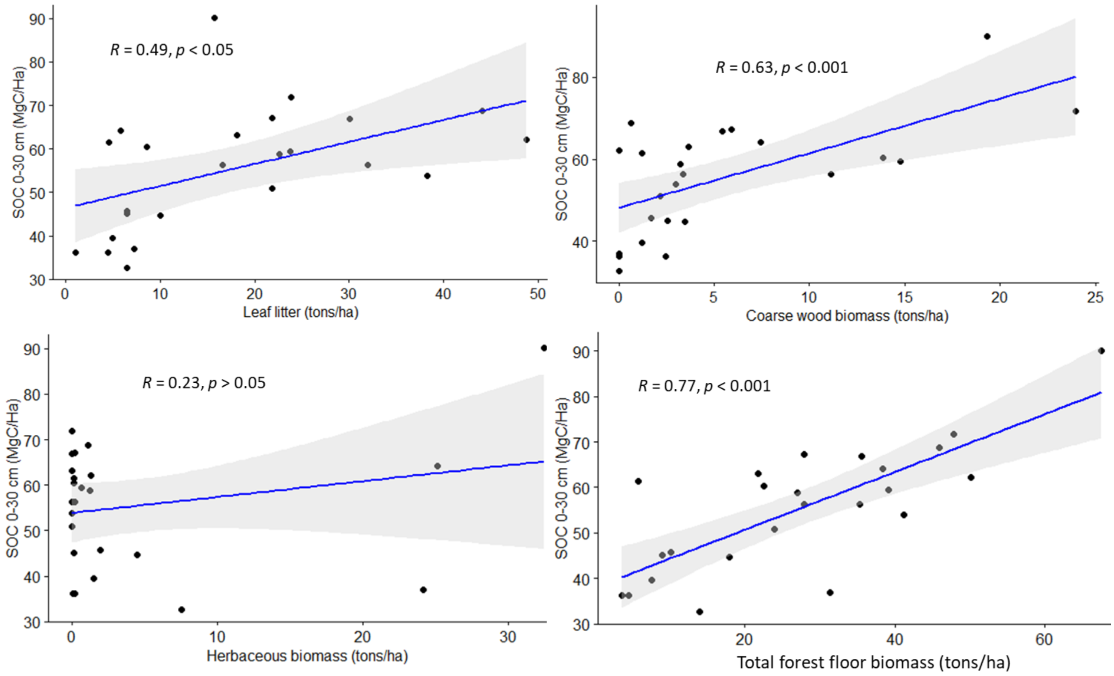

Figure 33. Relationship between SOC density (Mg ha^{-1}) with the corresponding forest floor biomass; litter, coarse wood, herbaceous and total floor biomass. The shaded area represents the 95 percent confidence interval around the regression line.

5.6 DISCUSSION AND CONCLUSION

Our study assessed the biomass accumulation and carbon pools in two high-altitude mountain forest ecosystems, namely temperate broadleaf forest and mixed conifer forest and showed their potentials as massive carbon sinks in the Himalayas. The results highlight the value of these forests in the Himalayan countries towards achieving a low carbon economy. As a limited number of species can adapt and grow at higher elevations, protecting the existing forests and intensifying forest management practices that are socially and ecologically sustainable and one which maximises forest productivity and carbon sequestration services are essential.

Our estimation on total carbon storage ranged from 303 to 367 Mg ha^{-1} for conifer and broadleaf forest, respectively. The major contribution to the forest carbon pool was from tree biomass in both the forest types. The tree biomass carbon for the broadleaf forest was 245 Mg ha^{-1} which was consistent with studies (190.74 to 234.07

Mg ha^{-1}) in similar forest types of Nepal Himalayas (Verma et al. 2012; Subedi and Shakya 1988). In the conifer forest, the tree biomass carbon was 131.37 Mg ha^{-1}, comparatively lower compared to the broadleaf forest, which is consistent with Bhutan's National Forestry Inventory (DoFPS 2018). We observed that higher tree biomass in the broadleaf forest is due to presence of old-growth mature oak trees compared to conifer forests. Majority of oak trees in the broadleaf forest have a DBH ranging from 60-190 cm reflecting the old-growth nature of the forest. Interestingly, in the case of conifer forests, strong preference of conifer species for rural timber could have possibly reduced mature tree density, thereby reducing the tree biomass. Similar results were also reported from Nepal Himalayas where extraction of biomass by people have led to lower above-ground biomass and carbon storage potentials (Suwal et al. 2014). Most of the conifer forests in Bhutan are inherently subjected to small scale selection cutting to meet timber demands of local communities (Moktan et al. 2009). The selection cutting was also apparent in our study plots as we recorded several old and freshly cut stumps and prevalence of thick layers of dead wood and leaf litter. Consequently, the tree biomass was lesser in conifer forest compared to the broadleaf forest. As a result, the contribution of AGB to total carbon pool from conifer forest was 44.32%, while it was 67.37% for the broadleaf forest.

The carbon stored in the herbaceous layer, dead wood and leaf litter were estimated at 0.79, 6.96 and 5.82 Mg ha^{-1} for broadleaf forest and 0.25, 18.51 and 14.18 Mg ha^{-1} for the conifer forest. The deadwood and leaf litter on the forest floor strongly correlated with higher SOC stocks. The relationship was particularly strong for the upper soil depths of up to 30 cm, indicating the importance of forest understorey structural components in the carbon cycle. The results indicate that proper management and accounting of dead wood and leaf litter are important for a reliable carbon stock assessment. Besides, their presence on the forest floor are also known to prevent soil erosion, add organic matter and nutrients to the soil and enhance habitat diversity. The importance of these lesser-known forest structural components should be given special consideration in the forest management plans should carbon management and biodiversity enhancement be the objective of forest management.

The second-largest SOC pool was found in the forest soils. The SOC density to a depth of 100 cm was estimated to vary from 107.18 Mg ha^{-1} to 138.84 Mg ha^{-1} for broadleaf and conifer forest, respectively. Our estimates fall within the range of 56.7-337.8 Mg ha^{-1} reported by Simon et al. (2018) for similar forest types of Bhutan. The mean SOC density obtained by our study also compare with other studies that reported a mean SOC density of 159.4 Mg ha^{-1} for *Quercus leucotricophora* and 118.8 Mg ha^{-1} for subtropical *Pinus roxburghii* forests (Sheikh et al. 2009); 112 to 247 Mg ha^{-1} for pasture and temperate forests (Singh et al. 2011), 127-149 Mg ha^{-1} for mixed broadleaf and 135 Mg ha^{-1} for pine forests of Nepal (Giri 2011). However, the mean SOC density obtained by our study was comparatively lower than some of the studies that report a mean SOC density of 405 Mg ha^{-1} from the foothill forests of Southern Bhutan Himalayas (Tashi et al. 2016). Conversely, our estimate of SOC density was higher than the national estimate of 58.06 Mg ha^{-1} for broadleaf and 59.87 247 Mg ha^{-1} for conifer forests of Bhutan (DoFPS 2018). Variations in sampling methods, forest types and climatic conditions (temperature and precipitation) could explain the differences in SOC density estimates (Simon et al. 2018). For instance, the southern foothills are geologically active environments, which are highly productive, climatically warm, and wet, and ecologically diverse compared to the inner Himalayan Mountains such as our study area, which could explain the variations in SOC stocks in the forest soils. For instance, studies in Western Himalayas have shown that tropical soils tend to have higher SOC along the rainfall gradient (Mehta et al. 2014). The variation can also be partly explained by the differences in soil sampling depths and accounting of larger soil coarse fraction (>2 mm) in the estimation. In the case of our study, the proportion of larger soil coarse fraction dominated the soil with values ranging from 12-70 % (average = 50.48%). Subsequent correction for the larger coarse fraction in the estimation of SOC could have resulted in differences in SOC estimates (Simon et al. 2018). Poeplau et al. (2017) showed the importance to properly account for coarse fractions to avoid overestimation of SOC, which can be more than 100 % in soils with a high proportion of coarse fractions. Our study also confirms the need for proper accounting of the coarse fraction to obtain reliable SOC stock estimates.

Along the soil depth gradient, SOC content declined significantly from the topsoil to the lower soil horizons in both the forest types with values ranging from 10% to 1.5%. The results are consistent with several other studies (Verma et al. 2012; Simon et al. 2018; Tashi et al. 2016). Soil organic carbon content even at a depth of 100 cm was as high as 1.5% to 2.6% for broadleaf and conifer forest respectively, illustrating the huge soil carbon content in the deep soils of these forests. Verma et al. (2012) report similar values of up to 1% SOC content at a soil depth of 100 cm in a high-altitude oak forest of Nepal Himalayas. Many studies estimate SOC stocks on soil samples collected from 0-30 cm depth (see DoFPS (2018), Meena et al. (2019)), which could underestimate soil carbon by not accounting for deeper SOCs. Deep soils at 30-60 cm and 60-100 cm, for instance, were found to contain a mean SOC density of 33.62 Mg ha^{-1} and 34.82 Mg ha^{-1} which constitutes about 55.62 % of the total soil carbon stock. The top 0-10 cm and 10-30 cm contained a SOC density of 26.01 and 28.48 Mg ha^{-1}. The differences in SOC density at varying soil depth is a result of thicker soil depth interval, increasing soil bulk density and carbon content (%), which are used to estimate the carbon stock.

Our results also showed SOC differences in the two forest types. The SOC stock in conifer forests was higher than the broadleaf forests. The variations in SOC could be a result of the differences in altitude and forest types (Simon et al. 2018; Tashi et al. 2016), varying land management practices such as forest grazing (Reeder and Schuman 2002; Liu et al. 2012) or combination of grazing and forest burning (Oliver et al. 2017). Similarly, variations in SOC stock due to different land-use practices such as forests, non-forests and agricultural lands are also reported (Dorji et al. 2014). Along the altitudinal gradient, SOCs are reported to increase progressively from valley bottom to upper mountain slopes generally climaxing at an elevation of 3200-3400 m (Simon et al. 2018; Tashi et al. 2016), the range found for the distribution of mixed conifer forests. We observed that conifer forest was very rich in leaf litter and deadwood components on the forest floor which consequently led to significantly higher accumulation of organic matter layers and SOC in the soils as compared to the broadleaf forest. Although published literature reveal differences in SOC as a result of forest grazing (Reeder and Schuman 2002; Jaweed et al. 2015; Hatton and Smart 1984; Hewins et al. 2018), results from our grazed and ungrazed plots did not show statistically significant differences in SOC concentrations. As our fenced plots were established only recently, we assume that the difference in SOC may not have developed. Higher sample numbers in association with intensive soil sampling in the experimental plots longer than 10-15 years are necessary to detect a change in SOC (Smith 2004; Garten and Wullschleger 1999). Further, it is also reported that changes in herbaceous vegetation can change the SOC (Frank et al. 1995; Dahlgren et al. 1997; Garten and Wullschleger 1999). In the case of our study, fencing and grazing exclusion resulted in strong dominance by *Yushania microphylla*, a bamboo species with a shallow and fibrous rooting system which could possibly be the reason for the maintenance of SOC levels at par with the grazed plots.

Our results on the total carbon storage of two important and most widely distributed high-altitude Himalayan forests serve as baseline information that has relevance for long-term monitoring of changes in carbon stored in these forests. We did not specifically investigate the carbon sequestration rates on a yearly basis, although the significant increment of carbon stock due to tree growth, leaf litter and seed fall was reported from a similar forest of Nepal (Verma et al. 2012). As the net primary productivity of these forests is very high, accounting the net biomass gained or lost on a yearly basis will be valuable for determining carbon sequestration rates. We recommend further studies on carbon sequestration rates of these forests through periodic monitoring and measurement of carbon stock. Forest management is crucial to balance forest productivity with carbon sequestration and storage. As highlighted from our study, the forest floor biomass such as herbaceous layer, deadwood and leaf litter were strongly linked to carbon storage, and thus their importance should be reviewed in the forest management practices. On this basis, we conclude that forest diversification, stand stratification and retention of forest floor biomass can be crucial forest management tools to enhance carbon absorption and storage. As forest soils constituted an important component of total carbon pool, land management campaigns including the planting of trees and grasses should be encouraged to prevent soil erosion and increase forest floor biomass which can directly increase the SOC stocks. We conclude that the Himalayan high-altitude forests are important carbon sinks and deserve special consideration for protection and sustainable management to enhance carbon storage in these forests.

5.6.1 ACKNOWLEDGEMENTS

This study was carried out as a part of the PhD thesis of the first author at the University of Adelaide, Australia, under the Commonwealth Government's Research Training Program Scholarships (RTP). The Royal Government of Bhutan, the University of Adelaide, and RTP of the Australian Commonwealth Government are gratefully acknowledged for their support. The fieldworks were supported by the Royal Government of Bhutan through Sloping lands in Transition (SLANT) project funded by Austrian Development Agency at Centre for International Forestry Research (CIFOR) and Forest Ecosystem Restoration initiative of the Korea Forest Service. The authors would like to thank Dr Himlal Baral, CIFOR, for his continuous support to this study. The authors would also like to specifically thank Evelina Facelli (Adelaide

University), Karma Tenzin, Suvaraj Rai, Rinchen Letro, Sonam Dorji, Tanka Raj Gurung, Bikash Gurung, Ganga Prasad and all the colleagues at Conifer Forest Research Sub-Centre, UWICER, Bhutan, for their support during the fieldwork. We are also grateful to Mr Jamyang and his team at the National Soil Service Centre, Simtokha, Bhutan for carrying out the soil sample analysis within a short time frame despite their busy schedule. Finally, we would like to thank the three anonymous reviewers for their valuable suggestions, which significantly improved the paper.

