Managing shifting agriculture in Northeast India to protect carbon and biodiversity

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Joli Rumi Borah

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DEDICATION

Dedicated to my koka (grandfathers):

Padmakanta Gogoi

&

Brojendra Nath Borah

For being the nurturing roots of my life

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Statement of Contribution

Chapter 2 in this thesis is a literature review on shifting cultivation in Northeast India. This has been submitted as an invited book chapter in Farmer Innovations and Best Practices by Shifting Cultivators in Asia-Pacific, Editor- Malcolm Cairns (Borah JR, Evans KL, Edwards DP. 2018. Jhum farmers' Innovations in Northeast India). I conceived the main idea and wrote the chapter. David P. Edwards, Karl L. Evans and many others provided useful comments on the draft. Prasujya Gogoi produced the map.

Chapter 3 assesses potential of REDD+ mechanism in protecting and enhancing carbon stocks in shifting cultivation. This was published in Ecological Applications (Borah JR, Evans KL, Edwards DP. 2018. Quantifying carbon stocks in shifting cultivation landscapes under divergent management scenarios relevant to REDD+. Ecological Applications. 28 (6): 1581-1593). I collected the data, analysed and wrote the manuscript. David P. Edwards and Karl L. Evans helped in conceiving the main idea, designing the study and assisted with the analysis and writing. Felix K. Lim and Matthew Hethcoat gave useful inputs on the analysis.

Chapter 4 examines bird diversity recovery following shifting cultivation. I collected the data, analysed and wrote the manuscript. David P. Edwards and Karl L. Evans helped in conceiving the main idea, designing the study and revised the manuscript. James Gilroy helped critically with analysis. David Farrow, Paul Elsen, Craig Robson, Umesh Srinivasan and Werzik helped in identifying unknown bird calls.

Chapter 5 deals with the recovery of avian phylogenetic and functional diversity following shifting cultivation. I collected the data, analysed and wrote the paper. Gavin Thomas provided the bird functional trait data. James Gilroy provided valuable inputs on analysis. David P. Edwards and Karl L. Evans helped in conceiving the main idea, designing the study and revised the manuscript.

David P. Edwards guided all the work herein and read this entire thesis. Karl L. Evans co-supervised and helped with designing the study, analysis and revised the initial drafts. This PhD was supported by ACCE (Adapting to the Challenges of a Changing Environment) Doctoral Training Partnership.

Thesis abstract

Shifting cultivation, a traditional farming method practiced pantropically, involves clearing a forest patch, cropping and subsequent abandonment on a rotational basis. With the declining rotation period, shifting cultivation is expanding into old-growth forests, thus becoming a major driver of carbon emissions through deforestation and forest degradation, and of biodiversity loss. The impacts of shifting cultivation on carbon stocks and biodiversity have rarely been quantified, and the potential for carbon-based payments for ecosystem services (PES), such as REDD+, to protect carbon and biodiversity in shifting cultivation landscapes is unknown. This thesis addressed these critical knowledge gaps and assessed carbon and biodiversity recovery following shifting cultivation and the potential of REDD+ mechanism in protecting and enhancing carbons stocks in shifting cultivation landscapes. I sampled carbon stocks and bird communities across farmland, regenerating secondary forest and old-growth forest in a shifting cultivation landscape in Nagaland, Northeast India. I then used these data to predict carbon stocks, species richness, phylogenetic and functional diversity at a landscape level under divergent management scenarios of REDD+. I showed that carbon stocks recovered substantially across regenerating secondary forest, with a 30-year fallow storing about half the carbon of an old-growth forest. However, bird species occurrence was higher in farmland and regenerating secondary forests than old-growth forest. I also showed that functional diversity in bird community was robust to shifting cultivation impacts, whereas loss in phylogenetic diversity from forest conversion to shifting cultivation recovered rapidly with increasing fallow age. Old-growth forests sustained a phylogenetically and functionally clustered bird community. In contrast, farmland maintained an over-dispersed bird community retaining diverse evolutionary lineages and functional groups. Scenario simulations suggested that sparing old-growth forests from deforestation and intensifying cropping in the remaining area of shifting cultivation is the most optimal strategy for carbon storage. In contrast, REDD+ interventions that maintain a mosaic landscape with farmland, regenerating forest and old-growth forest were optimal in preserving high levels of species abundance and phylogenetic and functional diversity. These findings highlight the conservation value of shifting cultivation and potential of conservation interventions to achieve both carbon and biodiversity co-benefits with carbon payments.

Contents

Chapter 1

Thesis Introduction

Tropical forests are important carbon sinks storing 55% of the world’s forest carbon (Pan et al. 2011), and thus contributing significantly to climate change mitigation. These are also the most biodiverse ecosystems on Earth (Pimm and Raven 2000), harbouring two-thirds of the global terrestrial biodiversity (Gardner et al. 2009). At the same time, tropical forests are threatened by high levels of deforestation and forest degradation due to land-use change (Fearnside 2000) . Tropical deforestation accounts for 6-17% of net global greenhouse gas emissions (Werf et al. 2009), which represents the second largest source of greenhouse gases after the burning of fossil fuels. In turn, biodiversity is suffering major losses to habitat loss, hunting, wildlife trade and climate change with tens of thousands of species at imminent risk of extinction (Tilman et al. 2017).

Agricultural expansion and intensification is one of the main drivers of large scale deforestation in the tropics (Gibbs et al. 2010), causing significant green-house gas emission and biodiversity loss (Houghton 2012). With growing human population and economic development, the potential trade-offs between producing enough food and sustaining biodiversity and ecosystem services have become a major conservation concern. Apart from settled agriculture, shifting cultivation, practised widely across (sub-) tropics, contributes significantly to deforestation, carbon emission and biodiversity loss in the region (Houghton 2012). At the same time, it also plays an important role in ensuring food security by providing subsistence to 200-300 million people across the world (Mertz 2009).

Shifting cultivation, also known as swidden or slash-and-burn cultivation, is a traditional farming method that has been practised over both tropical and temperate regions (Mediterranean, Northern Europe and parts of North America) until the 19th century. Currently, it is only prevalent in the (sub-)tropical regions of Africa, Asia, Latin America and parts of Oceania (Thrupp et al. 1997a), covering an area of 2.6 million km^2 across 64 countries (Silva et al. 2011, Li et al. 2014). This rotational agriculture typically involves cycles of burning and clearing forests (primary or secondary), cropping for a short time and subsequent abandonment of the site for vegetation regeneration (Ramakrishnan 2006). In the absence of any external inputs, the productivity of shifting cultivation mainly depends on the intrinsic soil fertility and nutrients released from the burnt biomass.

1.1 Impacts of shifting cultivation on carbon emission and biodiversity

In the last few decades, population growth, increasing demands for food production and scarcity of available land have driven significant changes in the traditional processes of shifting cultivation (Li et al. 2014). Particularly, cultivation cycles (cropping and abandonment) have become increasingly frequent, thus shortening the fallow period i.e. the time for vegetation recovery following cultivation from 30-40 years to 4-5 years in many regions (Klanderud et al. 2009). Incomplete fallow and soil recovery in shorter fallow cycles leads to decrease in crop yields in shifting cultivation. To compensate for this, farmers clear more forests expanding cultivation to a larger area to meet the growing demand for food, thus resulting in deforestation and forest degradation.

The recent trends of reducing fallow period and expansion to primary forest in shifting cultivation contribute considerably to carbon emission and biodiversity loss in the tropics. For instance, expansion of shifting cultivation to primary forest released 241, 205 and 295 Tg/years CO_2 in Asia, Africa and America respectively (Silva et al. 2011). Similarly, frequent cultivation cycles in shifting cultivation lead to forest degradation as it does not allow sufficient time for a complete recovery of the vegetation. The overall carbon pool in shifting cultivation increases with fallow age as secondary forest regrows on abandoned farmland (Kotto-Same et al. 1997a, Tschakert et al. 2007, Bruun et al. 2009). Although the estimates vary across sites, previous studies suggest that it takes at least 20-35 years for a site to accumulate the similar levels of biomass compared to an old-growth forest (Mukul and Herbohn 2016). The most vulnerable carbon pool to shifting cultivation is the above ground biomass whereas soil organic carbon remains mostly stable (Kotto-Same et al. 1997a) or rapidly recovers within the early successional stages (Bruun et al. 2006).

Shifting cultivation modifies the landscape into a mosaic of farmland, regenerating secondary forests and primary forest, thus impacting biodiversity. Species richness for most taxa is typically lower in fallow forests than primary forests (Scales and Marsden 2008). Species richness increases over time as fallow forests regenerate for plants (Toky and Ramakrishnan 1981a, Lawrence et al. 2005), amphibians (Pawar 1999), and reptiles and butterflies (Dunn 2004) and is highest in primary forest for most of these taxa (Bowman et al. 1990). However, birds do not show a clear response to fallow period. For instance, bird species richness and abundance were higher in primary forest than regenerating fallows in Asia-pacific region (Bowman et al. 1990, Blankespoor 1991a, Raman 2001a, Zhijun and Young 2003); similar in both habitats in the Colombian Amazon (Andrade and Rubio-Torgler

1994) and higher in fallow areas than primary forest in Costa Rica (Blake and Loiselle 2016) and Indonesia (Jones et al. 2003). The reasons for the lack of patterns may be the change in diverse vegetation (Scales and Marsden 2008), sampling scale and design or other habitat factors, such as distance to nearest forest and landscape configuration which varied across studies (Scales and Marsden 2008).

The time required for a complete recovery of species richness and community composition following shifting cultivation vary across sites. For instance, a fallow period of 20 to 40 years was long enough to accumulate similar levels of bird species richness with adjoining mature forest in Northeast India (Raman et al. 1998), 23 years in Bolivia (Kennard et al. 2002), 190 years in Colombia and Venezuela (Saldarriaga et al. 1988) and 25 years in China (Zhijun and Young 2003). Species composition of various taxa however can take 36- 90 years to reach the level of an old-growth forest (Karthik and Veeraswami 2009, Acevedo-Charry 2016). Among various foraging guilds, forest specialist birds such as large frugivores, canopy insectivores, branch gleaners and bark feeders are most affected by shifting cultivation (Thiollay 1995, Raman et al. 1998) whereas small frugivores, foliage insectivores and nectarivores thrive in regenerating forests (Thiollay 1995, Marsden et al. 2006). However, the trajectories of biomass and biodiversity recovery following shifting cultivation vary across sites and still poorly understood at a landscape level. Similarly, recovery of evolutionary lineages and associated ecological functions of communities in shifting cultivation landscapes are not well studied.

1.2 Managing shifting cultivation to improve conservation outcomes

With the marked and increasing contribution of shifting cultivation to forest transformation, it is vital to formulate strategies that simultaneously reduce emission and biodiversity loss and improve crop production in these landscapes. Effective management of shifting cultivation can involve a range of mechanisms such as sparing primary forests as protected areas, community forest reserves co-managed with local communities and improving efficiency in crop production. Large areas of primary forests are maintained as community forest reserves by local communities in shifting cultivation landscapes. Ensuring protection of these forests from clearing for shifting cultivation by integrating into protected area network or co-managing with local communities can thus play an important role in conserving biodiversity and ecosystem functions. Although regenerating secondary forests have typically lower biodiversity value than mature forests, secondary forests recover biodiversity and contribute significantly in providing ecosystem services as they regenerate (Sayer et al. 2017a). Regenerating secondary forests in shifting cultivation are, therefore, valuable in reducing emission and biodiversity loss and worth restoring. Moreover, integrating secondary forests in conservation planning can also protect these forests from further degradation by intensive logging or conversion to cash crop plantation.

Sparing primary or regenerating secondary forest for climate change mitigation and biodiversity conservation in shifting cultivation requires increasing crop yield in limited land. Increasing production per area by sustainable agricultural intensification can reduce the need for clearing additional forests for shifting cultivation (DeFries and Rosenzweig 2010). This can be achieved through nutrient supplementation (Tawnenga and Tripathi 1997), optimization of crop choice (Toky and Ramakrishnan 1981a), and improved fallow management (Grogan et al. 2012) in shifting cultivation. While intensifying shifting cultivation, conservation policies need to consider the potential impacts of such interventions on non-provisioning ecosystem services such as biodiversity, soil formation and water regulation apart from food production to ensure positive outcomes for both ecosystem services and wellbeing of the shifting cultivators (Rasmussen et al. 2018). Evaluating the trade-offs in a wider landscape context and identifying an appropriate balance between these two approaches is a major challenge while developing effective conservation policies in shifting cultivation landscapes.

1.3 Potential role of REDD+ in shifting cultivation

Conservation policies addressing biodiversity conservation and carbon emission simultaneously while ensuring local livelihoods such as REDD+ (Reducing emission from Deforestation and forest degradation) presents an opportunity for effectively managing shifting cultivation landscapes to reduce emission and biodiversity loss. REDD+ is a mechanism, initiated by UNFCC (United Nations Framework Convention on Climate Change), that provides financial incentives to developing countries to reduce carbon emission. As formulated in 2007 at the thirteenth session of the Conference of the Parties (COP-13), REDD+ focusses on five main interventions; reducing emissions from deforestation, reducing emissions from forest degradation, conservation of forest carbon, sustainable management of forests and enhancement of forest carbon (Gardner et al. 2012a) . These interventions can also potentially provide co-benefits for biodiversity by protecting the forest habitats for reducing carbon emission. Thus, REDD+ has a strong potential to protect primary forest from conversion into shifting cultivation, avoid forest degradation by maintaining a longer fallow period or to enhance forest carbon by protecting the older fallows from further clearing. As shifting cultivation is a dynamic and complex system, it is still not clear which of these activities can be targeted while implementing REDD+ in shifting cultivation.

REDD+ actions for shifting cultivation are debated on two sets of arguments. First, following the criticisms of shifting cultivation as a major driver of tropical deforestation and degradation, this suggests eliminating this system by providing incentives for transformation into settled agriculture or other types of intensive land use (Angelsen et al. 2008). This perception generated a drive for a transition away from shifting cultivation to settled agriculture across the world (Mertz 2009, van Vliet et al. 2012, Jakovac et al. 2016). For instance, government policies promoting cash crop plantation, marginalization of shifting cultivators and market development led to rapid conversion of shifting cultivation to permanent agriculture such as rubber, oil palm, fruit tree, pepper and tea in Southeast Asia, Central America and East Africa (Schmidt-Vogt et al. 2009, van Vliet et al. 2012). However, recent evidence suggests that intensive cropping systems and cash crop cultivation have many long-term negative impacts on forests, biodiversity, ecosystem services, and local livelihoods (Rasmussen et al. 2018).

The second argument for REDD+ implementation in shifting cultivation perceives shifting cultivation as a traditional suitable land use choice by farmers, harbouring rich biodiversity (Rerkasem et al. 2009). For instance, long fallow shifting cultivation landscapes are well adapted to the environmental (e.g. low fertility of soil), economic (e.g. limited access to market) and cultural conditions in tropical mountainous regions and can be sustainable at low population densities (van Vliet et al. 2012, Filho et al. 2015). This suggests that REDD+ should try making it sustainable by incorporating measures to improve yield for avoiding shifting cultivation expansion into primary forest and by rewarding for longer fallow period where farmland is available. The second argument has been widely accepted by scientific community, but it has not been incorporated into policy and decision making yet (Noordwijk et al. 2008). Thus, there is still uncertainty about the specific actions of REDD+ mechanism for shifting cultivation and its consequences to local livelihoods (Hett et al. 2012).

Although the nature and general impact of shifting cultivation have been widely studied, the understanding of changing shifting cultivation regimes and its impact on carbon stock and biodiversity, and the potential of REDD+ in providing carbon and biodiversity co-benefits is still limited. For effective planning, it is required to assess how REDD+ activities address the dynamics of shifting cultivation landscape. Whether it qualifies for deforestation or forest degradation is still debated (Mertz 2009). Similarly, considering forest degradation in terms of carbon stock may not have the desired effects of reducing carbon emissions in shifting cultivation, unless it is a very intensive system with short fallow periods and few large trees, which store most of the carbon (Leisz et al. 2007). Preventing more frequent clearing of primary or secondary forest for intensified agriculture may be beneficial if the REDD+ mechanism rewards for maintaining long fallow systems (Mertz 2009). Despite the limited number of studies assessing the potential of REDD+ in shifting cultivation, it seems to be feasible in providing economic gains for maintaining forests rather than clearing new land for cultivation (Bellassen and Gitz 2008).

Thesis overview

The overarching goal of this PhD work was to assess the impacts of shifting cultivation on carbon stocks and biodiversity and examine the effectiveness of divergent management strategies under REDD+ in protecting both in such landscapes. I assessed how carbon stocks and bird diversity (taxonomic, phylogenetic and functional diversity) recovered following shifting cultivation. I then used these data to simulate management scenarios under REDD+ to evaluate which intervention pathways are most effective in protecting maximum level of landscape carbon stocks and bird diversity. I present six chapters addressing these objectives based on existing literature and field data collected over two field seasons during 2015-2016 in three districts of Nagaland, Northeast India.

Northeast India is one of the most biodiverse regions in India and is a part of the Himalayan and Indo-Burma global biodiversity hotspots (Mittermeier et al. 2004). Shifting cultivation is a dominant cultivation system in the mountainous region of Northeast India, covering an area of 8771.62 km^2 (Rathore et al. 2012) and providing subsistence for some 620,000 families across the region (Ramakrishnan 1992). The proportionate area under Shifting cultivation is the highest in Nagaland covering 38.2% of the total geographical area and 71.2% of the agricultural area in the state (Pareta 2013). Nagaland thus provides a strong potential for climate change mitigation and likely co-benefits for biodiversity with a high emission mitigation potential under REDD+ (Murthy et al. 2013b).

Shifting cultivation is a dynamic and complex cultivation system practised by over 3000 ethnic groups in diverse environmental conditions across the world (Thrupp et al. 1997b). Thus, the farming practices, such as clearing, and cropping patterns and fallow management systems vary across tribes and sites. Shifting cultivation system in Northeast India, which is home to about 209 diverse ethnic tribes (Ghosh 1997), also differs across tribes in terms of land tenure system and crop and fallow management practices. Following this introductory chapter, in Chapter 2, I attempt to fill the need of a comprehensive review of the shifting cultivation process in Northeast India in the wider literature. I do so by giving a background of the shifting cultivation system in Northeast India and the innovative techniques that farmers adopt to address the challenge in declining crop yield.

As fallows regenerate in shifting cultivation, they sequester carbon stocks in recovering biomass. Carbon-based payment schemes such as REDD+ implementation requires a better understanding of what pathways will maximize carbon storage in these fallow forests relative to an old-growth forest. In Chapter 3, I estimate above-ground carbon stocks in various carbon pools such as trees, lianas, dead wood and leaf litter in shifting cultivation across farmland, regenerating fallows and old-growth forest. I use this data to model landscape-level carbon stocks under business-as-usual scenarios (expansion into the old-growth forest or decreasing fallow periods) and intervention scenarios that applies intervention through REDD+ to reduce deforestation of primary or secondary forest to identify the optimal management strategies for maximum carbon storage.

Successful implementation of REDD+ can also conserve biodiversity in shifting cultivation landscapes by protecting its habitats. As fallows with regenerating forest occupy a major portion of the shifting cultivation landscape (Finegan and Nasi 2004), their potential in biodiversity conservation is of fundamental importance. But whether the co-benefits for biodiversity are delivered will depend on how biodiversity recovers following shifting cultivation. Chapter 4 addresses this question by investigating bird species richness and community composition across farmland, regenerating secondary forests and old-growth forests. I use hierarchical occupancy modelling in a Bayesian framework to estimate species occurrence and simulate hypothetical landscapes under the divergent management scenarios of REDD+ to examine biodiversity recovery at a landscape level.

Taxonomic diversity measures such as species richness and composition may not represent the true extent of impacts of shifting cultivation on biodiversity as they do not consider influence of species’ life-history traits and evolutionary history on community response to land-use change (Edwards et al. 2014b). Phylogenetic and functional diversity measures are better predictors for assessing species response to land-use change as they incorporate the evolutionary history and ecological functions in a community (Chapman et al. 2018). In Chapter 5, I assess recovery of avian phylogenetic (PD) and functional diversify (FD) following shifting cultivation under the management scenarios of REDD+ to examine the effectiveness of each REDD+ strategy to protect PD and FD in bird communities.

Finally, in the final Chapter 6, I summarise the key findings of this research and highlight the conservation implications of the study. I also address the possible pathways for successful implementation of REDD+ in shifting cultivation landscapes.

Chapter 2

Shifting cultivation in Northeast India

Abstract

Northeast India, comprising eight Indian states, is unique in its geographical location, climate, and biodiversity. Shifting cultivation is predominant in the mountainous regions of Northeast India. It involves clearing a forest patch by slash and burn method, cropping for one or two years and subsequent abandonment on a rotational basis. However, increasing frequency of clearing and cropping due to land scarcity and population growth has resulted in incomplete soil and vegetation recovery and declining crop yield. Farmers in Northeast India have adopted various innovative ways to address the increasing pressure for food production and improve crop yield. This extensive literature review suggests that farmers have innovated techniques in various stages of the cultivation process, such as clearing, cropping and fallow management in Northeast India. They prevent soil erosion by using physical and mechanical barriers such as wooden logs and cover crops, optimise resource use by mixed cropping and sequential harvesting, manage fallows by retaining plants that aid in faster fallow and soil recovery and in some instances avoid burning by adopting slash and mulch system. These innovations imply that farmers are adapting to the changes and if applied widely, their effective innovations can contribute significantly in improving crop yield and avoiding forest loss in shifting cultivation landscapes. Therefore, incorporating farmer’s innovations in policies and wider application of effective innovations can pave the way for sustainable shifting cultivation in Northeast India.

1 Introduction

Northeast India, comprising eight states (Arunachal Pradesh, Assam, Manipur, Meghalaya, Mizoram, Nagaland, Sikkim and Tripura), which cover an area of 262,000 km^2 (Murthy et al. 2013a), is unique in terms of its geography, biodiversity and diverse ethnic communities (Deka and Sarmah 2010, Singh et al. 2010). The region shares its borders with China, Nepal, Bhutan, Myanmar and Bangladesh and is connected to the rest of India through a narrow 21 km wide land corridor (Maaker and Joshi 2007) (Fig 1). The region includes the Eastern Himalayas in the north (up to 7060 m elevation in Arunachal Pradesh), the Northeastern hills in the south (up to 3840 m elevation in Nagaland) and the Brahmaputra river valley in between (Mani 1974). Most of Northeast India receives high rainfall with Mawsynram in Meghalaya being the wettest place on Earth, which receives up to 11,871 mm average annual rainfall. Due to its unique location at the confluence of two biogeographical realms (Palearctic and Indo-Malay; Olson et al. 2001), wide elevational range and vegetation diversity (from tropical to alpine), Northeast India harbours high biological diversity and is a part of the Himalayan and Indo-Burma global biodiversity hotspots (Mittermeier et al. 2004). This region is also culturally diverse with about 209 different ethnic tribes (Ghosh 1997), who have been traditionally dependent on forest resources to varying degree for their subsistence or livelihood (Choudhury and Sundriyal 2003, Bhatt and Sachan 2004).

With the majority of its population living in rural areas, agriculture is the main source of livelihood in Northeast India (Johnson and Hutton 2014). While the people in the Brahmaputra valley practise settled agriculture, people in the mountainous regions of Eastern Himalayas and Northeastern Hills predominantly practise shifting cultivation. Shifting cultivation in Northeast India, locally referred to as ‘jhum’, dates back fifteen thousand years. Evidence suggests that the Garo tribe in Meghalaya, one of the earliest inhabitants in the region, practised jhum cultivation during the Neolithic period (1300-3000 BC) (Goswami 1972, Sharma 1976). They grew yam and other root crops in jhum fields initially and later started cultivating rice and millet (Roy 1981). Jhum cultivation continues to be a dominant cultivation system in Northeast India, covering an area of 8771.62 km^2 (85% of total cultivation area in Northeast India; Rathore et al. 2012) and providing subsistence for some 620,000 families across the region (Ramakrishnan 1992) (Appendix I, Text S2). Among the Northeastern states, proportionate area under shifting cultivation is the highest in Nagaland and lowest in Arunachal Pradesh (38.2% and 2.5% of total geographical area respectively; Tiwari 2007). The jhum process in Northeast India is similar to the classic system as described by Conklin (1961) for Asia. Farmers clear a forest patch by cutting and burning the vegetation during the dry months in winter. They grow a variety of crops such as rice, maize and different vegetables in the cleared site for one to two years, and then temporarily abandon the site for forest recovery. Farmers rotate from one site to another making use of the temporary increase in soil fertility from ash produced by burnt vegetation (Mishra and Ramakrishnan 1983a).

Although the fundamental cultivation process is similar, jhum system differs across tribes in Northeast India in terms of land tenure system and crop and fallow management practices. The mountainous region in Northeast India has a greater degree of autonomy than the rest of India under the nation's constitution. Village councils govern land ownership through tribe specific customary laws in this region (Devi 2005). Three broad patterns of land ownership exist in jhum areas of Northeast India (Thangam 1984); land can be owned collectively by the village community (e.g., Angami tribe in Nagaland, Devi 2005; Nishi tribe in Arunachal Pradesh, Mitra 1998) or by individual families (e.g. Tangkhul tribe in Manipur, Shah 2003; Mikir tribe in Arunachal Pradesh, Burmon 1977) or by the chiefs or village council who distribute it among households for cultivation through a lottery system (e.g., Konyak tribe in Nagaland, Krug et al. 2013). The last pattern of ownership is the most common across villages and tribes (Krug 2009a, Choudhury 2015). Across villages, farm plot sizes vary depending on the land tenure system (Krug et al. 2013). Crops grown, cropping patterns, length of fallow cycles and fallow management practices also vary across tribes (Kushwaha and Ramakrishnan 1987).

Over the last few decades, traditional processes of jhum have undergone significant changes due to population growth, increased demand for food production and scarcity of available land (van Vliet et al. 2012, Li et al. 2014). For instance, the population of Nagaland grew at the rate of 56.08% during 1981–91 and of 64.53% during 1991–2001(Agarwal and Kumar 2012). As a result, fallow periods have decreased from 30-40 years to just 4-5 years in many parts of the region to meet the growing demand for food (Maikhuri and Gangwar 1993, Klanderud et al. 2009). Soil fertility decreases with shortened fallow periods due to plant nutrient uptake, leaching and soil erosion from high rainfall. This may lead to two, not necessarily exclusive, consequences: (i) decreasing crop yields; and (ii) farmers clearing additional forest to expand jhum to meet the growing demand for food. Although historical data on jhum yield is rare, a few evidences suggest that crop yield declined significantly in many parts of Northeast India (Bela 2003). Although changes in jhum cultivation vary across villages, farmers in many villages have expanded jhum area into community forest reserves in Nagaland (personal observation). Thus, the declining yield and environmental impacts from changing trends in jhum cultivation generated a drive from policy-makers for a transition away from jhum to more intensive agricultural systems (Behera et al. 2016).

The majority of the development programmes initiated by the governments in Northeast India viewed shifting cultivation as a primitive, inefficient (due to lower yields) and environmentally unsustainable practice (due to deforestation, forest degradation and soil erosion) and therefore provided incentives for settled agriculture and perennial cash crops (Tiwari 2007). For instance, legislations such as the Jhum Land Regulation act (1948) and National Forest Policy (1952) aimed at rehabilitating jhum land and promoting terrace cultivation, animal husbandry, horticulture, permanent agriculture and cash crop cultivation (coffee, tea, black pepper, teak and rubber) (Ninan 1992). Governments also initiated various schemes such as ‘Control of Shifting Cultivation (1976-1977)’ and ‘Purchase of Land for Rehabilitation of Jhumias and Land-less Tribal (1985-86)’ to resettle jhum farmers in arable land across the Northeastern states (Gupta 2000). However, these policies and legislations have mostly been unsuccessful in replacing shifting cultivation with settled agriculture in this region (Kerkhoff and Sharma 2006). The main reasons behind this failure were the lack of understanding of the complexity of the jhum system and the important role of this system in the socio-economic life of tribal societies (Tripathi and Barik 2003). Topographical constraints (steep terrain and high elevation) and high rainfall, limited market access and the labour and technology intensive nature of settled agriculture also made it challenging to shift to settled agriculture (Choudhury et al. 2001, Choudhury and Sundriyal 2003).

Declining crop yield due to frequent fallow cycles and government incentives encouraging conversion of shifting cultivation to permanent agriculture across the world led to predictions in the 1960s of a likely collapse of the shifting cultivation system (Padoch et al. 2007a). Despite these, jhum cultivation still continues in many remote regions across Northeast India, ensuring food security to diverse ethnic groups. Farmers have, instead, responded to the increasing pressures on jhum cultivation by developing innovative techniques for rapid recovery of soil and vegetation and for increasing crop yields (Kerkhoff and Sharma 2006, Padoch et al. 2007a). Farmers' innovations can be any change in technique within the jhum system, often based on indigenous traditional knowledge, that increases crop yield and is sustainable in the long term. These innovations have evolved over time mainly to address the increasing environmental and economic pressures on jhum cultivation, such as declining crop yield. For instance, farmers in Northeast India use mechanical and biological barriers in cleared sites to prevent soil nutrient loss, grow a variety of crops simultaneously optimising resource use and manage fallows actively facilitating faster recovery (Rathore et al. 2010, Deb et al. 2013). These innovations by jhum farmers can therefore play an important role in sustainable management of shifting cultivation in Northeast India. Integrating effective innovations by farmers along with scientific evidence in management strategies will facilitate successful policy implementation to reduce deforestation from shifting cultivation without affecting crop production (He et al. 2009).

Fig 1. Location of the eight states in Northeast India; Assam, Arunachal Pradesh, Manipur, Meghalaya, Nagaland, Tripura and Sikkim

2. Jhum cultivation process in Northeast India

Each stage in the Jhum cultivation process, such as site selection, clearing and burning, cropping and fallow management is crucial in determining the annual crop yield. Jhum farmers carry out the cultivation process with careful consideration of local site-specific requirements and adopt various ways to ensure maximum yield.

2.1 Site Selection

Selecting suitable sites is critical in jhum cultivation as various physical and biological features such as soil quality, slope and fallow vegetation determine the annual yield for a site. Site selection generally starts during the winter months of October-November in Northeast India. Farmers choose sites based on their perception of various indicators of soil quality and potential agricultural productivity derived from their knowledge of the system (Saikia 2010). For example, farmers prefer sites with black soil which is highly permeable and rich in organic matter (Marten and Vityakon 1986, Tylor-Powell et al. 1991). Soil with earthworms is also preferred as they increase soil fertility by helping humus formation (Bhadauria and Ramakrishnan 1989, Singh 2009). Farmers generally choose fallow sites that contain certain plant species such as *Alnus nepalensis*, *Artemisia* sp, *Albizia* sp and *Macaranga denticulate* (Singh 2009, Saikia 2010). These species are early colonising, fast growing and enrich soil by fixing atmospheric nitrogen (Ramakrishnan 1993, Cairns 2007). Similarly, farmers in Arunachal and Manipur usually avoid sites that have *Quercus spp*. and *Pinus spp*. (Singh 2009) as these species are rich in secondary metabolites such as tannins and phenols that slow down leaf breakdown increasing soil acidity (Singh 2009). Areas with high abundances of ferns tend to be indicative of poor soil nutrients and therefore usually avoided by jhum farmers in Nagaland (Saikia 2010). Crop choice also influences selection of a site. For instance, Tangkhul farmers in Manipur grow groundnuts in loose soil but plant legumes (ricebean and soyabean) in soil with stones or gravels as the shallow rooted legumes can improve soil fertility by fixing nitrogen (Singh 2009).

2.2. Clearing and burning

Jhum farmers start clearing land during November-December (Maikhuri and Gangwar 1993). They usually prefer clearing secondary forests to primary forests as it involves less labour, easier burning of wood logs and lower concentrations of allelochemicals in soil (Singh 2009). The clearing pattern varies depending on elevation and availability of soil nutrients (Ramakrishnan and Patnaik 1992). At low elevation (100-1500 m asl), forest is clear cut and burnt to grow rice. However, at higher elevation (>1500 m asl), farmers generally retain the big trees in relatively less fertile site to facilitate regeneration (Ramakrishnan and Patnaik 1992). For instance, the Tangkhul (Manipur) and Yimchunger (Nagaland) tribes retain the trunks of many species, particularly Alnus nepalensis for subsequent coppicing and regrowth (Singh 2009). After drying the slashed vegetation for 3-4 weeks, farmers burn them before the onset of the monsoon (March-April). Fire breaks are created to ensure that fire does not spread to adjacent areas (Toky and Ramakrishnan 1981a). Monsoon rain helps in mixing ash with the top soil, thus making it temporarily fertile (Saikia 2010).

Photograph 1. A typical jhum cultivation landscape with undulating terrain in Kiphire district, Nagaland, India. (Photo- Lansothung Lotha)

Photograph 2. Trees and burnt logs retained in a jhum field in Phek district, Nagaland (Photo-Joli R. Borah)

2.3 Cropping

Sowing typically takes place in March-April and Jhum farmers sow seeds by dibbling or using broadcast methods (Saikia 2010). Maize and rice seeds are sowed at regular intervals among other crops whereas perennial crops such as ginger (*Zingiber officinale*), colocasia (*Colocasia esculenta*), tapioca (*Manihot esculenta*) and castor (*Ricinus communis*) are sown intermittently throughout the growing season (Toky and Ramakrishnan 1981a). Cropping in Northeast India is predominantly done for one year in a jhum cycle, rarely up to three years (e.g. Nocte tribe in Arunachal Pradesh; (Tawnenga et al. 1997, Tangjang 2009). Harvesting continues throughout the year as crops are sown at different times in a mixed-cropping system (Saikia 2010). The main crop (e.g. rice or maize) is usually harvested in September-October (Saikia 2010). Jhum farmers rarely apply fertilizers and pesticides in their fields (Joshi and Kar 1992).

2.4 Fallow

Fallow period in Northeast India varies across villages and tribes depending on population size, availability of land, elevation and crop choice (Ramakrishnan 1984). For instance, fallow period ranges from 5-6 years in Kohima district to 15-20 years in Phek district in Nagaland that differs in terms of tribe and land availability (personal observation). In both districts, however, fallow periods are reported to have reduced from an earlier longer cycle (20-30 years; personal observation). As a result, jhum farmers have adopted various techniques to facilitate faster recovery of vegetation and soil fertility in fallow sites (details below).

3 Jhum farmers' innovations in Northeast India

Farmers' innovations in jhum cultivation have developed over time in efforts to intervene and modify various stages of the cultivation process to address challenges of land scarcity and declining yield (Bhan 2009). Farmers adopt various techniques to improve crop yield by preventing soil erosion and weed infestation, modifying crop choice and managing fallow sites based on their experience and indigenous traditional knowledge.

3.1 Innovation in soil conservation

Steep terrain and high rainfall often make it challenging for jhum farmers to prevent soil nutrient loss through erosion, surface runoff and leaching (Mishra and Ramakrishnan 1983a). They have innovated various ways of conserving soil by using mechanical and biological barriers. For instance, Naga farmers (Nagaland) construct mechanical earth made barriers -called contour bunds- following the contour of a slope (Tiwari 2007). Farmers also create small holes along the contour line (dibbling) to absorb rain water and prevent soil erosion in the steep slopes (Kerkhoff and Sharma 2006, Singh et al. 2010). These contour bunds help to conserve the rainwater in-situ, prevent soil erosion and nutrient loss (Mishra and Rai 2013) and are also cost-effective (Belguami et al. 1994).

Similarly, Khasi farmers (Meghalaya) construct ridges and furrows that serve as channels for run-off water (Tiwari 2003). They place logs and poles across the steep slope to check soil erosion and water runoff (Singh and Sureja 2006). Farmers also grow cover crops such as maize, velvet bean (*Mucuna spp.*), yam (*Dioscorea spp.*), colocasia, job's tears (*Coix lacryma-jobi*) and ginger along the contour to reduce run-off (Saikia 2010, Deb et al. 2013). Monpa farmers (Arunachal Pradesh) use dry leaves of pine and Paisang tree as natural mulch by spreading them uniformly over the soil (Singh and Sureja 2006). Jhum sites with cover crops or logs and stone barriers face 2.5 times less soil erosion compared to sites with no barriers (Ngullie et al. 2006, Bhan 2009, Singh 2009). These barriers prevent surface run-off (Romkens et al. 1990), thereby increasing water infiltration and storage potential (Tribouillois et al. 2018). Thus, the mechanical and biological barriers are effective soil and water conservation strategies in jhum cultivation system (Nyssen et al. 2000, Jagger and Pender 2003).

3.2 Innovation in crop management

Mixed cropping, i.e. growing a variety of crops instead of a single crop at a time, is a characteristic feature of shifting cultivation across the world (Nye and Greenland 1960, Kleinman et al. 1995) and is prevalent in Northeast India (Appendix I, Text S1; Ingty and Goswami 1979, Tewari 1991, Ramakrishnan 1993). Growing several crops with diverse growth habits, root system and nutrient requirements enables optimal use of the available space and resources such as light, nutrients and water (Toky and Ramakrishnan 1981a, Ramakrishnan 1992). For example, Adi farmers (Arunachal Pradesh) grow up to 72 crops (Yumnams et al. 2011) and Naga farmers (Nagaland) cultivate up to 60 crops (Nakro 2011) at a site of 2 - 2.5 ha with simultaneous sowing and sequential harvesting (Ramakrishnan and Patnaik 1992). Farmers also grow different varieties of the same crop to avoid risks due to variation of annual rainfall. For instance, Nishi tribe (Arunachal Pradesh) cultivate seven different varieties of paddy and over 30 types of vegetables with varying resistance level to drought and high rainfall (Teegalapalli and Datta 2016a). Similarly, jhum farmers grow different crops in multiple sites with varying fallow period in the same year. For instance, Naga farmers grow grain crops such as maize and dry paddy in sites with long jhum cycles but perennials and tuberous vegetable crops in sites with short jhum cycles (Toky and Ramakrishnan 1981a). As soil in short cycles are relatively less fertile compared to long cycles, tuberous crops grown in these sites help in nutrient uptake. Perennial crops also work as cover crops and prevent soil erosion (Ramakrishnan 1984).

Farmers plant crops in distinct zones to optimise the available space based on crop requirement. For instance, the Tangkhul tribe (Manipur) plant maize, millet, sesame, rice bean, soyabean and chilli mainly at the centre of the field as main crop whereas cucumber and pumpkin are grown in the peripheral areas (Singh 2009). Similarly, the Sema tribe (Nagaland) grow brinjal and tomato generally in rocky areas, different varieties of beans next to tree trunks or stubs, yam, job’s tear, colocasia, ginger along the contour, tuber crops just above and leguminous crops immediately below the contour bund to reduce soil erosion (Tewari 1991, Singh 2009). Legumes such as Sesbania, Mucuna, Crotolaria, Cajanus, Indigofera and Mimosae are also planted along with major crops in jhum field for rapid restoration of soil nutrients by nitrogen fixation in short fallows (Patiram and Kumar 2005).

Mixed-cropping provides better returns in kilograms of grain per person hour of labour input compared to settled agriculture in mountainous regions (Thangam 1997). The maize and rice intercropping produced relatively higher yield than other crops in Nagaland (Quayyum and Muniruzzaman 1995). Mixed-cropping also provides a crop cover preventing nutrient loss by water run-off and leaching (Borthakur et al. 1978, Ingty and Goswami 1979), protection against disease and weeds (Ramakrishnan 1993, Kalita and Ram 2015), and facilitates biomass and nutrient recycling through crop and weed residues (Ramakrishnan and Patnaik 1992).

Mixed-cropping involves sequential harvesting of different crops throughout the year (Tewari 1991, Paini and Choudhury 2000). For instance, Angami and Yimchunger tribes (Nagaland) harvest cabbage and potato in May, millet and maize in August and dry paddy in October-November (personal observation). Similar practice exists among Nocte (Arunachal Pradesh), Tangkhul (Manipur) and Konyak (Nagaland) tribes (Bhan 2009, Singh 2009, Tangjang 2009). Sequential harvesting allows farmers to optimise land use. For example, the successive harvesting of cereals creates additional space and decaying plant debris produces organic matter and nutrients for the remaining perennial crops. This innovation also provides an all-purpose diet and ensures availability of food throughout the year (Singh 2009).

3.3 Innovation in weed management

Weed infestation is a primary cause for abandoning sites after a short cropping period (Warner 1991, Mertz 2002) as weeds increase with continuous cropping (Ekeleme et al. 2004, Singh 2009) . Weeds are especially prevalent during the monsoon season (April-June) in jhum fields of Northeast India. The most common weeds in jhum fields are *Imperata cylindrica*, *Digitaria sanguinalis*, *Eluesine indica*, *Borreria hispida, Ageratum conyzoides, Amaranthus viridis, Chromolaena odorata, Commelina benghalensis, Mimosa pudica, Ageratum conyzoides, Spilanthes paniculata, Eupatorium odoratum and Mikania micrantha* (Arunachalam et al. 2002, Debbarma and Singh 2007, Saikia 2010). Weeds compete with crops for soil nutrients and reduce crop yields in jhum fields (Toky and Ramakrishnan 1981a). Efficient weed control mechanisms determine fallow length as weed infestation is more severe in relatively shorter jhum cycle and studies suggest that a minimum of 10 years fallow period is required for natural elimination of weeds (Swamy and Ramakrishnan 1987, Roder et al. 1997).

Weeding in jhum cultivation is traditionally done from April to July, by hand or by using a traditional hoe or spade (e.g. Nocte tribe in Arunachal) (Tangjang 2009, Saikia 2010). Jhum farmers have also developed various innovative ways to prevent weed infestation in their fields. For example, jhum farmers across Northeast India manage weeds by leaving about 20% of the slashed weed biomass in situ to limit nutrient loss on steep slopes (Ramakrishnan and Patnaik 1992). Similarly, Monpa and Sherdukpan tribes (Arunachal Pradesh) use Oak leaves as a mulching material that helps in suppressing weed growth (Dollo 2007). Weed biomass retention in the jhum fields is efficient in preventing weeds from competing with crops while conserving nutrients (Ramakrishnan 2007, Bhan 2009). The retained weed biomass and natural mulch also provide a plant cover over the soil and contributes to nutrient cycling (Ramakrishnan and Patnaik 1992). Moreover, shading effects from coppice sprouts of retained trees may also suppress weed populations as reported elsewhere (Schmidt-vogt 1998, Chikoye et al. 2002, Ekeleme et al. 2004).