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Chapter 6: A 639-year monsoon (May-August) precipitation reconstructions indicate precipitation as an important factor controlling the growth of high-altitude Larix *griffithii* in Western Bhutan Himalayas

6.1 STATEMENT OF AUTHORSHIP

Statement of Authorship

Title of Paper	**639-years monsoon (May-August) precipitation reconstructions indicate monsoon (May-August) precipitation as the limiting factor for the growth of high altitude *Larix griffithii* in western Bhutan Himalayas**
Publication Status	☐ Published ☐ Accepted for Publication ☐ Submitted for Publication ☑ Unpublished and Unsubmitted w ork w ritten in manuscript style
Publication Details	The manuscript will be submitted for publication in geophysical letters

Principal Author

Name of Principal Author (Candidate)	Tshewang Dorji		
Contribution to the Paper	Conceptualization, methodology, data collection and analysis, funding acquisition, writing, and editing, incorporating reviewer comments		
Overall percentage (%)	75 %		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	20/4/2020

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);

ii. permission is granted for the candidate in include the publication in the thesis; and

iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

Name of Co-Author	Justin Brookes		
Contribution to the Paper	Conceptualization, Methodology design, fieldwork, data analysis, funding acquisition, overall supervision and editing and reviewing.		
Signature		Date	21 July 2020

Name of Co-Author	José M. Facelli		
Contribution to the Paper	Supervision, data analysis, writing, editing, and reviewing		
Signature		Date	4 August 2020

Please cut and paste additional co-author panels here as required

Name of Co-Author	Karma Tenzin
Contribution to the Paper	Fieldwork and data collection, tree-ring measurement, cross-dating & detrending analysis, writing, editing and reviewing.
Signature	Date 1/7/20

Name of Co-Author	Tshewang Norbu
Contribution to the Paper	Developing methodology, field work and logistics, funding acquisition and writing
Signature	Date 1/7/2020

Name of Co-Author	Dorji Dukpa
Contribution to the Paper	Fieldwork, data acquisition, supervision, writing and editing
Signature	Date 1/7/2020

Name of Co-Author	Markus Stoffel
Contribution to the Paper	Data analysis and methodology, supervision, writing, editing, and reviewing.
Signature	Date July 16, 2020

A 639-year monsoon (May-August) precipitation reconstructions indicate precipitation as an important factor controlling the growth of high-altitude *Larix griffithii* in Western Bhutan Himalayas

Tshewang Dorji [1,2], *, Karma Tenzin[2,6], Dorji Dukpa[2], Markus Stoffel[3,4,5], Tshewang Norbu[2], José M. Facelli[1] and Justin D. Brookes [1]

1. School of Biological Sciences, The University of Adelaide, South Australia 5005, Australia
2. Conifer Forest Research subcentre, Ugyen Wangchuck Institute for Conservation and Environmental Research, P.O.Box 2049 Bhutan
3. Climatic Change Impacts and Risks in the Anthropocene (C-CIA), Institute for Environmental Sciences, University of Geneva, Geneva, Switzerland.
4. Dendrolab.ch, Department of Earth Sciences, University of Geneva, Geneva, Switzerland.
5. Department of F.A. Forel for Environmental and Aquatic Sciences, University of Geneva, Geneva, Switzerland.
6. School of Ecosystem and Forest Sciences, University of Melbourne, Victoria, Australia.

* Corresponding author: tshewangdorji1@gmail.com

6.2 ABSTRACT

This study uses annual growth rings and dendrochronological techniques to reveal the growth of Himalayan larch (*Larix griffithii*) in response to precipitation and temperature. The longevity, distinct annual ring patterns, and sensitivity to climate variables, most particularly precipitation, make Himalayan larch valuable for the reconstruction of past climate. The chronology statistics of larch showed significant dendroclimatic potentials for reconstructing summer monsoon precipitation (May-August). Tree growth was positively correlated with monsoon rainfall and negatively correlated with summer temperature, which concurs with other studies in the Himalayas. We reconstructed 639 years of monsoon precipitation for Western Bhutan, which indicated annual to decadal rainfall variability, usually alternating between periods of wet and dry episodes. The magnitude of the variability increased after 1900, showing the significant changes in rainfall patterns during the twentieth century. Most

of these variabilities were strongly associated with the modes of natural variability of the global climate system such as the El Niño–Southern Oscillation (ENSO) and the Indian Ocean Dipole (IOD). Records of high rainfall (1905-07, 1938-42, 1954-56, 1967, 1998-2000, and 2007-09) and drought records (1981-83) indeed correspond to major La Niña and El Niño episodes, respectively, and stand out clearly in the tree ring chronology. Other years, such as 1914, received heavy rainfalls, and years like 1916-17 or 1958 were quite dry despite neutral ENSO conditions; here, the unusual records can be explained by the strong positive and negative Indian Ocean Dipole (IOD), respectively. Similarly, the major ENSO events of 1972-73 and 1997-98 did not result in droughts in our reconstruction as these years coincided with a positive IOD index, which brought rainfalls to the Indian subcontinent and over to Bhutan, thereby negating the effect of ENSO events. However, in years like 1967 and 2007-2008, when La Niña co-occurred with a positive IOD index, we reconstruct excessive monsoon precipitation. The results evidence that ENSO and IOD independently and synergistically influence the growth of trees by controlling the quantity of summer monsoon precipitation of Bhutan. Our larch chronology and summer precipitation reconstruction also captured some of the well-documented megadroughts in Asia as well as preserved potential dendroclimatic signals that are common and comparable to other precipitation-based reconstructions from the Himalayas.

6.3 INTRODUCTION

Climate change and global warming in recent decades have led to the significant warming of the land and sea surfaces which has the potential to destabilize atmospheric circulation (IPCC, 2019; Lima and Wethey, 2012; Sutton et al., 2007). Atmospheric circulation results from the thermal contrasts between land and sea surfaces and is an important determining factor of rainfall pattern on land (Totman Parrish and Curtis, 1982; Zhao et al., 2006). The temperature difference between the Himalayan landmass along with the Tibetan plateau and the surrounding oceans is an example of thermal contrasts that influences the global weather patterns and are responsible for monsoon circulation in Asia (Sato and Kimura, 2007). However, with climate change having a significant impact on atmospheric circulation, the Asian summer monsoon is adversely affected and has been documented to weaken gradually (Liu et al., 2019). This weakening also has effects on and causes large interannual variations in the summer monsoon over the Himalayas, a likely candidate of interest for climate change research. Adequate summer monsoon rains are crucial not only for agricultural productivity (Kumar and Raj Gautam, 2014; Lal, 2010) but also for the discharge of rivers that originate in the region (Gautam et al., 2014). In combination with the snowmelt and discharge from the glaciers, abundant rainfall during summers also help in maintaining the role of the Himalayas to act as a water tower feeding the river systems and downstream regions, thereby supporting unique biodiversity (Gautam et al., 2014) and providing lifeline and economy to the world's most populated mountain ranges. At the same time, however, and as glaciers lose mass at a staggering pace (Bolch et al., 2012; Huss et al., 2010), the role of the Himalayan region as the water tower of Asia is at risk (Immerzeel et al., 2010).

The significant variations in the summer monsoon precipitation in recent times has caused frequent floods and droughts that have severe economic and societal bearings on Himalayan residents (Kumar, 2006). Several regions of the wider Himalayan environment are politically fragile and less resilient to cope with the immediate effects of natural disasters and their longer-term, economic impacts (Gurung et al., 2017; Watanbe and Rothacher, 1996). In addition, climate change is likely to exacerbate the occurrence of intense rainfall elements, possibly leading to more landslides during the monsoon season, whereas the ongoing wasting of glaciers has (and will even more so in the future) contribute to catastrophic glacial lake outburst floods (GLOF) from glacierized environments (Schwanghart et al., 2016; Worni et al., 2014). As a consequence of the limited resilience and more limited adaptive capacities, the probability for natural hazards to become disasters is higher in the Himalayan region than in other more developed nations. On the other hand, frequent monsoon failures or the late onset of monsoon rains with related droughts have already led to marked swings in agricultural productivity that threaten the sustenance of life-supporting socio-economic activities in the region (Lin et al., 2015) and the existence of smallholders in remote mountain environments (Cook et al., 2010; Miyan, 2015; Wangdi et al., 2017).

Approaches that are dealing with climate adaptation in the mountains have to rely on reliable predictions of likely effects of future climate change on processes, including the Indian summer monsoon (Mathison et al., 2013). The complexity lies in the monsoon pattern itself, which is highly uncertain and accompanied by high internal and inter-annual variability, thereby often leading to faulty predictions which can create doubt in the utility of monsoon predictions for climate change adaptation and mitigation approaches in the mountains (Kumar, 2006). The absence of long-term meteorological records increases the uncertainty in monsoon predictions further. As most of the instrumental meteorological records only reach back to the late 19th century at best (and often much less so in mountain environments; Salzmann et al. (2013)), the reconstruction of paleoclimatic data for the pre-instrumental period is very crucial to improve our knowledge of ongoing climatic changes and modes of natural variability in a longer-term context. Therefore, the development of long-term, high-resolution, and well-calibrated climate reconstructions from the Himalayas is imperative to understand past climatic conditions and for better, informed predictions of future climate change.

Much progress has been made on the reconstruction of past climate using high-resolution climate proxies such as tree rings (Krusic et al., 2015; Sano et al., 2013; Singh et al., 2009), ice cores (Banerjee and Azam, 2016; Dahe et al., 2000), lake sediment cores (Lone et al., 2019; Sanwal et al., 2019) and laminated stalagmite records (Zhang et al., 2018). Reconstructions from diatoms (Fayó et al., 2018), fish otoliths (Darnaude et al., 2014) and permafrost (Saito et al., 2013) are also common elsewhere but have not been investigated in detail in the Himalayas. Tree-ring based reconstructions are widely used thanks to the longevity of many tree species in mountain environments and the sub-annual resolution of dendrochronological records. The growth responses of trees, expressed through the formation of annual rings, are sensitive to climatic conditions, most notably, atmospheric temperature and precipitation, especially at locations where one of these factors is limited, thereby controlled growth-ring formation and width (Fritts, 1976).

Dendrochronological research in the Himalayas has increased significantly over the years, and most notably after the 2000s (Chhetri and Shrestha, 2010). Several tree ring-based climate reconstructions have been produced spanning from present to pre-instrumental times (700-1000 years). Most of these studies were, however, concentrated in the western part of the Himalayas, such that research becomes much more scarce as one moves towards the eastern Himalayas (Krusic et al., 2015). Further, most of the studies in the Eastern Himalayas were conducted in Nepal. Tree-ring based temperature (Bhatta et al., 2018; Cook et al., 2003), precipitation (Gaire et al., 2017), and drought (Gaire et al., 2018) reconstructions were developed from conifer species such as *Picea smithiana*, *Juniperus recurva*, *Abies spectabilis*, *Tsuga dumosa*, and *Pinus wallichiana*. Recently, researchers also reported the high dendroclimatic potentials of *Larix griffithii* and *L. himaliaca* from Nepal Himalayas (Aryal et al., 2020). For Bhutan, only two reconstructions have been realized so far, one focusing on tree-ring cellulose $\delta^{18}O$ isotope of *Larix grifithii* to reconstruct summer precipitation (Sano et al., 2013) and the other based on tree-ring width records of conifers to reconstruct summer temperatures (Krusic et al., 2015).

Our study presents the most extensive monsoon precipitation reconstruction for Bhutan Himalayas with a further comparison of our reconstruction to those available in the region to identify common climatic signals captured by the trees growing at spatially different locations. The study is expected to dramatically improve understanding of altered monsoon precipitation patterns over the eastern Himalayas that are associated with natural disasters such as droughts, landslides, glacial lake outburst floods and ecosystem changes.

6.4 MATERIAL AND METHODS

6.4.1 Study site

The Kingdom of Bhutan is situated in monsoonal Asia where summer monsoon (May-September) usually contributes *c*. 80% to annual rainfalls. The country has a rich natural forest cover of more than 70% of its geographical area that is vital for maintaining healthy watersheds and river catchments and providing a suite of ecosystem services (Dorji et al., 2019; Sears et al., 2017). Despite allocating vast areas to forest cover and adopting a unique, carbon-neutral policy embedded within a development philosophy of Gross National Happiness, the kingdom has to bear the consequences of global climate change. As such, climate change and the increasingly erratic monsoon rainfall patterns observed in recent years have increased both the frequency of flash floods and accelerated the melting of glaciers, which in turn has resulted in an increase of glacial lake outburst floods (GLOF) and landslides (Ghimire, 2004). Furthermore, the prevalence of persistent heatwaves accompanied by more frequent monsoon failures (Wangdi et al., 2017) have propelled the wasting of glaciers further. The increasing frequency of natural disasters has raised the need to reconsider infrastructure development and natural disaster preparedness. Only within two decades, Bhutan experienced significant water-related disasters. The best example is the Lugge Tsho GLOF of 1994, where 21 people lost their lives and properties (Watanbe and Rothacher, 1996). Similarly, the monsoon floods of 2000, 2004, 2009, 2015 & 2016 and recent Cyclone Amphan (2020) are some major ones that destroyed lives and properties. At the same time, pre-monsoon droughts (Wangdi et al., 2017) and drying of water sources continue to disrupt the social harmony and threaten the country's key economic sectors like agriculture, tourism and hydropower (Hoy et al., 2015).

In Bhutan, systematic meteorological measurements reach back only to-1985 (NCHM, 2018), and records often are incomplete and of insufficient quality (Eguchi and Wangda, 2011). Climate reconstructions are, therefore, essential to bridge the gap between present and past climatic conditions and to put recent changes into perspective, thereby enabling better prediction and enhanced adaptation to climate change. Geographically, the country lies between the world's two most populous countries, China and India. Climatically, Bhutan is situated at the crossroads of two climate extremes: Chiranpunji (India) to the south with a warm climate and the world's most torrential rainfall recorded and the cold Tibetan plateau to the north with cold conditions and an environment lacking abundant rains. The wide climate variations and high mountain ranges in conjunction with the existence of barely disturbed old-growth forests, rich in conifer species, make Bhutan an ideal place for dendroclimatic studies (Krusic et al., 2015). In this paper, we developed a tree-ring chronology from Himalayan larch (*Larix griffithii)* to present multi-centennial monsoon precipitation (May-August) reconstruction for Bhutan. *L. griffithii* is the only deciduous conifer native to the Eastern Himalayas and is among the least studied species, despite its high dendroclimatic potential (Aryal et al., 2020; Bhatta et al., 2018). Trees of this species are long-lived, have distinct annual rings, and grow adjacent to the tree lines (3000 - 4100 m asl) where environmental conditions are limiting, which is an essential requirement for dendroclimatic studies. Further, the species are confined to smaller pockets close to alpine meadows and usually grow on dry sites where growth is limited by moisture. Trees growing in such limiting environments are known to preserve the unique signature of climate, particularly temperature and/or precipitation in their wood over the years.