Jhum farmers also apply common salt (NaCl) to manage broad-leaved weeds (Keitzar and Imliakum 1999, Saikia 2010). Salt lowers water potential and interference with the uptake of essential nutrients (Chatterjee et al. 2016). Studies suggest that common salt application is more effective in controlling weeds than hand weeding (Rathore et al. 2012, Kumar et al. 2016). Application of salt also increases yield without any harmful effect on soil pH, soil organic carbon and available nutrients (NPK) (Rathore et al. 2012, Chatterjee et al. 2016, Kumar et al. 2016). Crops such as paddy and maize are not affected from applied salt as it specifically targets broad-leafed plants.

Photograph 3. Maize and dry paddy are grown together in a mixed cropping system with wooden log placed strategically to prevent soil erosion in Kiphire district, Nagaland (Photo- Limthure Yimchunger)

3.4 Innovation in fallow management

Efficient fallow management is crucial in facilitating rapid fallow recovery. Farmers adopt various ways to manage fallow areas for faster vegetation growth and restoration of soil nutrients. They retain certain trees instead of clear-cutting while clearing the jhum fields. Jhum farmers also pollard (cutting trees at a certain height leaving the trunk to grow new sprouts) beneficial species to facilitate vegetation regrowth. For example, Angami, Chakhesang, Chang and Yimchunger tribes (Nagaland) retain and pollard *Alnus nepalensis* (Saikia 2010), Nocte and Nissi tribes (Arunachal Pradesh) pollard *Macaranga denticulate* (Nakro and Liezie 1999, Darlong 2004) and Khasi tribe (Meghalaya) pollards species such as *Calicarpa arborea, Albizia procera, Castanopsis tribuloides, Duabanga grandiflora, Gmelina arborea, Michelia champaca* etc (Deb et al. 2013). Similarly, Konyak tribe (Nagaland) pollards various species such as *Alnus nepalensis, Trema orientalis, Sapium baccatum* and *Schima wallichii* to help in coppicing (Tiwari 2007). Farmers also retain useful plants in fallows such as *Livistona jenkinsiana* in Arunachal Pradesh, *Aleurites spp* in Mizoram and broom grass (*Thysanolaena maxima*) in Meghalaya (Tiwari 2007, Bhan 2009).

Retained and pollarded trees provide mechanical support for crops such as *Cucumis sativus* and *Phaseolus vulgaris*, thus saving labour during cropping and later help in vegetation recovery by quick sprouting and maintaining seed banks in the fallows (Singh 2009). For example, Alder (*Alnus nepalensis*) is a native plant that colonises rapidly on highly degraded, unstable soils at high altitudes (800 - 3000 m). Root nodules of the Alder trees improve soil fertility by fixing atmospheric nitrogen in very acidic soils typical of Northeast India (Cairns 2004, Krug 2009a). Moreover, the decomposed leaf litter of Alder retain moisture and mulches by adding humus to the soil (Mishra 2016). It also meets the firewood demand and various domestic needs of farmers, such as charcoal burning and construction (Krug 2009a). Similarly, *Macaranga denticulate* is an early coloniser, fast grower and used for firewood, construction material and other domestic purposes (Bhan 2009). The retained trees are also used for timber and honey bee rearing.

3.5 Slash and mulch jhum cultivation

Jhum farmers in parts of Northeast India have innovated ways to modify traditional jhum cultivation into a fireless cultivation system. For instance, Tangkhul tribe (predominantly in Kalhang village, Manipur) practise a jhum system that does not involve burning (Singh 2009). Farmers slash a chosen site for cultivation in July, unlike the rest of Northeast India where forest is cleared and burned in November-December. They pile up the slashed vegetation along slope contours as vegetative bunds and plant crops immediately without any burning. The slashed unburnt debris provides natural mulch. Such fireless jhum cultivation improves soil quality and crop yield by enhancing soil surface aggregate stability and permeability (Carsky et al. 1998) and reducing run-off (Erenstein 2003). Organic carbon is also higher in the fireless jhum cultivation sites than burnt sites (Singh 2009). Experimental study in Nagaland suggests that application of rice straws as a mulch for the subsequent maize crop significantly increased soil fertility and yield by preventing soil erosion and improving water infiltration (Kumar et al. 2012). Similar practices of slash and mulch systems produced higher yield compared to slash and burn system in Papua New Guinea, Indonesia (Garrity and Lai 2001), Solomon Island (Kabu 2001) and Brazil (Comte et al. 2012).

Photograph 4. Angami Naga tribe innovated the Alder coppicing system by retaining and pollarding nitrogen fixing Alder trees that facilitates fallow regrowth in Khonoma village, Nagaland (Photo Joli R. Borah)

3.6 Innovation for biodiversity conservation

Jhum cultivation creates a mosaic habitat of farmlands, regenerating fallows and primary forests interspersed with each other and thus provides a heterogenous habitat supporting species that prefer diverse landscapes (Padoch et al. 2010a). Such habitat mosaics with short cropping periods and long fallows also maintain seed pools aiding faster regeneration of diverse secondary forests (Rerkasem et al. 2009). Biodiversity in these mosaics play an important role in fallow recovery. For instance, frugivorous birds such as bulbuls, barbet, green pigeons and orioles help in seed dispersal and faster recovery of fallows (Raman 2001b). The regenerating fallows also harbour mammals, such as barking deer, flying lemur, wild pig, squirrels and rodents (Wangpakapattanawong et al. 2010). Similarly, remnant forest patches or trees adjacent to farmland that are typical in many parts of Northeast India (e.g. Yimchunger, Pochury tribe and Konyak tribes in Nagaland; personal observation) play an important role as refuge for biodiversity. Garo farmers in Meghalaya plant fruit trees such as *Dendropthoe falcate* and *Morus macroura* in and around crop fields to attract natural predators such as birds to control crop pests (Sinha et al. 2008).

4 Why Jhum persists?

Jhum cultivation continues to be a dominant land-use system in many remote corners of Northeast India. Various factors influence farmer’s preference for jhum cultivation over alternative land uses in the region. In contrast to the perceptions that motivated government initiatives to replace jhum cultivation, scientific evidence suggests that it is well adapted to heavy rainfall and environmental conditions in mountainous regions (Ramakrishnan 1992, Goswami et al. 2012) and less harmful for the environment and biodiversity compared to permanent agriculture (Gadgil and Guha 1992, Mandal and Raman 2016). Jhum system supports high agro-diversity and ensures food security by optimising resource use (Borthakur et al. 1978, Choudhury and Sundriyal 2003), thus providing an all-purpose diet for farmers (Padoch et al. 2010b). Jhum cultivation with long cycles is also highly energy efficient as shown in studies from Northeast India (Mishra and Ramakrishnan 1982) and elsewhere across the world (Rappaport 1971, Uhl and Murphy 1981). For instance, a 10-year jhum cycle practiced by Nishi tribe in Arunachal Pradesh was highly energy efficient (Maikhuri and Ramakrishnan 1991). Jhum cultivation is also resilient to climatic fluctuations (Shimrah et al. 2015a) and supports both livelihoods and ecosystem services (Dressler et al. 2015).

Transition to alternative land use is challenging along steep slopes with high rainfall in mountainous region of Northeast India for various reasons. Economic efficiency of introduced commercial crops is determined by altitude and regional socio-cultural settings (Behera et al. 2016). Farmers adopted various traditional cash-crops such as areca nuts, citrus, ginger, banana, large cardamom, broom grass, betel leaf and beetle nut, cinnamon and fruit orchards to supplement their livelihood as a coping strategy in the face of growing food insecurity (Tripathi and Barik 2003, Behera et al. 2016). However, introduced cash crops such as rubber and oil palm do not grow at high elevation and in high rainfall areas, and is often limited by poor market accessibility in the areas where they were introduced (Krug et al. 2013). Promotion of horticultural plantations also had limited success due to failure in ensuring storage and effective marketing of produce. For instance, lack of storage and marketing facilities led to waste of excess produce in horticultural plantations such as citrus in Arunachal Pradesh (Arunachalam et al. 2002) and subsequent abandonment of plantations in Meghalaya (Choudhury et al. 2001).

Similarly, terrace cultivation, introduced across Northeast India in 1974-75, failed to gain wide acceptance among jhum farmers as it requires heavy inputs in terms of labour, fertilizers and water supply making it economically unsustainable (Ramakrishnan 1993, Mitra 1998). In Nagaland, terrace cultivation was adopted in Kohima, Zuhneboto and Peren districts where water is available but attempts to grow rice in terraces in districts such as Phek, Kiphire, Tuensang and Mon have been largely unsuccessful for lack of seed supply, fertiliser and assistance from the concerned government department (Ninan 1992). Moreover, settled cultivation could not provide similar levels of dietary variety as jhum cultivation, thus limiting farmer’s choice in the absence of accessible markets (Behera et al. 2016).

Jhum cultivation is also deeply linked with the socio-economic and cultural lives of farmers in Northeast India (Thangam and Mehta 1979). For example, Adi farmers (Arunachal Pradesh) celebrate thirteen different jhum-centric festivals and many rituals at various stages of the jhum cultivation process. Thus, instead of being merely a farming system, jhum cultivation signifies a long historical and cultural connection of the farmer with the land. It maintains social integration and equity (Shimrah et al. 2015a), and contributes to the overall stability and sustainability of the tribal societies (Cramb et al. 2009). A transition away from jhum may therefore lead to unintended social outcomes as evidenced in some parts of Northeast India (Teegalapalli and Datta 2016b) and in Indonesia (Carlson et al. 2012, Obidzinski et al. 2012). For example, a shift to settled wet-rice cultivation from jhum in Garo tribe (Meghalaya) increased gender inequality, reduced social cooperation and negatively affected traditional matrilineal inheritance by promoting male-biased land ownership (Agarwal 1987).

5 Jhum cultivation: the way forward

Farmer-led innovations combined with modern technology have a strong potential in improving the livelihoods of marginalized upland farmers, while balancing economic growth and conservation goals (Arunachalam et al. 2002). These traditional methods can also be further optimized with modern technologies to cope with current market-driven economy. Farmers' innovations can address the declining crop yield in jhum system in two possible ways for sustainable management of the system. As studies suggest a minimum jhum cycle length of 10 years for sustainable farming in the mountains (Ramakrishnan 1984), the fallow period can be maintained at 10 years or longer in areas where enough farmland is available to accommodate a longer cycle. Alternatively, in areas with limited land, jhum can be practiced at shorter cycles by incorporating additional inputs. Both pathways also require diversification of livelihoods to ease the pressure on jhum. Moreover, cropping period can also be extended from the typical one-year period in Northeast India to at least two years with use of inorganic and organic fertilizers, as evidenced by experimental studies in Mizoram (Tawnenga et al. 1997).

The effectiveness of jhum farmers' innovations in conserving soil and improving crop yield suggests that they are adapting to the changing trends of this cultivation system. These innovations in crop, soil and fallow management, if incorporated into agricultural policies and applied widely, can contribute significantly in sustainable management of jhum cultivation in Northeast India (Cairns and Garrity 1999, Arunachalam et al. 2002, Shimrah et al. 2015b). However, this requires a broader perspective of jhum cultivation as a complex and dynamic system to adapt to the ecological, socioeconomic and structural constraints in mountainous regions rather than a primitive agriculture. Extending institutional support to the farmers in terms of research, financial help and technical assistance to sustain farmers innovations can help in wide and effective implementation of such techniques (Goswami et al. 2012). Policies can also provide incentives for community-based sustainable land-use practices, such as agroforestry, soil conservation, cover crops, intercropping in jhum cultivation systems. These interventions also need to respect and legally protect customary rights and diverse tenure arrangements of jhum farmers. Instead of resettlement of jhum farmers, which leads to land exploitation, governments can formulate strategies for sustainable management of jhum cultivation with active participation by farmers. The use of participatory methods such as Participatory Rural Appraisal (PRA) can help in involving local people in planning and formulating such policies (Thrupp et al. 1997c).

Sharing information on effective innovations among jhum farmers can help farmers to learn from each other and devise appropriate strategies suitable for their local environment (Bhan 2009). For example, Alder coppicing was an innovation by Angami farmers from Khonoma village in Nagaland, which has been successfully adopted by farmers in many other parts of Nagaland. Similarly, plantation and commercialization of broom, previously a wild grass in Meghalaya, proved successful in improving soil fertility and generating livelihood (Tiwari and Kumar 2008). Farmers’ innovations across the world can also be communicated and integrated in jhum system if found suitable. For example, in the Peruvian Amazon, shifting cultivation farmers responded to growing land scarcity through diversification of land holdings, increased use of fallow products for generating income, and innovative use of biochar on charcoal kiln sites and home gardens (Coomes et al. 2017). Application of biochar derived from weed biomass can increase crop yield by enhancing soil fertility in Northeast India (Mandal et al. 2015), while simultaneously mitigating climate change by sequestrating carbon (Woolf et al. 2010). Similarly, farmers have integrated legumes and composting in Zambia (Chidumayo 1987) and pine needles and animal manure in Bhutan (Roder et al. 1992) to improve soil fertility. Adding both ash and compost in a 30-year fallow in Madagascar significantly increased crop yield for Maize by improving soil moisture, pH, organic matter and microbial activity (Gay-des-Combes et al. 2017).

Policy makers across countries in South and South-east Asia have also changed their negative perception towards shifting cultivation and implemented policies to improve shifting cultivation rather than replacing it. For example, farmers’ innovations on soil and water conservation have been collated as a resource material in Bangladesh (Khisa et al. 2006). In Bhutan, the forest department has initiated further research to identify crop and tree species that grow well following traditional controlled burning. Similarly, policy makers in Nepal have initiated dialogue with all stakeholders, including the members of indigenous communities, to address issues such as land rights and declining yield in shifting cultivation. The recently adopted Shillong Declaration 2004 in Northeast India also recognises the inherent mechanisms of jhum cultivation that makes it sustainable in the steep mountains of Northeast India. It recommended improving jhum cultivation by addressing land tenure security, market development and commercialization of jhum products and strengthening customary institutions instead of entirely replacing the cultivation system. The National Mission on Greening India also encouraged effective fallow management using scientific evidence and traditional indigenous knowledge (Deb et al. 2013).

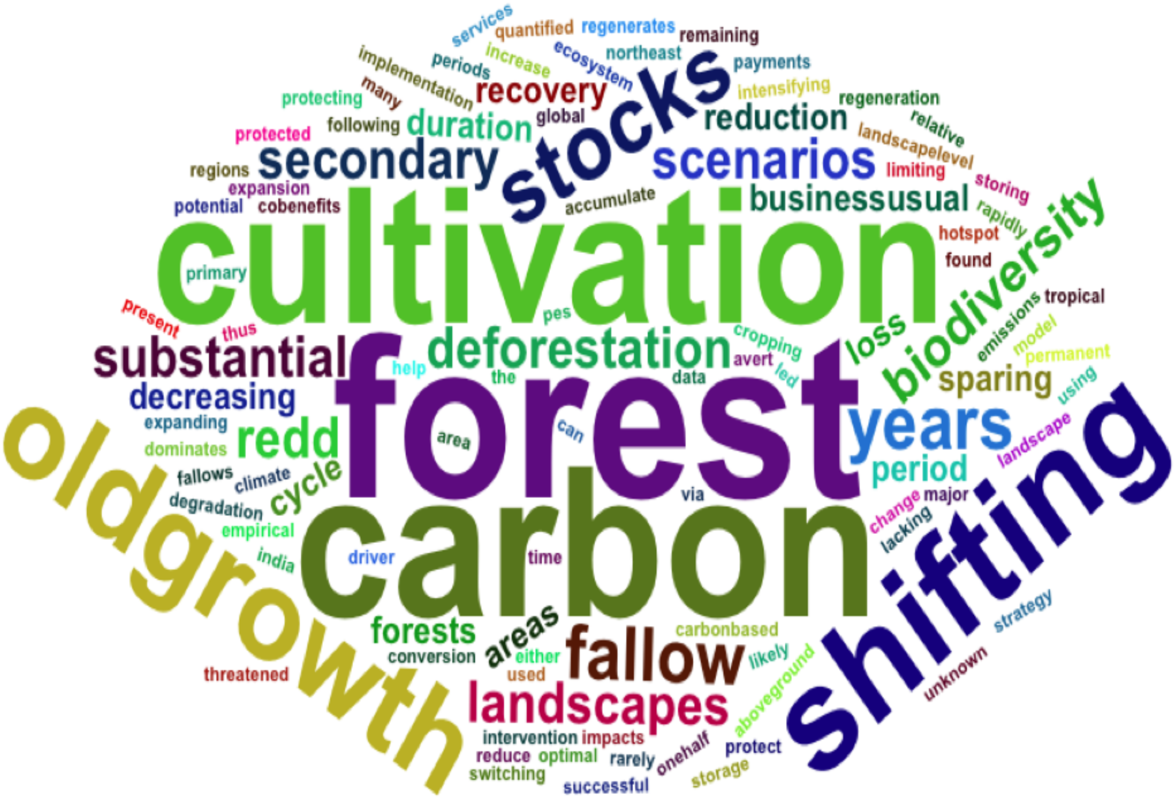
Similarly, Nagaland Environmental Protection and Economic Development has successfully incorporated farmers' innovations into its programmes (NEPED 2002). For example, experiments by NEPED with contour hedgerow intercropping, i.e. growing nitrogen-fixing shrubs as dense hedgerows along slope contours and planting crops between the hedgerows, have shown positive effects on soil fertility and crop yields (Bhatt and Laxminarayana 2010, Grogan et al. 2012). NEPED also facilitated an institutional framework for diversification of economic activity through active community involvement by forming village development boards (VDBs) across 1200 villages in Nagaland (Ramakrishnan 2006). These VDBs developed effective fallow management practices based on farmer's knowledge and experience of plant species that helps faster recovery. Thus, this project in Nagaland focusses on building upon farmers' innovations to manage jhum cultivation sustainably rather than replacing it (Ramakrishnan 2007) and these approached can be replicated in rest of Northeast India.

6 Conclusion

Jhum cultivation is a dynamic and adaptive farming system (Fujisaka et al. 1996). The number and arrangements of farmland, fallows at different stages of recovery, and old-growth forests and remnant trees or forest patches, as well as up to 60 crops grown in a mixed cropping system, generate a highly complex landscape (Padoch et al. 2010b). It also plays an important cultural role in local customs, traditions, and practices, besides offering economic security to farmers in Northeast India (Joshi and Kar 1992, Ninan 1992). Thus, a balanced approach to development that also recognises the merits of jhum is needed to design effective policies. Holistic management of jhum cultivation requires sincere will, appropriate planning, efficient implementation and active participation of all stakeholders. Understanding the farmers' perspectives and the intricacies of this complex farming system and incorporating their innovations into policies can pave the way for sustainable jhum cultivation in Northeast India.

Chapter 3

Quantifying carbon stocks in shifting cultivation landscapes under divergent management scenarios relevant to REDD+



1 Abstract

Shifting cultivation dominates many tropical forest regions. It is expanding into old-growth forests, and fallow period duration is rapidly decreasing, limiting secondary forest recovery. Shifting cultivation is thus a major driver of carbon emissions through deforestation and forest degradation, and of biodiversity loss. The impacts of shifting cultivation on carbon stocks have rarely been quantified, and the potential for carbon-based payments for ecosystem services (PES), such as REDD+, to protect carbon in shifting cultivation landscapes is unknown. I present empirical data on aboveground carbon stocks in old-growth forest and shifting cultivation landscapes in Northeast India, a hotspot of threatened biodiversity. I then model landscape-level carbon stocks under business-as-usual scenarios, via expansion into the old-growth forest or decreasing fallow periods, and intervention scenarios in which REDD+ is used to either reduce deforestation of primary or secondary forest or increase fallow period duration. I found substantial recovery of carbon stocks as secondary forest regenerates, with a 30-years fallow storing about one-half the carbon of an old-growth forest. Business-as-usual scenarios led to substantial carbon loss, with an 80% reduction following conversion of old-growth forest to a 30-years shifting cultivation cycle and, relative to a 30-years cultivation landscape, a 70% reduction when switching to a 5-years cultivation cycle. Sparing old-growth forests from deforestation using protected areas and intensifying cropping in the remaining area of shifting cultivation is the most optimal strategy for carbon storage. In areas lacking old-growth forest, substantial carbon stocks accumulate over time by sparing fallows for permanent forest regeneration. Successful implementation of REDD+ in shifting cultivation landscapes can help avert global climate change by protecting forest carbon, with likely co-benefits for biodiversity.

2 Introduction

Deforestation and forest degradation in the tropics contribute significantly to biodiversity loss and generate 12% of global annual anthropogenic carbon emissions (Van der Werf et al. 2009; Barlow et al. 2016). Shifting cultivation is the dominant land use across 2.6 million km^2 in the tropics, of which only 6%-19% is cleared annually for crop production (Silva et al. 2011). While this provides subsistence for 200-300 million people across 64 developing countries (Mertz et al. 2009; Li et al. 2014), it is also a major driver of carbon emissions (Fearnside 2000) and biodiversity loss (Ogedegbe & Omoigberale 2011; Ding et al. 2012). Reducing deforestation and forest degradation from shifting cultivation can thus play a key role in averting climate change and the global extinction crisis (Lawrence et al. 1998, Houghton 2012).

Shifting cultivation involves clearing a forest patch using slash-and-burn methods. Crops are grown on the cleared land for a few seasons, after which the farmland is left fallow for vegetation regeneration (Mertz 2009). During this fallow period, farmers cultivate other plots and return to clear the regenerated secondary forest in the original plot at the end of the rotation period (Mishra and Ramakrishnan 1983b). Historically, the fallow period lasted for 20-30 years allowing complete regeneration of secondary forest in tropical regions (Rerkasem et al. 2009, Poorter et al. 2016). However, due to increasing human population and more demand for farmland, fallow periods have reduced to just 2 to 3 years in many regions, which is insufficient for forest regeneration (Grogan et al. 2012). This causes more frequent rotation in existing shifting cultivation and further clearing of primary forest to compensate for decreasing yield, which leads to carbon emissions and biodiversity loss in shifting cultivation landscapes (Raman 2001b, Williams et al. 2008, Klanderud et al. 2009, Rossi et al. 2010, Jakovac et al. 2015).

Due to various socioeconomic factors, including human population growth, market development, and government policies, there is an increasing trend of transforming shifting cultivation landscapes to more profitable and intensive land uses, such as cash crop plantations (e.g., rubber; Brookfield et al. 1995) and permanent agriculture (De Jong et al. 2001). This trend is particularly evident in tropical Asia (van Vliet et al. 2012), although shifting cultivation is still widely practiced in remote mountains of Bangladesh, Laos, and Northeast India (Rasul and Thapa 2003). This transition from shifting cultivation to more intensive land uses can have drastic negative impacts on the environment leading to permanent deforestation and biodiversity loss (van Vliet et al. 2012). Therefore, finding alternative and more sustainable approaches to managing shifting cultivation landscapes is of utmost importance.

Few previous studies have assessed how changes in fallow period or the conversion of primary forest to shifting cultivation affect landscape-level carbon stocks (Mukul et al. 2016a, 2016b). There is an urgent need to do so given the widespread trend for reduced fallow periods (Metzger 2002, van Vliet et al. 2012) and marked expansion of shifting cultivation in recent decades (Castella et al. 2005; Hansen & Mertz 2006; Bogaert et al. 2008; Robichaud et al. 2009). Such assessments are critical to the development of carbon-based payments for ecosystem services (PES) schemes, such as the ‘Reducing Emissions from Deforestation and forest Degradation (REDD+)’ framework (Mertz 2009). REDD+ provides financial incentives to forest rich developing countries for reducing carbon emissions by avoiding deforestation and forest degradation, enhancing forest carbon stocks, and managing forests sustainably (UNFCCC 2010). REDD+ has the potential to avoid deforestation by protecting old-growth forests from shifting cultivation expansion, avoid forest degradation by maintaining a longer fallow cycle, and to enhance carbon stocks by permanent abandonment of older fallow sites or by moving back from short to long fallow cycles. These approaches might also provide co-benefits for biodiversity conservation, other ecosystem services and sustainable rural development (Gibbs et al. 2007, Phelps et al. 2012b). However, it is not clear which of these REDD+ pathways will maximize carbon storage in a shifting cultivation landscape.

Here, we examine how fallow period affects carbon stocks across regenerating secondary forests following shifting cultivation in Nagaland, Northeast India, which is of critical importance for global biodiversity conservation (Myers et al. 2000) and where shifting cultivation occupies nearly three quarters of agricultural area (Pareta 2013). We then use these data to model and compare landscape-level carbon stocks under two alternative management scenarios of shifting cultivation: (1) scenarios with reduced fallow periods or expansion into primary forest; and (2) intervention scenarios with efforts to protect forest carbon through mechanisms compatible with REDD+. We assess the relative effectiveness of these scenarios in retaining maximum levels of landscape carbon to identify the optimal allocation of efforts and resources under REDD+ in shifting cultivation landscapes.

3 Materials and methods

3.1 Study area

Our study region comprised three districts (Kiphire, Phek, and Kohima) in Nagaland, Northeast India (Appendix II: Fig. S1) across an altitudinal range of 1,487-2,652 m above sea level (asl; Appendix II: Table S1). These landscapes are within the Indo-Burma global biodiversity hotspot and specifically are part of the Fakim Wildlife Sanctuary and Saramati area Important Bird Area (#IN421; BirdLife International 2017). The major forest types of the sampling sites were subtropical broad-leaved wet hill forests (500-1,800 m asl), subtropical pine forests (1,000-1,500 m asl; to 1,645 m asl in our study area) and montane wet temperate forests (>2,000 m asl; Champion and Seth 1968). Annual rainfall varies from 1,800 to 2,500 mm (Statistical Handbook of Nagaland 2015). Shifting cultivation occupies 71.2% of the total agricultural area in Nagaland (Pareta 2013). Fallow period in this region varies from 6 to 27 years (personal observation). Common crops grown in shifting cultivation sites are upland rice (*Oryza sativa*), pearl millet (*Pennisetum glaucum*), maize (*Zea mays*), cassava (*Manihot esculenta*), ginger (*Zingiber officinale*), chili pepper (*Capsicum annuum*), sweet potato (*Ipomoea batatas*), and various pulses (Krug 2009b).

3.2 Sampling framework

I sampled in three shifting cultivation landscapes (Kiphire in 2015; Phek and Kohima in 2016), each separated by at least 25 km of mountainous terrain (Appendix II: Fig. S1). Each landscape comprised shifting cultivation farmland, fallows with regenerating secondary forests (abandoned farmland), and old-growth forests. Old-growth forests were sampled as control sites. They had no history of shifting cultivation but had low to moderate levels of disturbance from grazing and selective logging. Under the realistic assumption that adverse anthropogenic activities will not be entirely prevented under REDD+ management scenarios, these old-growth forests provide a robust estimate of how much carbon could be stored if land currently under shifting cultivation were allowed to regenerate fully and, conversely, the carbon stock that would be lost if shifting agriculture expands into previously unfarmed areas.

I defined the fallow period as the unfarmed interval between cropping periods, during which natural vegetation regenerates. Cropping period (one or two years in our study system) is the duration of cropping at a site following clearing. The entire duration of cultivation, that is, from cropping to the start of the next phase of clearing is termed as a cultivation cycle (cropping period + fallow period).

3.2.1 Determining the age of secondary forest. — I determined and verified the age of secondary forest regenerating during the fallow period (55 sampling plots) via interviews with members of each village and remote-sensing Landsat images (Landsat 5 TM32, USGS 2017). I conducted semi-structured interviews with famers managing each plot. I interviewed at least five farmers per village (range 5-9) and selected individuals whose age and experience enabled them to provide information on fallow age from first-hand experience. In most villages in this region farmers clear forest patches in groups, so I were able to verify these dates by independently cross-checking with a number of farmers from the same group.

I verified the interview-derived estimates using remote-sensing data to find the year when each sampling plot was last burnt. This is a strong signal of shifting cultivation because fires are always used to clear vegetation, and otherwise are extremely rare in this landscape (personal observation and data from semi-structured interviews). I used Landsat 5 Thematic Mapper (TM) 32-day raw composite images from USGS with band combinations of 7, 4 and 2 – which depicts vegetation as bright green and recently burnt areas as red. These Landsat data were available from 1988 to 2012 and could thus verify precisely the estimates provided by farmers when plots were between three and 28 years old. The difference between interview and Landsat derived estimates ranged from -7 to 9 years (mean absolute difference (± SD) 2.3 ± 2.5 years). Where there were discrepancies in ages between satellite data and interviews, I used the age estimates from satellite data for our analysis. However, for plots cleared after 2012 I used the interview-derived estimates on the assumption that recent memories of clearance dates will be very accurate.

3.3 Carbon sampling

I measured non-soil carbon stocks across three main habitat types: farmland, secondary forest (accounting for variation in age), and old-growth forest. I randomly selected 36 400 m × 400 m sampling squares across the three habitats in each of the three landscapes (15, 12, and 9 squares in Kiphire, Phek, and Kohima, respectively).

The number of squares in each district varied depending on the availability of fallow sites and adjacent old-growth forest sites (distance between fallow sites to the nearest primary forest across the three landscapes = 2410.5 ± 1748 m). Sampling squares were placed at least 300 m apart between different habitats and 400 m apart within the same habitat. As the sampling squares in old-growth forest sites were consistently in higher elevation compared to jhum farmland and fallows in each village, I sampled in multiple villages in each district with varying elevation (total 7) to avoid any systematic bias in terms of elevation across the various habitat types. Within each sampling square, I located three 10 m × 30 m sampling plots (n = 108; 3.24 ha sampled in total) that were at least 200 m apart (Appendix II: Fig. S1B, C, D).

I used a large number of relatively small plots across farmland, secondary, and old-growth forest rather than fewer bigger plots to better capture the small-scale heterogeneity in land-use history and topography (altitude and ruggedness; 1,487–2,652 m asl) typical of a shifting cultivation mosaic landscape in the study region (Yadav et al. 2012). Previous studies from such mountainous regions have derived reliable carbon estimates from plots of similar or smaller size: McEwan et al. 2011, 0.04 ha; Zeng et al. 2013, 0.04 ha; Hu et al. 2015a, 0.04 ha; Ali et al. 2014, 0.01 ha; Mukul et al. 2016a, 0.025 ha; and Gilroy et al. 2014a, 0.0075 ha. To ensure unbiased selection of plots, I walked 100 m perpendicular from the boundary into the focal habitat type. The resultant end point was used as the first corner of the 10 m × 30 m carbon-sampling plot and the second point was located 30 m to the left (i.e., roughly 30 m parallel to the habitat edge). The other two axes of the rectangular plot were parallel to these two randomly selected points. I followed this methodology consistently for all plots. Within each sampling plot, I first measured aboveground living biomass (trees and lianas) and dead biomass (deadwood and leaf litter) using a composite plot design (Appendix S1: Fig. S1E) and converted these biomass estimates to carbon stocks (see section “Estimating total carbon”).

I did not quantify soil organic carbon as studies from Northeast India indicate that soil carbon is resilient to land-use changes from shifting cultivation and recovers rapidly within the first two years of the fallow period (Lungmuana et al. 2017). In addition, studies from elsewhere in the tropics also suggest that forest age has negligible influence on soil carbon, which accumulates rapidly and then stabilizes following abandonment (Kotto-Same et al. 1997b, Martin et al. 2013a). Previous studies also show that tree root biomass in fallow sites can contribute considerably to total carbon as resprouting trees, that is typical of shifting cultivation systems, have larger roots than trees grown from seeds in shifting cultivation fallows (McNicol et al. 2015). As I did not incorporate root biomass in carbon stock estimation for lack of appropriate root: shoot ratio in the literature, it is likely that carbon stock recovery in fallows was underestimated to some extent. However higher carbon stocks in fallows further emphasizes the pattern of rapid recovery of carbon stocks as found in this study.

I took a space-for-time substitution approach to assess variation in carbon stock across fallow ages. This approach assumes that the observed spatial sequence truly represents a temporal sequence, such that sites in the sequence differ in age, but are similar in abiotic and biotic components and thus share a similar predictable history of regeneration (Johnson and Miyanishi 2008). To minimize any difference in successional history and thus trajectories of carbon accumulation, I sampled landscapes across similar topography, soil type, and land-use histories (derived from Landsat images and farmer interviews) as recommended by (Walker et al. 2010). I also sampled multiple replicates for younger age classes where variability in vegetation structure is high (Swamy and Ramakrishnan 1987).

3.3.1 Estimating live biomass. —I determined live biomass by measuring the diameter at breast height (DBH) and wood specific gravity of trees. I measured DBH at 1.3 m from ground level in each 10 m × 30 m plot for all trees larger than 5 cm DBH. I measured trees with 1–5 cm DBH in three subplots each of 2 m × 2 m in size (T1–T3, Fig. S1E) at 5-, 15-, and 25-m distance from the start of the plot, along the plot midline. To calculate wood specific gravity, I extracted tree cores from all trees larger than 5 cm DBH at 1.3 m with an increment borer (two threads, 5.15 mm diameter, 400 mm bit length; Haglöf, Sweden). The full core was placed in water for 30 min to fully hydrate it and the fresh volume (i.e., green volume) was then measured using the water-displacement method (Chave 2005). Cores were then oven dried at 101°–105°C (Williamson and Wiemann 2010) for 24 h and weighed. Finally, I calculated wood specific gravity (g/cm3) from the dry mass (g) to green volume (cm3) ratio (Chave 2005):

$$\text{Wood specific gravity} = \text{Wood oven dried weight}/\text{green volume}$$

The extraction of cores was not possible for small trees (1- 5 cm DBH), so for these individuals, I used the mean wood specific gravity calculated from large trees within the focal 10 m × 30 m plot.

The tree biomass was calculated as the mean estimate from suitable allometric equations generated from studies of harvested trees. I used five allometric equations generated for similar forest types to those in our study that incorporated information on DBH and wood specific gravity: two equations for trees in old-growth forest (Dung et al. 2012, Chave et al. 2014), and three equations for trees in secondary forest (Ketterings et al. 2001; Van Breugel et al. 2011; Chave et al. 2014; Appendix II: Table S2). I did not use equations that included height as a predictor as this is extremely difficult to measure accurately in closed canopy forests and on steep terrain. I did, however, calculate the biomass by measuring heights and DBH of 39 randomly selected trees (DBH range = 75.7- 206.9 cm) for which I was able to accurately measure height using a clinometer. For these trees, I compared biomass from the equation that incorporated height with biomass from the one that did not (both equations from Chave et al., 2014). I found that allometric equations with height generated slightly higher biomass estimates than equations without height (matched paired t test, $t = 2.25$, $P = 0.03$, RMSE = 6.07 Mg), suggesting that our estimates of biomass are conservative (lower carbon) across our plots. For trees with a DBH of 1–5 cm, I calculated tree biomass using the same allometric equations as those used for larger trees, because the few equations developed specifically for younger trees did not incorporate wood specific gravity as a predictor variable (Nascimento and Laurance 2002).

I measured the DBH at 1.3 m height of all lianas larger than 2 cm DBH in two 1 m × 30 m sampling subplots located on the plot sides (V1-2, Appendix II: Fig. S1E). I converted the liana DBH into biomass using five allometric equations for lianas that have been developed for tropical forests (Putz 1983, Gehring et al. 2005, Schnitzer et al. 2006, Sierra et al. 2007, Addo-Fordjour et al. 2012); Appendix II: Table S2). I used the mean of these five estimates as a measure of the biomass of each liana. I calculated subplot liana biomass by summing the biomass estimates of all lianas for each subplot. Finally, liana biomass for each plot was calculated as the average of the two subplot biomass estimates.

3.3.2 Estimating dead biomass.— I measured deadwood and leaf litter to estimate the carbon stock in dead vegetation in each plot. To estimate deadwood biomass, I recorded all standing and fallen deadwood larger than 5 cm DBH within each 10 m × 30 m sampling plot. I measured the diameter at both ends of the fallen dead wood and its total length (in all cases, these measurements were only taken for the section of deadwood inside each plot). For standing deadwood, I measured the diameter at the bottom of the deadwood and its height using either a measuring tape (when the top was accessible) or a clinometer (when the top was not accessible). When possible, I also measured the diameter at the top of the deadwood. I measured deadwood volume using the 'frustum of a cone' formula when diameter at the top and bottom could be measured:

$$V = \frac{\pi h}{3} \times (R^2 + r^2 + Rr)$$

where V is volume (cm3), h is height/length (cm), R is diameter of the base (cm), and r is diameter of the top (cm; Pfeifer et al. 2015).

When the top diameter could not be measured, I assessed volume using the formula for the volume of a cone (symbols denote the same parameters as the frustum equation)

$$V = \frac{\pi R^2 h}{3}$$

I assigned each standing and fallen deadwood into one of five decomposition classes ranging from class 1 (recently dead intact wood) to class 5 (almost decomposed) following Pfeifer et al. (2015). When deadwood was class 1, I extracted a wood core to calculate deadwood density. For the rest of the decay classes, I extracted wood density estimates for each class from the literature (Pfeifer et al. 2015) to estimate deadwood biomass.

I collected all leaf litter (fallen leaves, twigs, and grasses) from three 1 × 1 m subplots (L1–L3, Appendix II: Fig. S1E) centered within each 2-m^2 subplot (T1–T3, Appendix II: Fig. S1E) for each 10 m × 30 m plot. I measured total leaf litter volume in situ using a 'compression cylinder' (Parsons et al. 2009) and calculated the dry mass (oven dried to constant mass) of a 1 L subsample to estimate total dry biomass of leaf litter.

3.3.3 Estimating total carbon. — I used our four biomass estimates (living tree, lianas, deadwood, and leaf litter) to calculate biomass within each plot (Mg/ha). To derive an estimate of total carbon stock in each plot, I multiplied the plot level biomass estimate by 0.474, which is the wood carbon to biomass ratio for both living and dead carbon estimated by Martin & Thomas (2011).

3. 4 Statistical analyses

All analyses were conducted using R 3.3.1 software (R Development Core Team 2017). Prior to analysis, I confirmed that all data used in statistical tests did not violate the assumptions of normality and heteroscedasticity using Shapiro-Wilk and Levene’s tests, respectively. I log10- transformed the carbon estimates prior to analysis to meet the normality assumption of regression analyses. A Moran’s I test, implemented in the ape package (Paradis et al. 2004) in R software, confirmed that there was limited spatial autocorrelation in total carbon stock and this was not statistically significant (Moran’s I = 0.082, P = 0.08).

3.4.1 Variation in carbon stock across habitats and fallow period. — I constructed a linear mixed-effect regression (Lmer) model using the lme4 package (Bates et al. 2015) to examine differences in carbon stocks across the three habitats, that is farmland (n = 17 plots), secondary forest (n = 55), and old-growth forest (n = 36). I included habitat type and elevation as fixed effects. Similarly, to assess differences in carbon stock across fallow ages of secondary forest, Lmer models were fitted including fallow age and elevation as fixed effects. I included squares nested within landscapes as random intercepts in the model to control for multiple sites within each square. I fitted separate Lmer models for total, living, and dead carbon with the same fixed and random effects. ‘Elevation’ in both model sets was scaled by subtracting the mean and dividing by the standard deviation to facilitate model interpretation (Gelman 2007).

I conducted AICc-based multimodel inference using the function ‘dredge’ in the MuMIn package in R to run a complete set of models with all possible combinations of the fixed effects including their interaction terms. The function ‘r.squared’ in the same package was used to calculate marginal and conditional r^2 values for each model, which showed the percentage of variation explained by the fixed and random effects, respectively (Barton 2014). I used an information theoretical approach based on Akaike Information Criterion corrected for small sample sizes

(AICc) for model selection. The model with the lowest AICc value was chosen as the best-fit model (Burnham and Anderson 2002).

3.4.2 Predicting change in landscape-level carbon under hypothetical scenarios.— Our objective was to assess how carbon stocks change under alternative management systems that alter the fallow period in two different types of landscapes: (1) existing shifting cultivation that, at the start, contains farmland and various ages of regenerating secondary forest, but no old-growth forest (Scenarios 1 and 2, Fig. 1); and (2) pioneer shifting cultivation that, at the start, only contain old-growth forest (Scenario 3 and 4; Fig. 1). I considered a 5-year cultivation cycle as the shortest cycle, because studies show that, with fertilizer inputs, soil fertility restores within the first two years of fallow ensuring a 5-year cycle as a viable option for crop cultivation (Thomaz 2013, Lungmuana et al. 2017). I did not include conversion to permanent agriculture in scenario predictions as studies suggest that this is not sustainable in this region, in part due to severe soil erosion and nutrient depletion (Grogan et al. 2012), and there will often also be cultural impediments. I used empirical data from our models of carbon stocks in farmland, secondary, and old-growth forests to predict landscape- level carbon stocks under different management scenarios.

Scenario 1 applies to landscapes currently used for shifting cultivation and represents the current trend of decreasing fallow periods to meet growing food demands (no forest sparing, Fig. 1). Thus, it provides a scenario without any interventions to reduce carbon emissions. I assume that the initial cultivation cycle is 30 years (one year of cropping followed by a 29-year fallow period), with an equal area of land in each of the 30 possible states, that is farmland and secondary forest of each age class (1–29 years post farming). I predicted the change in landscape-level carbon when increasing demand for food is met by reducing the fallow period but without expanding cultivation to additional old-growth forests. I estimated carbon stocks when the original 30-year cycle is reduced to 15-year (Scenario 1.1), 10-year (Scenario 1.2), and 5-year (Scenario 1.3).

Scenario 2 also applies to a landscape currently used for shifting cultivation with a 30-year cultivation cycle. However, in this scenario, financial incentives are available to reduce the amount of land used for shifting cultivation, enabling remaining older fallows to regenerate (secondary forest creation and sparing, Fig. 1). Thus, this scenario reduces carbon emissions by avoiding forest degradation and enhancing forest carbon stocks making it relevant to conservation interventions through REDD+. Under this scenario, as fallow period declines, the older fallows are spared from cultivation by increasing agricultural intensity of a part of the landscape. Increased intensification (such as the use of chemical fertilizers) would enable food production to be maintained despite shorter fallow period (Lungmuana et al. 2017). I estimated carbon stocks when 50%, 67%, and 83% of the landscape were removed from shifting cultivation in 15-year (Scenario 2.1), 10-year (Scenario 2.2), and 5-year cultivation cycles (Scenario 2.3), respectively.

Scenarios 3 and 4 apply to landscapes originally covered by old-growth forest, but converted to a shifting cultivation landscape (i.e., pioneer shifting cultivation; Mertz, 2009). Scenario 3 describes the application of conservation interventions, such as protected areas, that limit further clearing of old-growth forest and associated carbon emissions for expanding shifting cultivation (old-growth forest sparing, Fig. 1). This scenario is thus relevant to REDD+ interventions to reduce emission from deforestation. This scenario also requires intensification as increasing land areas are spared from shifting cultivation with declining fallow period. I assessed three alternatives for this scenario: conservation of 50%, 67%, and 83% of the old-growth forest in 15-year (Scenario 3.1), 10-year (Scenario 3.2), and 5-year cultivation cycles (Scenario 3.3), respectively. Our final scenario (Scenario 4) occurs when old-growth forest is entirely cleared to create a shifting cultivation landscape, thus providing an additional business-as-usual scenario with no REDD+ intervention (shifting cultivation expansion, Fig. 1). The shifting cultivation landscape in Scenario 4 has a 30-year cultivation cycle, that is the same cycle as that is used for the baseline situation in Scenario 1 and 2. Across all scenarios, landscapes consist of 30 individual and uniform-sized parcels of land. Each parcel is either under shifting cultivation (farmland or fallow site), permanently abandoned regenerating secondary forest (Scenario 2 only), or old-growth forest (Scenario 3 and 4). To assess temporal variation in carbon accumulation across scenarios, I estimated landscape-level carbon after 30 years (i.e., the maximum fallow period across our scenarios) and after a shorter time frame of 5 years (Fig. S3), giving a snapshot of changes in carbon stocks following interventions. I calculated landscape-scale carbon using 1,000 simulations for each scenario. This was achieved by randomly allocating, with replacement, each land parcel an estimated amount of carbon from observed values for farmland and old-growth forest. For secondary forest, I cannot sample with replacement from observed carbon values for each fallow age as there is insufficient observation for each fallow age. I thus fitted a linear mixed-effect model of carbon as a function of fallow age (with landscape as a random effect) and sampled with replacement from the range of carbon values generated by the model (i.e., taking 95% confidence intervals of parameter estimates into account) for each age. I then summed the predicted carbon estimates together across the 30 sites to derive the predicted landscape-level carbon stock for each hypothetical scenario at the end of 5 and 30 years.

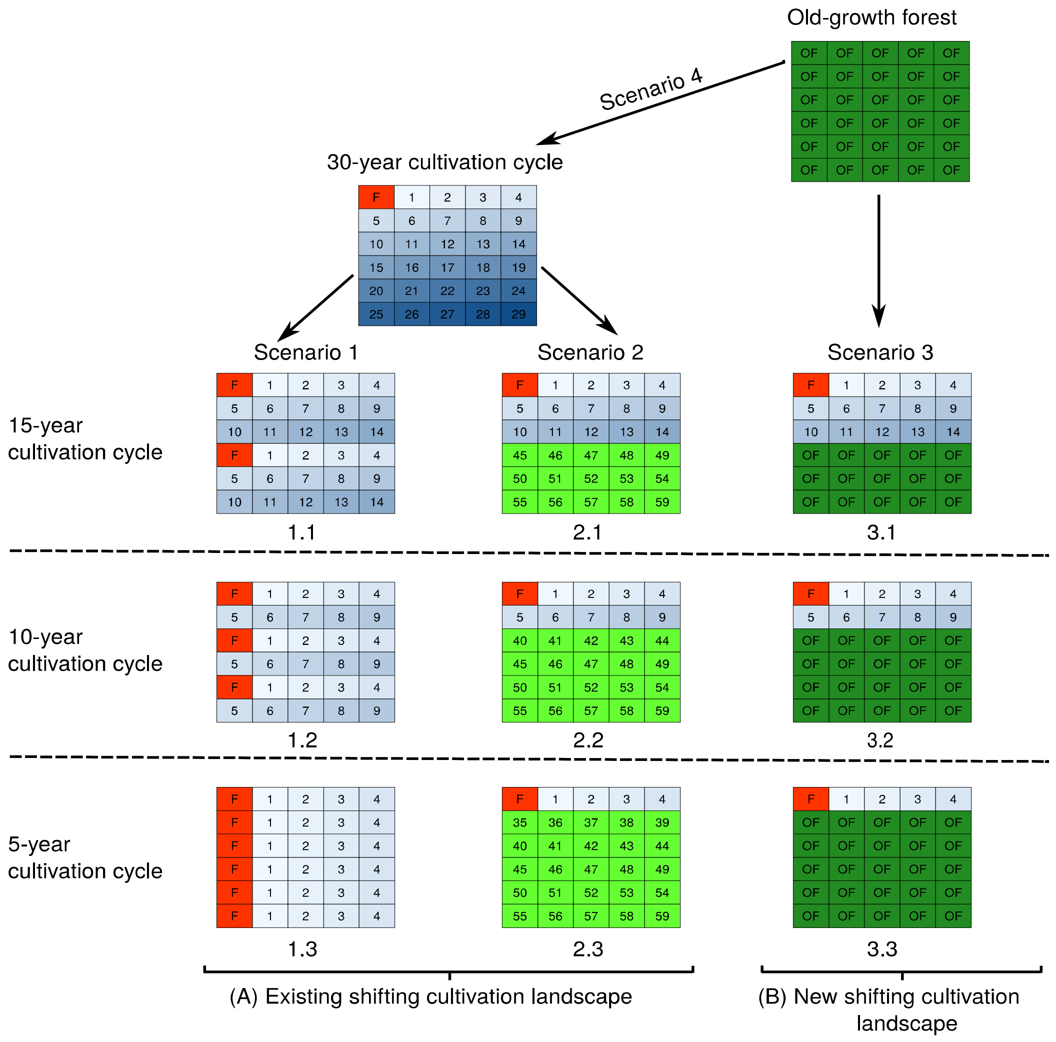


Figure 1. The four sets of management scenarios used to predict changes in landscape carbon in (A) no forest sparing (Scenario 1) and secondary forest creation and sparing (Scenario 2) and (B) new shifting cultivation landscape with old-growth forest sparing (Scenario 3) and shifting cultivation expansion (Scenario 4). Colours indicate habitat types: farmland (F, red), active fallows (1-29 years, different shades of blue), abandoned old fallows (>30 years, light green) and old-growth forests (OF, dark green). Numbers within cells denote the age of the secondary forests

4 Results

I measured a total of 3,160 stems (range 1.27– 280.36 cm DBH), of which 1,976 (62.5%) were from secondary forest and 1,184 (37.5%) were from old-growth forest. Stems were absent in our farmland plots. I also measured 128 lianas (75.7%, 24.3%, and 0% in old growth forest, secondary forest, and farmland, respectively), 226 standing deadwood stems (32.3%, 44.7%, and 23% in old-growth forest, secondary forest, and farmland, respectively), and 1491 pieces of fallen deadwood (54.4%, 22.4%, and 23.2% in old-growth forest, secondary forest, and farmland, respectively).

4.1 Variation in carbon stocks across habitats

The best-fit model for total carbon stock included habitat type as a fixed effect, with higher total carbon in old-growth forests than secondary forests and farmland (coefficient estimates ± SD, farmland= 0.99 ± 0.13, secondary forest=1.74 ± 0.08, old-growth forest = 2.48 ± 0.09; marginal R^2= 0.57, conditional R^2 =0.76; Fig. 2). For live carbon, the best model included both habitat type and elevation along with an interaction term between habitat type and elevation. This suggests that differences in live carbon stock across habitat types increased with elevation (coefficient estimates ± SD, farmland= 0.01 ± 0.11, secondary forest=1.39 ± 0.08, old-growth forest = 2.34 ± 0.09, elevation= 0.06 ± 0.14; marginal R^2= 0.81, conditional R^2 =0.85; Appendix S1: Fig. S2 (a)). Dead carbon stock showed no significant difference across habitat types (coefficient estimates ± SD, farmland= 1.10 ± 0.13, secondary forest=1.14 ± 0.08, old-growth forest =1.58 ± 0.09; marginal R^2 = 0.19, conditional R^2 =0.68; Appendix II: Fig. S2 (b)).

4.2 Variation in carbon stocks with fallow period

Total carbon stock increased exponentially with fallow age (coefficient estimate ± SD = 0.04 ± 0.01, Marginal R^2= 0.37, Conditional R^2 = 0.64; Fig. 2), with 30-year old fallow sites retaining 56.1% of the carbon stock (7.44 ± 0.32 Mg C/0.03 ha) recorded in old-growth forest (13.24 ± 1.80 Mg C/0.03 ha). Live carbon stock showed a similar trend (coefficient estimate ± SD = 0.05 ± 0.01, Marginal R^2 = 0.51, Conditional R^2 = 0.65; Appendix II: Fig. S2 (a)), but fallow age was not significantly associated with the amount of dead carbon (Appendix II: Fig. S2 (b))

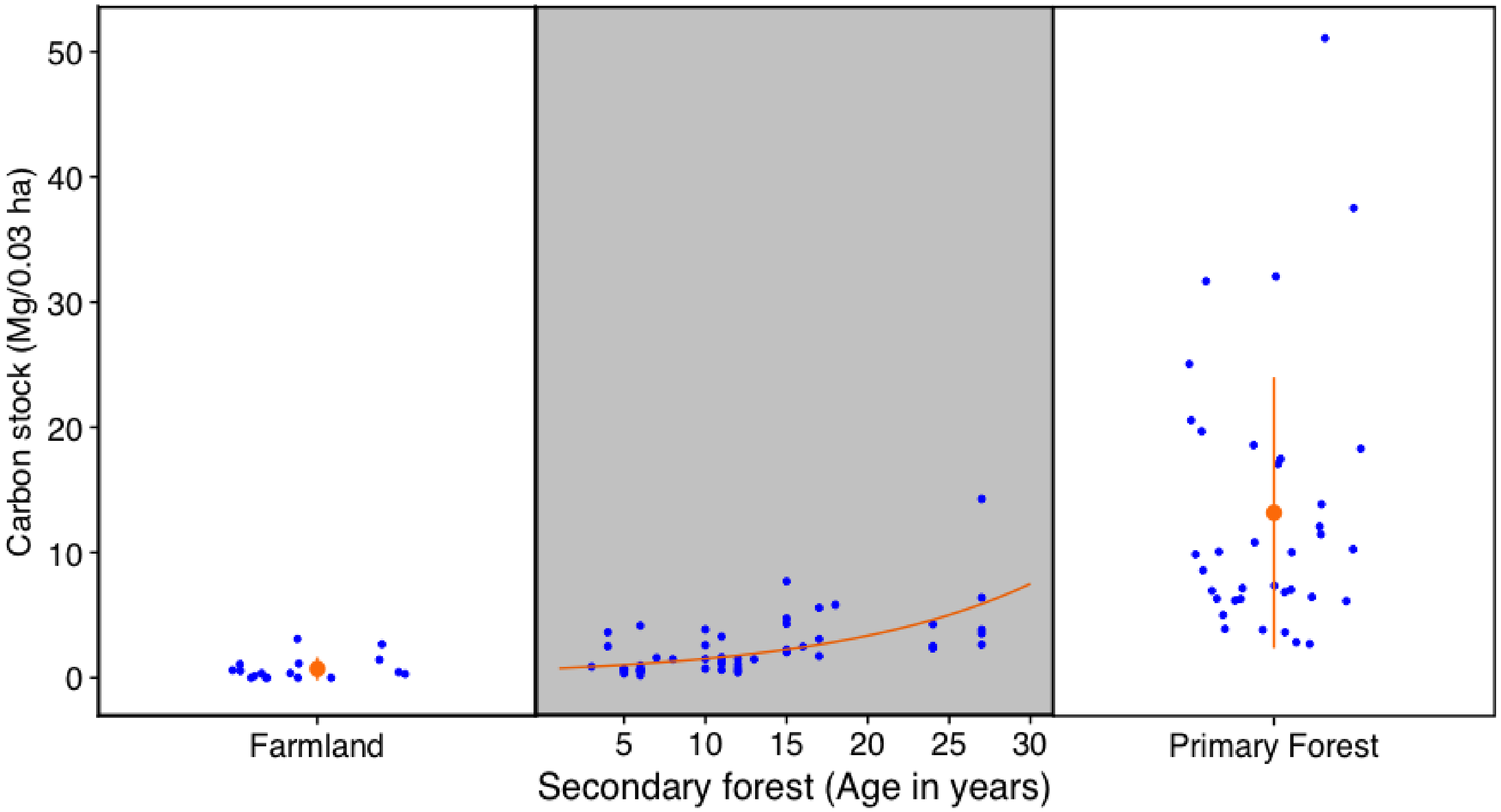


Figure 2. Plots showing total carbon accumulation across the three habitats, farmland, secondary forest with age, and old-growth forest plots in Nagaland, Northeast India. Black line in secondary forest (age in years) shows fitted linear mixed effect model.

4.3 Landscape-level carbon under alternative management scenarios

Under the no forest sparing scenario (Scenario 1), carbon stocks reduced by 56.3%, 64.8% and 71% from the 30-year baseline of 2699.7 ± 378.6 Mg C/30 ha (mean ± SD) in a 15, 10 and 5-year cycle, respectively (Scenario 1.1, 1.2 and 1.3; Fig. 3). Under the second business-as-usual scenario of shifting cultivation expansion (Scenario 4), 79.6% of the carbon stocks in the original old-growth forest landscape (13261.7 ± 1799.7 Mg C/30 ha [mean ± SD]) is lost.

In landscapes with already shortened fallow cycles (Scenario 1.3), REDD+ interventions can be applied for enhancement of carbon stocks by converting it from short to long fallow system. For instance, converting the 5-year fallow cycle to 10-, 15-, and 30-year cycles (from Scenario 1.3 to Baseline Scenario, Fig. 1) enhanced landscape carbon by 21.8%, 51.3%, and 246.4%, respectively. Applying REDD+ style interventions by secondary forest creation and sparing (Scenario 2) also increased carbon stocks substantially. Relative to a 30-years baseline landscape, these interventions increased carbon stocks by 46.4%, 77.8%, and 112.4% in 15-, 10-, and 5-year cycles, respectively (Scenario 2.1, 2.2, and 2.3, respectively; Fig. 3).

In pioneer shifting cultivation landscapes, intervention by old-growth forest sparing (Scenario 3; Fig. 3) reduced substantial carbon loss compared to the complete conversion of old-growth forest to a shifting cultivation landscape (shifting cultivation expansion, Scenario 4). Sparing 50% of old-growth forest (Scenario 3.1) reduced carbon loss by 83.6% relative to a landscape managed entirely as shifting cultivation with a 15-year cycle (Scenario 1.1). Similarly, protecting 83% of old-growth forest (Scenario 3.3) reduced carbon loss by 93% relative to a landscape managed entirely as shifting cultivation with a 5-year cycle (Scenario 1.3).

Overall, intervention by old-growth forest sparing (Scenario 3) held the maximum amount of landscape carbon (54.5%, 69.1%, and 84.3% carbon of an old-growth forest landscape in Scenario 3.1, 3.2, and 3.3, respectively), followed by secondary forest creation and sparing (29.7%, 36.1%, and 43.1% carbon in Scenario 2.1, 2.2, and 2.3, respectively). Maintaining a longer fallow cycle at 30 years also retained considerable amount of landscape carbon (20.3%). REDD+ intervention to convert from a short to long cultivation cycle sequestered the least amount of carbon (Scenario 1; 8.9% and 7.2% in 15- and 10-years cultivation cycle, respectively) when compared to an old-growth forest landscape.

The above estimates of changes in landscape-level carbon stocks are calculated at 30 years following the intervention. Carbon stocks showed similar patterns but less clear differences across scenarios after 5 years of management changes (Appendix II: Fig. S3). Intervention scenarios of old-growth forest sparing retained the highest amount of landscape carbon followed by secondary forest creation and sparing after 5 years (See Appendix II: Changes in carbon stocks five years after management changes for more details).

To test if high carbon estimates for old-growth forests have resulted in an overestimation of the benefits of sparing old-growth forest relative to those of secondary forest creation and sparing, I reran the simulations replacing our randomly selected primary forest carbon estimates with the median carbon estimates (which is a more conservative estimate being lower than the mean value and thus the value typically used in the random selection process) and with estimates from three comparable published studies that report lower values, that is (Mukul et al. 2016a) (321.29 Mg/ha), (Joshi et al. 2013) (355.09 Mg/ha), (Zhang et al. 2013) (376.6 Mg/ha). These studies were selected for comparison as the carbon stocks were estimated from old-growth forests in (sub-) tropical mountainous regions in Asia with minimal anthropogenic disturbances, which is similar to our study system. Simulation results (Appendix II: Fig. S5) show that even with the more conservative estimate (median instead of mean) of primary forest carbon from our study and estimates from other comparable studies, our conclusions on the most optimal scenarios under REDD+ do not change.

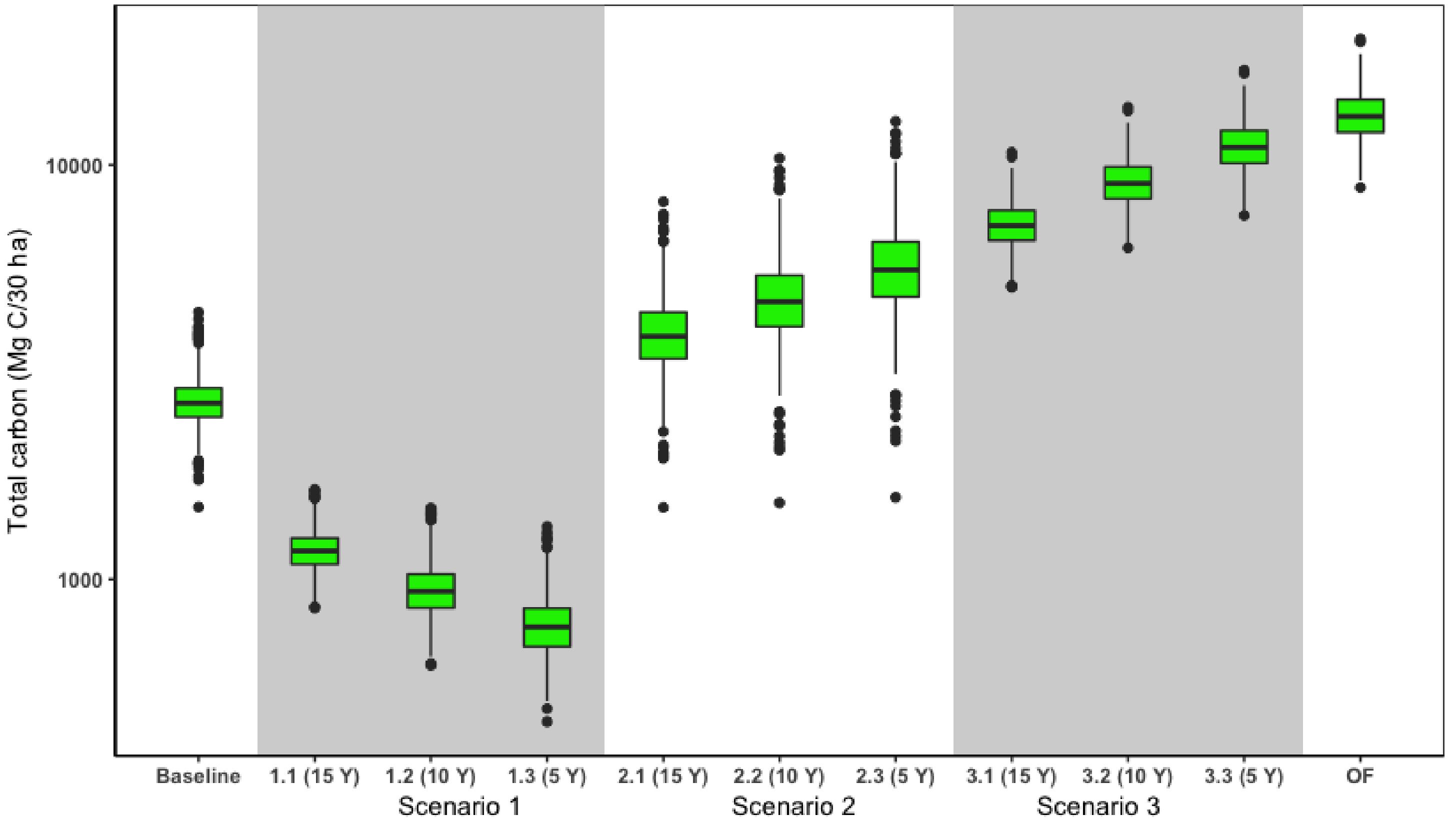


Figure 3. Boxplots showing the difference in landscape level carbon stock under three alternative management regimes of shifting cultivation at the end of 30 years relative to a baseline of 30-year cultivation cycle (Baseline) and old-growth forest landscape (OF): (1)'Business-as-usual' with no forest sparing in Scenario 1 (Scenario 1.1, 15-year cycle; Scenario 1.2, 10-year cycle; Scenario 1.3, 5-year cycle) ; (2) intervention scenarios by secondary forest creation and sparing in Scenario 2 (Scenario 2.1, 15-year cycle; Scenario 2.2,10-year cycle; Scenario 2.3, 5-year cycle) and old-growth forests sparing in Scenario 3 (Scenario 3.1, 15-year cycle; Scenario 3.2, 10-year cycle; Scenario 3.3, 5-year cycle).

5 Discussion

Finding an effective way to manage shifting cultivation without adversely affecting crop production is essential for climate change mitigation and biodiversity protection in forest-rich developing countries. Our study suggests that sparing old-growth forests by intensifying cultivation in a smaller area (Scenario 3) is the most optimal strategy under REDD+ in (sub-) tropical forests in mountainous regions. This scenario retained the maximum level of landscape carbon across all the scenarios. In existing shifting cultivation, REDD+ can enhance forest carbon by secondary forest creation and sparing (Scenario 2), which stored almost one-half of the landscape carbon compared to an old-growth forest. Maintaining a longer fallow cycle and moving from a short to long cultivation cycle also retained a considerable amount of landscape carbon (Scenario 1). Each of these scenarios is particularly relevant under the REDD+ mechanism for reducing carbon emission through avoided deforestation (Scenario 3), avoided forest degradation (Scenario 1, from 5-years to 10, 15-, 30-years cycles), and conservation and enhancement of forest carbon stock (Scenario 2). Thus, these scenarios illustrate the strong potential of REDD+ for protecting and enhancing forest carbon in shifting cultivation landscapes.

5.1 Carbon stock across habitat types

Although subtropical forests with diverse vegetation contribute considerably to the world's forest carbon stores (Lin et al. 2012), few studies have quantified carbon stocks in old-growth forests of the subtropics (Ngugi et al. 2014). I show that old-growth forests in our study area held the highest amount of aboveground carbon (441.4 ± 60 Mg C/ha) compared to other habitat types (i.e., farmland and secondary forest). This estimate of old-growth forest carbon is comparable to the carbon estimates reported from old-growth forests of Garhwal Himalayas in India (Joshi et al. 2013). However, old-growth forest carbon estimates from our study area are relatively higher than those reported by other studies from subtropical forests in India (Baishya et al. 2009) and elsewhere (Zhang et al. 2013, Mukul et al. 2016a). The relatively higher carbon estimates in our study can likely be attributed to the low levels of anthropogenic disturbance in the old-growth forests due to the remoteness and inaccessibility of the region, thus avoiding market-driven large-scale forest exploitation. Previous studies from similar sites in India that report lower carbon estimates also reported high levels of anthropogenic disturbances in their old-growth forest sites, including selective logging/timber extraction (Shaheen et al. 2008, Baishya et al. 2009).

Old-growth subtropical hardwood forests with minimal anthropogenic and environmental disturbances can accumulate very high levels of biomass as shown in tropical sites from South-East Asia (McEwan et al. 2011). The relatively undisturbed forests in our study site contained extremely large trees (maximum DBH measured 280.36 cm [measured above the buttress] unlike forests in previous studies where DBH of trees did not exceed 150 cm) (Shaheen et al. 2008, Baishya et al. 2009). Large trees contribute disproportionately to the carbon stock in primary forests (Sist et al. 2014, Hu et al. 2015b) and drive variation in aboveground carbon (Slik et al. 2013). As carbon estimates in steep terrain of montane subtropical forests are still underreported (Venter et al. 2017), our results indicate that old-growth forests with minimal anthropogenic disturbances in this montane region can accumulate substantially high levels of carbon stocks.

5.2 Carbon stock recovery across fallow ages of secondary forest

I found a positive association between fallow period and total carbon stock in regenerating secondary forest, as shown by other studies of recovery in shifting agriculture from tropical forests (Hughes et al. 1999, Read and Lawrence 2003, Pelletier et al. 2012, Chan et al. 2016) and, more generally, by studies of (sub-) tropical land abandonment (Gilroy et al., 2014a; Poorter et al., 2016). Our study also suggests that mature secondary forests reach about one-half of the levels (56%) of aboveground biomass in old-growth forest within 30 years. A similar time frame has been shown in tropical forests of Mexico (Salinas-melgoza et al. 2017), Colombia (Gilroy et al. 2014a), and the Brazilian Amazon (D’oliveira et al. 2011a).

The exponential increase in total carbon across fallow ages in our study can be influenced by the small-scale mosaic nature of the shifting cultivation landscape. Close proximity of old-growth or mature secondary forest to these fallow sites may help animal-induced seed dispersal (Cole et al. 2010), resulting in increasing rates of forest recovery over time once there has been some regeneration that encourages animals to use the plot. This can create a positive feedback loop with greater recovery leading to increased use by seed dispersing animals that leads to faster recovery. Moreover, regenerating vegetation provides increased protection to the soil from erosion (Tawnenga and Tripathi 1997), which is likely to be particularly important in the study area, which is characterized by steep terrain and high rainfall.

Increased protection from erosion is likely to lead to faster recovery, reducing potential for destabilization of young trees.

5.3 Potential of REDD+ in shifting cultivation landscapes

The growing demands for food production with increasing human population have led to either more frequent rotation in existing shifting cultivation systems or expansion of shifting cultivation into old-growth forest in the tropics (Robichaud et al. 2009). I show that both more frequent cultivation cycles and expansion into old-growth forest can reduce landscape carbon substantially. These adverse impacts of shifting cultivation make it crucial to implement conservation intervention such as REDD+ for both carbon and biodiversity conservation. Shifting cultivation is likely to have a relatively low opportunity cost of conserving forest under REDD+ as it is a subsistence-based farming and is mainly practiced in remote regions with limited market access and low crop yields (Borrego and Skutsch 2014). Therefore, REDD+ payments are likely to offset the costs of avoiding deforestation and forest degradation from shifting cultivation at relatively low carbon prices, as found in marginal cattle lands in the Tropical Andes (Gilroy et al. 2014a). This presents an opportunity for REDD+ to provide economically viable financial incentives to effectively manage these landscapes for protecting and enhancing forest carbon stock in shifting cultivation landscapes (Ziegler et al. 2012).

As old-growth forests are the most important terrestrial carbon sink (Pan et al. 2011) and harbor rich biodiversity (Gibson et al. 2011), including our study area within the Indo-Malayan global biodiversity hotspot and Eastern-Himalayan Endemic Bird Area, restricting further expansion of shifting cultivation to such forests would protect significant conservation values. I show that sparing old-growth forests as protected areas by intensifying cropping in a smaller area (Scenario 3) will be the most optimal strategy under REDD+ for carbon storage. Research from other tropical regions also suggests the importance of sparing old-growth forest matched within more intensive farming (Gilroy et al. 2014b, Luskin et al. 2017). Given the likely economic viability of REDD+ within shifting cultivation (Mertz 2009), this suggests the potential for substantial biodiversity protection within our biodiverse study region as a free co-benefit from protecting carbon stocks under REDD+ (Gardner et al. 2012b, Gilroy et al. 2014a).

In existing shifting cultivation landscapes without any old-growth forest, secondary forest creation through regeneration by increasing rotation frequency in a smaller area (Scenario 2) is the next most optimal pathway for REDD+ investment. As regenerating secondary forests store substantial carbon stocks (Bongers et al. 2015) and often harbor rich biodiversity (Gilroy et al. 2014a, Sayer et al. 2017b), this could provide co-benefits for both carbon and biodiversity (Gilroy et al. 2014a; Jantz et al., 2014; Pandey et al., 2014). However, such benefits may change seasonally given that in winter, Himalayan farmland is more diverse than is forest (Elsen et al. 2017).

Intensive cropping in a smaller area for sparing old-growth forest (Scenario 3) and secondary forest creation (Scenario 2) can be plausible by adopting farmer's innovations in soil conservation, crop and fallow management (Chapter 2). For instance, farmers in Khonoma village in Nagaland pollard N-fixing trees such as *Alnus nipalensis* instead of clear cutting, thus facilitating fallow and soil recovery. This innovation enabled them to produce enough food in a shorter fallow period (mostly every 6 years) compared to earlier long fallow periods. The spared secondary and old-growth forests in this village now constitute the Khonoma Nature Conservation and Tragopan Sanctuary that harbors significant level of biodiversity (Chase and Singh 2012).

Across the entire cropping area, REDD+ can also provide financial incentives to maintain a relatively longer fallow cycle (baseline scenario with 30-years cycle) or transform back from a short to long fallow cycle (5- to 10-, 15-, or 30-years cultivation cycles in Scenario 1) to avoid forest degradation. Such carbon enhancements have shown similar positive outcomes in South-East Asia, where many countries still prioritize replacing shifting cultivation with alternative land uses (e.g., cash crop plantations) of lower carbon and biodiversity values (Ziegler et al. 2012).

For successful implementation in shifting cultivation landscapes, REDD+ should however carefully consider the risks associated with intensification such as permanent loss of soil productivity due to frequent rotation, nitrate contamination due to excessive fertilization (Pei et al. 2015) and potential trade-offs between ecosystem service and farmer well-being (Rasmussen et al. 2018). To mitigate these risks, REDD+ needs to adopt sustainable intensification and strategic land-use planning that maintain the multi-functionality of the entire landscape mosaic in terms of water, nutrients, energy, carbon, and biodiversity. Promoting farmer's innovations for soil and crop management, utilizing crop varieties and livestock breeds with a high ratio of productivity and harnessing agro-ecological processes such as nutrient cycling, biological nitrogen fixation, allelopathy and predation can ensure such positive outcomes (Rockström et al. 2017). As these actions may involve additional costs such as extra labour costs, REDD+ needs to provide incentives to farmers for adopting sustainable intensification techniques. Providing equitable access to knowledge and resources such as land tenure, common property and markets and avoiding negative social and cultural impacts (Campbell et al. 2014) will be beneficial. These can be achieved through community development, social protection schemes and formation of institutions of farmers (Rockström et al. 2017).

To implement REDD+ within shifting agriculture landscapes in this region, it would be advisable to learn from the Khasi Hill Community REDD+ project (the first REDD+ project in India), which has aimed to reverse deforestation and degradation through forest protection and restoration measures in Meghalaya, Northeast India (Sun and Chaturvedi 2016). More generally, India has implemented several policies to reduce deforestation and forest degradation including community forest management, protected area management, and afforestation programs (Murthy et al. 2013c), with the Green India Mission focusing on protecting and enhancing both carbon stocks and biodiversity to avert climate change (Ravindranath and Murthy 2010). Learning from the successes and failures of these policies and from established REDD+ readiness activities and protocols (e.g., capacity building and carbon stock assessment) will likely facilitate optimal implementation.

While interpreting the scenario results, it is important to consider two key limitations of this study. First, the scenarios assume that a reduction in cropping area will not reduce crop yield as per hectare yields can be increased by adopting various crop management options. Previous studies from the study region have shown that similar levels of crop yield can be maintained in a smaller area by nutrient supplementation (Tawnenga et al. 1997), optimizing crop choice (Toky and Ramakrishnan 1981b), and improved fallow management (Grogan et al. 2012). Second, I did not account for varying opportunity costs of different REDD+ interventions. Although crop yield is assumed to remain constant, other aspects of opportunity costs, such as labor input and timber revenues, may differ depending on whether older fallows or old-growth forests are spared from shifting cultivation under REDD+ (Scenario 2 or 3 respectively). Similarly, carbon prices may also vary depending on whether existing carbon is saved by avoiding deforestation or degradation (e.g., sparing old-growth forest in Scenario 3) or enhanced by moving from short to long fallows (e.g., Scenario 1). However, many areas dominated by shifting cultivation are remote for large-scale timber and crop markets, while prices may vary, it is highly likely that they would remain low compared to the opportunity costs in less remote areas of the tropics (e.g., Borneo (Fisher et al. 2011), Indochina (Warren-Thomas et al. 2018)). Moreover, protected areas are unlikely to avoid all degradation and deforestation, so protection may reduce carbon loss to a slightly smaller extent than suggested by our models. Any such reductions in carbon savings seem likely to apply similarly to protection of old-growth and secondary forest. In addition, REDD+ interventions that work effectively with, and are supported by, local communities with an appropriate level of enforcement can be effective in preventing deforestation and degradation (Hayes and Persha 2010, Danielsen et al. 2011).

For successful implementation of REDD+, effective mechanisms to quantify reduction in carbon emission and carbon payments are prerequisites. Moreover, policymakers should also consider information on biodiversity distribution and threats to achieve carbon and biodiversity co-benefits while prioritizing areas for REDD+ projects (Gardner et al. 2012b). It is also important to secure land tenure, reform market policies to create market opportunities for farmers, and to organize training and community activities for active participation of local community in REDD+ (Thrupp et al. 1997c).

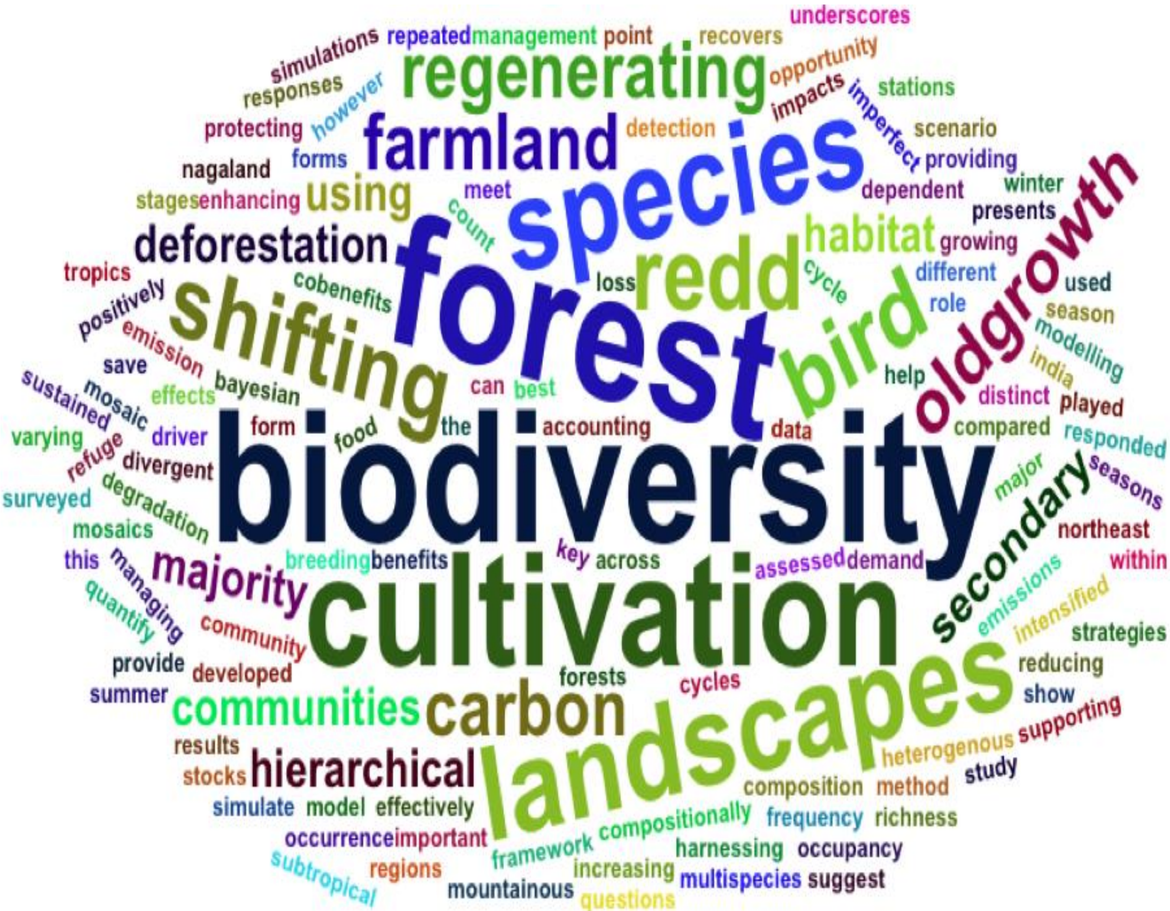
6 Conclusion

Shifting cultivation continues to be widely practiced in many remote montane regions of the (sub-) tropics, which also harbor much old-growth forest and biodiversity. The expansion of shifting cultivation into forests and the permanent transition of shifting cultivation into more intensive land-use systems both drive substantial carbon emissions and biodiversity loss. I suggest explicit pathways for implementing REDD+ to reduce deforestation and forest degradation from shifting cultivation, and successful implementation of these interventions will also likely provide cobenefits such as biodiversity conservation, provisioning of other ecosystem services, and sustainable rural development (Phelps et al. 2012a, Gilroy et al. 2014a, Mukul et al. 2016b). There is thus an urgent need to work with shifting cultivators through capacity building programs to implement these conservation strategies and to enable farmers to meet their production needs in a smaller area of land. Particularly fruitful in generating the income required could be the emerging

Bonn Challenge agenda for Forest and Landscape Restoration, and also India’s new tax revenue distribution reform (Busch and Mukherjee 2018).

Chapter 4

The roles of REDD+ in biodiversity recovery following shifting cultivation in Northeast India



1 Abstract

Shifting cultivation is a major driver of deforestation and biodiversity loss in the tropics. Increasing frequency of cultivation cycles to meet the growing demand for food has intensified its impacts on carbon emission and biodiversity. Reducing emissions form deforestation and forest degradation (REDD+) presents an opportunity to provide co-benefits for biodiversity while protecting and enhancing carbon stocks in shifting cultivation landscapes. Key questions are how biodiversity recovers and what forms of REDD+ would best save biodiversity within shifting cultivation landscapes. I assessed bird species richness and community composition using a Bayesian hierarchical modelling framework in Nagaland, Northeast India, and used these data to simulate responses of bird communities to divergent management strategies under REDD+. Bird communities were surveyed in summer and winter during 2015-16 in 108 stations across farmland, different stages of regenerating secondary forest and old-growth forest using repeated point count method. I developed a hierarchical multi-species occupancy model to quantify the habitat effects on bird occurrence while accounting for imperfect detection of species. The results show that majority of the species responded positively to farmland and regenerating secondary forests compared to an old-growth forest. However, old-growth forest was compositionally distinct in the breeding season supporting forest dependent species. Scenario simulations suggest that mosaic landscapes with farmland and regenerating forest at varying cultivation cycle sustained majority of the species in both seasons. This study underscores the important role played by the shifting cultivation landscapes as refuge for biodiversity providing heterogenous habitat mosaics. Effectively managing these landscapes under REDD+ can help in harnessing both carbon and biodiversity benefits in subtropical mountainous regions.

2 Introduction

Shifting cultivation is a dominant land-use regime in 64 developing countries (van Vliet et al. 2012, Li et al. 2014) across an area of 2.6 million km^2 in the tropics (Silva et al. 2011). It is a major driver of tropical deforestation and forest degradation (Bruun et al. 2009, Ogedegbe and Omoigberale 2011, Borah et al. 2018). As populations and thus food demands have expanded in shifting agricultural areas, there has been both continued expansion of shifting agriculture into old-growth forest and an increase in the frequency of shifting cultivation cycles of forest clearing and cropping (i.e. shortening rotation time) resulting in incomplete recovery of secondary forest vegetation (Grogan et al. 2012). Shifting cultivation thus modifies the landscape into a mosaic of remnant old-growth forests, regenerating secondary forests and farmland, affecting biodiversity (Raman et al. 1998). With shifting cultivation extensively transforming tropical and subtropical landscapes, it is crucial to (1) assess the impact of this cultivation system on biodiversity and (2) formulate strategies for protecting biodiversity in such landscapes.

The Reducing Emissions from Deforestation and Forest Degradation program (REDD+), initiated by United Nations Framework Convention on Climate Change (UNFCCC) presents an opportunity to protect biodiversity in shifting cultivation landscapes by protecting old-growth and secondary forests from further clearing (Borah et al. 2018). Because tropical forests harbour a significant proportion of global biodiversity (Pimm and Raven 2000) and are most threatened by deforestation and degradation (Gardner et al. 2012a), REDD+ has the potential to simultaneously protect biodiversity and carbon in such landscapes. Possible co-benefits for biodiversity from REDD+ are especially important given the lack of direct funding available for the conservation of biodiversity (Waldron et al. 2017). Effectively managing shifting cultivation landscapes under REDD+ will also contribute significantly in achieving the Aichi biodiversity targets as it integrates biodiversity values into development and poverty reduction strategies (Target 2), provides incentives for the conservation and sustainable use of biodiversity (Target 3) and reduces deforestation and forest degradation (Target 5). Moreover, adopting farmer's innovations with modern techniques REDD+ can facilitate integrating traditional knowledge and indigenous practices of local communities in biodiversity conservation strategies (Target 18).

Previous research suggests that sparing old-growth forests from deforestation and intensifying cropping in the remaining area of shifting cultivation under REDD+ can protect maximum levels of landscape carbon, and that in areas lacking old-growth forest, substantial carbon stocks can be stored by sparing old fallows for permanent forest regeneration (Borah et al. 2018). Maximising landscape-level carbon protection and accumulation via REDD+ interventions in shifting cultivation poses an opportunity to also conserve biodiversity as a co-benefit under REDD+ (Gilroy et al. 2014a, Deere et al. 2018). However, understanding the patterns of biodiversity recovery under different REDD+ strategies is a critical knowledge gap.

Species richness is typically lower in forests recovering from shifting cultivation than old-growth forests (review of 24 studies; (Scales and Marsden 2008), but not all taxa show consistent responses. Bird species richness, for example, follows this pattern in the Asia-Pacific region (Thiollay 1995, Raman et al. 1998, Zhijun and Young 2003) and Amazon (Andrade and Rubio-Torgler 1994, Borges 2007), yet avian biodiversity was higher in recovering secondary forest than in primary forest in Costa Rica (Blake and Loiselle 2016) and Indonesia (Jones et al. 2003). This inconsistency could in part arise because studies vary in their approach in assessing biodiversity, particularly in incorporating imperfect detection, which is likely to underestimate species richness across habitats (Mackenzie 2005).