6.4.2 Sampling strategy

Tree core samples were collected from western Bhutan (27° 40’ 36’’ N, 89° 16’ 11” E) at an elevation ranging from 3420-3510 m asl in early May 2019 (Figure 34). The sampling area is located at two days walking distance from the nearest motorable road en route to Jumolhari – one of the highest mountain peaks in Bhutan (7326 m).

The forest was purely dominated by *Larix griffithii* with some understorey shrubs. No evidence of human disturbance or grazing was recorded in the forest as the site is located far away from any settlement. The forest canopy was reasonably open during fieldwork, which could be due to the deciduous nature of the species. The climate in the study area is cold temperate to sub-alpine with four distinct seasons.

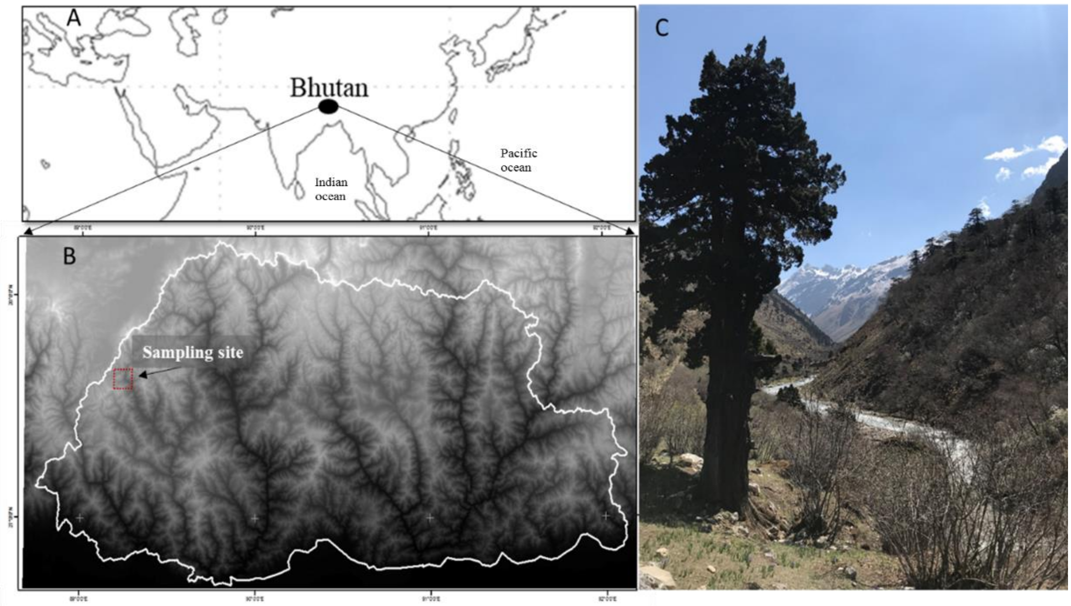

Figure 34. The Map shows the position of Bhutan in the Himalayas and its proximity to the Indian ocean in the south and pacific ocean in the south-east (A), Map of Bhutan showing the sampling area in Western Bhutan (B), and an overview of the study site (C).

Data recorded by the nearest meteorological station (1996-2017) situated about 50 km south of the study site showed a mean annual air temperature of 13.8°C and a mean yearly rainfall of 954 mm with a majority of precipitation occurring during the summer monsoon months (May-September). July and August were both the warmest and the wettest months with an average monthly temperature of 20°C and average monthly precipitation of about 228 mm and 256 mm, respectively (NCHM, 2018). Winters (December–February) are cold with an average monthly temperature of 6.7°C and dry with a mean monthly precipitation of less than 2 mm accounted by occasional snowfalls. Identifying the climate variables that control tree growth relies on long-term (>30 years), reliable climate data. As instrumental data in Bhutan reach back to 1985, but with major shortcomings (Eguchi and Wangda, 2011), we used the high-resolution, updated TS4.03 version climate data set from the Climate Research Unit (CRU) as it provides grid point temperatures on a 0.5° × 0.5° grid for the period 1901–2018 (Harris et al., 2020).

The selection of trees for sampling was concentrated, as far as possible, within a homogenous area of soil types, topographic features, and elevation range to exclude undesirable effects introduced by site variations. Because of the nature of the study aiming at constructing a long-time relationship between tree growth and climate variables, we purposely selected mature and old trees to obtain core samples. The cores were sampled at breast height (1.3 m). A total of 66 cores were collected from 33 healthy and dominant *Larix griffithii* trees (i.e., two increment core samples per tree).

6.4.3 Tree ring data and chronology development

The core samples were processed following well-established dendrochronological procedures (Bräker, 2002) in the dendrochronology lab of the Conifer Forestry Research Centre, Ugyen Wangchuck Institute for Conservation and Environmental Research, Bhutan. The process involved air drying and glueing of cores on wooden mounts, followed by sanding of core samples using a belt sander and successive grades of sandpaper (coarse to fine) until distinct annual rings were visible.

The growth rings of individual cores were counted under a stereo zoom microscope and marked with a pencil with dots to represent decades and centuries. The core samples were compared with one another for similarities in the patterns of ring widths (wide and narrow rings) using skeleton plots in dplR package in R dplR (Bunn and Korpela, 2019; Bunn, 2008). Based on these comparisons and the known year of formation of the outermost ring, each ring was assigned to the exact calendar year using a stereo zoom microscope. The core samples were then scanned with a high-resolution (1200 dpi) Epson expression 11000XL scanner. The scanned ring widths were measured to a precision of 0.01 mm using CDendro and CooRecorder. The measured ring-width series were then evaluated statistically for cross-dating and measurement errors by using the program COFECHA (Holmes, 1983). The program compares individual ring-width series against a master chronology, which is derived by averaging all the series. The program also evaluates errors in measurement through outliers and flags the tree-ring series with possible faults. The flagged tree-ring series were re-inspected and re-measured where necessary. Cores with weak correlations yielding questionable dating accuracy were removed from the analysis. As a result, a total of 29 ring-width series from a total of 19 trees were used in the development of the final chronology.

Each tree-ring series was detrended using a smoothing spline using the dplR package (Bunn, 2008) of R software (CoreTeam, 2019). This detrending removes “non-climatic variance” such as growth trends related to age or competition within stands while preserving the climate signals in the form of a ring-width index (Cook and Peters, 1981). A mean-value chronology was then built by averaging the ring width index of all series using Tukey’s biweight robust mean in dplR (Bunn and Korpela, 2019; Bunn, 2008). We also constructed a residual chronology by “prewhitening” the series before averaging; the latter differs from the standard chronology as the first order autocorrelations are removed by this process, thus leaving only white noise (Bunn and Korpela, 2019). To ensure adequate representation of sampled trees and to sufficiently reflect common climatic signals among the tree-ring series, we employed an expressed population signal (EPS) threshold ≥ 0.85 for the final chronology. In addition, we computed various descriptive statistics including first-order autocorrelation (ar1), a running mean interseries correlation coefficient (rbar) and signal-to-noise ratio (snr) with the aim to assess the quality of the chronology and its dendroclimatic potentials (Gaire et al., 2018).

6.4.4 Climate data

The biggest challenge of dendroclimatic studies in the Himalayas, and in particular in Bhutan, is the absence of long-term climate data (Krusic et al., 2015; Sano et al., 2013). Meteorological stations were established primarily in the 1980s, and most data was only available as of 1996 (NCHM, 2018). Further, most of the meteorological data were observed manually, and the series often contain missing observations that undermine the reliability of data (Eguchi and Wangda, 2011). To address this problem, we used the CRU high-resolution TS4.03, spatially gridded 0.5° × 0.5° climate dataset (Harris et al., 2020) containing monthly averages of climate variables from 1901 to 2018. A total of 13 grids surrounding the study area were selected for this study. The monthly temperature and precipitation values of these 13 grids were averaged for the period 1901-2018 and correlated with the RWI obtained from the residual chronology.

6.4.5 Climate-growth relationships

Spatial correlations were tested between the RWI of the larch chronology and climate variables using the KNMI climate explorer (climexp.knmi.nl) which contains climate data and analysis tools with relevance to high resolution paleoclimatology (Trouet and Van Oldenborgh, 2013). The growth response (RWI of the residual chronology) of trees to climate (atmospheric air temperature and precipitation totals from CRU TS4.03) were computed using the “numerical calibration of proxy-climate relationships” provided by the package treeclim (Zang and Biondi, 2015) in R (CoreTeam, 2019). The software offers static and moving bootstrapped response and correlation functions to compare climatic data at monthly resolution with tree-ring chronologies. By doing so, we explored the factors limiting tree growth as reflected in the tree-ring series, varying the time windows from monthly to seasonal and annual scales. Point-by-point correlations were computed between the tree-ring chronology and climate variables by using a window of 14 months (biological year from August of the previous year to current September where tree growth is assumed to have finished (Zang and Biondi, 2015).

6.4.6 Dendroclimatic reconstruction

Based on the climate-growth model, the best climatic variable that significantly represents growth mechanisms of *Larix griffithii* was selected for reconstruction. The best climate variable was modelled to tree-ring data using the skills function of treeclim package (Zang and Biondi, 2015) in R software (CoreTeam, 2019). We used the linear regression analysis (transfer function) to estimate or predict values of summer monsoon precipitation (SMP) from the corresponding value of the RWI of the corresponding year. The straight-line equation or model for the linear regression is: SMP = RWI*273.58 + (-24.81). The value of -24.81 is the intercept of the straight line, 273.58 is the slope and SMP is the predicted summer monsoon precipitation based on the corresponding value of the RWI of the same year. We evaluated the time stability of the model by deploying the calibration and verification methods. The split-half validation approach was adopted as it takes the maximum time available and divides the time window into two equal periods of verification and calibration periods. Thus, the calibration period was defined from 1959-2018. The calibrated and full models were subjected to various parameter estimates and goodness of fit, such as the Durbin-

Watson test for autocorrelated residuals, the reduction of error (RE) and the coefficient of efficiency (CE) (Cook et al., 1994) in treeclim. Positive values of RE and CE were taken as a basis for a reliable model with useful information and good reconstruction skills. The Durbin-Watson statistics was used to check for autocorrelation or similarities of the ring-width series over successive time intervals. Once the model was judged stable, it was used to reconstruct summer monsoon precipitation (May-August) based on the RWI chronology. The reconstruction of monsoon precipitation was realized for the period during which the EPS remained above the threshold of 0.85.

6.4.7 Structural Equation Model (SEM)

The structural equation model (SEM) is useful to understand direct and indirect influences of related climate variables such as temperature and precipitation and tree growth.

Further, a multiple linear regression model was developed to understand the influence of global weather events such as ENSO, IOD and PDO on the tree growth (RWI). The period 1950-2018 was considered into the analysis based on reliable climatic data after the 1950s. The analysis was carried out for the whole period as well as segregated into two equal periods to account for the climatic changes over time.

6.5 RESULTS

6.5.1 Development of tree-ring chronology

Based on the 29 tree-ring series obtained from 19 healthy and dominant trees, we developed a 639-year long tree-ring chronology spanning from 1379 to 2018 (Figure 35). The mean age of selected trees was 421 years, with the oldest tree having 719 rings at sampling height, whereas the youngest tree was 153 years old. The mean annual increment was 0.99 mm/year (± 0.26 SD). The widest ring widths were recorded in 1464-65, 1773, 1915, 1938, and 1942 while the narrowest ring widths were observed in 1441, 1546-47, 1552, 1917, 1972, 2013, and 2018. Series intercorrelation was 0.51, EPS of 0.851 and a signal-to-noise ratio (SNR) of 5.56 (Table 14).

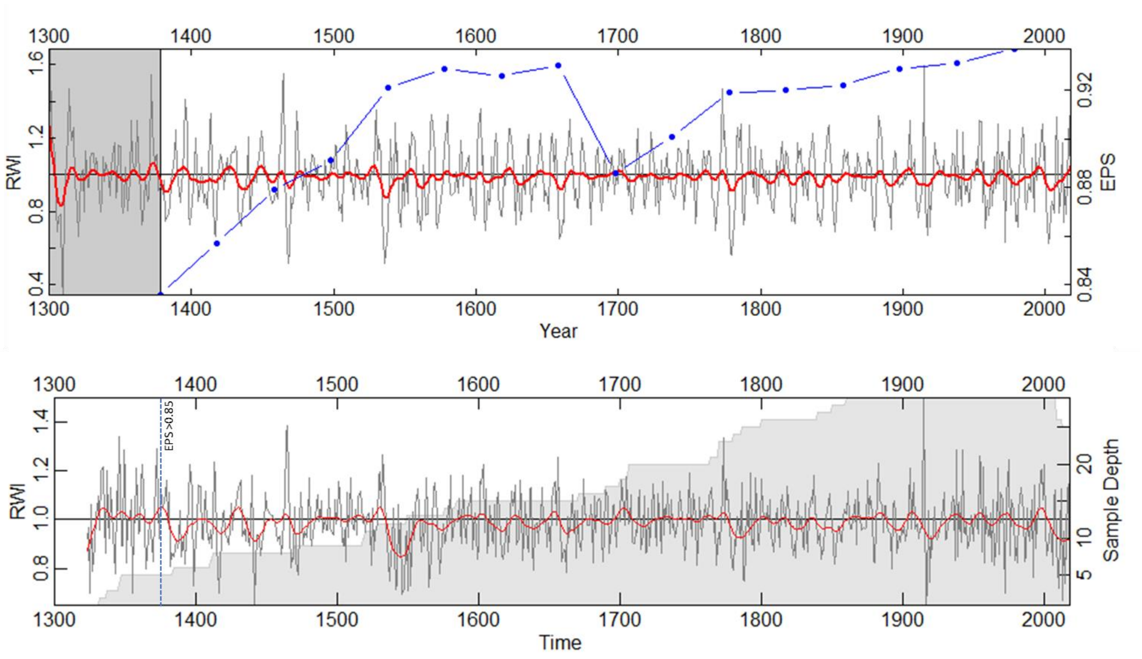

Figure 35. Himalayan larch chronology truncated at the expressed population signal (EPS) threshold of 0.85 (above) and residual chronology truncated at the EPS threshold of 0.85 (1379) along with sample depth. The red line shows the 20-year smoothing spline fitted to the ring width index (RWI).

Table 14. Descriptive statistics of Larix griffithii chronology obtained from Western Bhutan Himalaya.

Descriptive statistics	Value
No. of trees	19
No. of cores	29
Length of chronology	639 (1379 - 2018)
Mean age (in years)	421
Mean ring-width (in mm)	0.99
Standard deviation (SD)	0.26
Autocorrelation (AR1)	0.34
Inter-series correlation (Rbar)	0.51
Signal-to-noise ratio (SNR)	5.56

6.5.2 Growth response of larch trees to climate variables

The climate-tree growth analysis was realized with bootstrapped correlations and reveals that the larch trees from the Western Bhutan Himalaya are significant and positively correlated with May-August precipitation of the growth year ($p < 0.05$) (Figure 36). A similar trend was also observed with precipitation totals of March, April, and September of the current year; however, the correlation was less strong and not statistically significant. At the same time, a significant negative correlation was observed between July temperature and tree growth ($p < 0.05$) and a similar pattern of weak negative correlation observed between growth and current year's summer temperatures of June, August, and September; Figure 36). A weak, non-significant positive correlation was also observed between tree-ring growth and spring (April-May) temperatures, thus indicating that warmer temperatures in the early growing season may favour tree growth.

The spatial correlation maps computed between RWI and climate variables show that the trees respond to monsoon precipitation across Bhutan and, Sikkim, northernmost West Bengal and Bangladesh, thereby capturing the influence of the Indian summer monsoon from the Bay of Bengal. Likewise, the tree-ring records also capture, although less clearly, March-September temperature patterns of Bhutan (Figure 37).

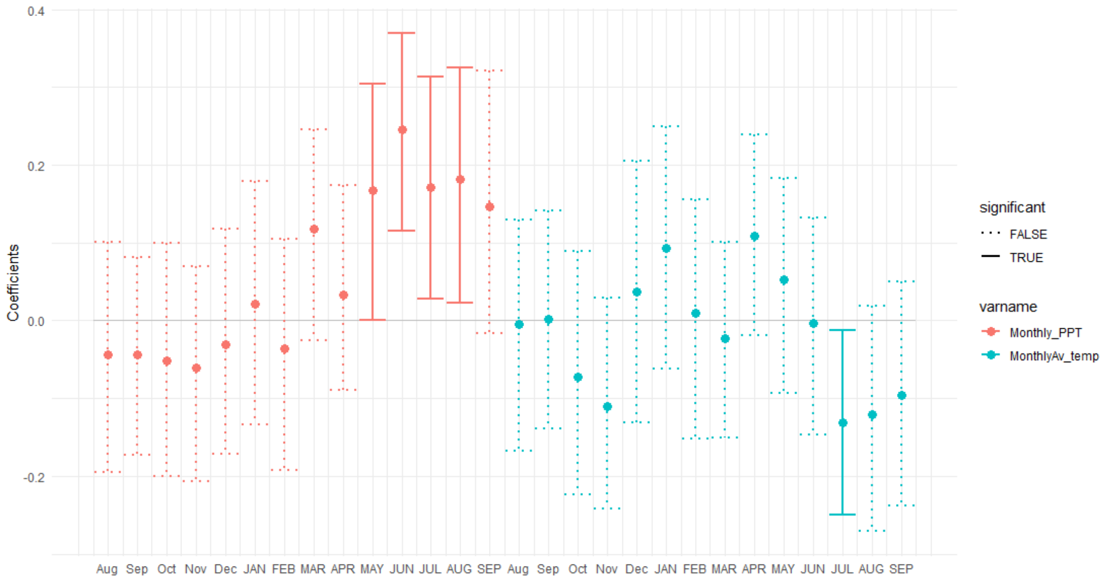

Figure 36. Bootstrap correlations calculated between the ring-width index of the Himalayan larch chronology and monthly precipitation and temperature data from CRU TS4.03 dataset covering the period 1901-2018. Thick lines indicate correlations that are significant at $p<0.05$. Months corresponding to the year before ring formation are marked with lowercase letters while the months of the current year are presented in capital letters.

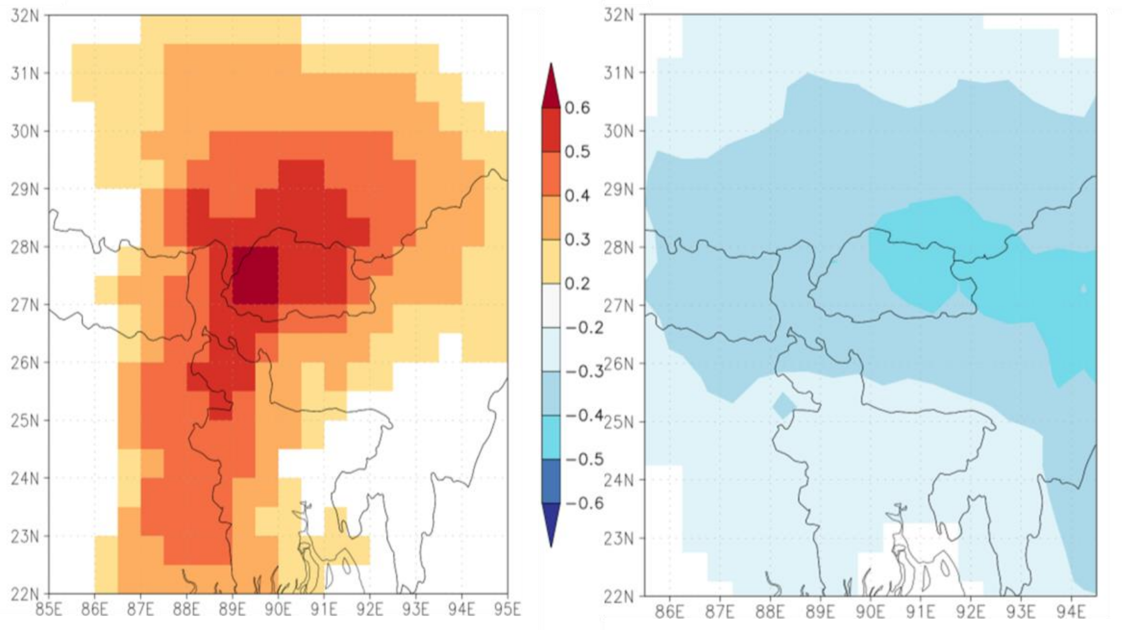

Figure 37. Spatial correlation maps (p <10%) between the ring width index of the Himalayan larch chronology and average May-August CRU TS4.03 precipitation (left) and temperature (right).

Structural equation models (SEMs) computed for the climatic variables from 1950-2018 confirmed that summer precipitation was the most influential variable controlling the radial growth of *Larix griffithii* (Figure 38). Despite the low R^2 values of the growth response variable, possibly due to the large variations in the precipitation over monthly, seasonal, and yearly scales, the significant *P*-values revealed an inherent relationship between rainfall and the tree growth response. The summer temperatures had a negative effect on the radial growth of *L. griffithii* that was evident for the full period (1950 -2018) (Figure 38A) but was non-significant for the period 1950-1984 (Figure 38B) and (1985-2018) (Figure 38C).

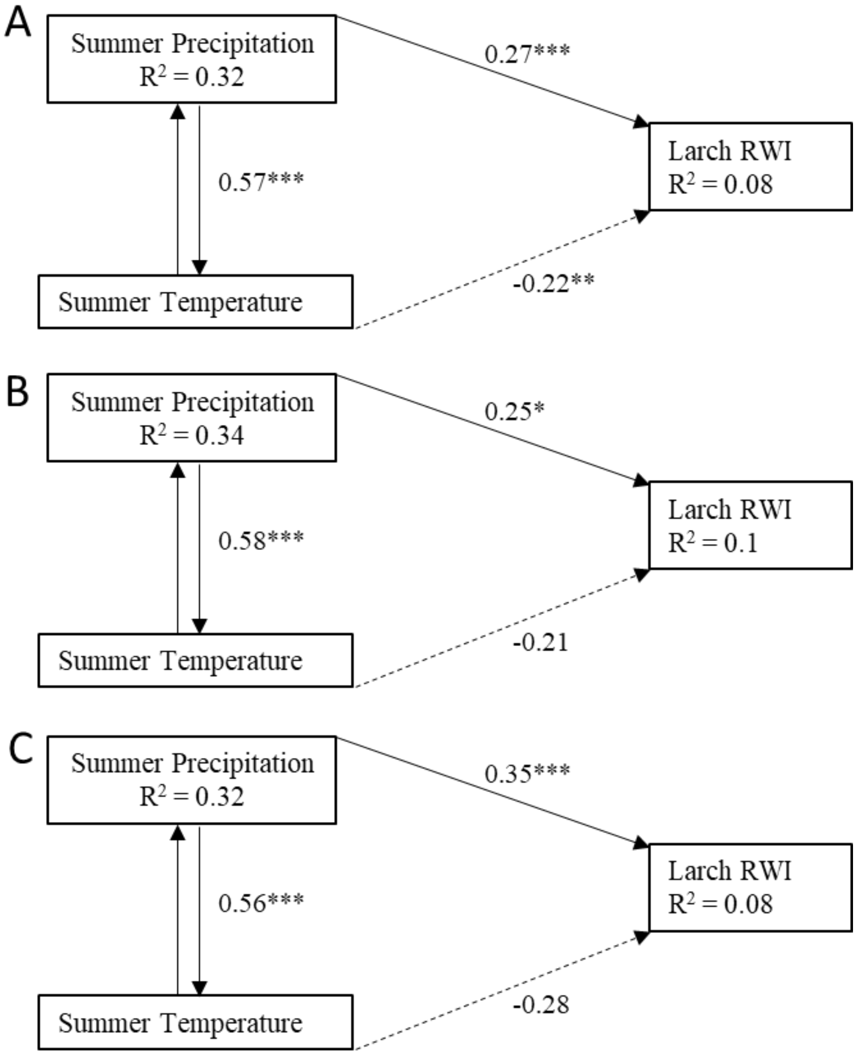

Figure 38. Structural equation models (SEMs) depicting the potential climate variables that can have a direct and indirect causal effect on radial growth of *Larix griffithii* for the full period 1950-2018 (A), sub-periods 1950-1984 (B) and 1985-2018 (C). Solid lines represent positive and dotted lines represent negative relationships indicated by the standardized coefficient values and significance. Multiple R^2 values are shown for the response variables box. Significance level are shown by $^{***}P < 0.001$, $^{**}P < 0.01$, $^{*}P < 0.05$.

6.5.3 Reconstruction of summer monsoon precipitation (May-August)

We selected the summer monsoon precipitation (May-August) as the most promising candidate for reconstruction due to its significant positive correlation with tree growth. Despite the absence of local meteorological stations data coupled with the high growth variations seen in the chronology, our model used to reconstruct the summer precipitation explains 27.1% of the variation in climate data and shows acceptable levels when subjected to the significant F-test ($F_{1,107}$ =39.79, p<0.001). The regression model yields positive RE (0.12) and CE (0.097) values as well, thus demonstrating considerable reconstruction skills, whereas the low value of the root-mean-square-error (RMSE =1.457) indicate the concentration of data points around the line of best fit—desirable for a robust prediction model. There was no evidence of autocorrelation of the residuals when model residuals were subjected to the Durbin-Watson test (p=0.26) (Figure 39).

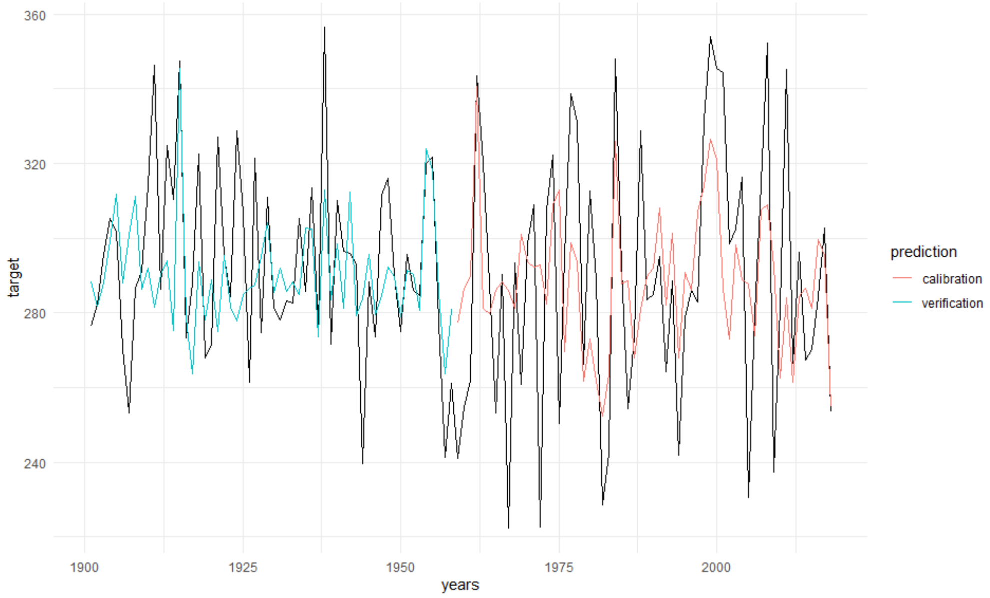

Figure 39. Actual and estimated mean May-August monsoon precipitation obtained for the verification and calibration periods.

The 639-year long reconstruction of monsoon precipitation reveals alternating patterns of intense and weak monsoon precipitation at annual to decadal timescales (Figure 40). Variability of monsoon rainfall was more important towards the latter part of the reconstruction (i.e. after 1900) than in preindustrial times. Although the reconstruction shows a consistent variation between wet and dry years, one can still observe individual years or periods sticking out of the series as they received more intense rainfall than unusual; these years include 1396, 1464-1465, 1532-33, 1656, 1773, 1775, 1914, 1954-55, 1967, 1984, 1999-2000, and 2007-2009. The drier years in the series are as follows: 1398, 1416, 1441, 1469, 1472, 1506, 1521, 1541, 1546-47, 1550-1552, 1565, 1572, 1617, 1638-39, 1756, 1768, 1787-88, 1917,1957, 1979, 1981-83, 2010, 2012, and 2018.