Determining the optimum strategy for biodiversity conservation in shifting cultivation landscapes also depends on the rate of biodiversity recovery as secondary forest ages. Studies suggest that secondary forests took at least 48 years to reach 80% of the species richness of old-growth forest, but with variation across taxa and locations (meta-analysis; Norgrove and Beck 2016). This average recovery time is much longer than the typical shifting cultivation cycle, which traditionally is around 20-30 years in most regions, but has frequently been reduced to less than five years due to increasing human population densities and associated demand for agricultural land (Schmidt-Vogt et al. 2009). REDD+ can potentially help protect old-growth forest from conversion to shifting cultivation or abandon older fallows permanently to maximise recovery of secondary forest – with these two contrasting approaches trading off against each other (Borah et al. 2018). The optimal strategy for conserving biodiversity will depend on whether priority species respond to the shifting cultivation cycle in a similar manner to the broader biodiversity measures assessed by previous work (e.g. (Raman et al. 1998), and on the precise rate and pattern of biodiversity recovery as secondary forest ages (Prateep and Wangpakapattanawong 2017).

Here I assess how bird species occurrence and community composition recover across fallow ages in a shifting cultivation dominated landscape of Nagaland, Northeast India. I then use these species-level results to uniquely simulate divergent management scenarios under REDD+ to quantify which land-use strategies would be most optimal in protecting landscape-level species occurrence and community composition in such landscapes. Nagaland is of critical importance for global biodiversity conservation (Myers et al. 2000) and shifting cultivation occupies nearly three quarters of agricultural area in the state (Pareta 2013). Nagaland provides a strong potential for climate change mitigation and likely co-benefits for biodiversity with an estimated high emission mitigation potential under REDD+ (Murthy et al. 2013b). Birds are important pollinators, predators and seed dispersers (Sekercioglu 2012) and thus are key indicators of ecosystem resilience to land-use (Barlow et al. 2007, Edwards et al. 2014b). Therefore, they are ideal in assessing biodiversity recovery following shifting cultivation and how REDD+ would affect landscape-level conservation values.

3. Methods

3.1 Study area

I sampled bird communities in three districts (Kiphire, Phek and Kohima) in Nagaland, Northeast India across an altitudinal range of 1487-2652 m asl (see Appendix II:Table S1 for plot details). These landscapes are within the Indo-Burma global biodiversity hotspot and specifically are part of the ‘Fakim Wildlife Sanctuary and Saramati area’ Important Bird Area (#IN421 Birdlife International, 2001). The major forest types of the sampling sites were subtropical broad-leaved wet hill forests (500 m-1800 m), subtropical pine forests (1000 m-1500 m) and montane wet temperate forests (>2000 m) (Champion and Seth 1968). Annual rainfall varies from 1800 mm to 2500 mm (Statistical Handbook of Nagaland, 2013). Shifting cultivation occupies 71.2% of the total agricultural area in Nagaland (Pareta 2013) with a fallow period varying from 6 to 27 years (personal observation).

3.2 Sampling framework

I sampled bird communities across three main habitat types during Jan-May 2016: old-growth forest, regenerating secondary forest (3-27 years after plot abandonment) and farmland (plots are cultivated for one or two years before being abandoned). I randomly selected thirty-six 400 m × 400 m sampling squares across the three habitats in each of the three landscapes (15, 12 and nine squares in Kiphire, Phek and Kohima, respectively). Sampling squares were placed at least 300 m apart between different habitats and 400 m apart within the same habitats. Within each square, three point-count stations of 100 m radius were established, spaced 200 m apart from each other (a total of 108 point-count stations across three landscapes).

I sampled birds using repeat-visit point counts at each station between 04:45 and 12:30 avoiding sampling in rain or strong winds. I did so in both the summer breeding season (April-May) and winter (Jan-Feb) when Palearctic migrants overwinter in the region. At each station, four point counts of 10 minutes duration were conducted on consecutive days, resulting in 171 point counts in total for the study (N=108 and N=63 point counts in summer and winter, respectively). At nine of our 171 point-counts, I was only able to make two visits during summer due to the early onset of the rainy season and associated flooding, but I deal with this in our modelling structure. Any bird seen or heard during the point count duration within a 100 m radius was recorded, with care taken to avoid double counting of the same individuals. The entire duration of point count at each station was recorded with a sound recorder (Olympus LS11) to allow unknown vocalisations to be subsequently identified using online reference material (xeno-canto.org) and assistance from regional experts. I randomized the sampling order of the plots to reduce bias due to survey time, while raptors and birds flying over the plots were excluded from the analysis. Nomenclature followed (Jetz et al. 2012) which was compiled from Birdlife International world list (version 3), Handbook of the Birds of the World (de Hoyo et al. 1992-2011) and IOC world list V2.7 (2010).

3.3 Modelling bird diversity across habitats and fallow ages

I modelled species-specific occupancy dynamics in a hierarchical community-modelling framework to examine variation in occupancy probability across habitat types and in relation to fallow age. The hierarchical community modelling approach allows estimation of both species-level and the community-level aggregated effects simultaneously, while accounting for imperfect detection probability (Zipkin et al. 2009). Thus, this approach improves the occupancy estimates of rare species resulting in more precise estimates of species richness by drawing information from community-level effects (Zipkin et al. 2009, Gilroy et al. 2014b). I modelled occupancy and detection for the two seasons (summer and winter) separately as the factors influencing species occupancy and detection vary across seasons (e.g.

behavioural differences in the breeding season and winter). I assume that our survey period for each season (winter-Jan-Feb; summer-April-May) was sufficiently short for the local bird community to experience any extinction or colonization events (i.e. no changes in species occurrence during repeated point counts across sites in each season). I also assumed that all species were correctly identified as 1) audio recordings were exhaustively investigated and compared with existing recordings from the region through xeno-canto 2) all point counts and processing of audio files were done by the same researcher (JRB), thus minimizing any possible bias in observer or skills.

3.3.1 Model specification. - I modelled the heterogeneity in detection across repeated visits in each point count location to estimate detection probability for each species when it is present at a location. I first created an observation matrix $X_{i,j}$ which denotes the number of times species i was detected at location j across K visits. The model then estimates a partially observed matrix of true occurrences $Z_{i,j}$ which indicate whether species i was actually present ($Z_{i,j}=1$) or absent ($Z_{i,j}=0$) at site j. If species i is detected at site j ($X_{i,j} > 0$), its occurrence in the site is known with certainty ($Z_{i,j}=1$), but if it is not detected at site j across K visits ($X_{i,j} = 0$) there are two possibilities: either species i is absent ($Z_{i,j} = 0$) or it is present but not detected ($Z_{i,j}=1$).. This model of true occurrence is specified as a Bernoulli trial where $\psi_{i,j}$ denotes the probability of occurrence of species i at site j.

$$Z_{i,j} \sim \mathrm{Bern}\ (\psi_{i,j})$$

$Z_{i,j}$ is estimated from the observed data $X_{i,j,k}$ for species i at site j during visit k. This observed data is also assumed to be Bernoulli random variables where $\theta_{i,j,k}$ is the detection probability for species i at site j on visit k, if species i is present at site j ($Z_{i,j}=1$).

$$X_{i,j,k} \sim \mathrm{Bern}\ (\theta_{i,j,k} * Z_{i,j})$$

I modelled both occupancy ($\psi_{i,j}$) and detection ($\theta_{i,j,k}$) as functions of species- and site-level covariates on a logit scale. As I want to assess how species respond to variation in habitats, I modelled occupancy probability $\psi_{i,j}$ as a linear function of habitat types. I included elevation (standardised and centred with zero mean) to control for variation in species occurrence across the elevational gradient. Landscape was incorporated as a random effect in the model (thus allowing three separate intercepts for the three landscapes where sampling was conducted) to account for potential autocorrelation in species distribution across landscapes.

Logit ($\psi_{i,j}$)= u_i. landscape$_j$ + $\alpha1_i$. habitat type$_j$ + $\alpha2_i$. elevation $_j$………… (Equation 1)

where habitat type is a factor specifying farmland (N=17), very young secondary forest (≤ 6 years, N=18), young secondary forest (7-14 years, N=18), advanced secondary forest (≥15 years, N=19) and old-growth forest (N=36), respectively; The coefficients $\alpha1$and $\alpha2$ denote the site-level effects of different habitat types and elevation and u_i denotes a random intercept with species- and landscape-level effects.

Similarly, I modelled detection probability θ for species i at sampling site j and visit k using a logit function. I expected detection probability to vary across species, and in relation to the time of point count visits as bird activity level peaks during early morning (Slagsvold 1977). Time was standardized and centred with zero mean before incorporating into the model. I also included weather conditions during the survey in the detection model as weather variations can change avian behaviour, thus influencing detection probability (O'Connor & Hicks, 1980).

Logit ($\theta_{i,j,k}$)= λ_i + $\beta1_i$. time$_{j,k}$+ $\beta2_i$. weather$_{j,k}$ …………….(Equation 2)

I formulated 11 sets of candidate models for occupancy and detection full models (Equation 1 and 2 respectively; see Appendix III: Table S1 and Table S2 for details) for summer and winter data separately. The best model was chosen by comparing Deviance information criterion (DIC; (Spiegelhalter et al. 2002) calculated from the posterior predictive distribution.

The hierarchical component was added to the models by drawing the distribution of species-specific coefficients of the occurrence model ($\alpha1$ and $\alpha2$) from community level 'hyper-prior' distributions. For example, I assumed that the parameter $\alpha1$ follows a normal distribution of mean $\mu_{\alpha1}$ and variance $\partial_{\alpha1}$ (i.e., $\alpha1_i$. ~ ($\mu_{\alpha1}$, $\partial_{\alpha1}$)), where $\mu_{\alpha1}$ and $\partial_{\alpha1}$ are drawn from a community-level normally distributed mean and gamma distributed variance across all species (i.e. $\mu_{\alpha1}$ and $\partial_{\alpha1}$ as hyper-parameters).

3.3.2 Prior distribution and parameter estimation:- I assigned non-informative and diffuse prior distributions for both community and species-specific model parameters to allow posterior distributions to be informed by the actual data rather than model assumptions. I assumed normal (0, 0.001) priors for species and site-level hyper-parameter means, and inverse gamma (0.001, 0.001) priors for hyper-parameter variances. Occurrence probabilities were modelled for a total of 113 species and 52 species in summer and winter community respectively that had been encountered more than 10 times in each season to avoid false convergence in the model due to rare species. I fitted our multi-species hierarchical community model using a Bayesian approach with JAGS version 4.3.0 (Plummer 2003) called from R version 3.5.1 (R Core Team 2018) and r2jags package. I ran the model using five Markov chains with 50,000 iterations, following a burn-in of 30,000 iterations and thinned by 5. I assessed the model convergence with visual diagnostics and the R-hat values for each parameter estimate (Gelman and Hill 2007). I validated model adequacy in describing data using posterior predictive checks (Kery and Schaub 2011) and examined model fit using Bayesian p-value (Gelman 2005) (Appendix III: Table S1 and Table S2).

3.4 Estimating species richness across habitats

I derived species accumulation curves to estimate the species richness of bird communities in each habitat for both seasons separately. This required accounting for potential species in the study area that were not detected during the survey. I used a data augmentation approach to estimate the true species richness (N) for a given habitat based on the observed species richness (n) which is assumed to come from a 'super-community' (S) containing many more species than the true community (N) (Dorazio et al. 2006). Thus, I created this super-community (S = n + nz) by augmenting data matrix ($x_{i,j}$) with all potential species (nz) to the observed species (338 and 399 species to the observed 113 and 52 species in summer and winter, respectively) by adding in 338 and 399 all zero encounter histories in summer and winter dataset, respectively, to include all possible species expected in the study region (total no of recorded species in Nagaland = 451(S);(Saikia and Saikia 2000)). Each of these encounter histories represented an additional undetected species that might have been present in the study region.

I modelled the occurrence of the undetected species by introducing a latent indicator variable (w_i), which is 1 when species i in the super-community is a member of the community available to be sampled and 0 otherwise. The indicator variable (w_i) is assumed to be independent Bernoulli-distributed random variables, with probability determined by a new parameter Ω. Values of w_i are known for all species detected within the survey (i.e. i = 1, 2, …n), but are unknown for the additional undetected species. I derived the true species richness N by estimating Ω as a latent process within the model, with inference based on the posterior distributions of occupancy and detection probabilities estimated from the observed data (Gilroy et al. 2014a).

I then used the fitted occupancy models to generate species accumulation curves for each habitat type. I also accounted for the uncertainty in estimates of the true species richness (N) by sampling the posterior-predictive distribution of the species accumulation curve using the model-derived species occupancy parameters (Dorazio et al. 2006)). A pool of random draws from the posterior distributions of species-level occupancy parameters (ψ_i) is generated for each habitat type. For each draw, I calculated a value of logit (ψ_i) for specific habitats by using the fitted occupancy model (Equation 1). The variation in elevation was incorporated here by randomly drawing elevation values from a uniform distribution of the sampled elevational range (1456 m- 2652 m).

I derived the predicted occupancy probability for species i (ψ_i) for each habitat by computing repeated random draws from the posterior distributions and transforming the resulting values to logit probability. I then used these values to create a predicted occurrence matrix Z with dimensions S * L, where L is the number of hypothetical sampling units within a given habitat. Each value of Z was computed as a single Bernoulli trial with ψ_I, which is the predicted probability of occurrence as drawn randomly for that species in that habitat type. All values of Z for a single set of random draws was summed to derive the number of species occurring in L sites. I produced a posterior-predictive sample of habitat-specific species accumulation curves by generating 1000 random samples of Z across a range of values of L for each habitat type and derived the medians and 95% confidence intervals.

3.5 Bird community composition across habitats

I evaluated bird community composition across the five habitat types (farmland, very young secondary forest, young secondary forest, advanced secondary forest and old-growth forest) by calculating Bray-Curtis similarity indices for the presence-absence species matrix and using nonmetric multidimensional scaling (NMDS) (Legendre and Gallagher 2001). I did so separately for summer and winter data as community composition may vary across seasons. To test for significant differences between habitat types, I employed multivariate analysis of variance via the ADONIS function (Oksanen et al. 2018) at 1000 permutations. I then applied ‘pairwiseAdonis’ function to make multilevel pairwise comparison in community composition across habitat types.

3.6 Landscape simulations under REDD+ management scenarios

To assess how bird species responds to divergent management scenarios under REDD+, I used our data-derived model to predict occupancy probabilities for species across simulated sets of hypothetical landscapes with alternative management systems. Following Borah et al. (2018) I considered two shifting cultivation systems (i) existing shifting cultivation which, at the start, contains farmland and various ages of regenerating secondary forest, but no old-growth forest (Scenario 1 and 2, Fig. 1); and (ii) pioneer shifting cultivation that, at the start, only contain old-growth forest (Scenario 3 and 4; Fig. 1). I built scenarios for each landscape with 30 individual and uniform-sized parcels of land. Each of these were under shifting cultivation (farmland or fallow site), permanently abandoned regenerating secondary forest (Scenario 2 only) or old-growth forest (Scenario 3 and 4).

Scenario 1 provides a scenario without any interventions with landscapes currently used for shifting cultivation. It represents the current trend of decreasing fallow periods to meet growing food demands (no intervention, Fig. 1): original 30-year cycle reducing to 15 years (Scenario 1.1), 10 years (Scenario 1.2) and 5 years (Scenario 1.3). Scenario 2 also applies to a landscape currently used for shifting cultivation. But this has financial incentives to reduce the amount of land used for shifting cultivation, enabling remaining older fallows to regenerate (sparing secondary forest, Fig. 1). Thus, as fallow period declines from 30-year cycle to a 15-year (Scenario 2.1), 10-year (Scenario 2.2) and 5-year cultivation cycles (Scenario 2.3), the older fallows are spared from cultivation by increasing agricultural intensity of a part of the landscape. Scenarios 3 and 4 apply to landscapes originally covered by old-growth forest but converted to a shifting cultivation landscape. Scenario 3 describes the application of conservation interventions, such as protected areas, that limit further clearing of old-growth forest for expanding shifting cultivation (sparing old-growth forest, Fig. 1) with three alternatives: conservation of half, two-thirds and 83% of the old-growth forest in 15-year (Scenario 3.1), 10-year (Scenario 3.2) and 5-year cultivation cycles (Scenario 3.3), respectively. Scenario 4 is when old-growth forest is entirely cleared to create a shifting cultivation landscape, thus providing an additional ‘business-as-usual’ scenario with no REDD+ intervention (shifting cultivation expansion, Fig. 1) (For more details see Chapter 3).

I generated 1000 replicate landscapes under each scenario. Occurrence probabilities for each species were predicted across each landscape by applying the fitted versions of the best model to each simulated site based on the habitat values assigned to each site (see methods 2.3.1). I accounted for uncertainty by randomly drawing from the posterior distributions of model parameters (μ and $\alpha1$ for habitat types, μ and $\alpha2$ for elevation, Equation 1 in methods). The difference in mean occupancy probability for each species in each scenario from the baseline landscape with 30-year cultivation cycle were used to assess how species respond to different management strategies. As I did not collect data for abandoned secondary forests older than > 30 years for Scenario 2 (15-, 10- and 5-year cultivation cycle), I used the model derived species occurrence information for advanced secondary forest (15-30 years) for these habitats, thus generating a conservative estimate of species occurrence for these scenarios.

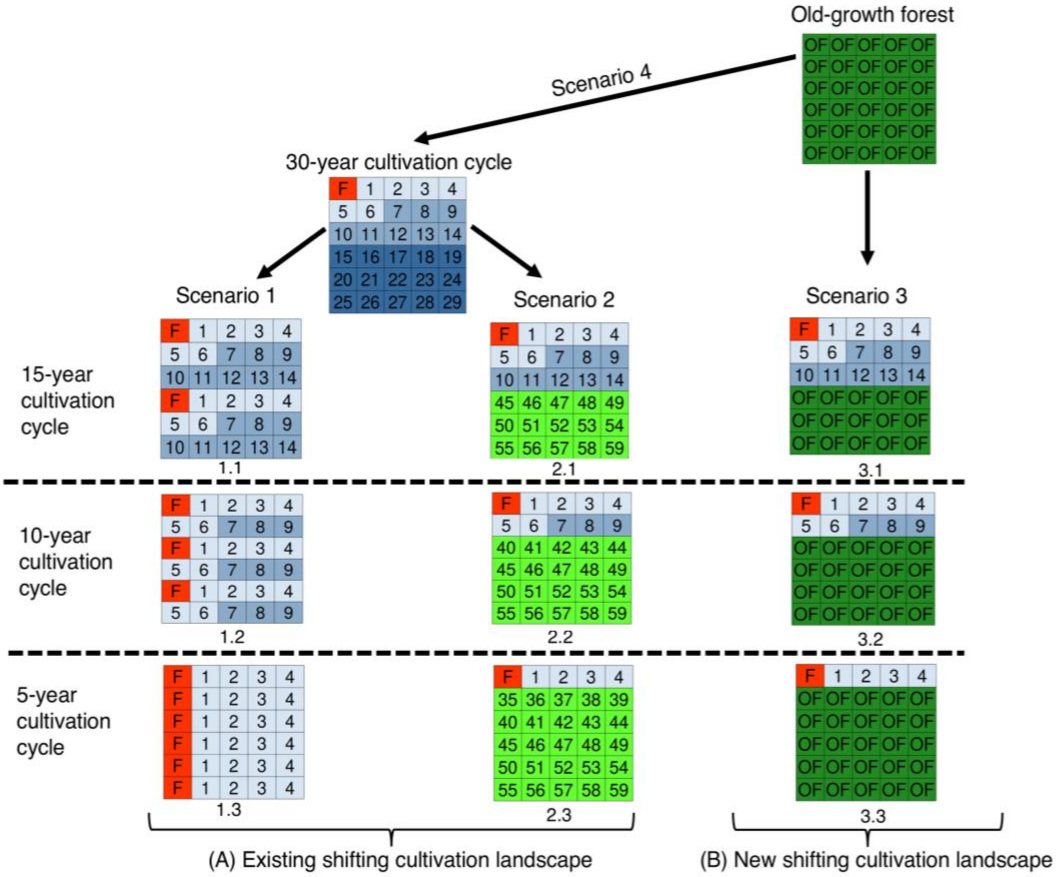


Figure 1. The four sets of management scenarios used to predict changes in bird occurrences in (A) no forest sparing (Scenario 1) and secondary forest creation and sparing (Scenario 2) and (B) new shifting cultivation landscape with old-growth forest sparing (Scenario 3) and shifting cultivation expansion (Scenario 4). Colours indicate habitat types: farmland (F, red), active fallows in different shades of blue, i.e. young secondary forest (1-6 years, light blue), young secondary forest (7-14 years, dark blue) advanced secondary forest (15-29 years, darkest blue) and abandoned old fallows (>30 years, light gree n) and old-growth forests (OF, dark green). Numbers within cells denote the age of the secondary forests.

4 Results

I observed a total of 7790 detections of 281 species across all habitats representing 45 families (257 and 193 species in summer and winter, respectively). Among these, one species was endangered, five were vulnerable, five were near threatened and the rest were least concern in the IUCN red list of threatened species. Occupancy was estimated for a total of 123 species (113 and 52 species in summer and winter, respectively) as the rare species encountered less than 10 detections were excluded to avoid uncertainty in the model prediction.

4.1 Community-level response to habitats

The posterior intervals for the habitat covariates of occupancy in summer and winter contain both positive and negative values (Fig 2, Appendix III: Table S3), suggesting variability in species occurrence in the community. In both summer and winter bird communities, the mean estimates of farmland and very young, young and advanced secondary forests were positive relative to an old-growth forest, but the 95% credible intervals overlapped with zero for all the habitat types except very young secondary forest in winter (Fig. 2). This suggests that occupancy probability on average was lower in old-growth forest than farmland and secondary forests. Within regenerating secondary forests, bird occurrences were similar across very young, young and advanced secondary forest in summer whereas very young secondary forest had on average a higher occupancy probability than young and advanced secondary forest in the winter community.

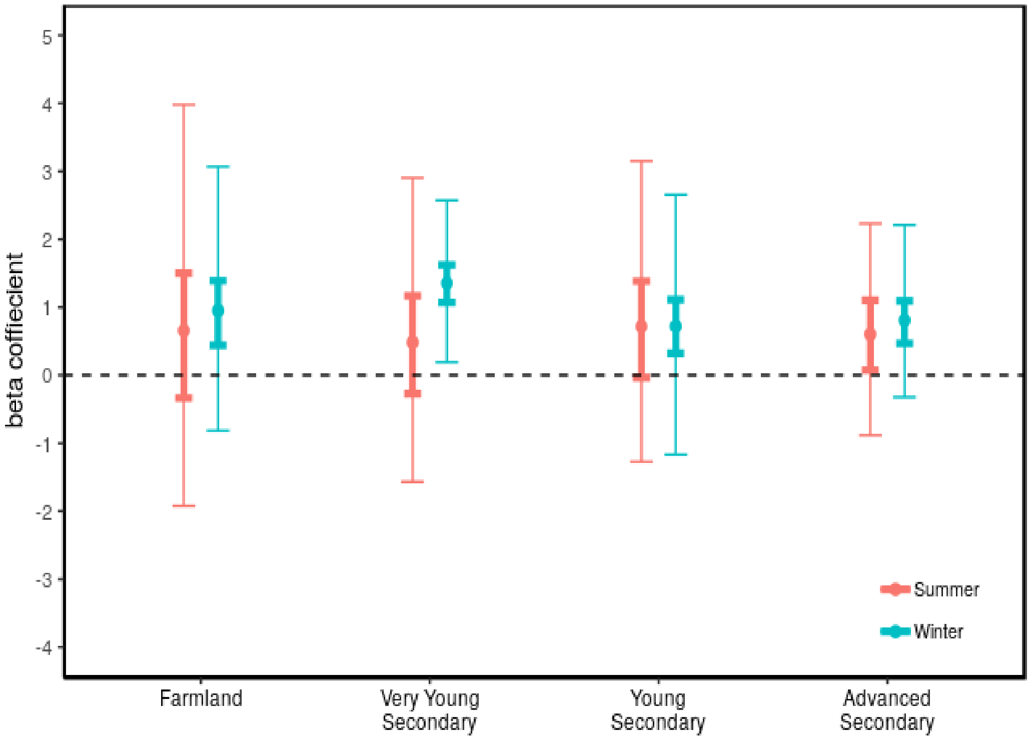


Figure 2. Estimated effects of different habitats on community-level bird occurrence probabilities relative to an old-growth forest (dashed line at zero) in summer and winter showing posterior means, 95% (thin bars) and 50% (thicker bars) Bayesian credible intervals across habitats.

4.2 Bird species richness and community composition

Species accumulation curves derived from the hierarchical model show lower species richness in farmland compared to old-growth forests but similar levels of species accumulation across very young, young and advanced secondary species in summer (Fig 3A, 3C, 3E and 3G). However, species richness was higher in very young secondary forest compared to an old growth forest in the winter bird community (Fig 3D). Similarly, observed species richness was significantly lower in old-growth forest than young secondary forest in summer (Tukey's HSD: $P = 0.002$) and very young secondary forest in winter (Tukey's HSD: $P = 0.002$) (Fig. 4C).

The NMDS ordination plots suggested distinct community composition across habitats in both summer (ADONIS: $r^2 = 0.14$, d.f. = 4, $P < 0.001$) and winter (ADONIS: $r^2 = 0.09$, d.f. = 4, $P = 0.005$) (Fig. 4A and 4B). Species' assemblages were significantly different in old-growth forest from farmland and secondary forests in summer (Pairwise Adonis: old-growth forest vs farmland: $r^2 = 0.13$, $P = 0.001$; vs very young secondary forest: $r^2 = 0.09$, $P = 0.001$; vs young secondary forest: $r^2 = 0.10$, $P = 0.001$ and vs advanced secondary forest: $r^2 = 0.06$, $P = 0.001$). Within secondary forests, species composition did not vary across very young, young and advanced secondary forests in both seasons suggesting that fallow age had little influence on community composition in regenerating forests (Fig. 4D).

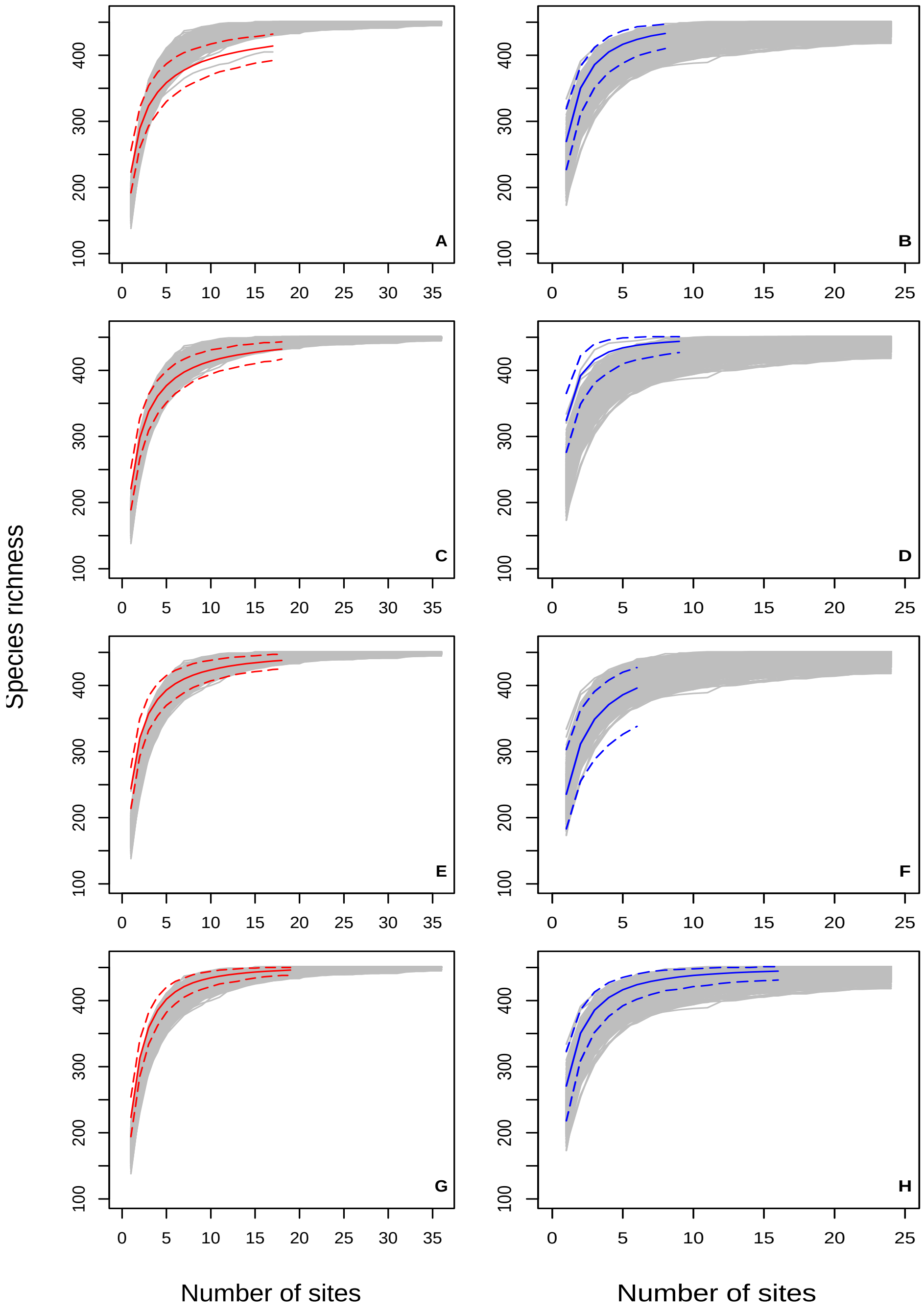


Figure 3. Species accumulation curves across farmland (A and B), very young secondary forest (C and D) young secondary forest (E and F), advanced secondary forest (G and H) with thick lines showing mean and dashed lines showing 95% credible intervals in summer (red lines) and winter (blue lines). Grey shaded area shows model-based predictions of species accumulation in old-growth forest.

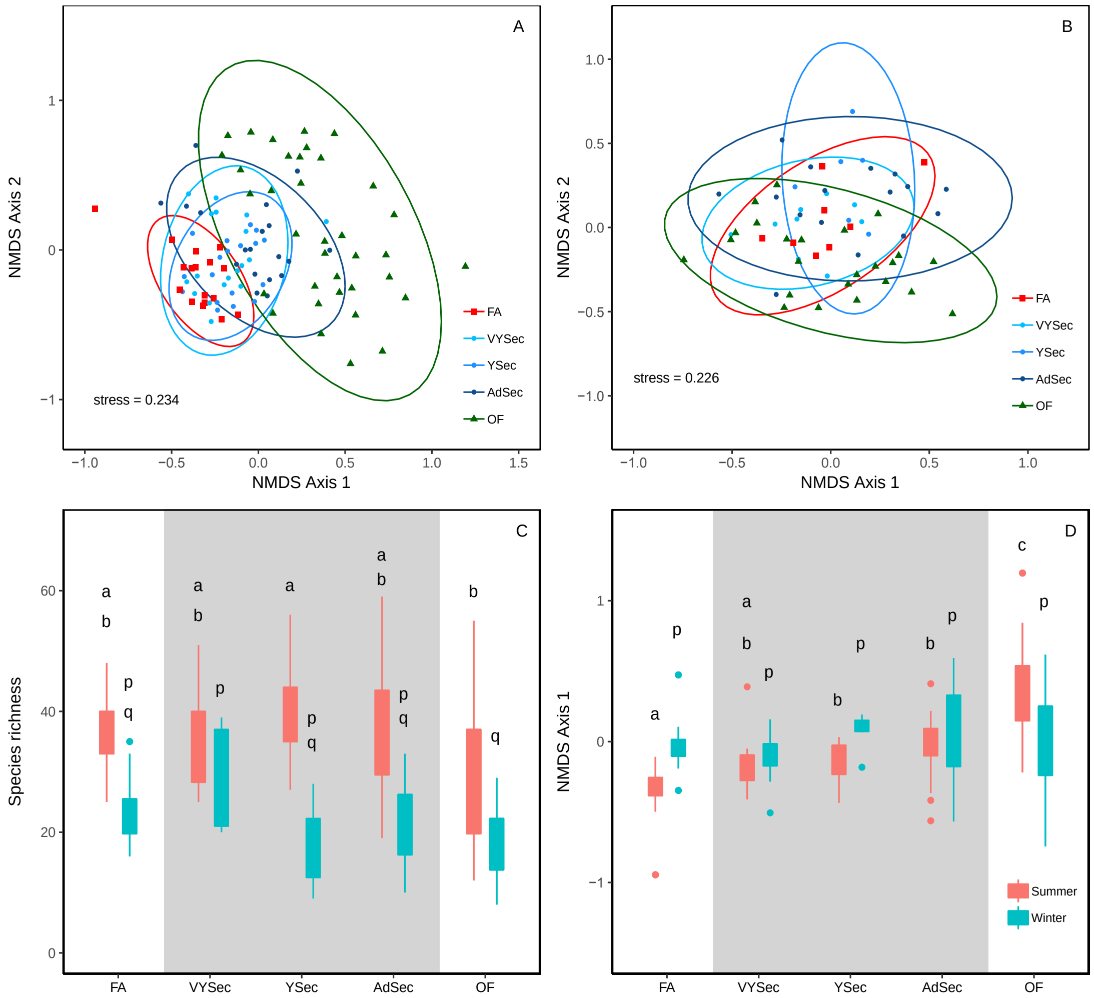


Figure 4. Ordination of point-level community composition using non-metric multidimensional scaling (NMDS) in summer (A) and winter (B) and pairwise comparisons of observed species richness (C) and community composition (D) across farmland (FA), very young secondary forest (VYSec), young secondary forest (YSec), advanced secondary forest (AdSec) and old-growth forest (OF) in Nagaland. Bird communities with different letters (a, b, c and p, q) show significant differences across habitats ($P \leq 0.05$).

4.3 Species-specific responses to habitat covariates

Mean species-level estimates of occurrence probability varied widely, ranging from 0.12 ± 0.15 to 0.88 ± 0.10 in summer and 0.22 ± 0.21 to 0.82 ± 0.15 in winter (Appendix III; Fig S1 and Fig S2). Mean species detection probabilities also showed high levels of heterogeneity (Appendix III: Fig S1 and Fig S2), but overall detection probability was low for majority of the species in both the summer and winter communities (92.9 % and 96.1% of the species had < 50% detection probability in summer and winter, respectively). Farmland had a positive association with 64.6% and 96.1% of the summer and winter bird community, respectively, relative to an old-growth forest (mean of posterior probabilities were higher than zero; Fig. 5; Appendix III: Table S3). Similarly, within secondary forests, 65.5%, 76.1% and 87.6% species responded positively to very young, young and advanced secondary forests in summer (Fig 5). The majority of the species in the winter community also showed preference to secondary forest relative to an old-growth forest (Fig 6). Among the species with higher occupancy probability in farmland and young and very young secondary forests were open habitat species such as Black-throated Prinia, Flavescent Bulbul and Red-vented Bulbul. Similarly, forest specialists such as Chestnut-headed Tesia, Pygmy Wren-babbler and Yellow-bellied Fantail showed considerable increase in occupancy probabilities in advanced secondary forest both in summer and winter (Fig 5 and Fig 6).

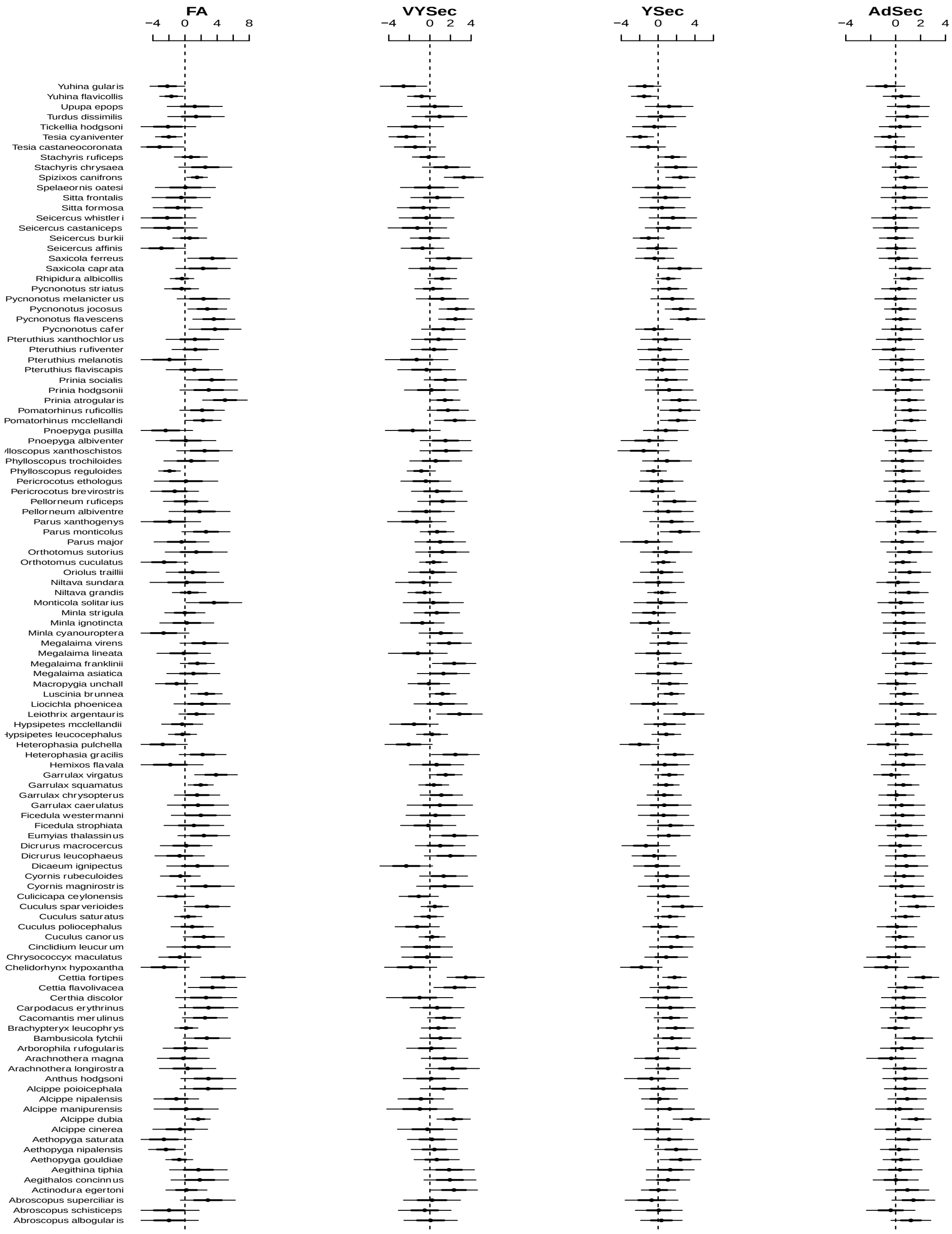


Figure 5. Effects of different habitat types (FA-farmland, VYSec- very young secondary forest, YSec-young secondary forest, AdSec-advanced secondary forest) relative to an old-growth forest (dashed line denoting zero) in the summer bird community in Nagaland. Black bars show 95% credible intervals (CI) with thick bars indicating 50% CI.

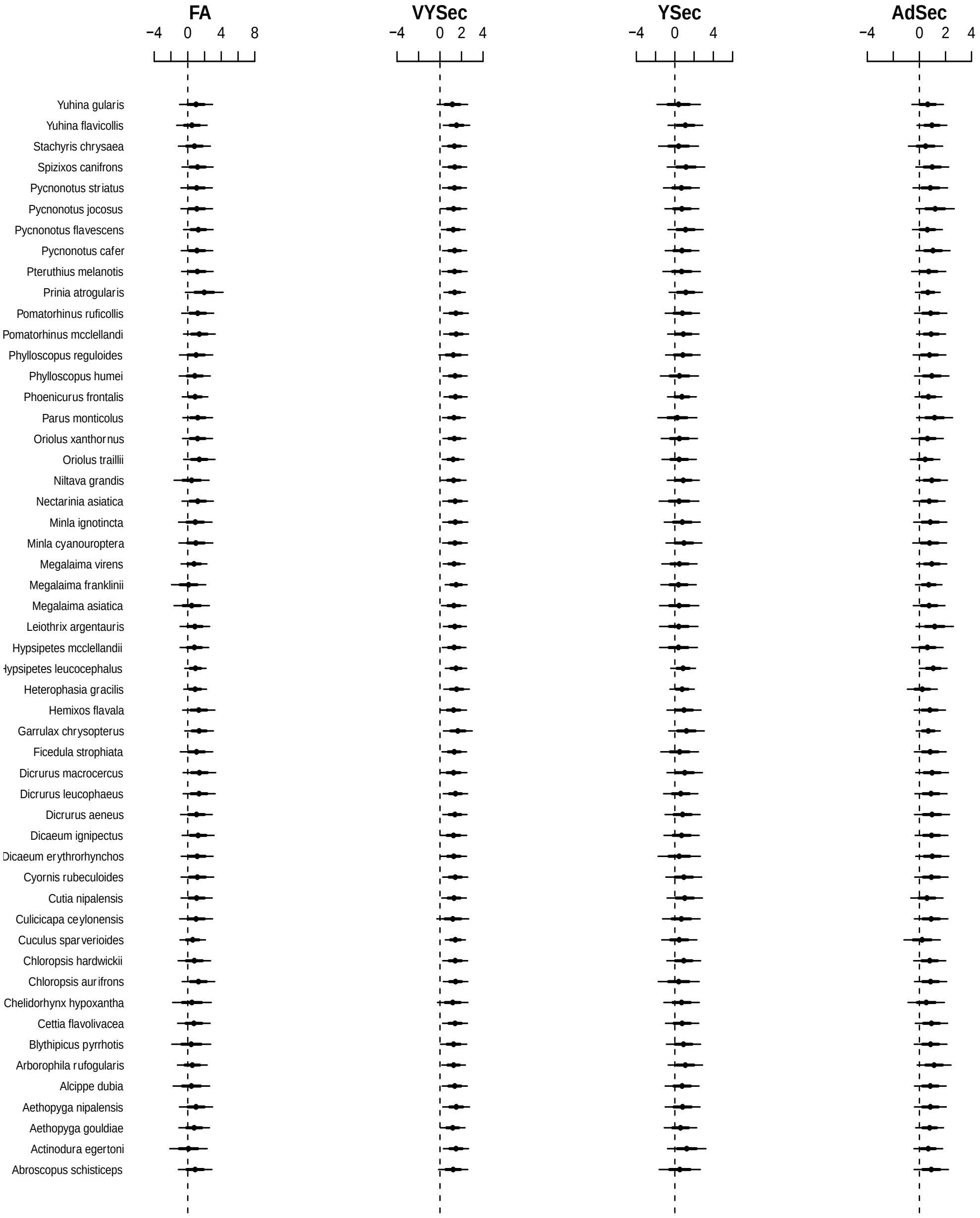


Figure 6. Effects of different habitat types (FA-farmland, VYSec- very young secondary forest, YSec-young secondary forest, AdSec-advanced secondary forest) relative to an old-growth forest (dashed line denoting zero) in the winter bird community in Nagaland. Black bars show 95% credible intervals (CI) with thick bars indicating 50% CI.