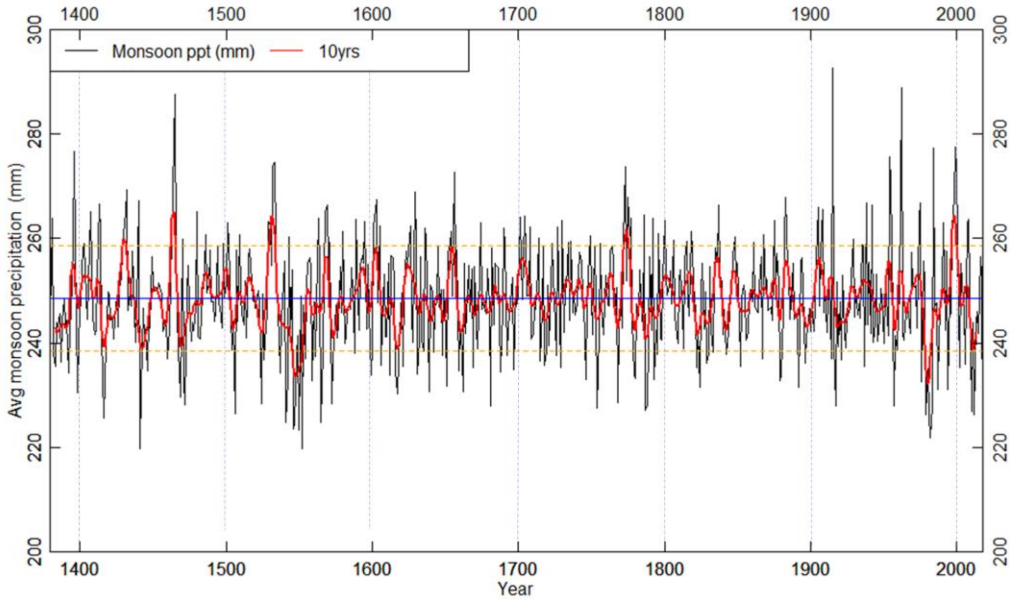

Figure 40. 639-year (1379-2018) monsoon (May-August) precipitation reconstruction showing variability in monsoon patterns. Note that both wet and dry extremes (black lines) tend to become more evident after the 1900s. The blue line depicts mean monsoon precipitation for 1901-2018, the dotted yellow lines highlight the mean ± 1SD and the red line is the 10-year low pass filter.

The multiple linear regression models in Table 15 explain the relationship between the monsoon precipitation and ENSO, IOD and PDO. We evidence a significant effect of ENSO on rainfall patterns for the period 1870-2018. We did not see any significant impact of all three independent variables on the rainfall pattern in the period 1870-1900. However, for the period 1900-1950, ENSO events seem to significantly predict summer monsoon precipitation, while for the period 1951-2018, both ENSO and IOD appeared to explain the summer precipitation variability significantly. Based on the results, we also observe that those years with less monsoon precipitation generally coincide with the El Niño events, while the years with abundant precipitation agree to a large extent with La Niña events (Figure 41). The Indian Ocean Dipole (IOD) plays a major role as well in the variability of monsoon rainfall over the Bhutan Himalaya (Figure 41); therefore, a positive IOD index will result in higher precipitation totals whereas a negative IOD index is associated with less rain. The effect is such that a positive IOD can even negate the negative effects of an El Niño event. In 1972 and 1997, for instance, when major ENSO events occurred, monsoon rainfall was still abundant thanks to the occurrence of a strong and positive IOD index.

By contrast to ENSO and IOD, we did not observe significant relationships between the Pacific Decadal Oscillation (PDO) and RWI (Table 15).

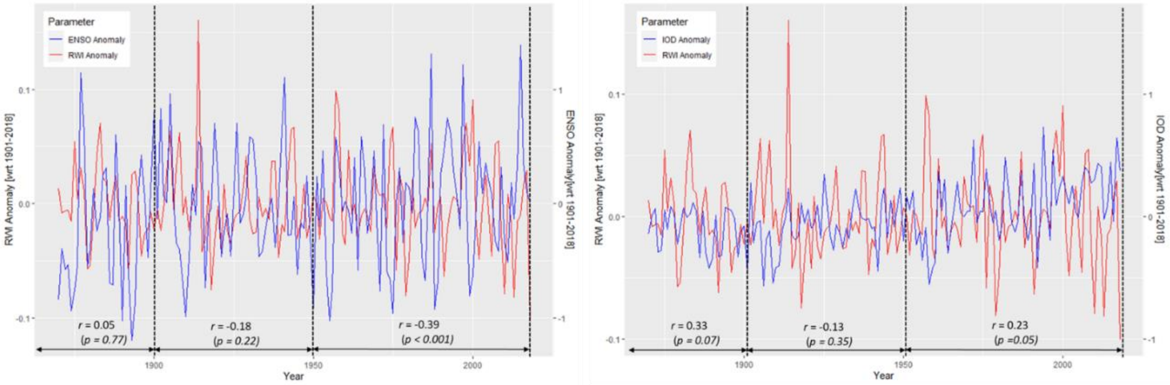

Figure 41. Relationship between the tree-growth represented by the ring-width index (RWI) and the El Niño Southern Oscillation (ENSO) anomaly (left), and with the Indian Ocean Dipole (IOD) anomaly along with correlation statistics and significance.

Table 15. Results of multiple regression models with tree-growth (RWI) modelled as the response variable and ENSO, IOD, and PDO as the predictor variable. Standard errors are reported in parentheses. *, **, *** indicates significance at the 90%, 95%, and 99% level, respectively.

Variables	Full period	1870-1900	1901-1950	1951-2018
Constant	0.995***	1.007***	0.993***	0.996***
	(0.004)	(0.01)	(0.006)	(0.006)
ENSO	-0.04***	-0.028	-0.031*	-0.037**
	(0.011)	(0.021	(0.015)	(0.013)
IOD	0.01	0.097	-0.011	0.001*
	(0.018)	(0.058)	(0.032)	(0.0005)
PDO	0.006	0.011	0.012	-0.002
	(0.005)	(0.009)	(0.006)	(0.007)
R-squared	0.12	0.15	0.118	0.211
Adjusted R-squared	0.1	-0.0005	0.06	0.174
Degree of freedom	3,114	3,17	3,46	3,64

6.6 DISCUSSION AND CONCLUSIONS

In this study, we present a 639-year-long reconstruction of annually resolved summer monsoon precipitation over the Bhutan Himalayas based on a tree-ring chronology of *Larix griffithii*. The use of such highly resolved climate proxies is crucial to understand past climate and to enhance knowledge on how modes of natural variability, along with climate change, are modulating monsoon precipitation and its variability over time. More in-depth understanding of these processes will also help to improve future predictions, especially in the Himalayas where a dearth of long-term climate information still persists. In our study, we used monsoon precipitation—a fundamental element of climate and the main contribution to the runoff in the Himalayas—to describe climate-tree growth relationships. The strong relationship between the ring width index (RWI) derived from the larch chronology and climate variables, most particularly May-August precipitation, was reflected by strong and significant chronology statistics pointing to a substantial dendroclimatic potential of the series. We evidence that growth responses of Himalayan larch growing in Western Bhutan are significantly positively correlated with summer precipitation but negatively correlated with summer temperature and that the signal recorded in the tree-ring records of Western Bhutan also is largely consistent across the country and into neighbouring Sikkim, northern parts of West Bengal and northernmost Bangladesh. As such, the spatial correlation map captured the precipitation patterns across Bhutan and to the Bay of Bengal in the south, thus revealing the influence of the Indian ocean and the Indian summer monsoon on precipitation.

Interestingly, our results on the positive larch growth response to precipitation and negative growth response to temperature were consistent with findings from other studies in *Larix griffithii* (Aryal et al., 2020; Bhatta et al., 2018) and *Abies spectabilis* (Gaire et al., 2017) from the Nepal Himalayas, *Cedrus deodara* from the Western Himalayas (Yadav et al., 2014) and for *Larix chinensis* sampled in the Qinling Mountains of China (Namieśnik et al., 2016). One possible explanation for this finding could be related to higher evaporation rates induced by high temperatures which in turn would lead to soil moisture limitations (Lakshmi et al., 2003) at the driest sites we sampled. Another possible explanation could be related to the inherent and directly inverse correlation between atmospheric temperature and precipitation as established by several studies based on meteorological observations (Cong and Brady, 2012; Łupikasza et al., 2016; Zhao and Khalil, 1993). Unique conditions with a warm and dry climate or cool and wet climate, and their transitions are known to occur in the Himalayan landmasses (Zheng et al., 2018). The markedly positive effect of summer monsoon precipitation (May-August) on tree growth and the negative effect of summer temperature (June-August) can be inherent characteristics of trees growing in limiting environments typical for sites located close to treeline of the Greater Himalayas, but also show the dependence of vegetation on the increasingly erratic monsoon precipitation from the Bay of Bengal. Our study confirms that monsoon precipitation and summer temperature are the two fundamental climate variables controlling tree growth through their direct and indirect effects on water availability (Aryal et al., 2020; Bhatta et al., 2018). Any amplification of the monsoon variability and excessive future climate warmings will not only have severe consequences on tree growth and the future distribution of important forest types in the higher Himalayas but will result in challenges and possible tensions when it comes to the allocation of water. Incidences like pre-monsoon failures, megadroughts and day-to-day rainfall variability—fewer rainy days but with the occurrence of more intense rainfall events or heatwaves—are just some of the new normal in the Himalayas that can disrupt the natural and human system (IPCC, 2019; IPPC, 2014).

The high variability in summer monsoon precipitation (May-August) was evident in our reconstruction at the interannual to decadal-scale, with marked fluctuations in reconstructed precipitation. These fluctuations have repeatedly resulted in extreme climatic conditions, including the occurrence of short-lived droughts and floods in Bhutan (Wangdi et al., 2017; Hoy et al., 2015; Watanbe and Rothacher, 1996). The erratic monsoon rainfall as well as its variability ultimately have an impact on agricultural productivity, and therefore also on prosperity or poverty. In addition, more than 80% of the annual precipitation in the Himalayan region originates from the summer monsoon and therefore is very crucial for glacier mass balance (Benn and Owen, 1998; Shrestha, 2000) and the recharging of watersheds of the often major river systems and life-supporting socio-economic activities of this highly populated part of the world (Menon et al., 2013).

The amplitude of monsoon precipitation variability has increased considerably since the early 20th century, as seen by the annual peaks and troughs in the ring-width reconstructions (see black graph in Figure 40). The repeated fluctuations in summer rain have had devastating impacts on the socio-economy and the livelihood of people living in the mountainous regions of Bhutan (Hoy et al., 2015; Watanbe and Rothacher, 1996). Our reconstruction thus represents a valuable contribution adding to the poor understanding of how monsoon is fluctuating and what are the drivers of these changes. Interestingly, our results are also in line with the high monsoon variability that is projected in climate models and the significant increase in day-to-day rainfall variability predicted for decades to come (IPCC, 2019). The recent changes in the amplitude of dry and wet monsoon seasons are predicted to exacerbate under unmitigated climate change, thus calling for massive investments in adaptation measures (Menon et al., 2013) despite the fact that Bhutan is one in a few countries reaching the targets defined at the Paris Conference of Parties (COP) in December 2015.

Past studies reconstructing changes in the Asian summer monsoon have been based on tree-rings (Gaire et al., 2017) and tree oxygen stable isotope chronologies (Sano et al., 2012) from the Central Nepalese Himalayas and the Western Himalayas (Singh et al., 2009; Yadav et al., 2014). These studies have highlighted a consistent, decreasing trend in monsoon precipitation after the 1980s, findings that are of huge concern in the wider Himalayan region. No such patterns were, however, observed in our reconstruction or other reconstructions based from oxygen stable isotopes from Bhutan, eastern Himalayas (Sano et al., 2013). The results of our tree-ring based record for Bhutan are in line with observations from Miyan (2015) who has highlighted the lesser incidence of decreasing rainfall episodes and modern droughts in Bhutan as compared to other neighbouring countries including Nepal or the Western Himalayas in more general terms. These findings, therefore, confirm that monsoon precipitation is not uniform between the Western and Eastern Himalayas, thus reflecting climate variations between the regions (Liu et al., 2011). Further investigation is, however, needed – by using a wider range of climate proxies – to improve the understanding of climate variations in the region.

The relative proximity of Bhutan to the Bay of Bengal, where the monsoon depression forms (Figure 42) may explain the differences in rainfall variability between the countries (Sano et al. (2013). The unique position of Bhutan located directly on the path of strong monsoon winds and tropical cyclones from the Bay of Bengal leads to heavy rainfall events that could have spared the country and the Eastern Himalayas from severe droughts. The summer cyclones “Amphan” (formed on16 May 2020 and dissipated on 21 May 2020) and Aila (formed on 25 May 2009 and dissipated on 27 May 2009) are recent examples of huge depressions that brought heavy rain showers along their path from the Bay of Bengal to Bhutan (Figure 42).

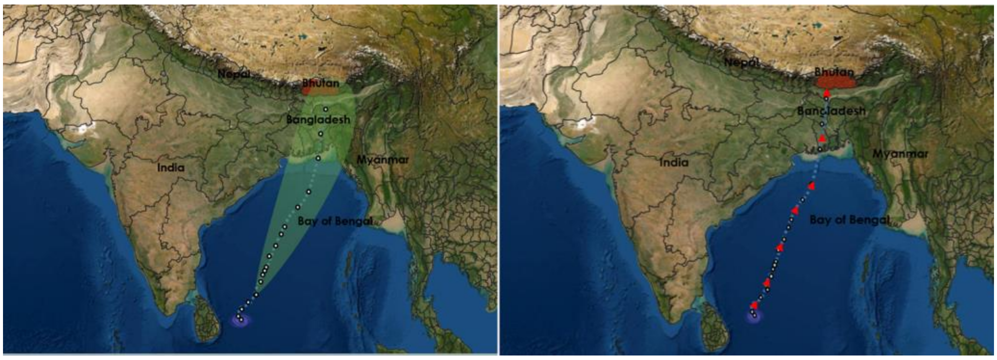

Figure 42. Maps of South Asia depicting the summer monsoon path from the Bay of Bengal and Bhutan's location on their path, which brings heavy monsoon rains. Shown in the picture are two of the many tropical cyclones-2020's cyclone Amphan (left) and 2009's cyclone Aila (right).

Nevertheless, our reconstruction captured many of the well-documented droughts in the Asian continent, namely the Ming Dynasty (1638-41), Strange Parallels (1756-68), East India (1790-96) and late Victorian Great (1876-1878) droughts defined in the regional drought atlas (Cook et al., 2010). Besides, the extreme drought that reportedly resulted in the rapid decline of Meyer spruce in China in the 1920s (Liang et al., 2003) was reflected in the Himalayan larch chronology as well, yet with less severe consequences. Furthermore, a comparison of our monsoon precipitation reconstruction (May-August) to other reconstructions from the Himalayas (Figure 43) revealed several areas of common trends and patterns in monsoon precipitation over the wider Himalayas. For instance, the dry years during the 1740-50s, 1790s, 1920-30s, 1950-60s, 1981-83, and 2014-16 were common between our and most other reconstructions. Similarly, the years with abundant summer monsoon rainfall (such as 1727-30, 1773-1777, 1900s, 1914-16, 1967, 2000s, and 2007-2010) were common for most of the reconstructions, thus indicating the presence of uniform climate signals preserved in trees growing across the Himalayas. Noteworthy, however, considerable variability is obvious regarding extremely dry and wet years reported in the different precipitation reconstruction from different locations in the Himalayas (Table 16).

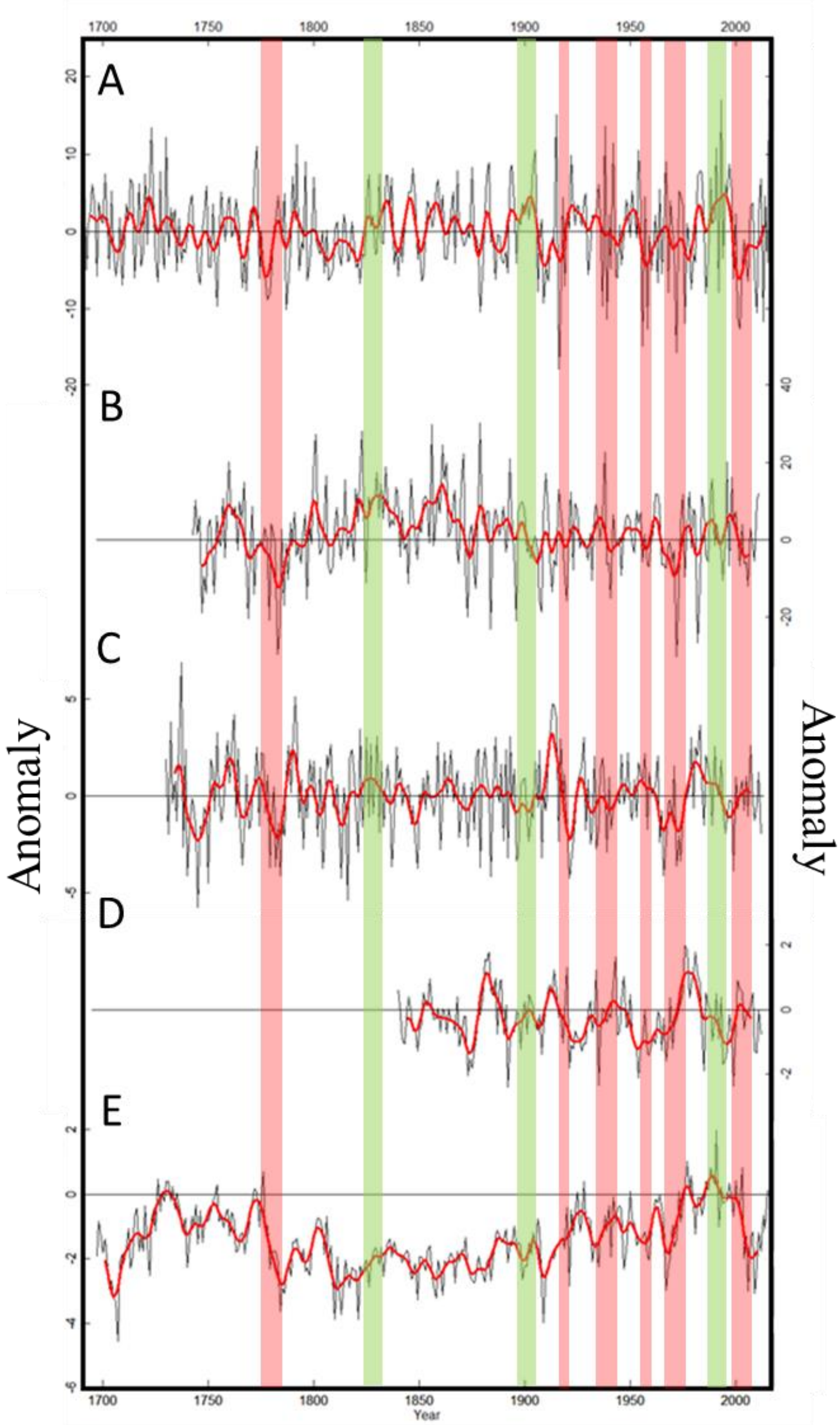

Figure 43. Comparison of the *Larix griffithii* monsoon precipitation reconstruction(this study) with records from other sites across the Himalayas: periods with common dry (light red band) and wet (light green) signals are highlighted for the last centuries common to all series. Comparison of the (A) reconstructed monsoon precipitation (anomaly) based on tree-ring records from the Bhutan Himalayas (this study) with (B) tree-ring oxygen isotope ($\delta^{18}O$) records (Sano et al. (2013) from the Western Bhutan Himalayas, (C) pre-monsoon (Feb-May) precipitation (Yadav et al., 2014) from the Indian Himalayas, (D) spring precipitation (Gaire et al., 2017) from the Nepalese

Himalayas, as well as (E) drought (scPDSI) reconstruction of the Central Himalayas (Gaire et al., 2018). Records were fitted with a 10-year low pass filter to highlight variations. All reconstructions were truncated at the 1690s for uniformity although some reconstruction including ours go back to the early 1400s. Data in the plot (B) were transformed to fit the scale of the other series.

Table 16. Comparison of studies reporting abnormal (arid and very wet) precipitation events in reconstructions realized in different locations of the wider Himalayan region.

Reference	**This study**	**Gaire et al. (2017)**	**Liu et al. (2011)**	**Yadav et al. (2014)**
Precipitation	May-August	Mar-June	Annual	Feb-May
Location	Western Bhutan	Western Nepal	South Tibet	Kumaon, Western Himalaya
Wet years	1396, 1432, 1464-1465, 1532-33, 1629, 1656, 1773, 1775, 1914, 1954-1955, 1967, 1984, 1999-2000, 2007-2009	1778-1886, 1850-1862, 1909-1917, 1971-1984, 2000-2008	1480-1535, 1626-1639, 1651-1671, 1704-1798, 1845-1872, 1888-1907, 1917-1940, 1994-2008	1782-1786, 1744-1748, 1812-1816, 1846-1850, 1920-1924, 1964-1968, 1972-1976, 1995-1999
Dry years	1398,1416, 1441, 1469, 1472, 1506, 1521, 1541, 1546-47, 1550-1552, 1565, 1572, 1617, 1638-39, 1756, 1768, 1787-88, 1917, 1957, 1979, 1981-83, 2010, 2012, 2018	1873-1877, 1921-1923, 1925-1929, 1951-1956, 1958-1962, 1994-1996	1536-1625, 1640-1650, 1672-1703, 1799-1844, 1873-1887, 1908-1916, 1914-1993	1744-1748, 1782-1786, 1812-1816, 1846-1850, 1920-1924, 1964-1968 1972-1976, 1995-1999

Our study also highlights that the most important drivers controlling annual monsoon rainfall variability over the Himalayas are associated with the El Niño Southern Oscillation (ENSO) and the Indian Ocean Dipole (IOD). We evidence that most years for which we reconstruct drier or wetter conditions coincide with El Niño and La Niña events, respectively. However, in years during which the IOD was highly positive, its influence on reconstructed monsoon precipitation was much stronger than that of ENSO events, thereby indicating the complex behaviour and control of Asian summer monsoon precipitation. One hypothesis for this ambiguity in the ENSO-monsoon relationship was explained by the high day-to-day and interseasonal variability of the monsoon that would nullify the remote effect of the ENSO (Kumar, 2006). In addition, the sensitivity of summer monsoon to local and regional water cycles, including tropical cyclones and the IOD, seem to play a significant role in influencing rainfall patterns over the Himalayas as well. As such, a positive IOD index will generally result in more monsoon precipitation, while a negative dipole is normally associated with lesser rain. As seen in this study, the IOD can, thus, either worsen or reduce the effects of ENSO. A positive IOD during the major El Niño events like 1972, 1983, 1994 and 1997 negated the effects of El Niño by bringing more precipitation to the Bhutan Himalayas, such that no incidences of water shortage were recorded in the tree-ring series. Likewise, the period 2007-2009 received heavy showers in the Himalayas as these years were associated with both a positive IOD index and the occurrence of a La Niña event. Therefore, the combined effect of the ENSO and IOD – by reinforcing or negating one another – seem to control tree growth independently and synergistically in the Bhutan Himalayas through their influence on the Asian summer monsoon and the resulting precipitation.

Past studies have indicated the effect of the Pacific Decadal Oscillation (PDO) on monsoon patterns in the Himalayas (Sano et al., 2013). However, such a relationship was neither observed in our reconstruction nor by Singh et al. (2009). The PDO operates on a timescale with longer periodicities as compared to the ENSO, which may explain the weaker correlation between the RWI and the PDO in our reconstruction exhibiting high annual-to-decadal monsoon variability. Analysis of spectral periodograms for reconstructed precipitation likewise reveals the significant short-term (i.e. yearly or 2-3 years) and decadal (above ten years) periodicity which confirms the linkage with short-term ENSO/IOD activity and longer-term Atlantic Multidecadal Oscillation, respectively (Figure 44). Similar findings were also reported by the precipitation reconstructions in the Himalayas (Gaire et al., 2017; Singh et al., 2009; Yadav et al., 2014) and on the Tibetan plateau (Fang et al., 2009; Wang et al., 2008).

We conclude that the *Larix griffithi* reconstruction of monsoon precipitation from the Western Bhutan Himalayas significantly contributed to a better understanding of drivers and consequences of intense and weak monsoon rainfalls and thereby propels knowledge of how changes in the ENSO and IOD have impacted on precipitation totals in the past. More research is needed to enhance our understanding of the crucial source of humidity and rainfall across the Himalayas further, and we call for more studies in other regions of the Himalayan range to complement this work, with the aim to provide decision-makers and the climate modelling community baselines for the projection of likely impacts of climate change on monsoon precipitation and the design of mitigation and adaptation strategies across the Third Pole.

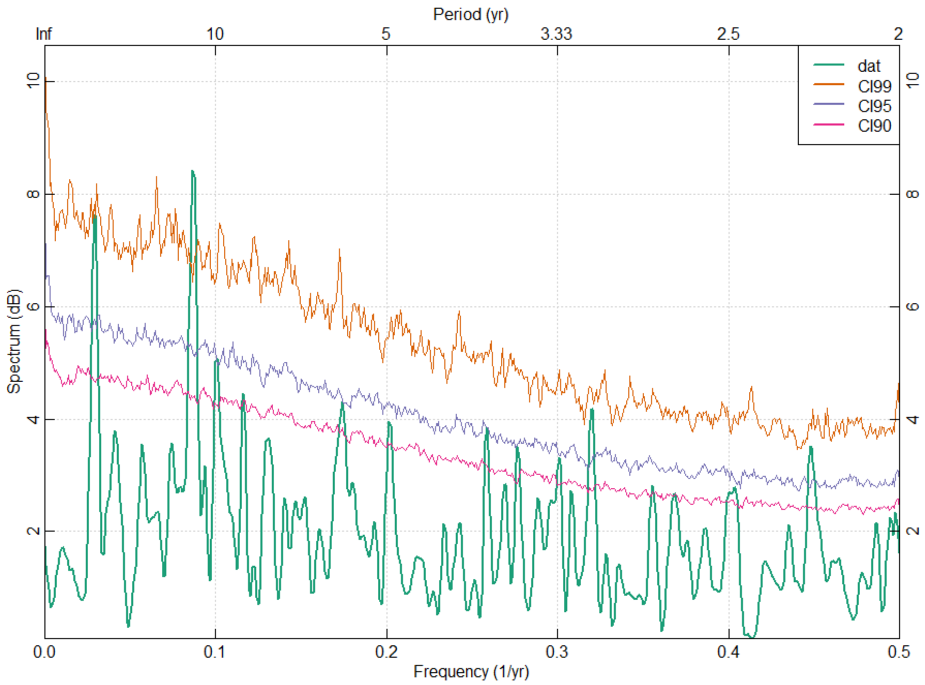

Figure 44. The spectral analysis of the reconstructed monsoon (May-August) precipitation series spanning the period AD 1379–2018. The confidence interval of 99% was determined as significant to test the periodicity of the peaks identified in the reconstruction.

6.7 ACKNOWLEDGEMENTS

The study is conducted as a part of the PhD thesis of the first author; hence, the University of Adelaide, RTP scholarship of the Australian Commonwealth Government and Royal Government of Bhutan is duly acknowledged for making the study possible. Special thanks to officials of the Ugyen Wangchuck Institute for

Conservation and Environmental Research (UWICER), Royal Government of Bhutan for facilitating our fieldwork in Bhutan. Specifically, we would like to thank Dr Kaka Tshering, head, Conifer Forest Research Subcentre, UWICER for his support while in Bhutan and Kuenzang Dorji for his assistance with GIS and the preparation of maps. The fieldwork was supported by the Royal Government of Bhutan through a project supported by the Forest Ecosystem Restoration initiative of the Korea Forest Service and Convention on Biological Diversity (CBD). The authors would also like to thank Paul J. Krusic, Cambridge University, Kathy Allen and Thiet Van Nguyen, Melbourne University, for their advice on crossdating and detrending of tree-ring samples. Pema Wangchuk and Rupali Dubey (who were interning at the UWICER), Bikash Gurung, Ganga Prasad Gurung, Tanka Raj Gurung and Sonam Dorji helped us with the coring and sample preparation in Dendrochronology Lab without whose support, it would be impossible to complete this arduous task.