4.4 Landscape scenarios under REDD+

The landscape scenario simulations show that occurrences of about half of the summer bird community increased in Scenario 1 with no forest sparing (48.7%, 46.9% and 44.2% of species in 15, 10 and 5-year cultivation cycles) and Scenario 2 with secondary forest creation and sparing (46.0% and 49.6% in 10 and 5-years cycles) relative to the baseline scenario of 30-year cultivation cycle (Fig 7A). This trend was more evident in the winter community with even greater proportion of birds benefiting in Scenario 1 and Scenario 2 (Fig 7B). In contrast, only a small proportion of the summer and winter bird community responded positively to Scenario 3 with old-growth forest sparing and the old-growth landscape in both the seasons.

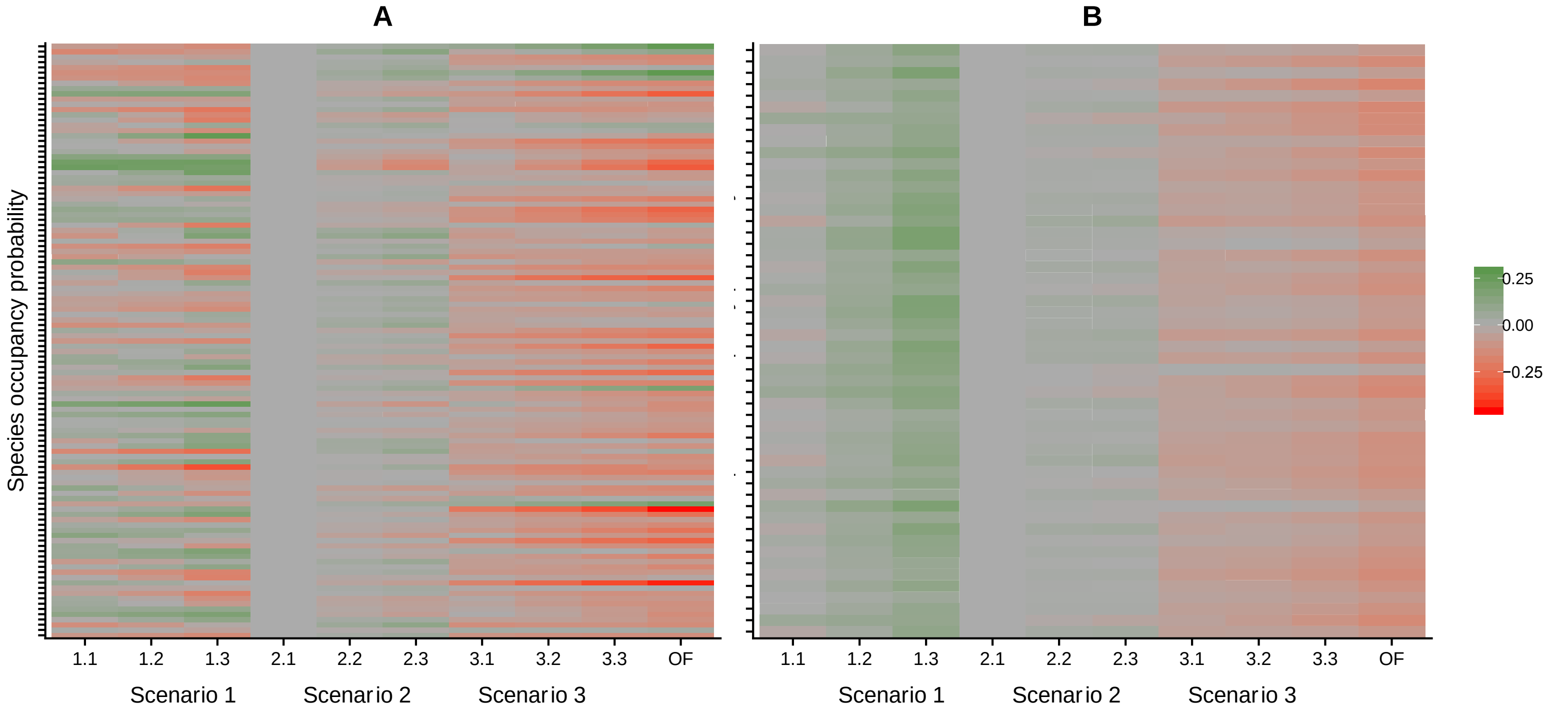


Figure 7. Predicted changes in mean species level occurrences relative to the baseline of 30-year cultivation cycle in summer (A) and winter (B) under the three alternative management regimes of shifting cultivation at the end of 30 years (1)No forest sparing in Scenario 1 (Scenario 1.1, 15-year cycle; Scenario 1.2, 10-year cycle; Scenario 1.3, 5-year cycle) ; (2) intervention scenarios by secondary forest creation and sparing in Scenario 2 (Scenario 2.1, 15-year cycle; Scenario 2.2,10-year cycle; Scenario 2.3, 5-year cycle) and (3) old-growth forests sparing in Scenario 3 (Scenario 3.1, 15-year cycle; Scenario 3.2, 10-year cycle; Scenario 3.3, 5-year cycle). The colour gradient shows mean occupancy probability in each scenario relative to the baseline landscape.

5 Discussion

Understanding biodiversity responses to shifting cultivation is crucial in formulating strategies for biodiversity protection. Although biodiversity response to land-use changes in the tropics has been widely studied, few have examined bird community recovery from shifting cultivation accounting for imperfect detection under diverse management strategies. This study in a biodiversity hotspot region of Northeast India suggests that farmland and regenerating secondary forests sustained higher levels of bird species richness than old-growth forests in shifting cultivation landscapes. However, old-growth forests harboured a distinct bird community with forest dependent species, particularly during the breeding season. Landscape simulations under REDD+ strategies reveal that Scenario 1 with no forest sparing and Scenario 2 with secondary forest creation and sparing were the most optimal strategy for supporting highest number of relatively common species.

5.1 Bird community response to shifting cultivation

This study reveals that farmland and regenerating secondary forests had higher bird occurrences than old-growth forests, which is consistent with studies in multi-strata agroforestry systems in India (Elsen et al. 2017) and elsewhere (Harvey and González Villalobos 2007, Van Bael et al. 2007, Mulwa et al. 2012, Buechley et al. 2015). However, few previous studies in shifting cultivation show contrasting patterns of biodiversity response with old-growth forest supporting higher species richness than farmland and regenerating secondary forest (Raman et al. 1998) . This could be partly explained by the presence of large trees retained by farmers in farmland and young regenerating forests, which is typical of the shifting cultivation landscapes in my study region. These trees can facilitate both open habitat and forest dependent species in farmlands and regenerating forests, thus increasing the species richness in farmland and young secondary forests (Sayer et al. 2017a).

Within secondary forests, I did not find any strong evidence of influence of fallow age on bird occurrence. This is also in contrast to biodiversity recovery pattern in shifting cultivation systems in Northeast India (Raman et al. 1998) and elsewhere (Bowman et al. 1990, Blankespoor 1991b, Zhijun and Young 2003, Marsden et al. 2006) that found an increase in bird species richness, abundance and diversity with increasing successional age. Most of these studies, however, were conducted in relatively lower elevations (<1000 m asl; but see (Zhijun and Young 2003), whereas resource use by bird communities can be different in higher elevations similar to our study region (1460-2600 m asl), influencing their responses to land-use change (Dehling et al. 2014).

Community composition in summer was distinct across farmland, secondary and old-growth forests mirroring findings from previous studies (Raman 2001a). In contrast, winter communities were much more dispersed showing no strong associations with habitats. This is because the majority of birds in the study region undergo seasonal migration (Rasmussen and Anderton 2005) and thus respond more strongly to elevational gradient rather than habitats. Thus, farmland and secondary forests supported a considerable proportion of winter bird communities, some of which are dependent on old-growth forest during the breeding season as evidenced in previous studies (Laiolo et al. 2004, Elsen et al. 2017).

Two major factors that can explain the observed patterns of biodiversity response in shifting cultivation are management effects and landscape configuration (Norgrove and Beck 2016). Shifting cultivation is a mosaic system with farmland, regenerating forest and old-growth forest interspersed with each other providing heterogeneity of habitats. This in turn can increase biodiversity through an accumulation of species associated with the different habitat types in such landscapes (Borges 2007, Devictor and Jiguet 2007). For instance, non-forest and open-habitats birds that colonize farmlands provide additional species richness in farmland and secondary vegetation as has been observed in India (Raman et al. 1998) and Amazon (Andrade and Rubio-Torgler 1994). This can be due to an increase in landscape complementation (Dunning et al. 1992, Brotons et al. 2005), with different habitat types preferred by birds at different times and providing complementary resources, such as food and nest sites during summer and winter. Landscape heterogeneity might also increase biodiversity by influencing interspecies interactions. For instance, increasing landscape compositional heterogeneity can reduce dispersal rates between patches of the same habitat type, that can indirectly increase biodiversity by reducing competition (Fahrig et al. 2011).

In addition, management decisions made by the local communities can influence biodiversity recovery by determining the rate of fallow recovery in shifting cultivation (Borges and Stouffer 1999, Zhijun and Young 2003, Norgrove and Beck 2016). For example, repeatedly cultivated sites had a distinct bird species composition and lower species richness compared with fields cultivated once in the Brazilian Amazon (Borges 2007). Similarly, the various stages of the cultivation process, such as farm size, retaining certain trees while clearing, and distance to old-growth forest also vary across villages. These factors, in turn, can lead to varying degrees of biodiversity recovery across sites.

5.2 Potential of REDD+ for biodiversity conservation in shifting cultivation

REDD+ holds particular potential for providing biodiversity co-benefits while protecting carbon stocks in shifting cultivation landscapes (Gardner et al. 2012a). Tropical and sub-tropical mountainous regions, such as our study site, with high levels of both carbon (Borah et al. 2018) and biodiversity provide an opportunity for harnessing win-win outcomes under REDD+.

The scenario simulations show that Scenario 2 with secondary forest creation and sparing benefited the highest proportion of the breeding bird community in summer whereas majority of the winter bird community persisted in both Scenario 1 with no forest sparing and Scenario 2. This suggests that, in existing shifting cultivation landscape without any remaining old-growth forest, sparing older fallows from shifting cultivation provides benefits for biodiversity while enhancing forest carbon (Borah et al. 2018) and thus is a good candidate for REDD+ investment. This also lends support to findings from previous studies that secondary forest protection can harbour a species rich community and can be effective in protecting significant numbers of endemic species (de Lima et al. 2013).

Although the Scenario 3 where old-growth forest is spared, and the old-growth forest landscape did not support as many species as Scenario 1 and 2, these Scenarios with varying proportion of old-growth forest are essential for survival of forest specialists which are of high conservation concern. Primary forests harbour 88% of the restricted-range species globally (Long et al. 2010), suggesting a link between forest carbon stock and endemic species in many regions across the world (de Lima et al. 2013). This was evident in this study as a significant number of threatened and restricted-range species were predominantly found in the old-growth forest in this study region but were excluded from the simulation analysis as they were rarely encountered. Thus, sparing primary forest from conversion under REDD+ is likely to protect these species while storing the maximum level of landscape carbon (Borah et al. 2018).

Although REDD+ has a strong potential to protect biodiversity by reducing carbon emissions, it needs careful planning and implementation to ensure biodiversity co-benefits. For instance, interventions focussed only on carbon sequestration may prioritise only high carbon areas irrespective of its significance to biodiversity (Gardner et al. 2012a), thus not necessarily delivering significant positive outcomes for biodiversity (Dickson and Kapos 2012, de Lima et al. 2013). In addition, unlike standard carbon estimates irrespective of locations, biodiversity is not distributed evenly across forests and might not be directly comparable across sites making it complicated to assess the contribution of REDD+ targets (Dickson and Kapos 2012). There may also be significant environmental and economic trade-offs while optimising carbon emission reductions with biodiversity co-benefits (Phelps et al. 2012b), such as risks of conversion of natural forest to high carbon plantations or displacement of deforestation and forest degradation (leakage) to low carbon but high biodiversity areas (Gardner et al. 2012a).

To ensure delivery of biodiversity co-benefits under REDD+, it is critical to incorporate biodiversity as an equal priority with carbon stock protection and enhancement in REDD+ planning. Data on biodiversity distribution, threats and its response to land-use change should be assessed while identifying priority areas for REDD+ implementation (Gardner et al. 2012a). Moreover, standard protocols for biodiversity monitoring should be included along with carbon stocks assessment (Dickson and Kapos 2012). Such monitoring efforts need to focus on species of high conservation concern and other diversity measures that incorporate their role in ecosystem functions and evolutionary history such as functional and phylogenetic diversity to get a better understanding of biodiversity responses to land-use change. Similarly, it is crucial to incorporate seasonal variation in biodiversity response as evident from this study and previous findings from elsewhere (Elsen et al. 2017). The existing monitoring efforts can be used for data on biodiversity for monitoring and tracking impacts of REDD+ with minimal additional cost (Dickson and Kapos 2012). REDD+ initiatives should also effectively address and integrate key safeguards during designing and implementing REDD+ to avoid possible negative outcomes for biodiversity (Pistorius et al. 2011).

Northeast India, a biodiversity hotspot with 66% of its total geographical area under forest cover has immense potential in harnessing REDD+ benefits for reducing carbon emission and biodiversity loss from shifting cultivation (Murthy et al. 2013b). The opportunity costs of REDD+ implementation is likely to be relatively low due to limited market access and challenges for adopting alternative livelihoods in these landscapes. However, about 93% of forests in Nagaland are owned and managed by village councils (Bhupathy et al. 2013) and thus vary in their protection status.

Incorporating these old-growth forest areas in protected area networks or co-managing with the local community can be effective in protecting the restricted-range and threatened bird species.

Two key limitations of the study should be considered while interpreting the results. Low detection probability across all habitats limited the hierarchical model from strongly teasing apart different covariate effects, thus influencing the habitat effects on rare species. Low detection probability of bird species can be partly due to the subsistence hunting, which is prevalent in the study region (Bhupathy et al. 2013). Hunting pressure can have a significant effect on species richness as shown elsewhere (Naughton-Treves et al. 2003) as well as bird behaviour influencing detection probability. However, I sampled a large number of replicates across three landscapes with a varying degree of hunting pressure to account for variation in species detection and occurrence caused by hunting. Second, the simulated scenarios assume that crop yield per hectare can be maintained or increased in spite of reducing cropping area by adopting various crop management options such as nutrient supplementation (Tawnenga and Tripathi 1997), optimizing crop choice (Toky and Ramakrishnan 1981a) and improved fallow management (Grogan et al. 2012).

The potential of agricultural landscapes in supporting high levels of biodiversity is increasingly being recognised in recent studies from the tropics and subtropics (Daily et al. 2001, Johnson et al. 2006). Heterogenous agricultural landscapes sustained high biodiversity for various taxa, such as plants, reptiles, amphibians, birds and mammals, in other parts of India (Ranganathan et al. 2010) and elsewhere in the tropics and subtropics (Mendenhall et al. 2016). As protected areas span only 13% of the global land surface, and the level of effective protection also varies across these (Ellis et al. 2010), managing unprotected lands such as shifting cultivation landscapes efficiently can help in mitigating both carbon emission and the current biodiversity crisis (Gardner et al. 2009, Perfecto et al. 2009)

6 Conclusion

Shifting cultivation is a dominant land-use system in developing countries that also harbour much of the world’s terrestrial biodiversity. This study reveals that shifting cultivation landscapes sustain high levels of biodiversity and can be equally effective in biodiversity protection compared to old-growth forests. However, old-growth forests are indispensable in sustaining forest-dependent species of high conservation concern. Successful implementation of REDD+ needs equally prioritising carbon storage and biodiversity co-benefits, monitoring varying biodiversity response across seasons and community management systems, and incorporating safeguards to avoid potential harmful effects on biodiversity.

Chapter 5

Recovery of avian phylogenetic and functional diversity following shifting cultivation in Northeast India

1 Abstract

Shifting cultivation is a major driver of tropical deforestation and biodiversity loss and the rate of forest conversion in many regions are increasing due to declining soil fertility and growing demand for food. Reducing Emissions from Deforestation and Forest Degradation (REDD+) scheme can potentially protect forests and biodiversity while funding for carbon storage and enhancement in shifting cultivation landscapes. A key question is how phylogenetic diversity, the total evolutionary history shared across all species within a community, and functional diversity, the diversity in functional traits, are impacted by shifting cultivation and in turn will be best conserved under divergent management scenarios of REDD+. Focusing on bird communities in shifting cultivation-dominated Nagaland, Northeast India, and using a Bayesian hierarchical analytical framework, I first show that phylogenetic diversity declined from forest conversion to shifting cultivation whereas functional diversity was robust to shifting cultivation impacts. Old-growth forest sustained both phylogenetically and functionally clustered breeding bird community while bird community in farmland and secondary forests were over dispersed. Similarly, evolutionary distinctiveness and evolutionary distinctiveness rarity were also maintained across all habitats. Scenario simulations suggest that maintaining a heterogenous landscape with farmland, regenerating secondary forest and old-growth forest is the most optimal strategy to protect phylogenetic and functional diversity. This underscores the conservation value of shifting cultivation landscapes and potential of REDD+ in achieving both carbon and biodiversity benefits in these regions.

2 Introduction

Shifting cultivation is a dominant tropical land-use regime across 2.6 million km^2 (Silva et al. 2011) in 64 developing countries (Mertz 2009, van Vliet et al. 2012, Li et al. 2014). This cultivation method involves clearing a forest patch by slash and burn followed by cropping and subsequent fallowing on a rotational basis. However, recent trends of increasingly frequent clearing and cropping cycles to meet growing food demands in shifting cultivation have led to deforestation, forest degradation and biodiversity loss (Bruun et al. 2009, Ogedegbe and Omoigberale 2011, Ding et al. 2012, Chapter 2).

Given the overlap between shifting agriculture and hyper-diverse tropical forests, it is crucial to assess biodiversity loss and recovery following shifting cultivation and to find strategies for sustainable management of shifting cultivation landscapes. A few studies have assessed biodiversity recovery patterns in shifting cultivation landscapes showing a loss of species richness with forest clearance followed by increasing richness with fallow age for plants, amphibia and birds in tropical and subtropical forests of Asia (Pawar 1999, Dunn 2004, Lawrence et al. 2005). However, diversity measures such as species richness may underestimate the true extent of impacts of land-use change on biodiversity (Mouillot et al. 2013) as they fail to capture changes in communities' response in terms of ecological functions and evolutionary history (Edwards et al. 2014a).

Phylogenetic (PD) and functional (FD) diversity measures can provide a better understanding of community response than species richness by providing information on the breadth of evolutionary history and ecological functions in a community (Chapman et al. 2018). Functional diversity incorporates functional differences among species based on their ecological and life history traits (Petchey and Gaston 2006). Therefore, it indicates the range of the functional roles played by the species in a community that reflects the biodiversity-ecosystem function relationships (Schmid et al. 2005). Similarly, phylogenetic diversity represents the diversity of evolutionary lineages and associated ecological functions that can contribute to ecosystem stability (Bregman et al. 2014). Therefore, considering phylogenetic and functional diversity responses is essential to gain a complete understanding of the impacts of land-use change on biodiversity (Mouillot et al. 2013).

Previous studies show a substantial loss of phylogenetic and functional diversity with forest conversion to farmland (e.g. Edwards et al. 2014b, Chapman et al. 2018) and subsequent return of phylogenetic and functional diversity as farmland is abandoned and secondary forests regenerate (e.g. Edwards et al. 2017, Sayer et al. 2017). However, the recovery patterns of avian phylogenetic and functional diversity may vary depending on type and intensity of agricultural systems. For instance, diversified agricultural systems with many crop types, diverse vegetation and surrounding tree cover supported higher level of phylogenetic diversity than monoculture plantations (Frishkoff et al. 2014). No previous study has assessed the phylogenetic and functional diversity impacts of shifting cultivation, despite its prevalence pan-tropically.

REDD+ (Reducing Emissions from Deforestation and Forest Degradation), an initiative by United Nations, presents an opportunity to protect biodiversity as a co-benefit of payments to slow deforestation and forest degradation, and to enhance forest recovery in shifting cultivation landscapes. In shifting agricultural landscapes, REDD+ strategies of sparing primary and secondary forests from further clearing are predicted to be most effective in protecting and enhancing forest carbon stocks (Borah et al. 2018). REDD+ also offers strong carbon-biodiversity co-benefits of forest recovery on farmland for birds, dung beetles and amphibians in the tropical Andes (Gilroy et al. 2014a, Basham et al. 2016) and of avoiding deforestation for birds, dung beetles and plants in the Amazon (Ferreira et al. 2018). However, a key question remaining is how a diverse array of REDD+ strategies would affect phylogenetic and functional diversity, including within shifting cultivation.

This study examines the phylogenetic and functional diversity of bird communities following shifting cultivation in Nagaland, Northeast India, which is one of the global biodiversity hotspots (Myers et al. 2000) and where shifting cultivation occupies 71% of its agricultural area (Pareta 2013). Birds are key indicators of biodiversity response to land-use change (Edwards et al. 2014b), have a global phylogeny (Jetz et al. 2012), and well-known functional traits making them an ideal study group. I first understand the impacts of the shifting cultivation cycle on avian phylogenetic and functional diversity and then, using data-derived models to simulate divergent landscape scenarios under REDD+, assess which land-use conservation strategy would be most optimal in protecting both avian phylogenetic and functional diversity.

3. Methods

3.1 Study area

I sampled bird communities in three shifting cultivation landscapes (Kiphire, Phek and Kohima districts) in Nagaland, Northeast India (Appendix II; Fig S1) across three habitat types (old-growth forest, regenerating secondary forest and farmland) in an altitudinal range of 1487-2652 m asl (see Appendix II; Table S1 for plot details), following (Borah et al. 2018). I sampled bird communities at points within randomly selected thirty-six 400 m × 400 m sampling squares across three main habitat types that were spaced ≥ 300 m apart between different habitats and ≥ 400 m apart within the same habitats. Age of the regenerating secondary forests was estimated through informal interviews with experienced farmers and cross-checked using Landsat images, taking the Landsat verified value when reported age differed (Borah et al. 2018) (Appendix II: Text S2).

3.2 Bird surveys

I sampled birds using repeat-visit point counts at three sampling points within each square with 200 m spacing between points to ensure community independence. I visited each point on four consecutive mornings for counts of 10-minute duration (04:45-12:30). I did so in both the summer breeding season (April-May) and winter (Jan-Feb) when Palearctic migrants overwinter in the region; resulting in 171-point counts in total for the study (N = 108 and N = 63 point counts in summer and winter, respectively). The difference in sampling effort between the two seasons was due to civil unrest interrupting sampling in one of the landscapes (Kiphire) in winter season. I avoided sampling in rain or strong winds and varied the routes taken by the observer each day to ensure that each point was visited both early and late in the sampling window. At nine of the 171 points, I was only able to make two visits in summer due to the early onset of the rainy season and associated flooding. The entire duration of the point count at each station was recorded with a sound recorder (Olympus LS11) to allow unknown vocalisations to be subsequently identified using online reference material (xeno-canto.org) and via assistance from regional experts. I restricted the analyses to detections within an estimated 100 m radius, excluding records of highly mobile or transient species (large raptors and birds flying over the plots). Nomenclature followed Jetz et al. 2012 which was compiled from Birdlife International world list (version 3), Handbook of the Birds of the World (del Hoyo et al. 1992-2011) and IOC world list V2.7 (Dec 29, 2010).

3.3 Measures of phylogenetic and functional diversity

I used tree-based approaches for computing phylogenetic and functional diversity metrics for each community. A total of 500 phylogenetic trees were obtained for the observed bird community from birdtree.org (Jetz et al. 2012) based on Hackett et al (2008) backbone. I calculated six abundance-weighted measures of phylogenetic diversity and two measures of evolutionary uniqueness for each sampling point. Phylogenetic diversity (PD) is the sum of evolutionary history in a community (Faith 2013). Phylogenetic Mean Pairwise Distance (pMPD) is the average distance on a phylogenetic tree between all individuals of a community indicating phylogenetic clustering in species (Webb et al. 2002). Phylogenetic Mean Nearest Taxon Distance (pMNTD) is the average distance on a phylogenetic tree between individuals and their closest non-conspecific relative (Webb et al. 2002). Standard effect size of PD (sesPD), of MPD (sespMPD) and of MNTD (sespMNTD) are measures that incorporate species richness into PD, MPD and MNTD respectively, as PD is correlated with species richness (Swenson 2014). Positive values of sesPD indicate higher PD than expected by chance for a given species richness, while negative values indicate lower PD than expected by chance. Similarly, higher sesMPD values indicate phylogenetically more even communities, whereas lower values indicate more phylogenetically clustered communities (Webb et al. 2002). Higher sesMNTD than expected for a given species richness suggests that closely related individuals do not co-occur in the community, and lower sesMNTD than expected suggests co-occurrence of closely related individuals.

I also calculated two measures of evolutionary uniqueness i.e., evolutionary distinctiveness (ED) and evolutionary distinctiveness rarity (EDR). ED is the amount of unique evolutionary history represented by a species in a phylogenetic tree (Jetz et al. 2014) and Evolutionary distinctiveness rarity (EDR) assigns ED evenly across a species' global range (Edwards et al. 2017). High ED and EDR therefore imply importance of conserving species that are evolutionarily unique and that have high extinction risk due to small global range size (for full description of PD and FD metrics see Appendix IV; Text S1).

The six metrics of phylogenetic diversity were calculated using the picante package (Kembel et al. 2010) in R version 3.5.1 (R Core Team 2018). To calculate the standard effect size (ses) for PD, MPD, and MNTD, I compared the observed community against 999 null communities generated using null models with an independent swap algorithm to draw species at random from the regional species pool while maintaining species richness (Gotelli 2000). Metrics were abundance-weighted using the highest count of a species during a single visit to each point (i.e. across the four point-count repeats). I derived a single measure of ED for each species from a global phylogeny of birds (Jetz et al. 2014), which shows the distinctiveness of each species at a global level. I then calculated the mean ED for species recorded at each sample point in the study. Mean EDR was calculated for each species by dividing the mean ED by its global range size (km^2) (values form (Jetz et al. 2014).

I quantified six abundance-weighted measures of functional diversity equivalent to measures for phylogenetic diversity using the functional dendrogram method following (Chapman et al. 2018)– Functional dendrogram (FD), functional Mean Pairwise Distance (fMPD), functional Mean Nearest Taxon Distance (fMNTD) and standard effect sizes of each of these metrics, i.e. sesFD, sesfMPD and sesfMNTD. FD is the sum of branch lengths in a dendrogram generated from functional trait differences (Petchey and Gaston 2006), thus higher FD implying greater differences between species (Chapman et al. 2018). Similarly, communities with positive values of sesFD showed higher functional diversity than expected by chance, and relative of their species richness, whilst communities with negative sesFD had lower levels of functional diversity than expected by chance, and relative of their species richness. Similarly, I used fMPD and fMNTD as measures of community relatedness equivalent to pMPD and pMNTD, aiding direct comparison of phylogenetic and functional diversity.

To calculate the measures of functional diversity, I first compiled a functional trait matrix for three life-history trait categories, i.e., diet type, foraging strata and morphological traits (body mass, bill length, width and depth and length of tarsus, wing and tail) for all species (N= 123) by extracting information from literature on functional traits (Wilman et al. 2014, Cooney et al. 2017). I chose traits that were important in terms of avian resource-use, thus influencing ecosystem function and processes (Flynn et al. 2009, Luck et al. 2013, Edwards et al. 2014b). As the morphological trait measurements were correlated with each other (see Appendix IV; Fig. S1), I performed principal components analysis (PCA) to calculate a new trait matrix of independent transformed coordinates (Appendix IV: Fig. S2). I extracted the first principal component (PC) with 4.42 eigen value that accounted for 73.6 % of the total variation in the morphological trait data and corresponded to dispersal ability and foraging method. I also added four PCs of variables corresponding to bill shape extracted from (Cooney et al. 2017). For further explanation and rationale, see Appendix IV: Text S2.

I derived the functional diversity (FD), functional Mean Pairwise Distance (fMPD) and functional Mean Nearest Taxon Distance (fMNTD) from the functional dendrogram by using function ‘FD_dendro’ in package ‘fundiv’ and functions ‘mpd’ and ‘mnyd’ in ‘picante’ package in R respectively. The standard effect sizes of each of the three functional metrics (sesFD, sesfMPD and sesfMNTD) were calculated similarly using ‘picante’ package (Kembel et al. 2010).

3.4 Modelling species response to habitat type and fallow age

I fitted a Bayesian hierarchical model to estimate species-specific occupancy dynamics to habitat characteristics, allowing estimation of both species-level and the community-level aggregated effects simultaneously while incorporating imperfect detection of individuals. Apart from habitat type, I included elevation (standardised and centred with zero mean) as a fixed effect and ‘Landscape’ as a random effect in the model to account for variation in species occurrence across the elevational gradient and potential autocorrelation in species distribution across landscapes respectively. Occupancy and detection for the two seasons (summer and winter) were modelled separately. I added the hierarchical structure at the community level by specifying all model parameters as random effects drawn from the community level ‘hyper-prior’ distributions. I assigned non-informative and diffuse normal (0, 0.001) priors for hyper-parameter means, and inverse gamma (0.001, 0.001) priors for variances. The model was fitted using JAGS version 4.3.0 (Plummer 2003) called from R version 3.5.1 (R Core Team 2018) and r2jags package. To minimize model uncertainty, I excluded species detected fewer than ten times in each season from this analysis, retaining a total community of 113 and 52 bird species in summer and winter, respectively. I ran the model using three Markov chains with 50,000 iterations, following a burn-in of 30,000 iterations and thinned by five (see Chapter 4 for more details).

To calculate measures of PD and FD across habitat types, I first generated occurrence probabilities for each species based on the habitat characteristics at each sampling point. These were then converted into site-level presence-absence values via individual Bernoulli trials of each species at each site (following (Gilroy et al. 2014a). As both PD and FD measures require abundances of species, I randomly sampled from the observed abundances (> 0) for every presence of each species at each site in the simulated community matrix. Phylogenetic and functional diversity measures were then produced from 100 replicates of each community abundance matrix at each sampling site.

3.5 Assessing avian phylogenetic and functional diversity under REDD+ management scenarios

I used a simulation approach to generate hypothetical bird communities under divergent management scenarios of REDD+ to assess how bird communities recover in terms of PD and FD in shifting cultivation. The fitted Bayesian hierarchical models were used to predict occupancy probabilities for species across simulated sets of hypothetical landscapes with alternative management systems. Following (Borah et al. 2018) I considered two shifting cultivation systems i) existing shifting cultivation which, at the start, contains farmland and various ages of regenerating secondary forest, but no old-growth forest (Scenario 1 and 2, Fig. 1); and ii) pioneer shifting cultivation that, at the start, only contain old-growth forest (Scenario 3 and 4; Fig. 1). I built scenarios for each landscape with 30 individual and uniform-sized parcels of land. Each of these were under a combination of shifting cultivation (farmland or fallow sites; all scenarios), permanently abandoned regenerating secondary forest (Scenario 2 only) and old-growth forest (Scenarios 3 and 4).

Scenario 1 provides a scenario where fallow period decreases to meet growing food demands without any interventions (no forest sparing, Fig. 1), with the original 30-year cycle reducing to 15 years (Scenario 1.1), 10-year (Scenario 1.2) and 5-year (Scenario 1.3). Scenario 2 applies to a landscape where financial incentives from REDD+ enable fallows to permanently regenerate (secondary forest creation and sparing, Fig. 1). Thus, as the fallow period declines from a 30-year cycle to a 15-year (Scenario 2.1), 10-year (Scenario 2.2) and 5-year cultivation cycles (Scenario 2.3), the older fallows are spared from cultivation by increasing agricultural intensity of a part of the landscape. Scenarios 3 and 4 apply to landscapes originally covered by old-growth forest but converted to a shifting cultivation landscape. Scenario 3 describes the application of conservation interventions, such as protected areas, that limit further clearing of old-growth forest for expanding shifting cultivation (sparing old-growth forest, Fig. 1) with three alternatives: conservation of half, two-thirds and 83% of the old-growth forest in 15-year (Scenario 3.1), 10-year (Scenario 3.2) and 5-year cultivation cycles (Scenario 3.3), respectively. In Scenario 4, old-growth forest is entirely cleared to create a shifting cultivation landscape, thus providing an additional scenario with no REDD+ intervention (shifting cultivation expansion, Fig. 1).

To assess PD and FD under the REDD+ management strategies, I first predicted species occurrence probabilities for 1000 replicate landscapes under each scenario. In each simulation, I generated a predicted community Ž by populating an occurrence matrix with dimensions $N \cdot L$, where N is the total number of species detected in the study and L is the number of sites in the simulated scenario (Zipkin et al. 2009). Each element of Ž is computed as a single Bernoulli trial with probability drawn by sampling from the posterior distributions of model parameters for each species (Gilroy et al. 2014b). As PD and FD measures require abundances of species, I randomly sampled from the observed abundances (> 0) for every presence of each species at each site in the simulated matrix. These simulated abundance metrics were then used to calculate each phylogenetic and functional diversity metric for each scenario landscape.

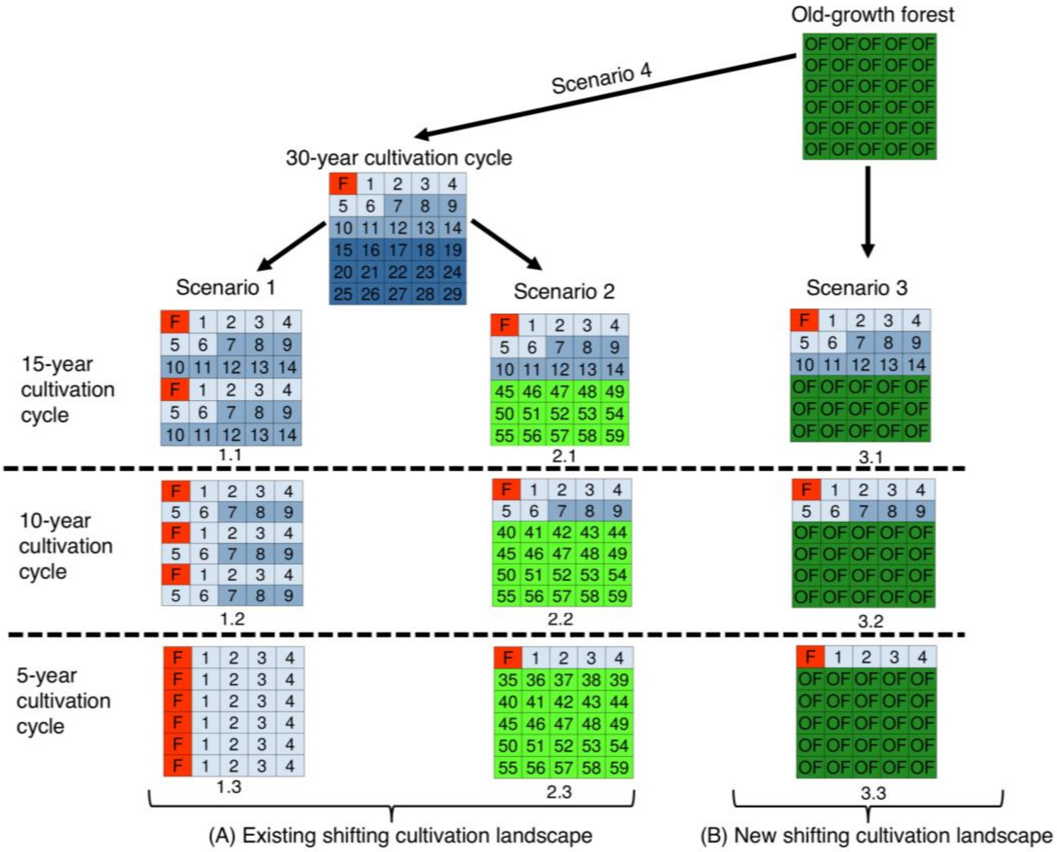


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Figure 1. The four sets of management scenarios used to predict changes in avian phylogenetic and functional diversity in (A) no forest sparing (Scenario 1) and secondary forest creation and sparing (Scenario 2) and (B) new shifting cultivation landscape with old-growth forest sparing (Scenario 3) and shifting cultivation expansion (Scenario 4). Colours indicate habitat types: farmland (F, red), active fallows in different shades of blue, i.e. young secondary forest (1-6 years, light blue), young secondary forest (7-14 years, dark blue) advanced secondary forest (15-29 years, darkest blue) and abandoned old fallows (>30 years, light green) and old-growth forests (OF, dark green). Numbers within cells denote the age of the secondary forests.

4. Results

I observed a total of 7790 detections of 281 species across all habitats representing 45 families (257 and 193 species in summer and winter, respectively). Among these, one species was endangered, five were vulnerable, five were near threatened and the rest were least concern in the IUCN red list of threatened species. The analysis was however conducted for 123 (113 and 52 species in summer and winter, respectively) with more than 10 detections in both seasons (for full species list see Appendix IV; Table S3).

4.1 Bird phylogenetic diversity across habitats

Bird phylogenetic diversity (PD) was higher in old-growth forest compared to farmland and secondary forest in both summer and winter (Fig 2A and Fig 3A). However, when accounting for species richness (sesPD), all three habitat types had similar levels of phylogenetic diversity in both seasons (Fig 2B and Fig 3B). Similarly, PD and sesPD did not differ across very young, young and advanced secondary forest in summer (Fig 2A-2B). In the winter bird community, PD was lower in young secondary forest but similar across all three secondary habitats after corrected for species richness (sesPD). Phylogenetic mean pairwise distance (pMPD) and its standard effect size (sespMPD) were lower in old-growth forest than farmland and secondary forest in summer (Fig 2C-2D), whereas the opposite pattern was found in winter bird community (Fig 3C-3D). Within secondary forest, pMPD and sespMPD increased with increasing fallow age in summer (Fig 2C-2D) whereas in winter, both pMPD and sespMPD were lower in young secondary forest compared to very young and advanced secondary forests (Fig 3C-3D). Mean nearest taxon distance (pMNTD) and sespMNTD were also lower in old-growth forest compared to farmland and secondary forest (Fig 2E-2F) in summer. Winter bird community showed similar pattern for pMNTD but sespMNTD did not vary across farmland, secondary forest and old-growth forest (Fig 3E- 3F).

Evolutionary distinctiveness (ED) had a weak negative correlation with species occurrences in all habitats except young secondary forest in summer (Appendix IV; Fig. S3; farmland: r = - 0.15, P < 0.001, very young secondary: r = - 0.13, P < 0.001, advanced secondary: r = - 0.11, P < 0.001; old-growth forest: r = - 0.09, P < 0.001). Similarly, species occurrence was negatively correlated with Evolutionary Distinctiveness Rarity (EDR) across all the habitats (Appendix IV; Fig S3; farmland: r = - 0.06, P = 0.01, very young secondary: r = - 0.12, P < 0.001, young secondary: r

= - 0.15, P < 0.001, advanced secondary: r = - 0.10, P < 0.001; old-growth forest: r = - 0.03, P = 0.01). In contrast, EDR had a weak positive correlation with species occurrence in very young (r = 0.09, P = 0.03), young (r = 0.12, P = 0.04), advanced secondary (r = 0.09, P = 0.01) and old-growth forest (r = 0.06, P = 0.01) (Appendix IV; Fig S4). ED was, however, not significantly correlated with species abundance of winter bird community in any habitat type (Appendix IV; Fig. S4).

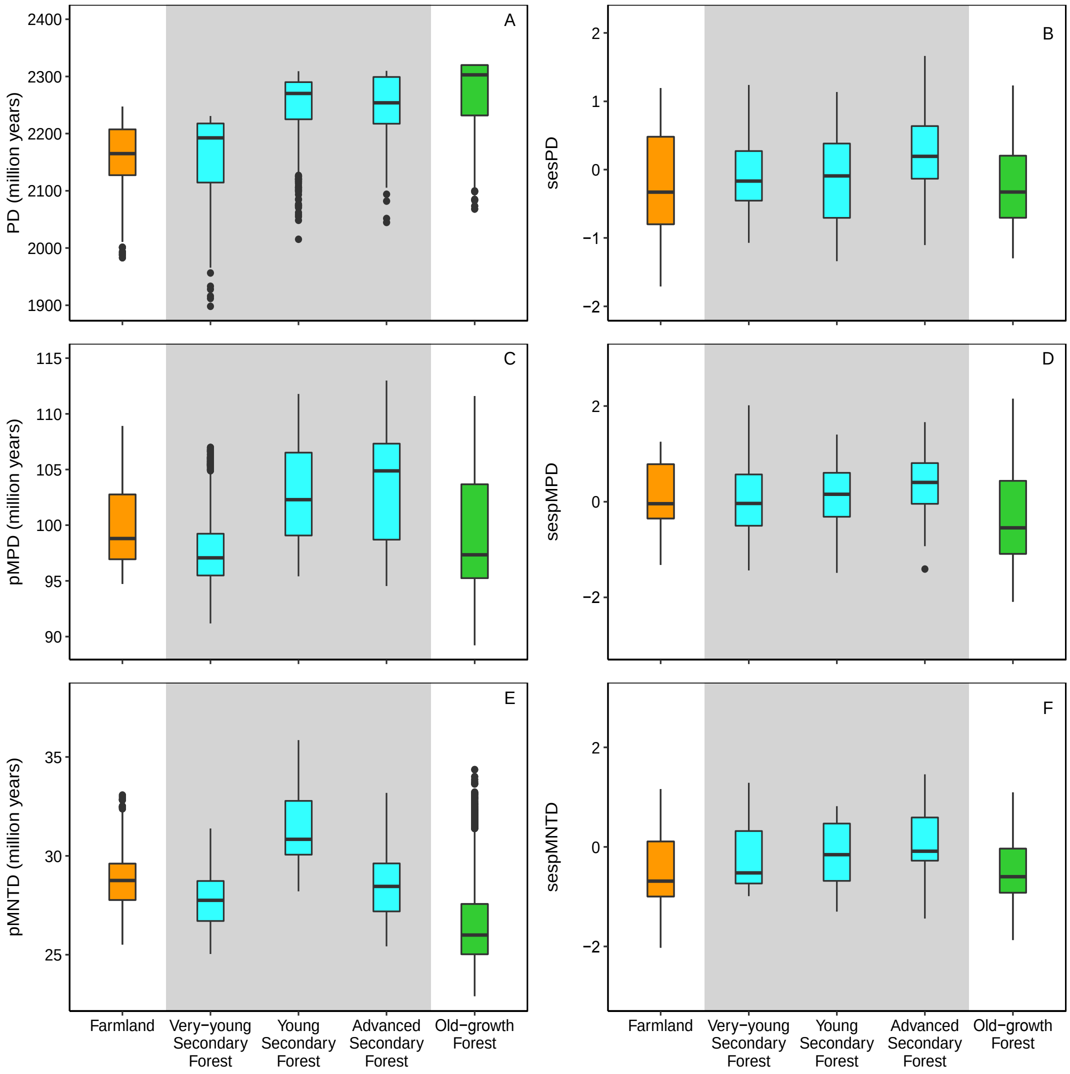

Figure 2. Phylogenetic diversity indices, Phylogenetic diversity (PD); standard effect size of PD (sesPD); mean pairwise distance (pMPD) and standard effect size of MPD (sespMPD) for point level model communities across habitat types in summer. The bottom and top of the box plots represent the first and third quartiles, respectively, the heavy line represents the median, and the points represent outliers across 1000 simulations (variation due to uncertainty in species occurrence at the point level).

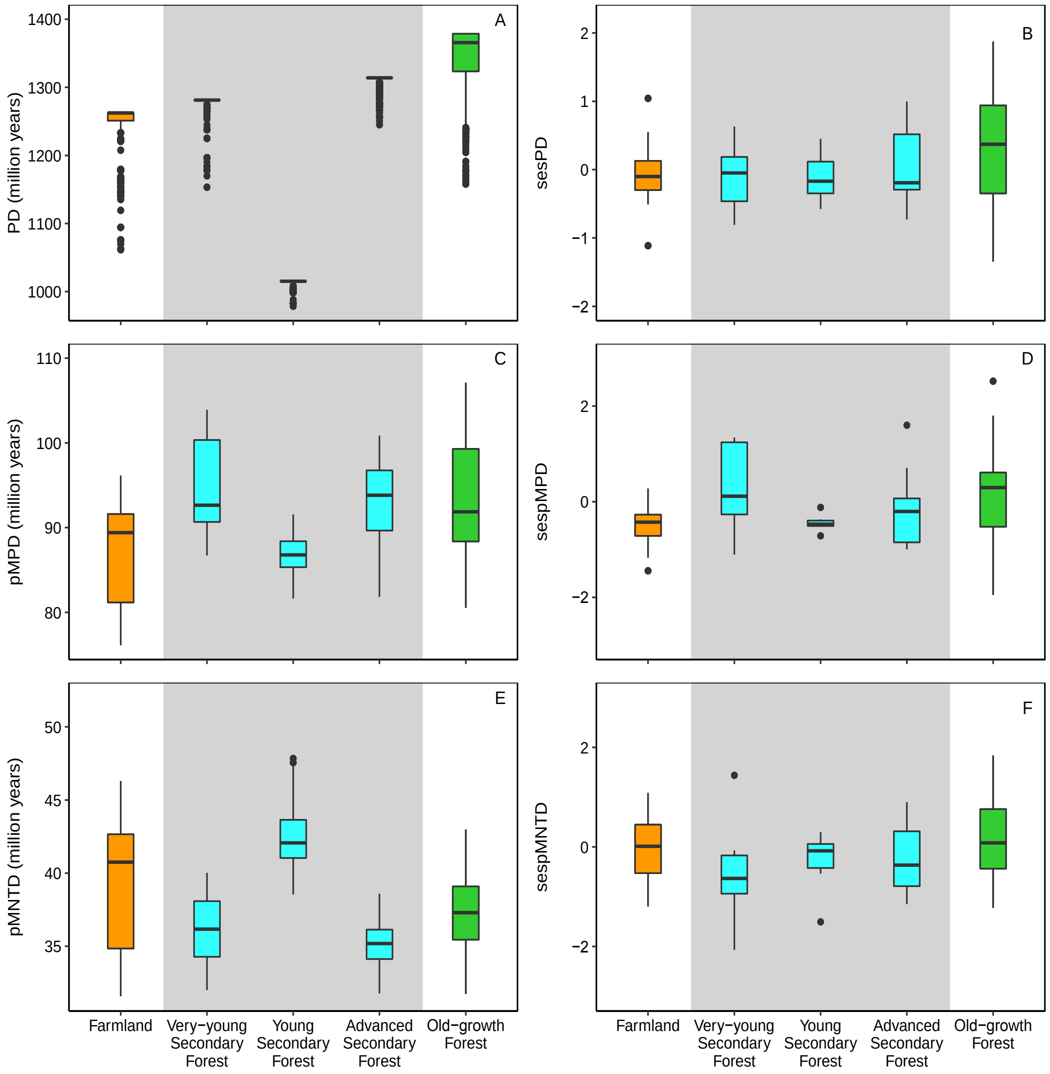

Figure 3. Phylogenetic diversity indices, Phylogenetic diversity (PD); standard effect size of PD (sesPD); mean pairwise distance (pMPD) and standard effect size of MPD (sespMPD) for point level model communities across habitat types in winter. The bottom and top of the box plots represent the first and third quartiles, respectively, the heavy line represents the median, and the points represent outliers across 1000 simulations (variation due to uncertainty in species occurrence at the point level).

4.2 Bird functional diversity across habitats

Old-growth forest retained a slightly higher functional diversity (FD) compared to farmland and secondary forest in both summer and winter (Fig 4A and Fig 5A). However, FD was similar across all habitats when accounted for the species richness (sesFD) in summer (Fig 4B). In contrast, sesFD was lower in old-growth forest compared to farmland in winter bird community (Fig 5B). Secondary forest showed no variation in FD and sesFD during summer, whereas very young secondary forest retained a higher sesFD in winter compared to young and advanced secondary forest (Fig. 5B).

Functional mean pairwise distance (fMPD) did not show any variation across habitats (Fig 5C), whereas sesfMPD was higher in farmland and secondary forests than old-growth forest in both seasons (Fig 4D and 5D). Similarly, old-growth forest maintained a lower functional mean pairwise distance (fMNTD) and standard effect sizes of MNTD (sesfMNTD) compared to farmland and secondary forest in both seasons (Fig. 4E and Fig. 5E). There was no variation in fMNTD within secondary forest (Fig. 4E), whereas sesfMNTD decreased with increasing fallow age from very-young to advanced secondary forest (Fig. 4F) in the summer bird community.

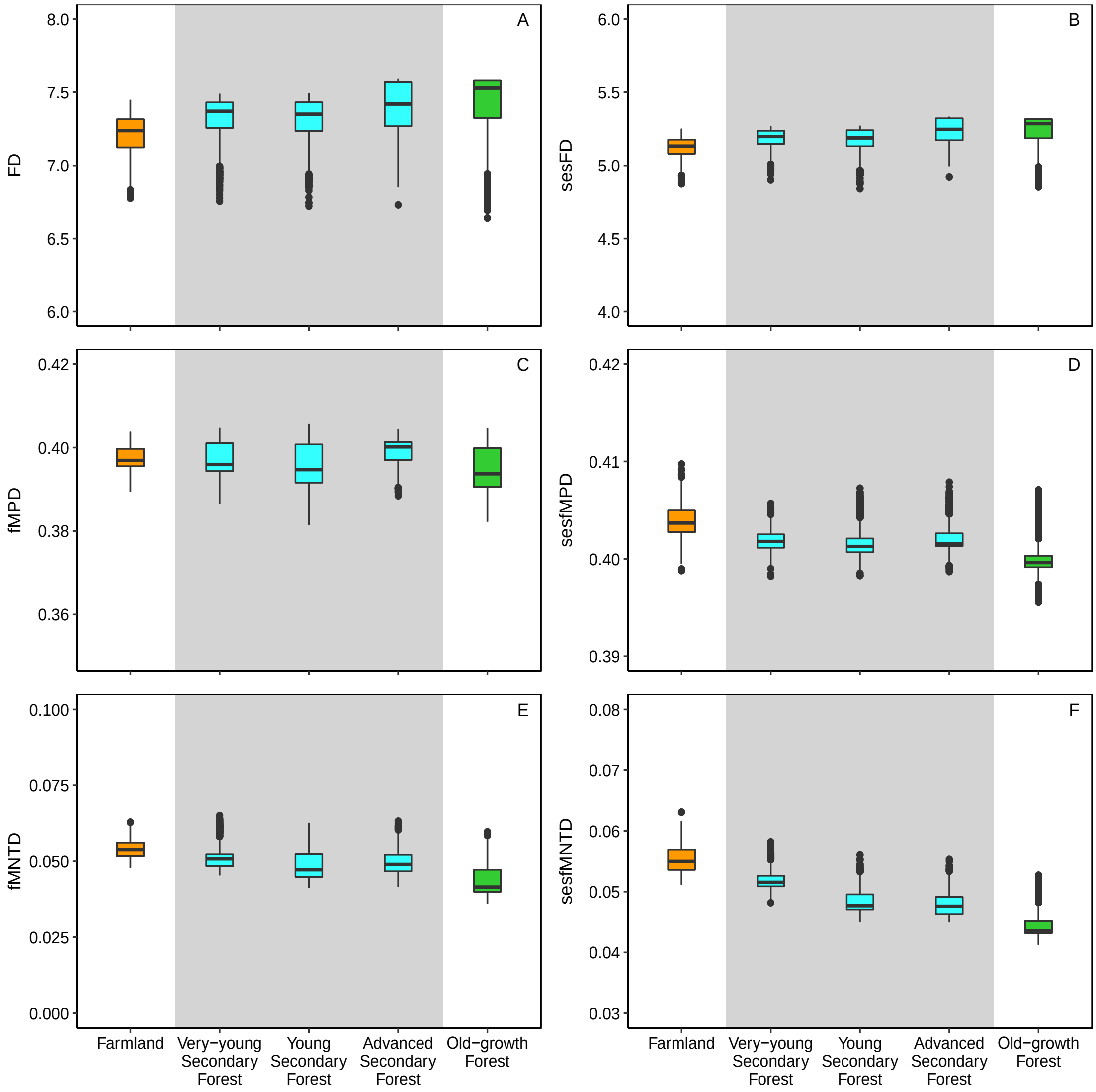

Figure 4. Functional diversity indices for point level model communities across habitat types in summer; functional diversity (FD), functional mean pairwise distance (fMPD), and functional mean nearest taxon distance (fMNTD) and standard effect sizes of each (sesFD, sesfMPD and sesfMNTD). The bottom and top of the box plots represent the first and third quartiles, respectively, the heavy line represents the median, and the points represent outliers across 1000 simulations (variation due to uncertainty in species occurrence at the point level).

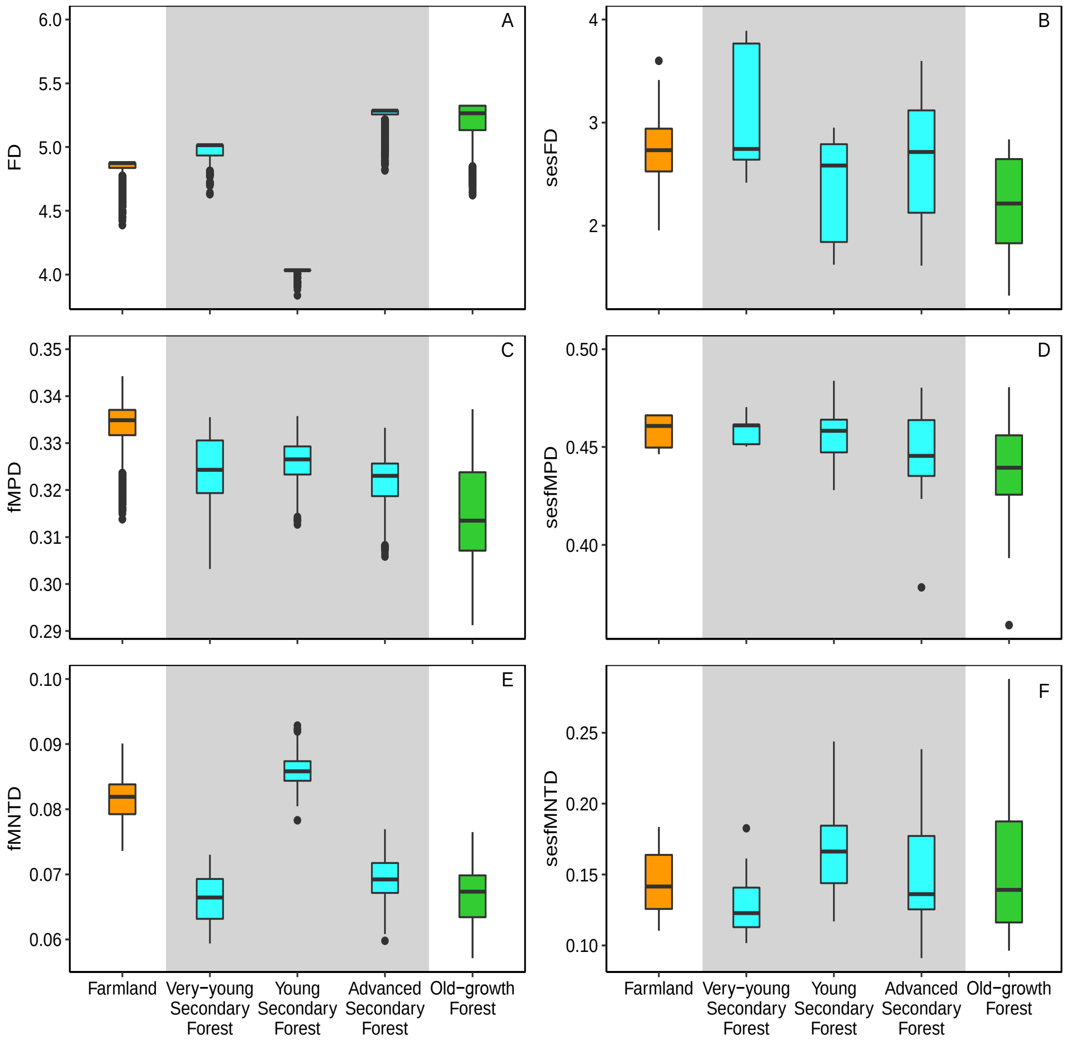

Figure 5. Functional diversity indices for point level model communities across habitat types in winter; functional diversity (FD), functional mean pairwise distance (fMPD), and functional mean nearest taxon distance (fMNTD) and standard effect sizes of each (sesFD, sesfMPD and sesfMNTD). The bottom and top of the box plots represent the first and third quartiles, respectively, the heavy line represents the median, and the points represent outliers across 1000 simulations (variation due to uncertainty in species occurrence at the point level).

4.3 Recovery of landscape level phylogenetic and functional diversity under REDD+

Landscape simulations showed that Phylogenetic diversity (PD) was highest in Scenario 2 with secondary forest creation and sparing (Scenario 2.3; Fig 6A) and Scenario 3 where old-growth forest is spared (Scenario 3.1; Fig 6A). However, phylogenetic diversity responded differently to the number of cultivation cycles and proportion of secondary forest or old-growth forest spared within the same scenarios. Both phylogenetic Mean Pair-wise Distance (pMPD) and Mean Nearest Taxon Distance (pMNTD) were the lowest in the old-growth forest landscape compared to rest of the scenarios (Fig 6C and Fig 6E). Scenario 2 and Scenario 3 at 10-year cultivation cycle (Scenario 2.2 and Scenario 3.2, respectively) had higher pMPD compared to landscapes with 5-year and 15-year cultivation cycles (Scenario 2.3 and Scenario 2.1, respectively; Fig 6C). Similar pattern was also observed for pMNTD across the Scenarios (Fig 6E). The standard effect sizes of each of these metrics, sesPD, sespMPD and sespMNTD were mostly similar across all the scenarios .

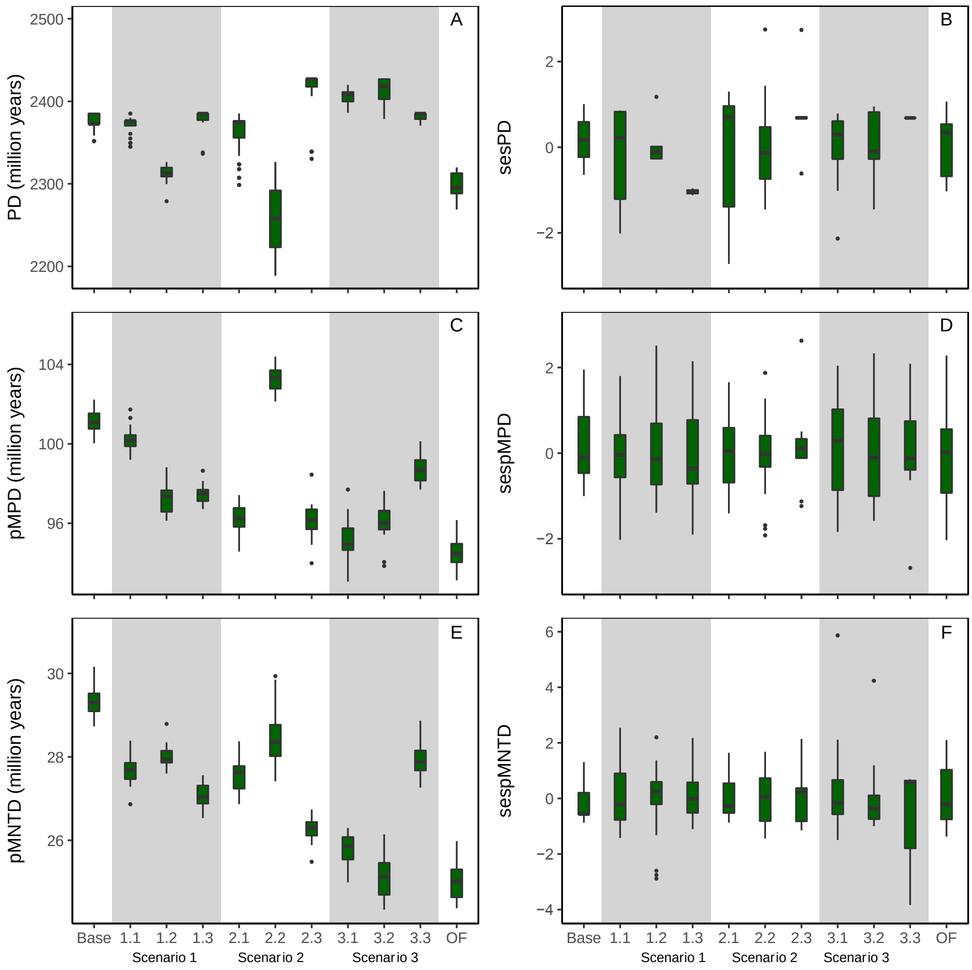

Figure 6. Predicted changes in phylogenetic diversity in summer under the four alternative management regimes of shifting cultivation at the end of 30 years (1) No forest sparing in Scenario 1 (Scenario 1.1, 15-year cycle; Scenario 1.2, 10-year cycle; Scenario 1.3, 5-year cycle) ; (2) intervention scenarios by secondary forest creation and sparing in Scenario 2 (Scenario 2.1, 15-year cycle; Scenario 2.2,10-year cycle; Scenario 2.3, 5-year cycle) and old-growth forests sparing in Scenario 3 (Scenario 3.1, 15-year cycle; Scenario 3.2, 10-year cycle; Scenario 3.3, 5-year cycle). Mean values from 1000 randomisations under each scenario indicated by points, with error bars representing 95th percentiles

Scenario simulations of functional diversity showed that the old-growth forest landscape had a lower FD compared to the Baseline and rest of the scenarios. However, Scenario 3 that maintains a mosaic system of farmland, regenerating forest and old-growth forest was the most optimal strategy under REDD+ to sustain the functional diversity.

FD and sesFD were highest in Scenario 3 where old-growth forest is spared at 5-year cultivation cycle (Scenario 3.1) and the lowest in old-growth forest landscape in summer (Fig. 7A -7B). Functional Mean Pairwise Distance (fMPD), Mean Nearest Taxon Distance (fMNTD) and standard effect sizes of both fMPD and fMNTD were the lowest in Scenario 3 (Scenario 3.1) and highest in Scenario 1 with no forest sparing at 10-year cultivation cycle (Scenario 1.2, Fig 7C- 7F). However, sesfMPD increased as fallow period declined in Scenario 3 from 15-year cycle (Scenario 3.1) to 5-year cycle (Scenario 3.3) and proportion of old-growth spared increased in the landscape. Similarly, fMNTD increased from 15-year to 5-year cultivation cycle in Scenario 2 suggesting functional over-dispersion with increase in proportion of secondary forest.

Simulations for functional diversity in winter bird community showed that all the scenarios maintaining similar levels of functional diversity (Fig S4A – S4B). Scenario 3 with old-growth forest sparing at 5-year cultivation cycles had the lowest level of fMPD and fMNTD indicating an accumulation of functionally similar species with increase in proportion of old-growth forest in shifting cultivation landscape (Fig S4C - Fig S4D). None of the standard effect sizes of the functional metrics showed any variation across the scenarios.

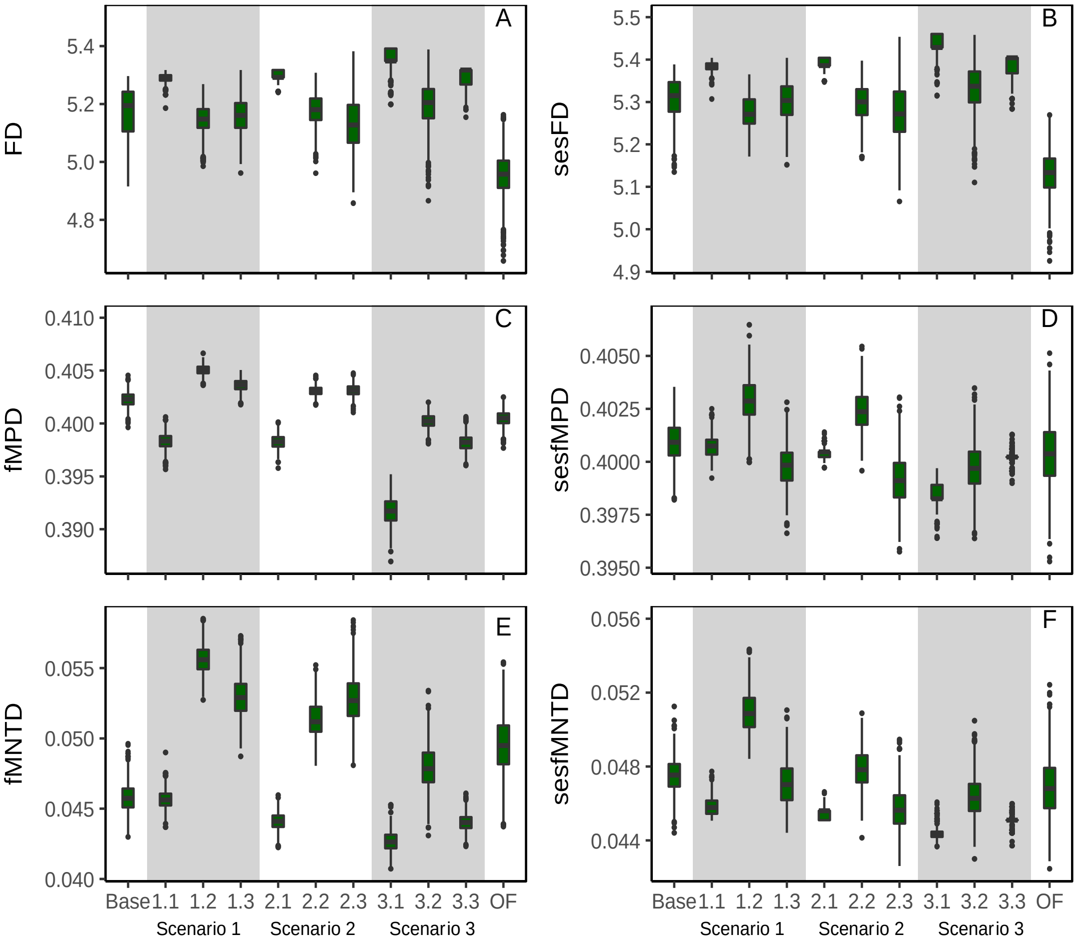

Figure 7. Predicted changes in functional diversity in summer under the four alternative management regimes of shifting cultivation at the end of 30 years (1) No forest sparing in Scenario 1 (Scenario 1.1, 15-year cycle; Scenario 1.2, 10-year cycle; Scenario 1.3, 5-year cycle) ; (2) intervention scenarios by secondary forest creation and sparing in Scenario 2 (Scenario 2.1, 15-year cycle; Scenario 2.2,10-year cycle; Scenario 2.3, 5-year cycle) and old-growth forests sparing in Scenario 3 (Scenario 3.1, 15-year cycle; Scenario 3.2, 10-year cycle; Scenario 3.3, 5-year cycle). Mean values from 1000 randomisations under each scenario indicated by points, with error bars representing 95th percentiles.

5 Discussion

As shifting cultivation cycles are becoming increasingly frequent leading to incomplete forest recovery, a key question is how this impacts biodiversity conservation. This is the first empirical study assessing recovery of avian phylogenetic and functional diversity following shifting cultivation. The results reveal that forest conversion to shifting cultivation caused decline in phylogenetic diversity and it increased with fallow age in secondary forest. However functional diversity was robust to shifting cultivation. Old-growth forest supported phylogenetically and functionally clustered breeding bird community that can contribute to resilience from potential disturbances in the system. The REDD+ scenario simulations suggest that phylogenetic and functional diversity can be best conserved by maintaining a heterogenous mosaic of farmland and secondary forest with large proportion of advanced secondary forest or old-growth forest.

5.1 Bird phylogenetic diversity in shifting cultivation

This study shows that phylogenetic diversity of the breeding bird community declined from forest conversion to shifting cultivation but recovered as secondary forest regenerated across fallow ages. This mirrors findings from previous studies that showed decline in phylogenetic diversity from forest conversion to agriculture (Frishkoff et al. 2014, Prescott et al. 2016). The rapid recovery of phylogenetic diversity in secondary forest was probably due to species gains as reported by previous studies elsewhere (Edwards et al. 2017). The habitat heterogeneity, typical of shifting cultivation mosaics, allow species to utilize different resources, thus accumulating additional species of diverse evolutionary lineages (Frishkoff et al. 2014).

A decrease in the mean pairwise distance (pMPD) and mean nearest taxon distance (pMNTD) in old-growth forest suggested phylogenetic clustering in the summer bird communtiy with more species of shared evolutionary history co-existing together, as shown in studies elsewhere (Prescott et al. 2016). This pattern is, however, not maintained in the winter bird community, potentially due to a shift in the community composition during winter (Chapter 4) adding distantly related species to the community (Dehling et al. 2014).

The overall maintenance of evolutionary distinctiveness and evolutionary distinctiveness rarity in shifting cultivation landscapes indicates persistence of a similar number of unique lineages across all habitat types. This is in contrast with previous studies that suggest a decline in evolutionarily distinct species in farmland compared to old-growth forest (Frishkoff et al. 2014, Edwards et al. 2017). As shown in previous studies in the study region, farmland and regenerating secondary forest supported many forest-dependent species due to presence of retained trees and close distance to old-growth forests (Chapter 4). These factors might also have contributed in addition of evolutionarily unique species. The weak positive correlation between species occurrence and mean evolutionary distinctiveness across secondary forest and old-growth forest in winter lends support to previous studies suggesting that the mean evolutionary distinctiveness also recovers rapidly in secondary forest communities (Edwards et al. 2017).

5.2 Bird functional diversity in shifting cultivation

Shifting cultivation maintained similar levels of functional diversity in farmland and secondary forest compared to an old-growth forest. This can be attributed to the mosaic nature of shifting cultivation landscapes (Clough et al. 2009, Perović et al. 2015). Low-intensity and patchy fire, typical of shifting cultivation, generates environmental heterogeneity and thus supports diverse functional groups by facilitating resource partitioning (Sitters et al. 2016). This is likely to positively influence ecosystem services such as seed dispersal and insect control by sustaining species with varying dietary niche in these landscapes (Sitters et al. 2016). The adjacent old-growth forest in these landscapes can also act as population sources providing resources for species that utilise both forest and farmland (Sekercioglu et al. 2007, Prescott et al. 2016, Norfolk et al. 2017). Low fMPD and fMNTD in old-growth forest suggest functional clustering in the community, with multiple species of similar functional roles in the old-growth forest community, which mirrors earlier findings elsewhere (Hidasi-Neto et al. 2012). This has significant positive conservation implication as co-existence of similar species can increase functional redundancy making the community in old-growth forest more resilient to future disturbances (Laliberté et al. 2010). In contrast, the higher fMPD and fMNTD than expected in farmland compared to regenerating secondary forest and old growth forest suggests over-dispersion with farmland retaining functionally different species. This can be mainly attributed to resource partitioning in species to avoid interspecific competition.

Within secondary forest, fallow age did not have any strong influence on functional diversity suggesting that functional groups are robust to changes caused by shifting cultivation. This contrasts with studies in secondary forest in which showed a substantial recovery of FD over time in other taxa (Audino et al. 2014). However, fMNTD declined along the successional gradient from very young to advanced secondary forest in summer. This is probably due to colonization of closely related species to utilise available resources as secondary forest matures.

5.3 Recovery of phylogenetic and functional diversity under REDD+

Our results suggest that shifting cultivation landscapes can play a vital role in conserving high levels of phylogenetic and functional diversity and evolutionarily unique species, mirroring findings for carbon (Borah et al. 2018; Chapter 3) and species richness (Chapter 4). Thus, these landscapes hold a strong potential in harnessing carbon and biodiversity benefits from carbon-based payments for ecosystem services such as REDD+. Because of the low profitability of shifting cultivation in the study region, REDD+ is likely to offer economic alternatives that promote protecting and enhancing forest carbon at a cheaper opportunity cost (Gilroy et al. 2014b).

Scenario 3 with old-growth sparing at 5 and 15-year cultivation cycle sustained the highest levels of both phylogenetic and functional diversity in summer. This suggests that maintaining a heterogenous landscape with farmland, regenerating secondary forest and old-growth forest is the most optimal strategy to protect phylogenetic and functional diversity as supported by previous studies in traditional agroecosystems elsewhere (Martin et al. 2012). These landscapes also allowed closely related species to coexist in the community by complementing ecosystem services. In areas without any old-growth forest, protecting the regenerating secondary forest within the mosaic by intensifying cropping in a smaller area is a good candidate for REDD+ investment. However, bird community in winter had similar levels of phylogenetic and functional diversity across all the scenarios indicating the importance of assessing seasonal variation in bird community response while designing conservation interventions (Elsen et al. 2017).

REDD+ interventions can assess impacts of management by monitoring certain indicator groups that represent general patterns of phylogenetic and functional diversity (Mayfield and Levine 2010, Trindade-Filho et al. 2012). For instance, restricted-range species are a good candidate to represent bird functional diversity (Trindade-Filho et al. 2012). Similarly, evolutionarily distinct species with high trait complementarity can be used to prioritize conservation of bird phylogenetic diversity. This approach will ensure the conservation of ecosystem functioning, stability and the unique evolutionary history of bird communities in the long term (Naeem and Wright 2003).

Although only 1% of the world's bird species primarily prefer agricultural areas, nearly one-third of all birds occasionally use such habitats (Sekercioglu et al. 2007), often providing important ecosystem services, such as pest control, pollination, and seed dispersal (Van Bael et al. 2007, Perfecto et al. 2009). This study illustrates the potential of shifting cultivation systems to retain a similar level of bird phylogenetic and functional diversity as in old-growth forest. This suggests that these landscapes are of critical importance to the conservation of biodiversity and ecosystem functioning in the subtropical mountains. However, these landscapes are increasingly converted to commercial plantation such as oil palm and rubber (van Vliet et al. 2012), which can cause a drastic loss of both phylogenetic and functional diversity (Edwards et al. 2017, Chapman et al. 2018). Shifting cultivation landscapes, with high degree of habitat heterogeneity, are an important refuge for biodiversity (Padoch et al. 2010a) and if managed effectively can play an important role in mitigating biodiversity loss and green-house gas emission.

This study comes with a few caveats. First, I chose to apply the hierarchical modelling approach to only the species that were detected more than ten times during the entire sampling period to reduce uncertainty in model prediction. This restriction may exclude evolutionarily distinct and functionally unique and rare species, thus influencing the diversity measures (Leita et al. 2016). In addition, I did not explicitly account for subsistence hunting that is prevalent in the study region (Bhupathy et al. 2013) and can have a significant effect on bird detection. However, I sampled a large number of replicates across three landscapes with a varying degree of hunting pressure to account for variation in species detection and occurrence caused by hunting. Finally, the simulated scenarios assume that crop yield per hectare can be maintained or increased despite reduced cropping area by adopting various crop management (Toky and Ramakrishnan 1981a, Tawnenga and Tripathi 1997) and improved fallow management techniques as shown in studies from the region (Grogan et al. 2012).

6. Conclusion

This study reveals that shifting cultivation sustained a high level of functional diversity in the breeding bird community, indicating minimal impact on ecosystem functioning, whereas phylogenetic diversity recovered rapidly in regenerating secondary forest. Heterogenous landscapes with varying extent of farmland, regenerating secondary forest and old-growth forest were most optimal under REDD+ in harbouring maximum levels of phylogenetic and functional diversity, and closely related species that maintain system resilience and ecosystem functioning. Wintering bird community, however, retained similar levels of phylogenetic and functional diversity across all the scenarios implying importance of considering seasonal variation in biodiversity response in conservation planning. Nevertheless, the results of this study show the high conservation value of shifting cultivation and the potential for achieving carbon and biodiversity co-benefits under REDD+ by effectively managing these landscapes.

Chapter 6

Thesis Discussion

Finding an effective strategy to reconcile the interrelationship between forests, agriculture and sustainable development is essential in ensuring food security, tackling climate change and halting the current biodiversity crisis. Despite the global trend towards a transition from subsistence to commercial agriculture (Padoch et al. 2010a, Jakovac et al. 2016) shifting cultivation continues to be a dominant agricultural system in many remote areas in South and South-east Asia, as well as in Africa, where market access is limited, and multi-functional land uses remain as the most suitable option to farmer’s socio-economic life (van Vliet et al. 2012). Undefined land tenure systems, naturally low fertile soil and soil erosion due to high rainfall also make transition to settled agriculture challenging in these regions (Chapter 2). In addition, a shift from shifting cultivation to commercial agriculture, such as cash crops, may offer immediate economic benefits, but such agricultural intensification leads to long-term negative impacts on forests, biodiversity, ecosystem services, and local livelihoods (van Vliet et al. 2012, Teegalapalli and Datta 2016a).

The major reason behind the transition away from shifting cultivation is the wide spread perception that it is a primitive farming system and a driver of tropical deforestation and degradation due to shortening fallow period (Mertz 2009, van Vliet et al. 2012, Jakovac et al. 2016). However, long-term empirical evidence of shortening fallow period is rarely found in literature. For instance, farmer surveys from field sites in my study region suggest that fallow period does not show any clear pattern of change. Although farmers have been clearing forests and farming more frequently in some villages, fallow period has not changed or rather increased in other villages. In addition, the assumed link between fallow period and decline in crop yield is not well established (Mertz 2009). Recent scientific evidence instead suggests that shifting cultivation is less harmful to the environment and biodiversity than compared to permanent agriculture (Mandal & Raman 2016, Borah et al 2018 / Chapter 3). This farming system is well adapted to heavy rainfall and environmental conditions in mountainous regions (Goswami et al. 2012), Thus, the earlier narrative of shifting cultivation as a ‘problem’ and the need to replace it entirely by alternative systems was due to lack of a holistic understanding of the system. Thus, there is an urgent need for policies to focus on effectively managing shifting cultivation for forests and biodiversity conservation rather than replacing it by commercial agriculture.

A core mechanism for ensuring forest protection and food security in shifting cultivation landscapes are carbon-based payments for ecosystem services (PES), which offer payments to nations to retain forests (UN-REDD 2017). PES programs such as REDD+ have been increasingly used as key mechanisms for slowing the rate of global climate change and biodiversity loss by protecting and enhancing forest carbon and sustainable management of forests (UN-REDD 2017). However, majority of the research focused on avoiding deforestation from permanent agriculture and plantations, slowing the rate of forest degradation from selective logging, and regenerating carbon stocks on abandoned farmland. The key knowledge gap tackled in this thesis was whether REDD+ has a potential role to play in protecting carbon and biodiversity in shifting agriculture.

In Chapter 3, I assess the potential of REDD+ mechanism in protecting and enhancing carbons stocks in shifting cultivation landscapes. I showed that carbon stocks recover substantially as secondary forest regenerates following shifting cultivation, with a 30-year fallow storing about half the carbon of an old-growth forest, as has been found in other studies (D’oliveira et al. 2011b, Salinas-melgoza et al. 2017). My assessment of diverse management strategies suggests that sparing old-growth forests from conversion into shifting cultivation by intensifying cropping in a smaller area is the most optimal strategy under REDD+ for protecting landscape carbon. In existing shifting cultivation system without any old-growth forest, REDD+ funding can be invested in sparing older fallows, which also stores significant amount of landscape carbon, for permanent forest regeneration. Maintaining a longer fallow cycle, for instance at 15-year cultivation cycle can also sequester considerable levels of carbon compared to landscapes with short fallow cycles (5- and 10-year cultivation cycles). This chapter thus shows strong potential of REDD+ in protecting landscape level carbon stocks and presents explicit pathways that can be most optimal in doing so.

Shifting cultivation modifies the landscape into a mosaic of farmland, regenerating and old-growth forests, thus, influencing biodiversity patterns (Scales and Marsden 2008, Borah et al/Chapter 2). In Chapter 4, I show that overall bird species occurrence was higher in farmland and regenerating secondary forests than old-growth forest. Old-growth forest, however, harboured a distinct community from farmland and secondary forests in the breeding season. Scenario simulations suggest heterogenous landscapes with farmland and regenerating secondary high levels of bird diversity including forest associated species of conservation concern.

This underscores the importance of shifting cultivation landscapes in sustaining high levels of biodiversity as shown in previous studies elsewhere (Norgrove and Beck 2016) and in other taxa (Bowman et al. 1990, Klanderud et al. 2009). This study also shows the potential of conservation interventions to achieve biodiversity co-benefits with carbon payments.

Biodiversity conservation strategies also increasingly target maintaining evolutionary history and the resilience of ecosystem function, instead of just species richness (Mouillot et al. 2013). The most commonly used metrics to assess these are phylogenetic diversity (PD) i.e. the diversity in evolutionary history of species and functional diversity (FD), i.e. diversity of functional traits across species. Assessing recovery of bird PD and FD across farmlands, regenerating secondary forests and old-growth forest in shifting cultivation revealed that shifting cultivation landscapes maintained similar levels of functional diversity compared to old-growth forests whereas loss in phylogenetic diversity from forest conversion to shifting cultivation recovered rapidly with increasing fallow age (Chapter 5). Old-growth forests supported a phylogenetically and functionally clustered bird community whereas farmland had an over-dispersed bird community retaining diverse evolutionary lineages and functional groups. REDD+ interventions that maintain a mosaic landscape with farmland, regenerating forest and old-growth forest were optimal in preserving high levels of phylogenetic and functional diversity. This signifies the importance of incorporating these diversity metrics in conservation policies to identify conservation targets (Chapman et al. 2018).

Reconciling shifting cultivation, carbon and biodiversity under REDD+

With the rising human population, mitigating climate change and biodiversity loss while ensuring food security is a global conservation concern (Tscharntke et al. 2012). Carbon-based payment schemes such as REDD+ potentially can achieve this threefold objective by effectively managing shifting cultivation in the tropical and subtropical mountains (Ziegler et al. 2012). These areas have the potential to harness conservation gains at minimal cost as shifting cultivation is a subsistence-based farming and is mainly practised in remote regions with limited market access and low crop yields (Borrego and Skutsch 2014). However, successful implementation of REDD+ in shifting cultivation landscapes need reliable carbon and biodiversity monitoring protocols, safeguards to avoid leakage, addressing tenure insecurity and efficient coordination across various stakeholders (Visseren-Hamakers et al. 2012).

Balancing carbon and biodiversity under REDD+

Balancing carbon and biodiversity benefits from REDD+ can be challenging as unlike carbon dynamics, biodiversity responses to land-use change are variable across time and space (Martin et al. 2013b). Moreover, impacts of biodiversity and carbon may not always be correlated as species richness and carbon do not necessarily have a linear positive relationship as evidenced in this study (see Chapter 3 and 4) and elsewhere (Ferreira et al. 2018). This potential trade-off between carbon and biodiversity also varies across local- and landscape-level making it challenging for carbon-focussed policies to deliver biodiversity co-benefits. Comparing spatial distribution of biodiversity and its threats, responses to land-use change and management against spatial priorities for carbon investment can be effective in minimizing these potential trade-offs between carbon and biodiversity (Gardner et al. 2012a). These carbon-biodiversity spatial overlays can help identify either carbon-neutral solutions that offer varying additional benefits for biodiversity, or opportunities where relatively minor adjustments to primary carbon objectives can deliver disproportionate benefits for biodiversity (Venter et al. 2009). For instance, strategic zoning by equally prioritizing carbon and biodiversity protection was able to achieve agricultural development in >56,000 km^2 land with significantly low impact on biodiversity and carbon in a potential carbon and biodiversity conflict area in Australia (Morán-Ordóñez et al. 2016).