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Chapter 7: General Discussion and Conclusion

The research undertaken in this PhD thesis has shown the invaluable nature of temperate forest of Bhutan Himalayas for their role as a source of rural livelihood, global carbon sink and indicators of changing climate (Figure 45). Despite their significance, the forests are facing a consistent threat, and several aspects related to their ecology, conservation and sustainable management are primarily overlooked (Zobel and Singh 1997). The major bottleneck is the limited scientific research and information that guides sustainable forest management (FAO 2000). The complexity is magnified by the increasing anthropogenic pressures on the natural resources, which is a direct outcome of population growth and economic development (Singh 1998; Ranjan 2018).

Being located on the geologically fragile Himalayan landscape with low adaptive opportunities, climate change, and related impacts also pose a considerable challenge for Bhutan. Flooding events and droughts have become a 'new normal' caused by highly erratic rainfall while human pressures, forest fires, regeneration failures and pest outbreaks are degrading the natural forests (Tshering and Tshering 2018; Gyeltshen and Tenzin 2018). The warming of Himalayas, at a rate higher than the global average, has already seen glaciers retreating at an alarming pace (Bolch et al., 2012; Bajracharya et al., 2006; Ding et al., 2006). The warming can present significant challenges to high-elevation forests and alpine ecosystems (Singh et al. 2010). The upward shifting of forests as a result of warming would gradually narrow down the habitat of treeline species that are already approaching the threshold of climatic limits (Dubey et al. 2003; Singh et al. 2012). There is already evidence of shrubs encroaching into the alpine meadows and altering the biodiversity and ecosystem functions (Brandt et al. 2013).

The sustainable management of natural resources that is resilient to climate change will depend on readily available information, resources, and human capacity. Forests can play a cushioning role in sustaining livelihoods, sequestering atmospheric carbon, and addressing the long-term goal of mitigating climate change. Forest management that is strongly founded on science-based knowledge and which is flexible enough to incorporate changes from experience and prevailing conditions is required. Such a management approach can ensure the sustenance of ecosystem services from the forest and enhance people's wellbeing while reducing social vulnerability.

The value of Bhutan's unique forest, which covers 71% of the total area has been highly recognized globally (Sargent et al. 2009). First, the forests serve as a storehouse for invaluable ecosystem services (Chapter 2). The ecosystem services are strongly linked to rural wellbeing and poverty alleviation (Sandhu and Sandhu 2014) and form a crucial component in fulfilling the millennium development goals in Bhutan. However, there exist some areas of mismatch in the priorities of ecosystem services between local forest users and forest managers which can result in competing interests and conflicting objectives of forest management. There is a need for awareness programs and environmental education to value the full range of ecosystem services by all the stakeholders. Any differences in the preferences over forest goods and services should be addressed through proper consultation and collective decision making so that the interests of all stakeholders are integrated into existing forest policies and local forest management plans. Such a process will also provide clear directives for long-term conservation goals. Many ecosystem services were becoming scarce and vulnerable to human pressures and climate change (e.g., freshwater, timber). As most of the population and urban centres in Bhutan are concentrated in the valleys, this can lead to localized overuse of forest goods and services surrounding the human settlements. There is a need to identify threatened ecosystem services and develop management plans to restore them. Ideally, this will be informed by research and community consultation. Not much information is currently available in the field of cultural/spiritual and recreation/ecotourism types of ecosystem services and how they will respond to climate change. These areas are crucial avenues for future research and development as more people are realizing the benefits of nature and will be increasingly availing these services for peace and tranquillity from their modern lives.

Secondly, the protection of existing forests and restoration of degraded ecosystems occupy a top priority for the continued provision of ecosystem services in the mountainous countries. The primary threat to high-altitude forests, most notably the oak (*Quercus semecarpifolia*) dominated forests, is the inadequate forest recruitment, which required proper assessment and documentation (Chapter 3). The need for a nationwide assessment of oak regeneration was also highlighted by local stakeholders, including forest managers and local communities (Chapter 2). Our study indicated that sapling and tree recruitment—which is the fundamental requisite for forest sustenance—currently poor in these forests. The chronic absence of oak recruitment in the Bhutanese forests is similar with the results from studies conducted in the central Himalayas (Shrestha et al. 2004; Shrestha and Paudel 1996; Singh et al. 1997) and western Himalayas (Singh and Rawat 2010; Singh and Rawat 2012). Forest over-grazing combined with inadequate canopy gaps and human pressures appear to be the crucial factors hampering forest regeneration. There is a significant gap in recruitment spanning over several decades, which is presented by small-class seedlings and old trees, but the absence of individuals in between the seedling and tree stages. Even undisturbed forests located far away from human settlements did not give adequate oak recruitment indicating that non-management or "let nature take its course" is not good enough. Continuing with no interventions can severely affect the oak forests distribution and provision of ecosystem services. The oak forests are gradually being replaced by pine and other vegetation types (Naudiyal and Schmerbeck 2017; Bisht and Kuniyal 2013; Singh et al. 1984). There are already experiences from neighbouring countries that indicate declining ecosystem services such as freshwater (Singh and Pande 1989), carbon capture (Pandey et al. 2019), nutrient cycling (Singh et al. 1984) and wildlife populations (Mcshea et al. 2007) as a result of changing forest composition. It is time for bold collective partnerships between forest managers and local communities and modifications to existing forest policies and management plans to suit the changing needs. Sustainable forest management focussing on the creation of canopy gaps through thinning/tending practices and controlling grazing pressures could be a way forward and should be given special consideration. The periodic removal of trees as a part of the thinning regime will meet the local demand for timber and fuelwood while promoting forest growth and productivity through reduced competition for space and nutrients. The forest growth and carbon sequestration potentials will also be enhanced from the managed forests (Schroeder 1991). Primarily, the effort should arise from local-community, and the role of forest managers and researchers should be that of facilitators or educators. Creation of awareness programs and educating the public on all aspects of forest ecology, ecosystem services and restoration should form the core component of a successful community-based sustainable forest management (Figure 45).

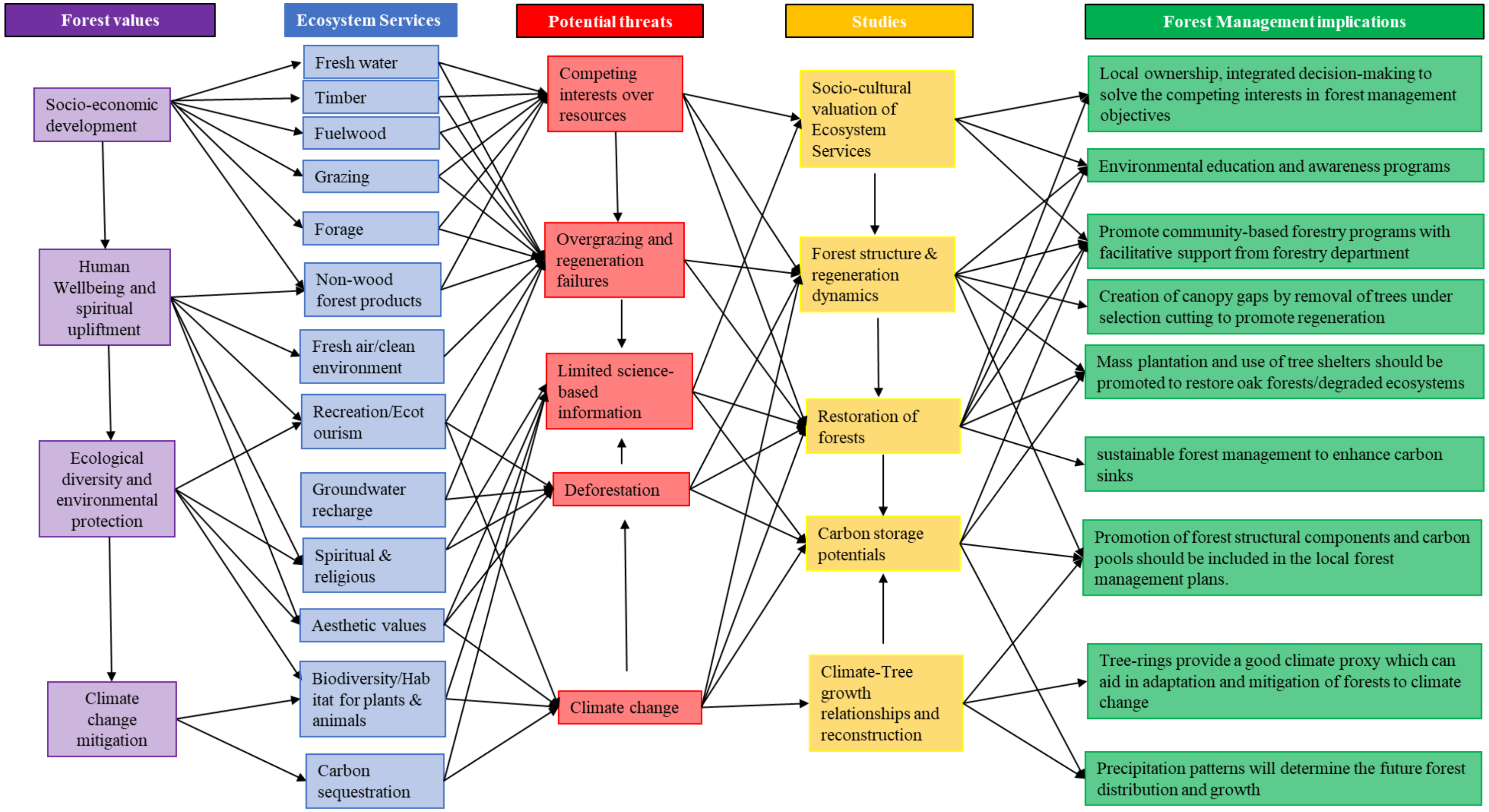

Figure 45. Structural framework depicting the importance of Himalayan forests and threats that undermine their sustainability. It also shows how the PhD research is contributing to better understanding and management of forest resources in Bhutan.

There also exists complication over the ownership of forest and its resources between centralized and decentralized control. Such confusion often leads to “tragedy of commons” and degradation of forest resources. Such issues should be solved through the implementation of community-based natural resource management, the best example being community forestry, and transfer of ownership from central government to local communities (Figure 45). This process would aid in the formulation of local specific forest management plans and sustainable utilization of resources. The process will also ensure sustainable reforestation and restoration of the forests as local communities better understand their own needs and environment.

With climate change posing a massive challenge to forest ecosystems, forest management will require continuous modifications and support of forest managers (Jandl et al. 2019). Not many species or forest types can adapt and grow at high elevation mountain ranges where environmental conditions are limiting. Besides, the upward shifting of forests along the elevation as a response to warming can pose considerable challenges to the forests growing at the tree line. The species with invasive nature (e.g., *Pinus wallichiana*) are known to expand their distribution to higher altitudes (Naudiyal and Schmerbeck 2017; Singh et al. 1984), while those with naturally less adaptive capacity (e.g., *Quercus semecarpifolia*) (Bisht and Kuniyal 2013), will shrink as a result of the expansion (Saran et al. 2010). The upward shift beyond the tree line will also be limited by poor edaphic and climatic conditions which combined with physiological morphological characters of some plants, for example, oaks with big acorn sizes, make them vulnerable to climate change (Bisht and Kuniyal 2013). Research elsewhere has indicated the need for adaptive forest management to save important forest types (Jandl et al. 2019).

The first step towards adaptive forest management is to supplement and assist forest regeneration artificially in those ecosystems where natural regeneration is poor. Sustainable forest management practices such as thinning and tending operations to maintain good forest health should be implemented thereafter. As grazing dominates the Himalayan forest ecosystem, grazing protection viz. use of tree shelters, in combination with artificial regeneration, would be beneficial (Chapter 4). Tree shelters protect the species of interest from grazing while allowing grazing to occur in a normal way, thus also promoting the local economy, unlike the plantation and fencing off grazing animals. Our study highlighted the benefits of tree shelters to significantly improve seedling survival and growth and recommend their use in restoration programs. We do not recommend complete exclusion of grazing animals from the forest as it leads to a strong dominance of unwanted vegetation on the forest floor.

Instead, more applied research on forest grazing tailored with restoration techniques would be meaningful. Specifically, research on natural means of deterring herbivory such as growing thorny bushes along with the species of choice can offer a cost-effective way of restoration. Future restoration works should focus on techniques that are practicable and cost-effective and include the goal of biodiversity conservation, carbon sequestration and livelihood sustenance.

The Himalayan forests are vital repositories of carbon and can play a pivotal role in mitigating global carbon emissions. As shown by our study, forests store substantial quantities of carbon ranging from 303-367 Mg ha^{-1} forming massive carbon sink (Chapter 5). All the structural components of the forest, e.g., tree biomass, belowground biomass, herbaceous layer, deadwood and soil form important carbon pools, and their management should be considered in the local forest management. Forest soils have high SOC density stored at varying soil depths and are vital ecosystem services regulating atmospheric carbon and mitigating climate change. Good soil conservation practices can improve carbon stocks, soil fertility and plant productivity (Doelkar and Wangmo 2018). Changes in forest management can affect ecological processes that can result either in carbon loss or enhanced storage (Mayer et al. 2020; Ontl and Schulte 2012; Achat et al. 2015). Therefore, short-term forestry decisions should not override the long-term impacts on soil carbon. Himalayan forest management is challenged by increasing population and poverty, livelihood necessity and socio-economic conditions that often lead to the weakening of social and regulatory functions of local forest management institutions (Ranjan 2018). This institutional degradation can lead to deforestation or localized exploitation of resources that can affect the ecological processes and SOC storage potentials of forests. Approaches to deal with these problems should lie in mass afforestation and plantation of barren lands surrounding the human settlements so that the pressure on the natural forests can be reduced. Community forestry initiative, which is gaining popularity, should be focused on the pursuit of enhancing carbon storage. The forest surrounding human settlements can be managed under sustainable management plans that will save the natural forests from degradation.

Many of the high-altitude forests have attained the old-growth status. Although many believe that young forests are more efficient in terms of carbon absorption and storage than the old forests, there is evidence that suggests old forests as equally capable of storing carbon as young forests (Carey et al. 2001; Luyssaert et al. 2008; Stephenson et al. 2014). Furthermore, the biodiversity conservation values of old forests should not be overlooked. Sustainable forest management that maintains forest productivity and biodiversity that is resilient to changing climate will be highly valuable under the future climate. Such a process will also improve the forests' capacity to absorb and store carbon from the atmosphere apart from meeting the economic, social, and ecological goals (Candell and Raupach 2008).