It is also essential to adopt safeguards for potential leakage as conservation interventions can indirectly displace land use pressures or extractive activities outside the area of management (Boyle et al. 2012). Across shifting cultivation landscapes, leakage might occur if a transition away from shifting cultivation encourages import of food from outside, driving agricultural expansion at the cost of forest elsewhere instead. Similarly, regulating timber extraction from fallow forest could lead to unintended rise in net greenhouse gas emissions by replacing wood products with more emissions-intensive alternatives such as concrete, steel or plastics. Evaluating the potential direct and indirect impacts of REDD+ actions across multiple spatial scales will help to avoid unintended leakage in these landscapes.

Socio-economic implications for shifting cultivators

The indigenous communities in shifting cultivation-dominated landscapes depend heavily on forest resources for their livelihood and play an important role in forest protection and management (Murthy et al. 2013b). The REDD+ mechanism, if not designed carefully, could potentially increase the vulnerability of these farmers to negative socio-economic changes. For example, a complete shift away from shifting cultivation will not be viable as subsistence production of staple foods such as rice and maize are important for farmers in these regions with limited market accessibility (Mertz et al. 2009).Similarly, complex land-tenure system across shifting cultivation landscapes (Chapter 2, (Padoch et al. 2007b) will make it difficult for equitable benefit sharing among communities (Mertz et al. 2009). Ensuring carbon payments to farmers in the developing nations that have high levels of corruption is also challenging. Therefore, involving the indigenous people in the policy formation and decision-making process and ensuring their livelihood and food security are important pre-requisites for implementing REDD+ (Mukul et al. 2016b).

Potential of REDD+ in harnessing carbon and biodiversity co-benefits in India

Home to 200 million people dependent on forest resources in a mega-biodiversity region, conservation of forest and biodiversity has been India’s national priorities. India has a strong legal, policy, and institutional framework for forest and biodiversity protection that also recognize the rights of the indigenous communities (Sharma and Kohli). For instance, Wild Life Protection Act (1972) implements conservation, management and development of wildlife and its habitat in the country and National Forest Policy (1988) ensures involvement of people for sustainable forest management with a slightly stronger emphasis given to ecological security, while ensuring sustenance and livelihood security. Similarly, the Scheduled Tribes and Other Traditional Forest Dwelling Communities (Recognition of Forest Rights) Act (2006) recognises the forest rights of indigenous communities including the communities widely practicing shifting cultivation in Northeast India.

India has increasingly adopted PES approaches to generate incentives for forest protection and restoration. For example, a recent reform in India’s tax revenue has now initiated a system of ecological fiscal transfers through which the amount of tax revenue to state governments is determined by its forest cover, thus providing an incentive to protect and restore forests (Busch and Mukherjee 2018). The first of its kind in the world, this allows assessment of how payments for protecting and restoring forest can be effective at a national level and thus can facilitate REDD+ implementation in the country (Busch and Mukherjee 2018).

India has played a strategic role in REDD+ negotiation at international level and was instrumental in expanding REDD to incorporate enhancement of carbon stocks in developing countries (thus making REDD+) (Negi and Giessen 2018). India's first REDD+ project (The Khasi Hill Community REDD+ project; 2011-2015) can be a model for community participation in shifting cultivation areas. This initiative facilitated regeneration of a ~27000-hectare community owned area and generated a net carbon benefit of 223,263 tCO_2 in the Garo Hills of Northeast India (Poffenberger 2012). Similar approaches with active involvement of indigenous communities and aligning rural livelihood options with forest conservation can be effective in managing shifting cultivation in the region.

India has advanced in REDD+ readiness stage with Forest Survey of India (FSI) conducting a national forest inventory programme since 2002 under the Ministry of Environment Forest & Climate Change (Government of India). REDD+ methodologies for mapping drivers of forest change, development of baselines, quantification of emissions, stakeholder engagements and establishment of monitoring structures have been developed in 2017 (Burnwal 2017). Recently in 2018, India has submitted the proposed REDD+ strategy with forest reference levels for REDD+ result-based payments to UNFCC. Capacity building programs for REDD+ implementation have also been underway across various parts of India (Behera 2016).

The first REDD+ project in India, the Khasi Hill Community REDD+ project was conducted in Meghalaya by Community Forestry International (CFI) in 2011-2015 (Sun and Chaturvedi 2016). The project engaged ten indigenous Khasi communities covering an area of 27000 hectare for both forest protection and restoration through assisted natural regeneration. At the same time, this project also aimed to improve the livelihoods of 4,400 households living below the poverty line through carbon revenue sharing and other income-generating activities. This project is expected to reduce 860,104 tonnes of CO_2 emissions and generate over $7.5 million for Khasi Hills communities over its 30-years lifespan (Carbon Offsets To Alleviate Poverty report 2017). The Khasi Hill Community REDD+ project, one of the first REDD+ initiatives in Asia to be developed and managed by indigenous communities on communal lands has shown strong potential for such approaches to be successfully implemented in shifting cultivation areas of Northeast India. Nagaland with sixteen diverse Naga tribes that mostly practice shifting cultivation on community owned land can potentially replicate similar approaches under REDD+ to harness co-benefits for carbon and biodiversity while improving livelihoods.

Conclusion

The results from this thesis reveal that shifting cultivation maintains high levels of carbon stocks, avian species abundance and functional and evolutionary diversity, suggesting potential win-win outcomes for conservation interventions. Sparing old-growth forest from conversion into shifting cultivation by intensifying cropping in a smaller area was the most effective strategy in protecting and enhancing landscape carbon (Chapter 3). However, scenarios that retained heterogenous habitats with farmland and regenerating secondary forests benefited avian biodiversity, both in terms of species abundance (Chapter 4) and phylogenetic and functional diversity (Chapter 5). This implies the high conservation value of shifting cultivation landscapes in storing carbon and sustaining species diversity. With careful introduction of REDD+ and similar PES schemes, which account for the cultural diversity and societal inequalities, there is a strong potential for major carbon and biodiversity benefits in shifting cultivation.

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Appendix I.

Text S1. Crops grown in shifting cultivation landscapes in NE India

Cereals: Upland rice (*Oryza sativa*), maize (*Zea mays*), sorghum (*Sorghum bicolor, Sorghum vulgare*), job's tears (*Coix lachruma-jobi*), yam (*Dioscorea spp*.), pearl millet (*Pennisetum glaucum*), finger millet (*Eleusine coracana*), foxtail millet (*Setaria italica*), barley (*Hordeum vulgare*), wheat (*Triticum aestivum*).

Legumes and oil seeds: Sesame (*Sesamum indicum*), Black gram (*Phaseolus mungo*), Castor bean (*Ricinus communis*), Perilla (*Perilla frutescense*), French bean (*Phaseolus vulgaris*), Black sesame (*Hyptis spicigera*), Cowpea (*Vigna unguiculate*), Soybean (*Glycine max*), Rice bean (*Vigna umbellate*), Sem bean (*Dolichos lablab*),mustard (*Brassica juncea*), niger (*Guizotia abyssinica*), pigeon pea (*Cajanus cajan*), winged bean (*Psophocarpus tetragonolobus*), Indian bean (*Lab-lab purpureus*), rajma bean (*Phaseolus vulgaris*), sword bean (*Canavalia gladiate*).

Leaf, fruit and vegetables: Prince's-feather (*Amaranthus hypocondriacus, A. paniculatus*), pigweed (*Chenopodium album*), watermelon (*Citrullus lanatus*), pumpkin (*Cucurbita pepo, C. maxima, C. mosschata*), cucumber (*Cucumis sativus*), okra (*Hibiscus esculentus*), tomato (*Lycopersicon esculentum*), sponge gourd (*Luffa cylindrica*), pointed gourd (*Trichosanthes dioica*), snake gourd (*Trichosanthes anguina*), sweet gourd (*Momordica cochinchinensis*), banana (*Musa sapientum*), eggplant (*Solanaum melongena, S. xanthocarpum, S. indicum., S. berbisetum*), sugarcane (*Saccharum officinarum*), bitter melon (*Momordica charantia*), roselle (*Hibiscus subdariffa*), upland cotton (*Gossipium hirsutum*), buckwheat (*Phaphda teeta, P. meetha*), coriander (*Coriandrum sativum*), bottle gourd (*Lagenaria siceraria*), spinach (*Spinacia oleracea*), wax gourd (*Benincasa hipsida*), tukey berry (*Solanum torvum),* black nightshade (*Solanum nigrum*), *Solanum kurzii, Plantago major, Spilanthes paniculata, Clerodendrum viscosum,* Ash gourd (*Benincasa hispida*), broccoli (*Brassica oleracea*), cabbage (*Brassica oleracea*), cauliflower (*Brassica oleracea*), cho-cho (*Sechium edule*), kankro (*Momordica dioica*), *Cylanthera pedata, Moghania vestita*, pea (*Pisum sativum*), amaranth (*Amaranthus spp)*, lai (*Brassica juncea*), lafa (*Malva verticillata*), puroi sag (*Basella rubra*), sorrel (*Rumex rasicarius*).

Tuber, root and rhizomes: *Manihot esculenta, Colocasia anticuorum, Dioscorea bulbifera , Dioscorea allata, Colocasia esculenta*, chives (*Allium tuberosum*), East

Indian arrowroot (*Curcuma angustifolia*), elephant-foot yam (*Amorphophallus bulbifer*), giant taro (*Alocasia macrorrhiza*), greater yam (*Dioscorea alata*), lesser yam (*Dioscorea esculenta*), onion (*Allium cepa*), potato (*Solanum tuberosum*), potato yam (*Dioscorea bulbifera*), radish (*Raphanus sativus*), sweet potato (*Ipomoea batatas*), taro (*Alocasia indica*), turnip (*Brassica rapa*) Mann bada (*Allium spp.)*, Mann Chhota (*Allium spp*.).

Spices: Tabasco pepper (*Capsicum fructescence*), chili pepper (*Capsicum annum*), ginger (*Zingiber officinale*), garlic (*Allium sativum*), *Amaranthus viridis, Amaranthus spinosus,* cilantro (*Eryginum foetidum*), Chameleon plant (*Houttuynia cordata*), garlic chives (*Allium hookeri*), Indian pepper (*Zanthoxylum rhetsa*), winged prickly ash (*Zanthoxylum armatum*), Indian bay leaf (*Cinnamomum tamala*), Bengal cardamom (*Amomum aromaticum*), black cardamom (*Amomum subulatum*), Nepal camphor (*Cinnamomum glanduliferum*), Alpinia spp (*Alpinia calcarata, A. malaccensis*), galangal (*Kaempferia galangal*), wild pepper (*Piper nigrum, Piper longum, Piper peepuloides*), turmeric (*Curcuma longa*).

Source: Kushwaha & Ramakrishnan 1987; Singh & Sureja 2006; Singh 2009; Krug et al. 2013; Shimrah et al. 2015; Wangpan & Tangjang 2015.

Table S1. Various estimates of the area under jhum cultivation and tribes involved in this cultivation system across Northeast India. Sources: [1] Thangam 1984; [2]Task Force, Ministry of Agriculture Report on Shifting cultivation in India 1983; [3]Satapathy & Sarma 2003; [4]National Remote Sensing Agency(NRSA) 2003

State	Area under jhum/year1 (km2)	Area under jhum/year2 (km2)	Area under jhum/year3 (km2)	Area under jhum/year4 (km2)	Families2	People1	Tribe1
India	50000	9956	22690	18641	607536	3000000	
Assam	700	696	3100	3931	58000	403000	Dimasa (Kachari), Garo, Kachari, Karbi, or Mikir, Khasi, Kuki, Lalung and Naga
Andhra Pradesh	173	500	1030	7	23200	116000	Bagata, Jatapus, Konda Dhoras, Konda Kapus, Konda Reddi, Mukha Dhora, Samantha, Savaras and Valmiki
Arunachal Pradesh	920	700	2610	1613	54000	270000	Adi, Aka, Dafla or Nissi, Hill Miri, Khowa, Mishmi, Miji, Nocte, Tangsa and Wancho
Madhya Pradesh	81	125	0	20	2500	14000	Hill Maria
Manipur	600	900	3600	4817	70000	300000	Anal, Chothe, Hmar, Kabui, Kacha, Naga, Lamgang, Mao Maring, Paito Simte, Tangkhul, Thadou, Valphui and Zou
Meghalaya	760	530	2650	744	52290	350000	Hmar, Jaintia, Khasi, and Mikir
Mizoram	616	630	450	4018	50000	260000	Chakma, Hmar, Lakher, Mizo, Pawi and Riang
Nagaland	735	192	6330	1918	116046	400000	Naga, Kuki and Mikir
Orissa	5298	5298	1840	1177	141000	706000	Shuiya, Bondo Poraja, Didayi, Gadaba, Juang, Khond, Koya, (Lanjia) Saora and Paroja
Tripura	223	223	1080	396	43000	100000	Chakma, Halam, Jamatia, Lushai, Mag, Naotia, Riang and Tripuri

Appendix II.

Text S1. Changes in carbon stocks 5 years after management changes

At the end of 5 years, carbon stocks reduced by 18.4%, 30.8% and 71% from the baseline of 2687.9 ± 357.3 Mg/30 ha (mean ± SD) in a 15, 10 and 5-year cycle (Scenario 1.1, 1.2 and 1.3 respectively; Appendix II: Fig S3) under the business-as-usual scenario of no forest sparing. The second business-as-usual scenario of shifting cultivation expansion (Scenario 4) resulted in 79.6 % loss of the carbon stocks compared to an original old-growth forest landscape (13221.9 ± 1736.2 Mg/30 ha (mean ± SD)).

Landscape carbon increased under REDD+ interventions by secondary forest creation and sparing by 18.3 %, 23% and 26.3% in a 15-year (Scenario 2.1), 10-year (Scenario 2.2) and 5-year cycle (Scenario 2.3) respectively (Appendix II: Fig. S3). In pioneer shifting cultivation landscapes, only 15.7% carbon stock is lost when intervention is applied by sparing old-growth forest (50%, 66.6% and 83% of the landscape is protected in Scenario 3.1, 3.2 and 3.3 respectively; Appendix II: Fig. S3). Protecting 50% of old-growth forest whilst the rest of the landscape is managed with a 15-year shifting cultivation cycle (Scenario 3.1; Appendix II: Fig. S3) reduces carbon loss by 80.3% relative to a landscape managed entirely as shifting cultivation with a 5-year cycle (Scenario 1.3; Appendix II: Fig. S3).

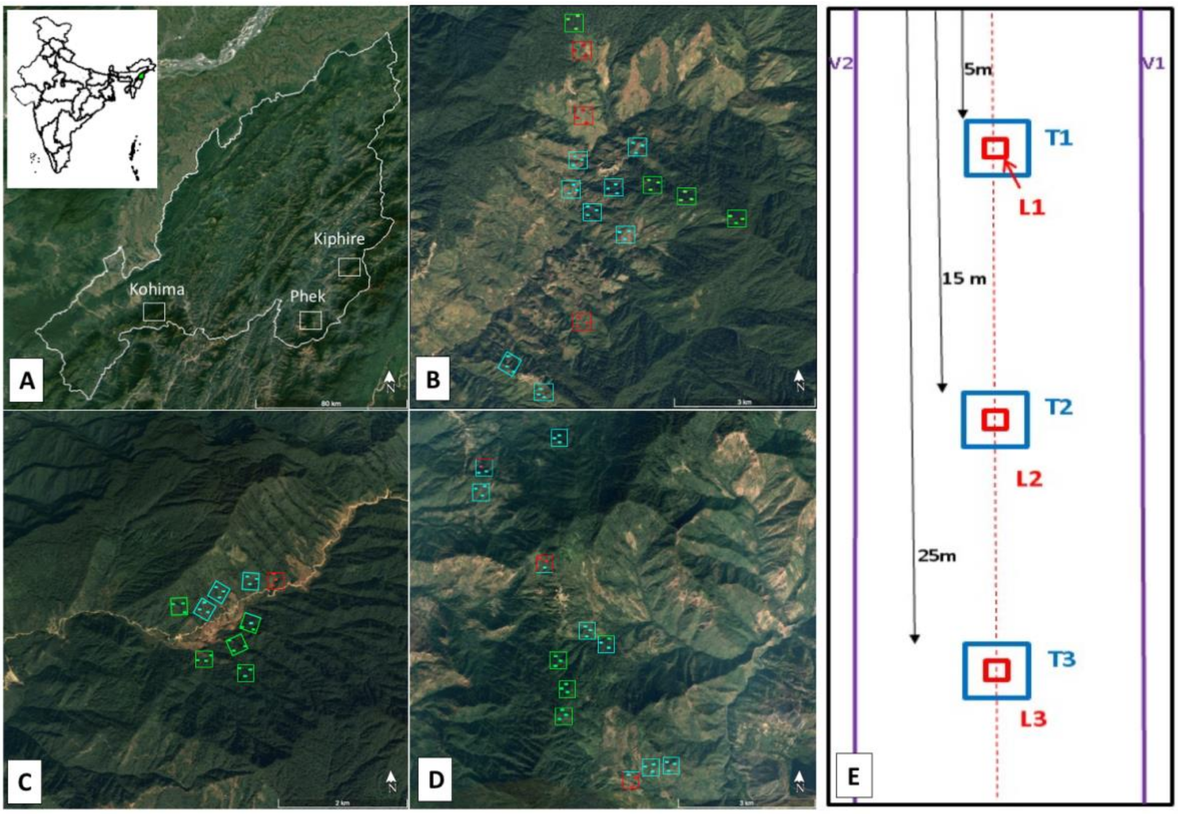

Figure S1. Details of study location and sampling design. The study took place in Nagaland, Northeast India (A), with carbon sampling in a total of 36, 400 m × 400 m squares across three landscapes: Kiphire (B), Kohima (C) and Phek districts (D). Colour of the squares denotes the three habitat types: farmland (red), regenerating secondary forest (blue) and old-growth forest (green). Each sampling square consisted of three 10 m × 30 m plot (E) containing three 2 m^2 sampling sub-plots (T1-3), within each of which there was a 1 m^2 central plot (L1-3), and two 1 m × 30 m sub-plots along the plot margins (V1-2).

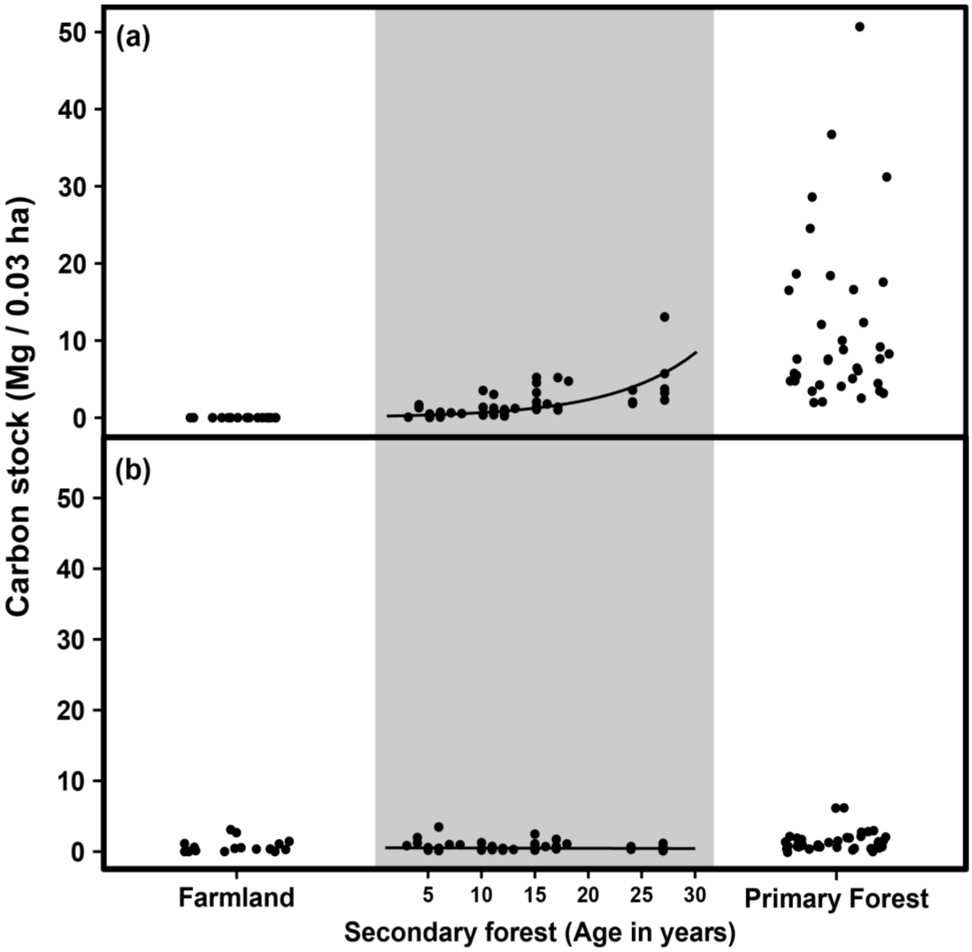

Figure S2. (a) Live and (b) dead carbon accumulation across the three habitats, farmland, secondary forest with age, and old-growth forest plots in Nagaland, Northeast India. Black line in secondary forest (age in years) in shows fitted linear mixed effect model.

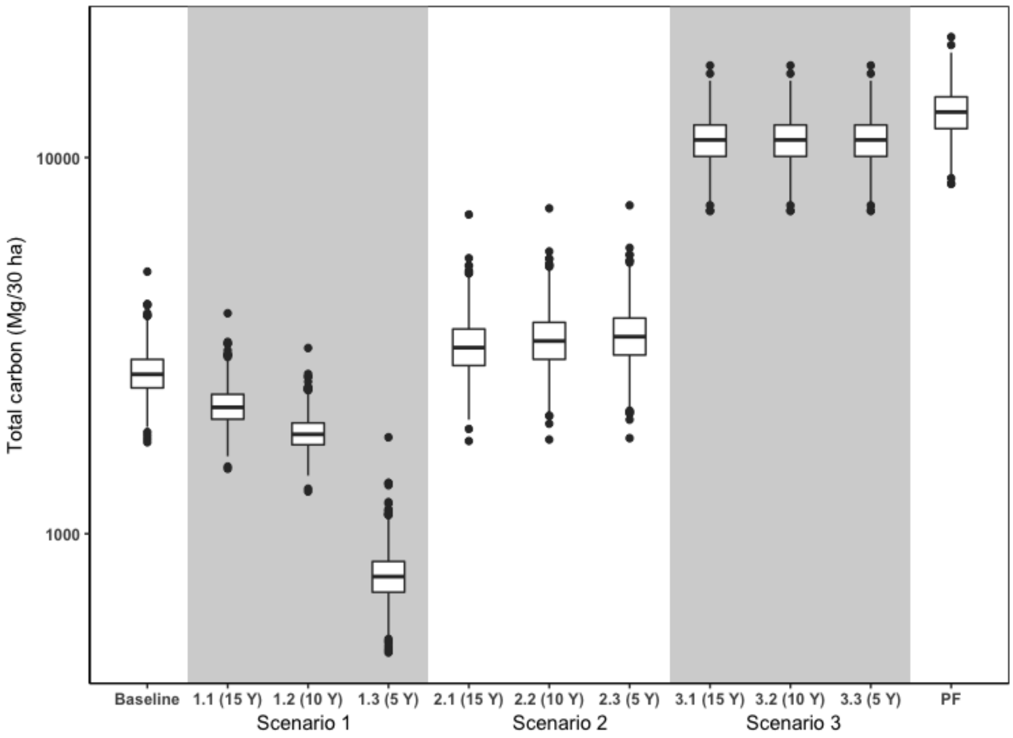

Figure S3. Boxplots showing the difference in landscape level carbon stock under three alternative management regimes of shifting cultivation at the end of 5 years relative to a baseline of 30-year cultivation cycle (Baseline) and old-growth forest landscape (PF) (i)'Business-as-usual' scenario with no forest sparing in Scenario 1 (Scenario 1.1, 15-year cycle; Scenario 1.2, 10-year cycle; Scenario 1.3, 5-year cycle) (ii) REDD+ intervention by secondary forest creation and sparing in Scenario 2 (Scenario 2.1, 15-year cycle; Scenario 2.2,10-year cycle; Scenario 2.3, 5-year cycle) and by restricting shifting cultivation expansion with protection of old-growth forest in Scenario 3 (Scenario 3.1, 15-year cycle; Scenario 3.2,10-year cycle; Scenario 3.3, 5-year cycle).

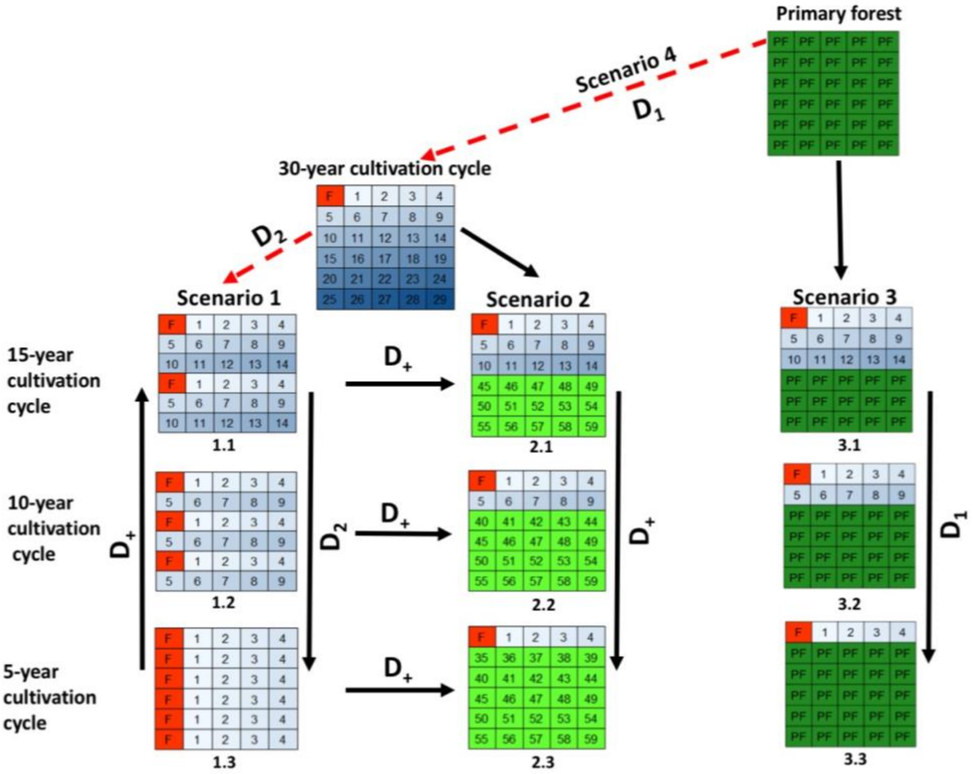

Figure S4. Potential pathways for REDD+ investment to protect and enhance forest carbon stocks in a shifting cultivation landscape under the four sets of management scenarios- avoided deforestation (D_1), avoided forest degradation (D_2) and enhancement of forest carbon (D_+). The scenarios depict no forest sparing (Scenario 1), secondary forest creation and sparing (Scenario 2), new shifting cultivation landscape with sparing old-growth forest (Scenario 3) and shifting cultivation expansion (Scenario 4). Colours indicate habitat types: farmland (F, red), active fallows (1-29 years, different shades of blue), abandoned old fallows (>30 years, light green) and old-growth forests (PF, dark green). Numbers within cells denote the age of the secondary forests.

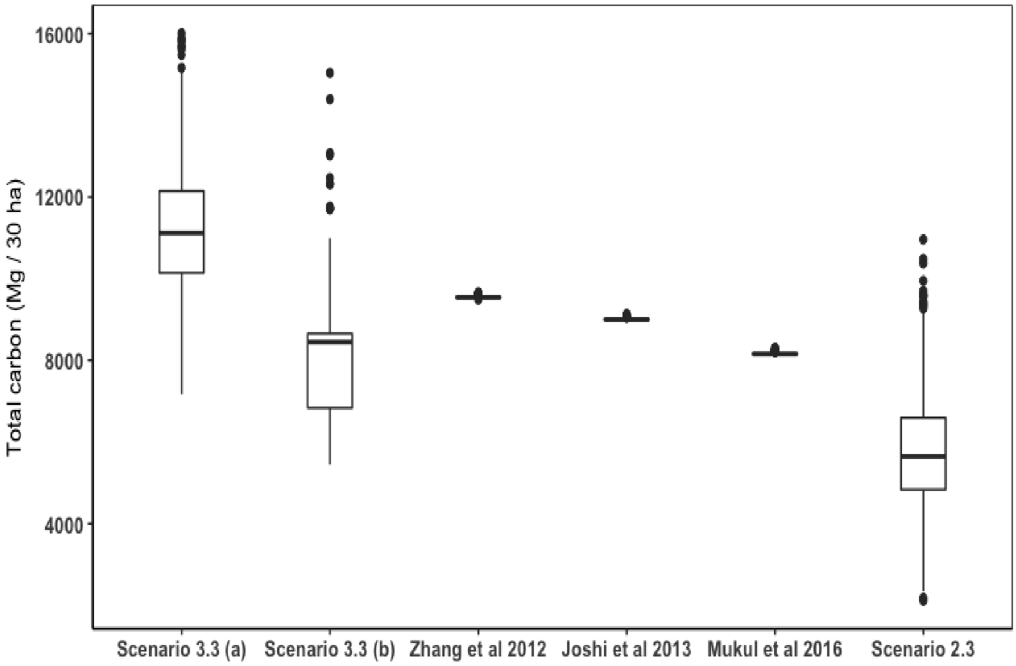

Figure S5. Comparison of total carbon stocks in old-growth forest sparing (Scenario 3.3) and the next best scenario of secondary forest creation and sparing (Scenario 2.3) between this study and three previous studies from montane Asia (Zhang et al. 2013, China; Joshi et al. 2013, India and Mukul et al. 2016, Philippines). Scenario 3.3 (a) and Scenario 3.3 (b) were simulated with mean and median carbon estimates from old-growth forest of this study, respectively.

Table S1. Study site details and sampling effort in Nagaland, Northeast India.

Landscapes	Kiphire	Kohima	Phek
Villages	Fakim, Tsundang, Thanamir	Dzuleke	Wazeho, Zhipu, Washelo
Elevation (m)	1722-2652	1716-2001	1487-2309
Location (lat, long)	25°48′18.58″ N	25°39′13.76″ N	25°36′21.00″ N
	94°58′40.28″ E	94°43′31.92″ E	94°01′32.43″ E
Sampling points - Farmland	9	3	5
Sampling points - secondary forest	24	10	21
Sampling points - old-growth forest	12	14	10
Total number of sampling points	45	27	36

Table S2. Allometric equations used for biomass estimation of trees, roots and liana sampled in Nagaland, Northeast India; where, AGB=Above ground biomass for individual tree, DBH=diameter at breast height, ρ=wood specific gravity, E= (0.178*TS-0.938* CWD-6.61*PS)*10^{-3}, E-measure of environmental stress, TS- temperature seasonality, CWD-climatic water deficit, PS-precipitation seasonality, diameter at 30 cm=1.235 DBH+0.002 $(DBH)^{2}$ and BA-Basal area

Component	Source	Region	Equation	Sample size
Trees, old-growth forest	Dung et al.2012	Vietnam	AGB = 0.3429 (DBH)2.3028 (ρ)1.2901	201
	Chave et al.2014	Pantropical	AGB= exp [-1.803-0.976 E + 0.976 (log ρ) + 2.673 (log DBH) -0.0299(log DBH)2]	
Trees, Secondary forest	Ketterings et al.2001	Indonesia	AGB= exp [-2.207 + 2.62 (ln DBH) + (ln ρ)]	29
	Van Breugel et al.2011	Central America	AGB = exp [-1.130+ 2.267 (ln DBH) + 1.186 (ln ρ)]	244
	Chave et al.2014	Pantropical	AGB=exp [-1.803-0.976 E+0.976 (ln ρ) + 2.673 (ln DBH) -0.0299 (ln DBH)2]	4004
Liana	Putz 1983	Amazon	AGB = exp [0.12+0.91 log (BA)]	17
	Gehring et al.2005	Amazon	AGB = exp [-7.114+2.276 ln (diameter at 30 cm)]	561
	Schnitzer et al.2006	S. America	AGB=exp [-1.484+2.657 (ln DBH)]	424
	Sierra et al.2007	Colombia	AGB= exp [0.028+1.841 (ln DBH)]	33
	Addo-Fordjour and Rahmad 2013	Malaysia	AGB=0.262+1.934 (DBH)	60

Table S3. Details of the top three best models based on AICc values obtained from linear mixed effect model applied to assess change in total, live and dead carbon stock across the three habitats, i.e. farmland, secondary and old-growth forest

Model	Habitat	Elevation	Habitat * Elevation	df	Log Likelihood	AICc	delta	weight	Marginal R2	Conditional R2
Lmer (log10(Total carbon) ~ Habitat + (1| Landscape/Square)	+	NA	NA	6	-54.74	122.3	0.00	0.59	0.57	0.76
Lmer (log10(Total carbon) ~ Habitat + Elevation + (1|Landscape/Square)	+	0.09	NA	7	-54.24	123.6	1.28	0.31	0.57	0.76
Lmer (log10(Total carbon) ~ Habitat + Elevation + (1|Landscape/Square) + Age*Elevation	+	0.19	+	9	-53.09	126.0	3.72	0.09	0.59	0.75
Lmer (log10(Live carbon) ~ Habitat + Elevation+ Habitat* Elevation+ (1| Landscape/Square)	+	0.07	+	9	-49.57	119	0.00	0.49	0.81	0.85
Lmer (log10(Live carbon) ~ Habitat + (1| Landscape/Square)	+	NA	NA	6	-53.39	119.6	0.65	0.36	0.79	0.85
Lmer (log10(Live carbon) ~ Habitat + Elevation + (1| Landscape/Square)	+	0.04	NA	7	-53.18	121.5	2.50	0.14	0.79	0.85
Lmer (log10(Dead carbon) ~ Habitat + (1| Landscape/Square)	+	NA	NA	6	-46.01	104.9	0.00	0.72	0.19	0.65
Lmer (log10(Dead carbon) ~ Habitat + Elevation+ (1| Landscape/Square)	+	0.001	NA	7	-46.01	107.2	2.29	0.23	0.19	0.65
Lmer (log10(Dead carbon) ~ Habitat +Elevation+ Habitat * Elevation+ (1| Landscape/Square)	+	-0.21	+	9	-45.67	111.2	6.32	0.03	0.20	0.67

Table S4. Details of the top three best models based on delta AIC values obtained from linear mixed effect model applied to assess change in total, live and dead carbon stock fallow ages in secondary forest.

Model	Age	Elevation	Age* Elevation	df	Log Likelihood	AICc	delta	weight	Marginal R2	Conditional R2
Lmer (log10 (Total carbon) ~ Age + (1|Landscape/Square)	0.589	NA	NA	5	- 6.948	25.1	0.00	0.728	0.374	0.640
Lmer (log10 (Total carbon) ~ Age + Elevation + (1|Landscape/Square)	0.588	-0.019	NA	6	-6.936	27.6	2.50	0.208	0.378	0.638
Lmer (log10 (Total carbon) ~ Age + Elevation + (1|Landscape/Square) + Age*Elevation	0.636	-0.122	0.135	7	-6.813	30.0	4.89	0.063	0.379	0.634
Lmer (log10 (Live carbon) ~ Age + (1| Landscape/Square)	0.71	NA	NA	5	-21.79	54.8	0.00	0.629	0.51	0.65
Lmer (log10 (Live carbon) ~ Age+ Elevation + (1| Landscape/Square)	0.69	-0.15	NA	6	21.33	56.4	1.62	0.280	0.53	0.65
Lmer (log10 (Live carbon) ~ Age + Elevation + Age* Elevation+ (1| Landscape/Square)	0.75	-0.25	0.17	7	-21.14	58.7	3.86	0.091	0.54	0.65
Lmer (log10 (Dead carbon) ~ (1| Landscape/Square)	NA	NA	NA	4	-17.29	43.4	0.00	0.56	0.00	0.49
Lmer (log10 (Dead carbon) ~ Age + (1| Landscape/Square)	-0.06	NA	NA	5	17.17	45.6	2.17	0.19	0.01	0.51
Lmer (log10 (Dead carbon) ~ Elevation + (1| Landscape/Square)	NA	0.06	NA	5	-17.24	45.7	2.31	0.18	0.00	0.49

Appendix III.

Table S1. Model selection statistics for the fixed and random effects; Gelman Rubin convergence diagnostic (Rhat), Bayesian p value and deviance information criterion (DIC) for summer. For each community, the best model was selected as the model that minimizes the hierarchical penalized measure of model fit, DIC.

Rank	Model	Rhat	Bayesian p	DIC
1	ψ (habitat + landscape[R])	1.007	0.647	0.431
2	ψ (habitat); θ (time + weather)	1.013	0.661	0.434
3	ψ (habitat); θ (time)	1.006	0.666	0.434
4	ψ (habitat); θ (weather)	1.008	0.618	0.437
5	ψ (habitat)	1.008	0.635	0.438
6	ψ (habitat + landscape[R]); θ (time + weather)	1.005	0.686	0.439
7	ψ (habitat + landscape[R]); θ (time)	1.005	0.695	0.440
8	ψ (habitat + elevation + landscape[R]); θ (time)	1.007	0.699	0.441
9	ψ (habitat + elevation + landscape[R]); θ (time + weather)	1.006	0.694	0.441
10	ψ (habitat + landscape[R]); θ (weather)	1.006	0.643	0.442
11	ψ (habitat + elevation + landscape[R]); θ (weather)	1.016	0.648	0.443

Table S2. Model selection statistics for the fixed and random effects; Gelman Rubin convergence diagnostic (Rhat), Bayesian p value and deviance information criterion (DIC) for winter. For each community, the best model was selected as the model that minimizes the hierarchical penalized measure of model fit, DIC.

Rank	Model	Rhat	Bayesian p	DIC
1	ψ (habitat + landscape[R])	1.027	0.602	0.456
2	ψ (habitat); θ (time + weather)	1.027	0.646	0.460
3	ψ (habitat); θ (time)	1.012	0.632	0.461
4	ψ (habitat + landscape[R]); θ (time)	1.023	0.647	0.466
5	ψ (habitat + landscape[R]); θ (time + weather)	1.030	0.658	0.466
6	ψ (habitat + elevation + landscape[R]); θ (time + weather)	1.030	0.651	0.467
7	ψ (habitat)	1.023	0.591	0.468
8	ψ (habitat); θ (weather)	1.039	0.602	0.468
9	ψ (habitat + elevation + landscape[R]); θ (time)	1.006	0.646	0.469
10	ψ (habitat + elevation + landscape[R]); θ (weather)	1.057	0.607	0.476
11	ψ (habitat + landscape[R]); θ (weather)	1.023	0.612	0.476

Table S3. Posterior means and standard deviations for hyper-parameters representing community-wide responses to a range of habitat predictor variables in summer and winter. All predictors were centred and standardized prior to analysis, allowing direct comparability of effect sizes. Values for ‘FA’ (farmland), ‘VYSec’ (very young secondary forest), ‘YSec’ (young secondary forest) and ‘AdSec’ (advanced secondary forest) show the mean change in species occurrence probabilities relative to sampling points in old-growth forests.

Season	Predictor variable		Mean	SD	95% Posterior intervals	
Summer	$\alpha 1$	FA	0.66	1.49	-1.92	3.98
	$\alpha 2$	VYSec	0.48	1.13	-1.57	2.90
	$\alpha 3$	YSec	0.72	1.12	-1.27	3.15
	$\alpha 4$	AdSec	0.60	0.79	-0.88	2.23
Winter	$\alpha 1$	FA	0.95	0.94	-0.82	3.07
	$\alpha 2$	VYSec	1.35	0.58	0.19	2.57
	$\alpha 3$	YSec	0.72	0.92	-1.17	2.65
	$\alpha 4$	AdSec	0.81	0.62	-0.32	2.21

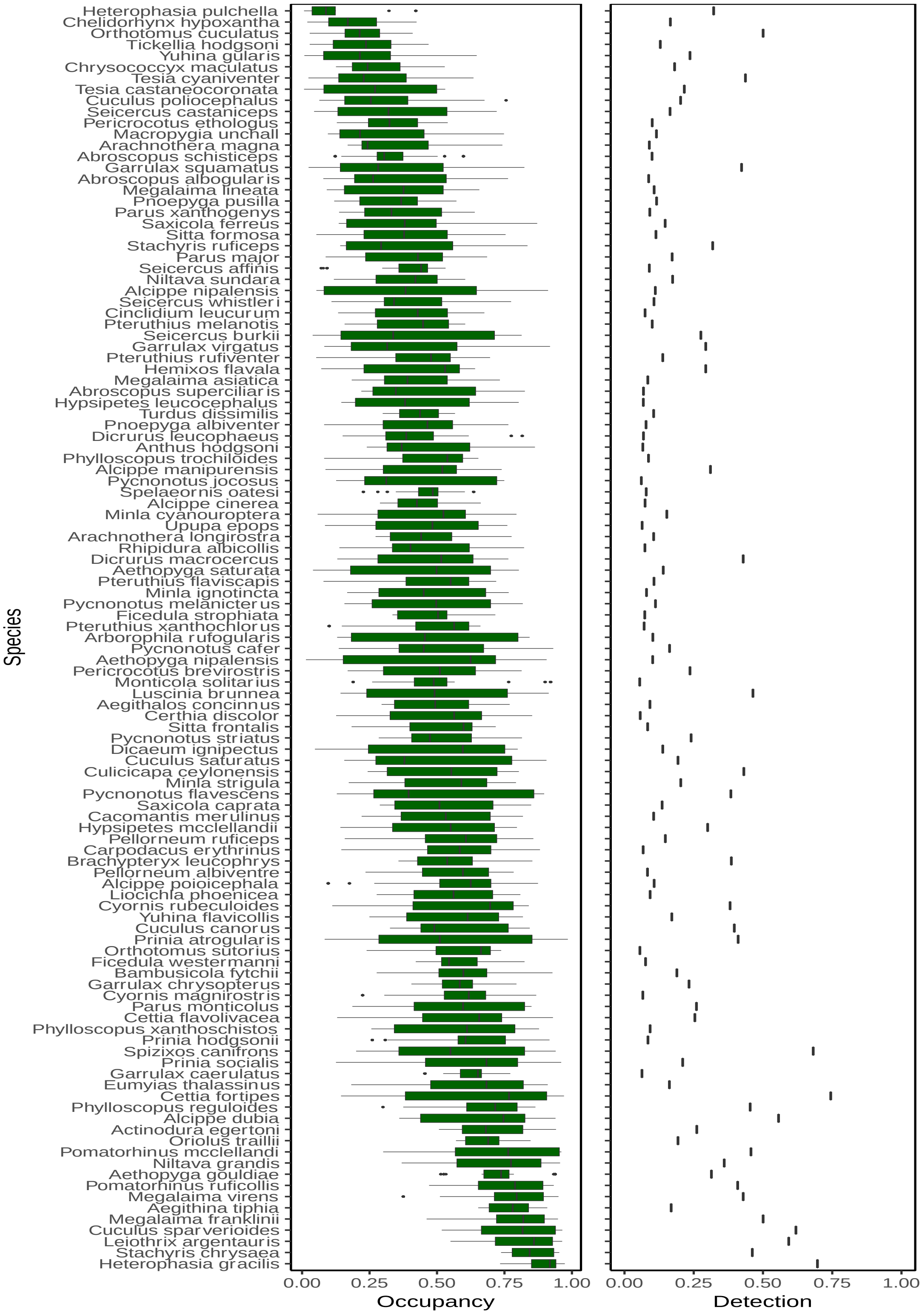

Figure S1. Species-level mean occurrence and detection probabilities in summer bird community in Nagaland.