Finally, changing climate conditions in the Himalayas will severely impact species distribution and forest growth. Increasing temperature and changing precipitation patterns can affect both people and the environment. Trees growing at the tree lines will be particularly vulnerable to climatic changes as they are already growing in harsh and limiting edaphic conditions. A small change in climate can affect their growth and hence their survival and future distribution. However, the sensitivity of trees of the high-altitude forest to climate provides opportunities to study changing climate spanning several centuries. A better understanding of past climate enables us to investigate how to present ecosystems came to existence and how they will respond in the future. Although it is a formidable task to predict future climate and its effect on species distribution, there are few possibilities through climate proxies, e.g., tree-ring study, by which past climate can be determined and how this has shaped species distribution. As shown by our study (Chapter 6), the growth of a tree line conifer *Larix griffithii* is significantly controlled by the summer monsoon precipitation. This unique relationship enabled us to understand the past climate, much before the instrumental recording. We reconstructed the summer monsoon precipitation of Bhutan for the past 639 years, which is the most extensive reconstruction available for Bhutan so far.

Our study provided new insights into past climate patterns and how they have changed with time. These insights can help us to prepare and deal with changing climate in future, most notably the inter-annual variability of rainfall. More climate reconstructions spanning over both temporal and spatial scales using different tree species would be desirable to fill in the information gap. Future works from this study should focus on the use of reconstructed climate data to model and predict precipitation patterns in both present and future climate change scenarios and how these changes can affect the distribution of high-altitude forests. The future climate change at high altitude is predicted to be well above the adaptive capacity of the forests, which will have consequences resulting in local extinctions of vulnerable species, loss of essential ecosystem services and functions, including a reduction in carbon stocks and sequestration potentials (Keenan 2015; Seppälä 2009).

A holistic approach to sustainable forest management that incorporates active stakeholder participation, adjustments to forest management policies or practices based on knowledge and experiences that will suit a wide range of potential future climate scenarios will become increasingly desirable in future (Seppälä 2009). The information generated on climate and climate-growth models will be highly valuable in the field of forest management for effective adaptation to climate change.

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Appendices

Appendix 1. Major life-forms of trees recorded during the nationwide survey. The values represent the relative density of individual species within the plots based on their basal area.

Species	P1	P2	P3	P4	P5	P6	P7	P8	P9	P10	P11	P12	P13	P14	P15	P16	P17	P18	P19	P20	P21	P22	P23	P24	P25	P26	P27	P28	P29	P30	P31	P32	P33	P34	P35	P36	P37	P38	P39	P40	P41	P42	P43	P44	P45	P46	P47	P48	P49	P50	P51	P52	P53	P54	P55	P56	P57	P58	P59	P60	P61	P62	P63	P64	P65	P66	P67	P68
Evergreen Broadleaf																																																																				
Quercus semecarpifolia	76.9	90.8	89.9	83.2	76.6	94.2	52.3	90.2	56.7	87.1	45.9	87.1	95.9	94.3	84.1	95.1	70.4	96.1	100	78.8	86.7	97.6	51.7	71.4	100.0	94.1	100.0	100.0	73.1	96.0	88.2	80.7	58.8	85.0	82.9	94.3	96.3	88.4	100	87.9	81.3	95.2	97.7	95.3	87.0	95.3	54.7	64.1	75.7	99.8	89.6	88.6	87.0	46.6	51.7	78.9	93.9	98.7	99.9	89.1	92.0	95.2	71.4	97.8	79.9	83.7	44.0	92.3
Rhododendron arboreum	19.6	9.2	10.1		3.5	0.7	0.2		1.7	0.4	0.6	0.6	0.7	0.2									7.1	4.7		2.6				2.2	3.7	0.1								5.1	17.3		0.7		1.8	1.3	3.4	9.1	0.5			1.0	3.6	7.1	4.7		4.1	1.3		1.7		4.8	19.6	0.6	7.0	15.6	8.8	2.1
Ilex dipyrena					1.7	0.7	6.3	1.1	1.7					0.3										1.7					0.4		1.0	0.5		2.1	0.3	1.4				0.2	1.0	3.4	0.3	1.2				0.1	10.4			1.1	6.4	0.1									1.5				21.5	
Quercus oxyodon							2.1	0.8	17.1	2.1																							23.9	4.7																																		
Quercus glauca																							31.9	3.8														2.3													3.4	5.1																				
Quercus lamellosa						0.27																																																													
Persea clarkeana					3.8	0.4	28.6							0.8																																																					
Persea duthiei												0.6		1.4																																																					
Persea sp																														1.6	4.3	3.7		1.8		0.8																																	
Rhododendron kesangia																																	0.4																																		
Rhododendron keysii																																																											0.1									
Rhododendron trifolia																							0.3																																												
Symplocos dryophila						0.5		1.0	5.1	2.0																															0.4		0.3					0.2																		
Symplocos paniculata													1.7										0.5			0.6																																									
Symplocos ramosissima						0.1	0.2	1.9	4.8																							0.7	0.9																																	
Enonymus lucidus							1.1																																																													
Euonymus sp									0.2						1.1								0.5	0.3																																											
Euonymus tingens							3.3								0.9																					0.2																															
Eurya acuminata					2.2	1.8	0.3	1.2	2.6																						1.1		0.2	0.9									1.4						7.5											0.7								1.3
Benthamedia capitata											8.52																																																									
Myrsine semiserrata					0.5	1.0						0.5		0.7																																																					
Osmanthus suavis																																												1.1								0.2	2.9											1.2				
Pieris formusa																																																		0.2						11.3	0.1				8.0		0.9					
Viburnum cylindricum												1.1																																																							
Deciduous broadleaf																																																																				
Acer campbellii							1.5																							0.3					1.3					3.3												1.5																1.5
Acer hookeri														2.3																																																					
Acer sikkimense	2.1					0.2																																																													
Alnus nepalensis												7.59																																																							
Betula utilis	1.4																																																															0.5				
Carpinus vimnea																																																					0.1														
Corylus ferox																													0.2						0.3																																
Coriaria nepalensis											5.98																																																									
Cotoneaster sp																					1.8																																															
Docynia indica																		0.6																																																	
Enkainthus deflexus																																																				0.3															
Fraxinus paxina																																		4.2																																	
Gamblea ciliata																								3.1							1.8												0.0	0.3																							
Hydrangea heteromalla																																																																			14.5	
Lindera pulcheerima							4.0	1.7																																																												
Lindera sp					1.7																0.4																												0.0																		
Litsea sp.																																											0.7																								
Lyonia ovalifolia				2.7	5.3			2.1	8.0	6.9	27.3	2.5	1.7										6.9			2.2																																									
Magnolia campbellii									2.0	1.6																																																									
Malus baccata															1.3										0.5																																									
Rhus chinensis											2.3																																																									
Populus ciliata																																													0.1																						
Prunus cerasoides																	0.5	0.6					1.1						0.3					0.2	5.9																																	
Quercus griffithii					4.79						9.3																																				7.6																				
Sorbus capitata																																																															0.4					
Sorbus microphylla																																																															1.8					
viburnum nervosum															0.6	0.2	0.3	2.6			11.1	2.4																													0.3																	
Evergreen Conifers																																																																				
Tsuga dumosa															1.0									3.9					26.0			15.0	15.9		7.5		3.7						0.2	2.1	0.6	1.8	29.4	7.7	0.5											1.0			0.1		3.9		11.1	0.0
Taxus baccata																								10.9																3.0			0.1																		0.6			3.1					2.9
Pinus wallichiana															4.9	4.7	12.2																			0.8									8.4		3.7	6.6							46.2	43.6	9.8	1.9							0.0		5.8	0.6		
Cryptomeria japonica				8.0																																																															
Juniperus recurva																																						0.5																											3.4			
Picea spinulosa															6.0		16.6			21.2																									2.1	1.6	1.3	9.0			10.1	4.7								7.0			1.2					
Pinus bhutanica				6.1																																																															
Deciduous conifer																																																																				
Larix griffithii																																						11.1																													
Grand Total	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100

Supplementary files

Figure S1. Free grazing in the forest is a traditional right in Himalayas. Herd of livestock grazing in our study area.

Figure S2. Three types of shelters and control used in the restoration experiment of brown oak forests.

Figure S3. Fieldwork in progress, researchers marking the plot boundary prior to data collection.

Figure S4. Fieldwork in 2018, researchers measuring seedling collar diameter from a mesh tree shelter.

Figure S5. Fencing leads to strong dominance by bamboo, which can hamper seedling survival and escalate restoration costs involved in weeding. The picture compares fenced area to an adjacent grazed area where tree shelters were used.

TableS 1**.** Summary table of F-tests of the fixed effects of treatment, location and year on seedling height, seedling collar diameter and seedling sturdiness quotient. Tests are based on Type II sums of squares and error degrees of freedom are approximations using Satterthwaite's method (hence non-integer denominator degrees of freedom). P-values < 0.05 are bolded.

Fixed effect	Mean square	Degrees of freedom	F-ratio	P-value
Seedling Height				
Location	0.19	2, 2.3	3.29	0.209
Treatment	0.34	3, 6.9	5.79	**0.027**
Location*Treatment	0.04	6, 7.2	0.62	0.709
Year	22.5	3, 4.3	387.01	**<0.001**
Location*Year	0.08	6, 4.6	1.41	0.369
Treatment*Year	0.56	9, 365	9.59	**<0.001**
Location*Treatment*Year	0.14	18, 342	2.36	**0.002**
Seedling Collar diameter				

Location	0.36	2, 62	6.13	**0.004**
Treatment	0.25	3, 122	4.16	**0.007**
Location*Treatment	0.1	6, 125	1.72	**0.12**
Year	8.39	3, 6.8	141.77	**<0.001**
Location*Year	0.46	6, 7.1	7.82	**0.007**
Treatment*Year	0.24	9, 366	4	**<0.001**
Location*Treatment*Year	0.04	18, 356	0.62	0.886
Seedling sturdiness quotient				
Location	0.17	2, 10.3	2.05	0.179
Treatment	0.97	3, 7.7	11.36	**0.003**
Location*Treatment	0.14	6,8.2	1.62	0.255
Year	5.34	3, 8.8	62.58	**<0.001**
Location*Year	0.43	6, 9.1	5.03	**0.015**
Treatment*Year	0.65	9, 385	7.63	**<0.001**
Location*Treatment*Year	0.15	18, 380	1.75	**0.03**

TableS 2**.** Comparison of environmental parameters inside the protex tubes with outside natural conditions using one-way ANOVA. ns = not significant, * = significant at p<.05.

Variables	**Protex**	**Outside**
Temperature (°C)		
Average	8.4 1 (1.51) ns	8.50 (1.45)
Summer (Max temperature)	27.27 (0.83)*	24.78 (0.48)
Winter (Min temperature)	-6.47 (0.24) ns	-5.99 (0.15)
RH (%)		
Average	89.08 (1.76) ns	86.20 (2.56)
Summer	93.01 (0.58)*	95.88 (0.94)
Winter	83.80 (1.48)*	75.62 (1.95)
Soil measurements		
Soil Moisture content (%)	15.56 (1.07)*	12.04 (0.80)
Soil Hardness (mm)	7.09 (0.59)*	9.66 (0.54)

TableS 3**.** The results from the mixed effects models comparing the fixed effects of forest type, grazing and their interactions.

Fixed effects	Estimate	Standard Error	t value	P- value
Litter thickness (cm)				
Forest	2.6667	1.4778	1.804	**<0.01****
Grazing	2.4722	1.4778	1.673	**<0.05***
Forest : Grazing	0.3611	2.09	0.173	0.86
Organic matter thickness (cm)				
Forest	9.0722	2.2527	4.027	**<0.001*****
Grazing	2.4333	2.2527	1.08	0.08
Forest : Grazing	0.5667	3.1858	0.178	0.85
Soil hardness				
Forest	-9.556	1.492	-6.406	**<0.001*****
Grazing	-9.639	1.492	-6.462	**<0.001*****
Forest : Grazing	8.806	2.109	4.174	**<0.001*****
Soil Moisture content (%)				
Forest	6.122	3.165	1.934	**<0.05***
Grazing	-1.028	3.165	-0.325	0.43
Forest : Grazing	-1.511	4.476	-0.338	0.74

TableS 4. Summary table of F-tests of the fixed effects of forest type, grazing, and soil depth on soil bulk density, soil carbon % and soil organic carbon (Mg/ha). Tests are based on Type II sums of squares and error degrees of freedom are approximations using Satterthwaite's method (hence non-integer denominator degrees of freedom). P-values < 0.05 are bolded.

Fixed effect	**Mean square**	**Degrees of freedom**	**F-ratio**	**P-value**
Bulk Density (g/cm^3)				
Forest type	0.01	1, 22	0.43	0.521
Grazing	0.02	1, 22	1.54	0.228
Soil depth	0.5	3, 66	31.53	**<0.001*** **
Forest type*Grazing	0.02	1,22	1.07	0.311
Forest type*Soil depth	0.01	3, 66	0.5	0.687
Grazing*Soil depth	0.01	3, 66	0.82	0.488
Forest type*Grazing*Soil depth	0.01	3, 66	0.39	0.758
SOC content (%)				
Forest type	38.04	1,22	25.98	**<0.001*** **
Grazing	3.07	1.22	2.09	0.162
Soil depth	149.58	3, 66	102.16	**<0.001*** **
Forest type*Grazing	5.62	1, 22	3.84	0.063
Forest type*Soil depth	18.28	3, 66	12.48	**<0.001*** **
Grazing*Soil depth	3.97	3, 66	2.71	0.052
Forest type*Grazing*Soil depth	9.93	3, 66	6.78	**<0.001*** **
SOC density (Mg ha^{-1})				
Forest type	605.04	1, 22	5.39	**0.030***
Grazing	3.17	1, 22	0.03	0.868
Soil depth	912.72	3, 66	8.13	**<0.001*** **
Forest type*Grazing	11.38	1, 22	0.10	0.753
Forest type*Soil depth	28.34	3, 66	0.25	0.859
Grazing*Soil depth	58.32	3, 66	0.52	0.670
Forest type*Grazing*Soil depth	70.25	3, 66	0.63	0.600