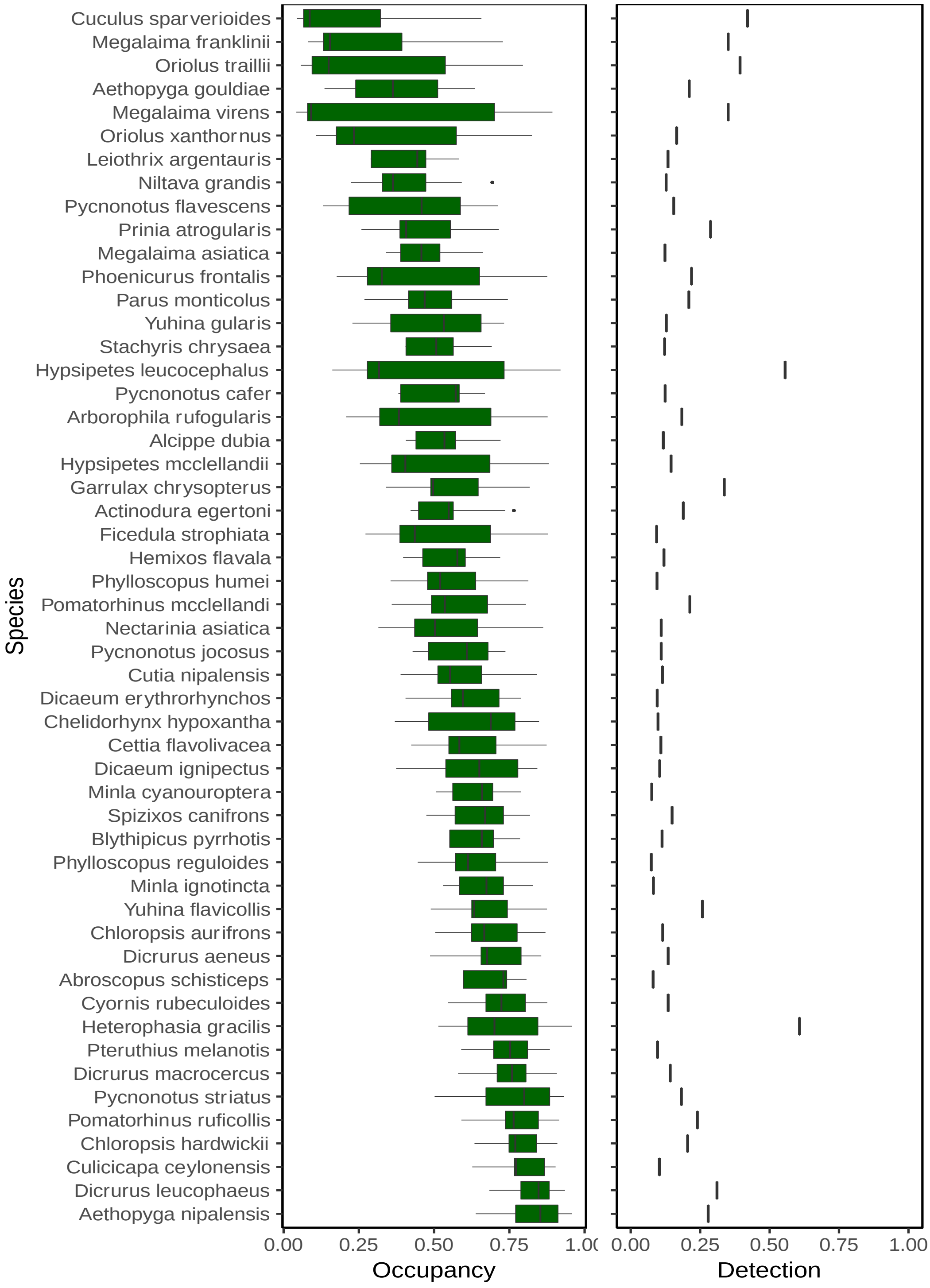

Figure S2. Species-level mean occurrence and detection probabilities in winter bird community in Nagaland

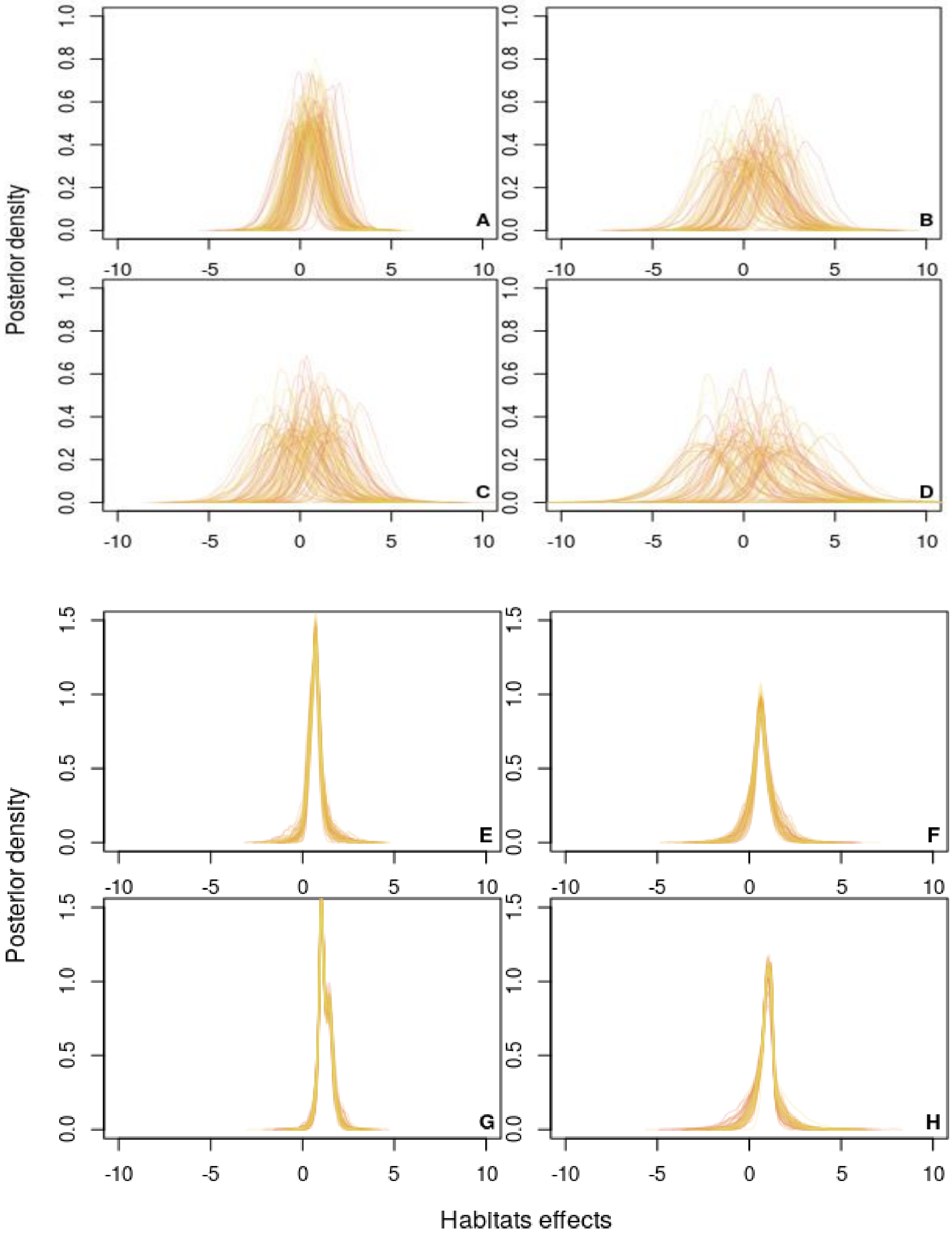

Figure S3 Posterior probability distributions of model parameters determining species occupancy across farmland (A and E), very young secondary forest (B and F), young secondary forest (C and G) and advanced secondary forest (D and H) in summer (A to D) and winter (E to H) bird community. The spread of each distribution indicates the level of uncertainty surrounding parameter estimates.

APPENDIX IV

Text S1. Description and calculations of the phylogenetic and functional metrics.

Phylogenetic metrics

I used six metrics to examine patterns of phylogenetic diversity across communities and REDD+ management scenarios.

1. **Phylogenetic Diversity (PD)** implies the total evolutionary history in a community which is equivalent to Faith's Index measured in millions of years. PD is calculated by summing the branche lengths in a phylogenetic tree that connects all species in a community (Faith 1992). This represents the sum of evolutionary history in the observed community.
2. **Standard Effect Size of PD (ses PD)** is measured as the proportionate difference between observed PD and the null expectation given the species richness of the sampled community (compared against 999 null communities for every phylogenetic tree in this study). It is derived from 999 randomizations in which N species were randomly drawn from a pool containing all species for each season in the study, where N was the number of species detected within the observed community.
3. **Mean Pairwise Distance (MPD)** is the average distance on a phylogenetic tree among all species of a community, implying the average number of years of evolutionary history separating all species in a community (Webb et al 2002). High MPD values indicate that species in the community are widely distributed across clades and thus have long diverged from other species in the community, whereas Low MPD values indicates phylogenetically clustering among species.
4. **Standard Effect Size of MPD (sesMPD)** is MPD accounting for species richness and is evaluated relative to an abundance-weighted null distribution for each community sample similar to sesPD. Positive values of sesMPD imply that communities have higher MPD (less closely related species) than expected for that given species richness – assuming that species were drawn at equal numbers randomly from the regional pool of species – and the opposite applies for negative sesMPD values (Swenson, 2014).
5. **Mean Nearest Taxon Distance (MNTD)** is the average number of years separating each species from its closest non-conspecific relative in the

community (Webb et al 2002). High MNTD values suggest that individuals within families that are closely related do not co-occur in the community, while low MNTD values suggest the opposite pattern.

6. **Standard Effect Size of MNTD (sesMNTD)** is MNTD considering species richness and is calculated the same way as sesPD. Positive values of sesMNTD imply that communities have higher MNTD (phylogenetically more evenly distributed within families or genera) than expected for that given species richness and the opposite (phylogenetically more clustered species within families or genera) applies for negative sesMNTD values (Swenson, 2014).

Evolutionary distinctiveness measures

1. **Evolutionary Distinctiveness (ED)** is a measure of unique evolutionary history represented by a species in a phylogenetic tree. It is calculated by adding up the total length of edges in a phylogenetic tree and dividing each edge is by the number of descendants for that species. Global evolutionary distinctiveness (ED) measures how isolated a given species is on the global phylogeny of 9,993 species (Arroyo-Rodriguez et al 2012) whereas local ED measures the evolutionary isolation of a species within the observed community (Cosset et al 2017). Species from monotypic families have high ED, whereas species with many close relatives have low ED. I calculated ED values for each species encountered in the study by taking the mean value from all 500 trees, using the ‘fair proportion’ metric (Jetz et al 2014). I then calculated the mean ED across each sampled and simulated community.
2. **Evolutionary distinctiveness rarity (EDR)** indicates the extent to which ED is concentrated in space (Edwards et al 2015) and is measured by dividing the ED by a species’ global geographic range size (Jetz et al 2014). A high EDR value for a given species therefore implies both high importance for the conservation of evolutionary diversity, and a high risk of extinction associated with a small global range size (Jetz et al 2014). I calculated mean EDR for each species by dividing the mean ED by its global range size (km2) using values taken from Jetz et al 2014.

Functional metrics

1. **Functional diversity (FD)** is the sum of branch lengths in a dendrogram generated from functional trait differences (Petchey & Gaston, 2002). Higher

FD implies greater differences between species representing the diversity of ecological interactions present within communities (Srivastava et al., 2012).

2. **sesFD** is a measure of functional diversity compared to expected functional diversity by a process of random assembly from the overall regional species pool (Edwards et al 2012, Prescott et al., 2016). It is calculated as, sesFD = [observed FD – mean expected FD]/standard deviation of expected FD. Observed FD represent values calculated from raw data whilst expected FD was calculated per point from 1000 random communities comprised of all sampled species in each season. I used an independent swap algorithm within the randomizeMatrix function (picante R package) (Gotelli, 2000; Kembel et al., 2010) to maintain species richness and frequency within each point.
3. **Functional Mean Pairwise Distance (fMPD)** is the average distance on a functional dendrogram among all species of a community, implying widely or closely they are spaced in the functional trait space. High fMPD value will suggest that species are functionally diverse in the community whereas low fMPD will indicate functional clustering with multiple species with similar functional traits co-occurring in the community.
4. **Standard Effect Size of functional MPD (sesfMPD)** is MPD accounting for species richness and is evaluated relative to an abundance-weighted null distribution for each community sample. Positive values of sesfMPD suggest that communities have less functional overlap than expected for that given species richness and the opposite applies for negative sesfMPD values.
5. **Functional Mean Nearest Taxon Distance (fMNTD)** indicates how closely or widely a species is distributed in the functional trait space from its closest non-conspecific relative in the community. High fMNTD values suggest that closely related species do not co-occur in the community, low fMNTD suggests functionally similar species co-existing in the community.
6. **Standard Effect Size of functional MNTD (sesfMNTD)** is functional MNTD as expected for the given species richness. Positive values of sesfMNTD imply that communities are functionally more evenly distributed within families or genera than expected for that given species richness and negative sesfMNTD values suggest that species are functionally more clustered within families or genera.

Text S2. Morphometric traits

The morphometric traits were measured from museum specimens (n = 5139) and obtained primarily from the avian skin collection at the Natural History Museum, Tring, and the Manchester Museum (Chira et al 2018, Cooney et al 2017). Where available, one mature male per species was selected for scanning as males are generally better represented in the collections than females. When undamaged males were unavailable, females were preferentially chosen over unsexed specimens. I used Principal Components Analysis (PCA) to reduce intercorrelation between the morphometric traits and extract ecologically meaningful axes of body measurements (bill length, bill depth, bill width, wing length, tail length and tarsus length) that represents dispersal ability and resource use. Before analysing, I log-transformed (natural log) all traits to reduce the disproportionate impact of a few very large species (Pigot et al. 2016). I extracted the first axis which explained the majority of the variation (73.6%) and corresponded to bill morphology.

The information on bill shape was taken using 3D scans of museum study skins comprising >2,000 species (>97% of extant genera) representing the full range of bill-shape diversity. Morphologically homologous points (‘landmarks’) on bills were placed using a bespoke crowdsourcing website (http://www.markmybird.org) and the bill-shape morphological space was quantified using Procrustes superimposition and principal component analyses. The first eight principal component (PC) axes explained >99% of the total variation in bill shape. PC1 (58% of overall shape variation) described the volumetric aspect ratio from elongated to stout bills and captures the range of shape variation encompassed by standard linear measurements (length, width and depth) (Cooney et al 2017).

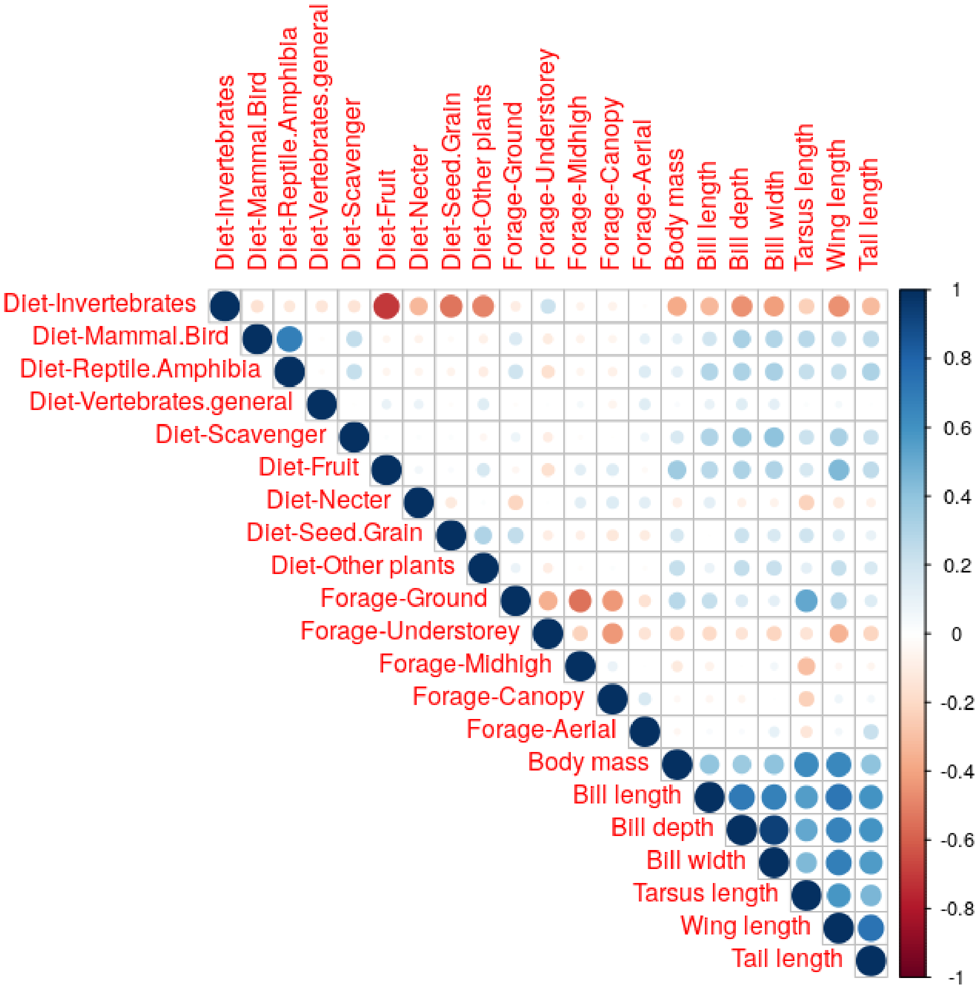

Figure S1. Correlation between all 21 species traits. Larger circles represent greater correlation between two traits and colour indicates the relationship (blue = positively correlated, red = negatively correlated). Darker colours depict more correlated traits.

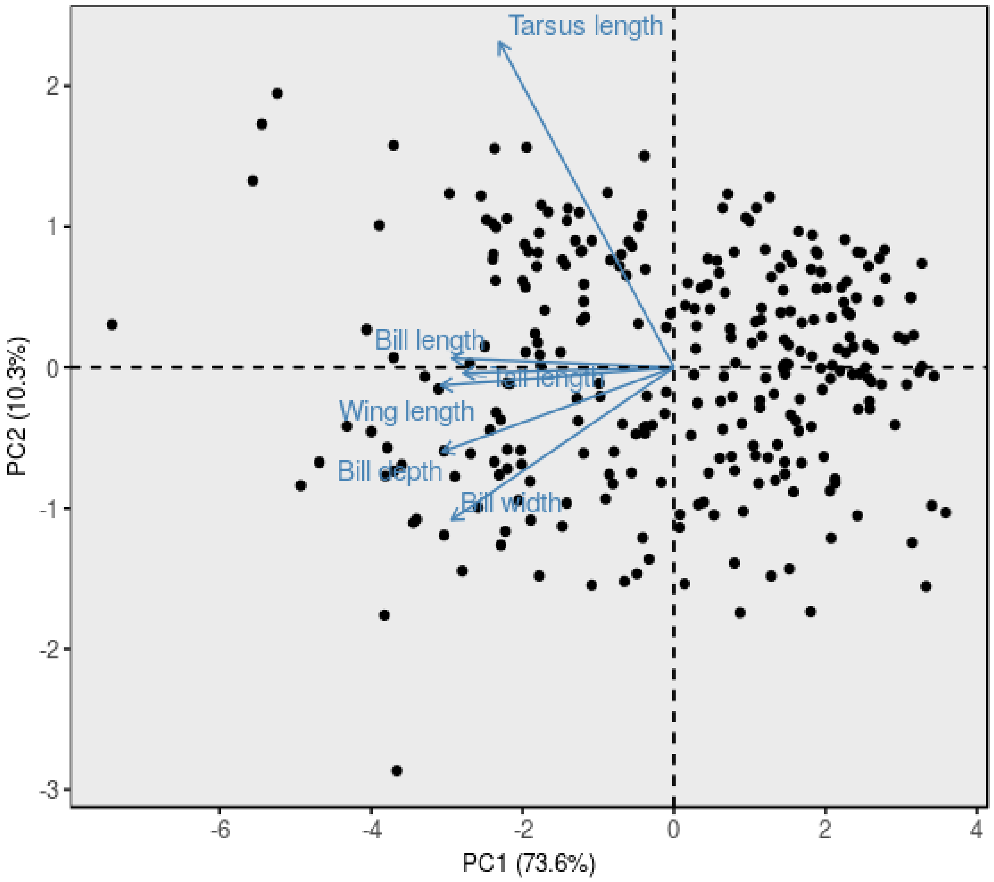

Figure S2. Ordination (PCA) of bird species by the seven morphological traits described in Table S2. The first axis with an eigen value 4.42 explained 73.6 % of the variation in the original data.

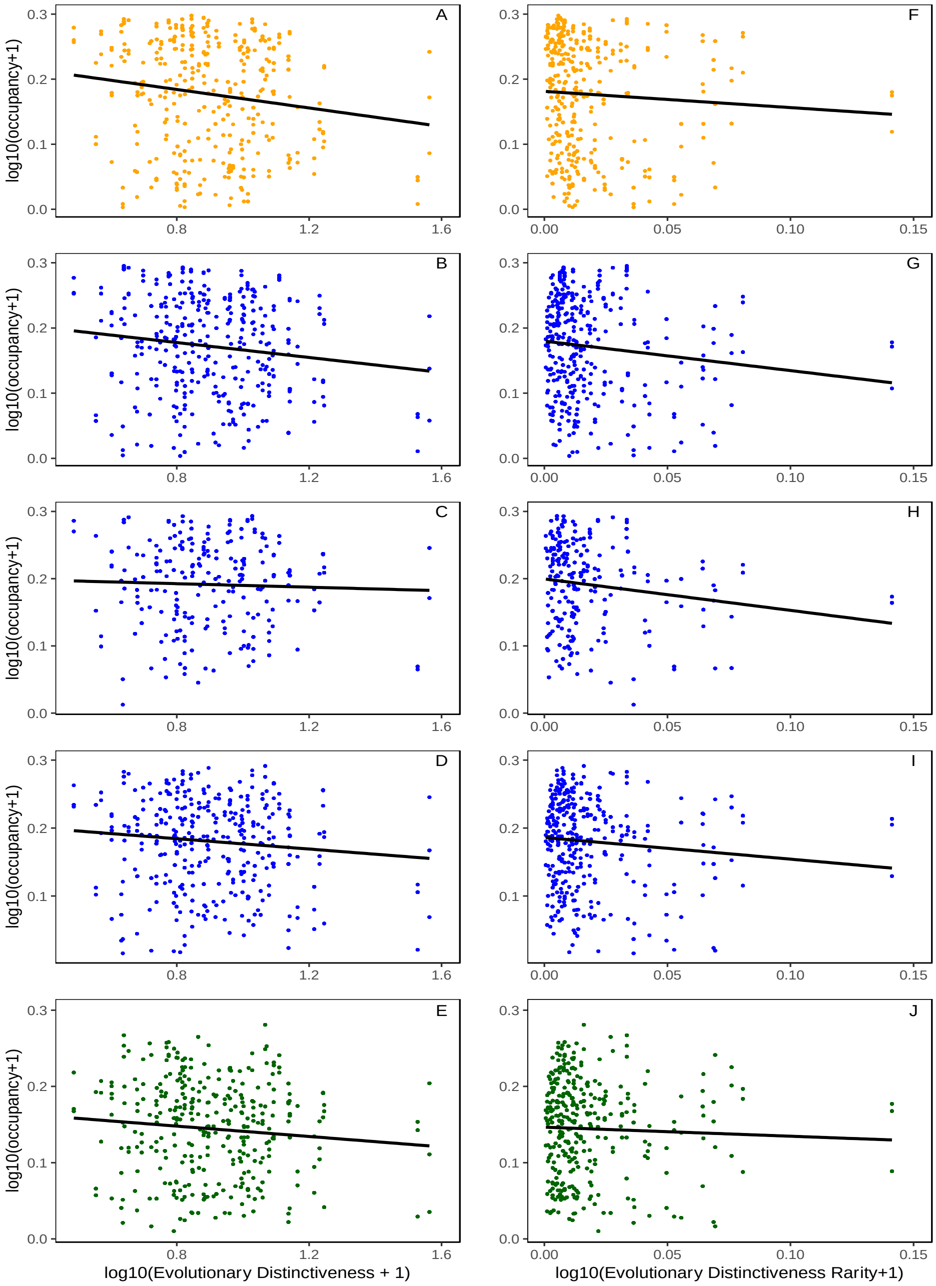

Figure S3. Species occurrence probabilities related to species evolutionary distinctiveness across farmland (A and F), very young secondary forest (B and G), young secondary forest (C and H), advanced secondary forest (D and I) and old growth forest (E and J) in summer

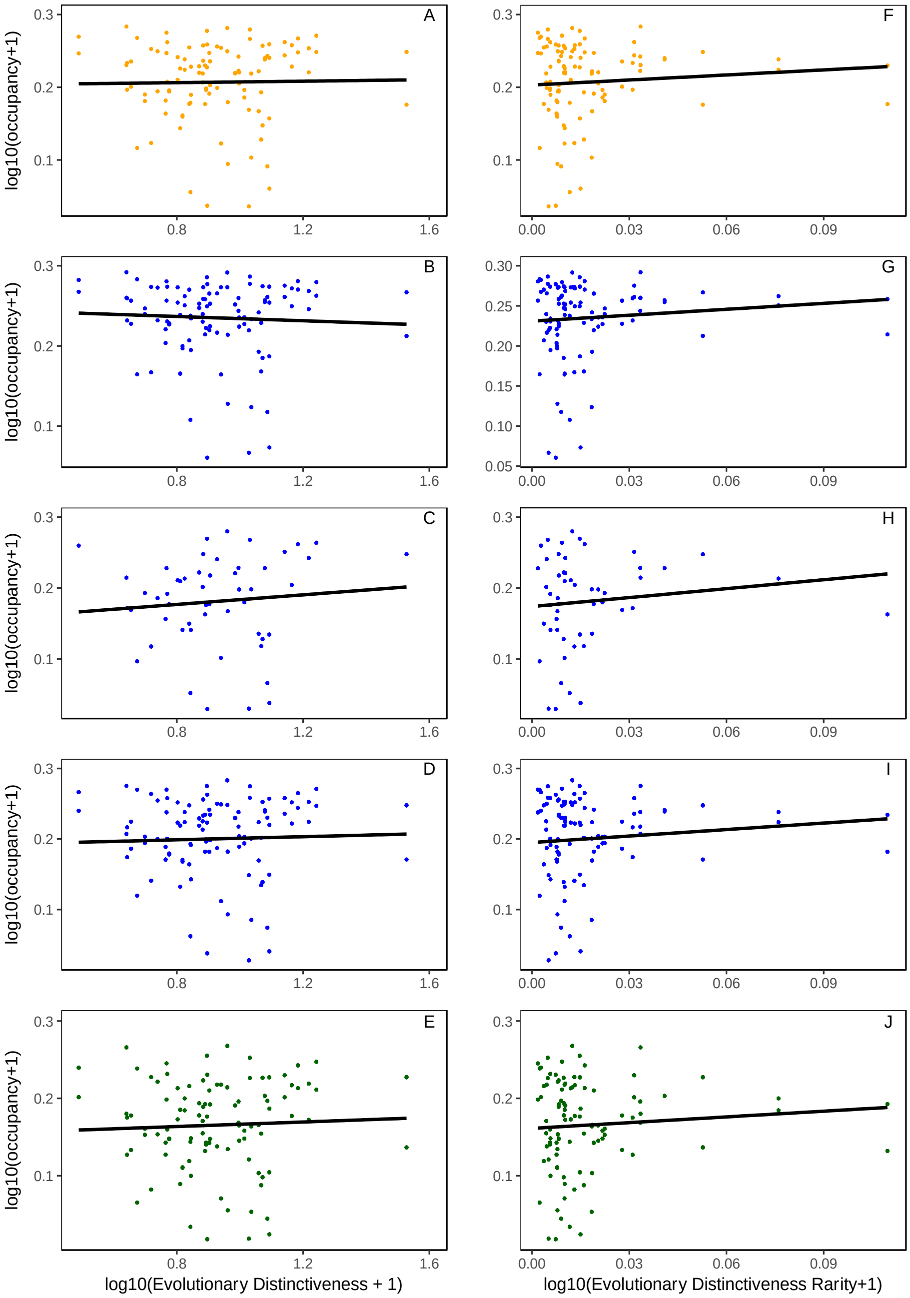

Figure S4. Species occurrence probabilities related to species Evolutionary Distinctiveness Rarity (EDR) across farmland (A and F), very young secondary forest (B and G), young secondary forest (C and H), advanced secondary forest (D and I) and old growth forest (E and J) in winter.

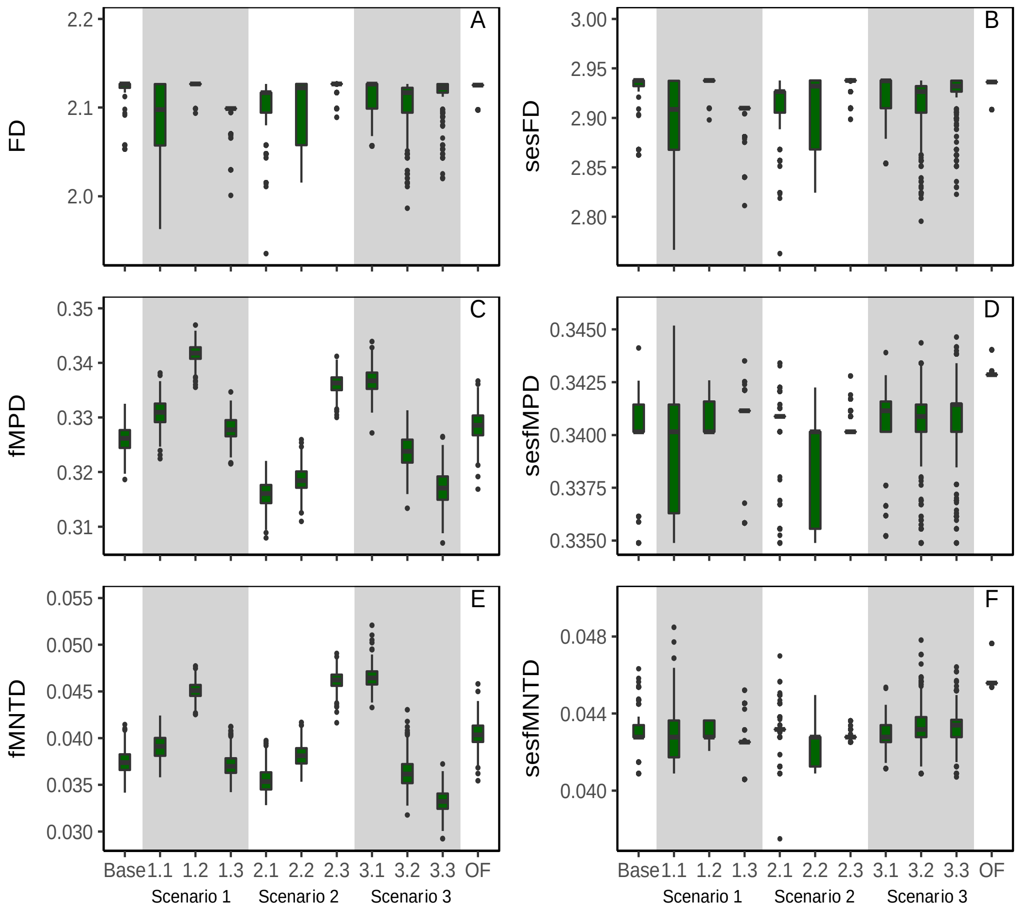

Figure S5. Predicted changes in functional diversity in winter under the four alternative management regimes of shifting cultivation at the end of 30 years (1)'Business-as-usual' with no forest sparing in Scenario 1 (Scenario 1.1, 15-year cycle; Scenario 1.2, 10-year cycle; Scenario 1.3, 5-year cycle) ; (2) intervention scenarios by secondary forest creation and sparing in Scenario 2 (Scenario 2.1, 15-year cycle; Scenario 2.2,10-year cycle; Scenario 2.3, 5-year cycle) and old-growth forests sparing in Scenario 3 (Scenario 3.1, 15-year cycle; Scenario 3.2, 10-year cycle; Scenario 3.3, 5-year cycle). Mean values from 1000 randomisations under each scenario indicated by points, with error bars representing 95th percentiles.

TABLE S1. Species richness and abundance in the overall bird communities across two seasons, summer (N=108) and winter (N=63). In Nagaland, Northeast India Total and mean species richness and abundance are shown for the entire dataset (Overall) and for each habitat type (OF= old-growth forests; SF=secondary forests; FA= farmland). Mean species richness and abundance are calculated per point (Overall birds) ± the standard error (SE).

	Habitat	Summer		Winter	
		Total	Mean (± SE)	Total	Mean (± SE)
Species richness	Overall	257	34.12 ± 0.99	193	20.82 ± 0.93
	FA	152	36.00 ± 1.47	95	23.25 ± 2.44
	SF	223	36.74 ± 1.24	159	22.52 ± 1.45
	OF	185	29.22 ± 1.98	128	17.83 ± 1.16
Abundance	Overall	4466	41.35 ± 1.41	1579	25.06 ± 1.21
	FA	769	45.23 ± 2.30	226	28.25 ± 3.07
	SF	2436	44.29 ± 1.75	847	27.32 ± 1.93
	OF	1261	35.02 ± 2.85	506	21.08 ± 1.43

TABLE S2. Information about the 22 resource-use traits used in the functional trait matrix. Traits are categorised into diet (i.e. what food they eat), foraging substrate (i.e. where they forage) and morphological traits (i.e. physical attributes). Measurement specifies the way traits are measured. Functional significance describes reveals the significance of these traits for ecosystem processes and services (based on Sekercioglu (2006) and Luck et al. (2012)). Data for the trait matrix was extracted from global trait matrix (Wilman et al 2014), Chira et al [in press]) and Handbook of the Birds of the World Alive (http://www.hbw.com/).

Category	Trait	Measurement	Functional significance
Diet	Invertebrates	Binary	Pollination and seed dispersal ability; trophic process, e.g. population control and pests and degree of dietary specialization
	Mammal/birds	Binary	
	Reptiles/amphibians	Binary	
	Vertebrates-general	Binary	
	Scavenger	Binary	
	Fruit	Binary	
	Necter, pollen	Binary	
	Seed, grains	Binary	
	Other plant material	Binary	
Foraging substrate	Ground	Binary	Rate of resource use and degree of habitat specialization
	Air	Binary	
	Understorey	Binary	
	Arboreal bark	Binary	
	Arboreal canopy	Binary	
Morphological	Body mass (g)	Continuous	Foraging behavior, rate of resource use, aerial and dispersal ability
	Bill length (mm)	Continuous	
	Bill width (mm)	Continuous	
	Bill depth (mm)	Continuous	
	Bill shape (mm)	Continuous	
	Wing length (mm)	Continuous	
	Trasus length (mm)	Continuous	
	Tail length (mm)	Continuous	

TABLE S3. List of bird species from the overall bird communities with their respective local evolutionary distinctiveness (ED) and evolutionary distinctiveness rarity (EDR). The taxonomy is from that of Jetz et al. (2012) and species are listed from most to least evolutionarily distinct.

Serial No	Scientific name	ED	EDR
1	*Upupa epops*	35.59	0.01
2	*Chelidorhynx hypoxantha*	32.71	0.13
3	*Melanochlora sultanea*	20.17	0.08
4	*Pteruthius xanthochlorus*	16.60	0.09
5	*Culicicapa ceylonensis*	16.47	0.03
6	*Erpornis zantholeuca*	16.39	0.04
7	*Arachnothera longirostra*	16.04	0.03
8	*Myzornis pyearsrhoura*	15.61	0.27
9	*Dicrurus aeneus*	15.52	0.03
10	*Chrysococcyx xanthorhynchus*	15.43	0.03
11	*Chrysococcyx maculatus*	15.42	0.04
12	*Psarisomus dalhousiae*	14.91	0.04
13	*Chloropsis hardwickii*	14.27	0.04
14	*Dicaeum melanoxanthum*	13.89	0.08
15	*Cuculus fugax*	13.79	0.02
16	*Arachnothera magna*	13.62	0.05
17	*Chloropsis aurifrons*	13.59	0.04
18	*Pteruthius melanotis*	12.86	0.08
19	*Alcippe poioicephala*	12.84	0.04
20	*Haematospiza sipahi*	12.79	0.15
21	*Pteruthius rufiventer*	12.73	0.18
22	*Pteruthius flaviscapis*	12.73	0.03
23	*Cinclidium frontale*	12.73	0.24
24	*Dicrurus remifer*	12.15	0.04
25	*Alcippe rufogularis*	11.91	0.09
26	*Centropus sinensis*	11.89	0.02
27	*Aegithina tiphia*	11.88	0.02
28	*Stigmatopelia chinensis*	11.53	0.02
29	*Glaucidium cuculoides*	11.51	0.02
30	*Oriolus traillii*	11.40	0.04
31	*Ficedula strophiata*	11.38	0.04
32	*Alcippe cinerea*	11.31	0.17
33	*Oriolus xanthornus*	11.22	0.03
34	*Abroscopus superciliaris*	11.01	0.03
35	*Abroscopus schisticeps*	10.99	0.1
36	*Alcippe castaneceps*	10.84	0.06

37	*Hypsipetes mcclellandii*	10.80	0.03
38	*Niltava macgrigoriae*	10.8	0.05
39	*Ficedula tricolor*	10.78	0.04
40	*Niltava grandis*	10.68	0.04
41	*Alcippe chrysotis*	10.66	0.12
42	*Ficedula hyperythra*	10.61	0.03
43	*Cinclidium leucurum*	10.48	0.03
44	*Hemipus picatus*	10.47	0.02
45	*Parus monticolus*	10.46	0.05
46	*Pericrocotus solaris*	10.45	0.02
47	*Pycnonotus melanicterus*	10.32	0.04
48	*Pycnonotus atriceps*	10.32	0.03
49	*Aethopyga siparaja*	10.13	0.02
50	*Niltava sundara*	10.11	0.03
51	*Otus spilocephalus*	10	0.03
52	*Pycnonotus flavescens*	9.86	0.05
53	*Garrulus glandarius*	9.83	0.01
54	*Parus major*	9.76	0.01
55	*Dicrurus leucophaeus*	9.75	0.02
56	*Pericrocotus cinnamomeus*	9.68	0.02
57	*Cuculus sparverioides*	9.67	0.02
58	*Zoothera dixoni*	9.67	0.07
59	*Sitta himalayensis*	9.66	0.12
60	*Cuculus poliocephalus*	9.41	0.02
61	*Orthotomus cuculatus*	9.36	0.02
62	*Alcippe dubia*	9.33	0.06
63	*Cyornis concretus*	9.2	0.04
64	*Enicurus schistaceus*	9.11	0.03
65	*Cacomantis merulinus*	9.09	0.02
66	*Tickellia hodgsoni*	9.07	0.11
67	*Aegithalos concinnus*	9.05	0.03
68	*Sitta frontalis*	8.97	0.02
69	*Hemixos flavala*	8.94	0.05
70	*Spizixos canifrons*	8.9	0.08
71	*Mycerobas affinis*	8.88	0.06
72	*Abroscopus albogularis*	8.87	0.03
73	*Bambusicola fytchii*	8.79	0.11
74	*Monticola rufiventris*	8.7	0.03
75	*Blythipicus pyearsrhotis*	8.65	0.03
76	*Ficedula hodgsonii*	8.59	0.04
77	*Certhia discolor*	8.49	0.16
78	*Certhia manipurensis*	8.49	0.11
79	*Ficedula monileger*	8.42	0.05
80	*Parus xanthogenys*	8.34	0.05
81	*Parus spilonotus*	8.33	0.04
82	*Phylloscopus maculipennis*	8.3	0.05

83	*Aethopyga ignicauda*	8.28	0.06
84	*Phylloscopus pulcher*	8.27	0.04
85	*Brachypteryx montana*	8.23	0.02
86	*Aethopyga saturata*	8.21	0.03
87	*Cuculus micropterus*	8.15	0.01
88	*Aethopyga gouldiae*	8.15	0.02
89	*Pericrocotus brevirostris*	8.13	0.03
90	*Aethopyga nipalensis*	8.12	0.03
91	*Pnoepyga pusilla*	8.07	0.02
92	*Brachypteryx leucophrys*	8.01	0.02
93	*Chaimarrornis leucocephalus*	7.92	0.02
94	*Pnoepyga albiventer*	7.82	0.06
95	*Melophus lathami*	7.73	0.02
96	*Pericrocotus ethologus*	7.72	0.02
97	*Phoenicurus frontalis*	7.71	0.03
98	*Myiomela albiventris*	7.7	0.89
99	*Streptopelia tranquebarica*	7.7	0.01
100	*Timalia pileata*	7.69	0.03
101	*Mycerobas melanozanthos*	7.62	0.04
102	*Tarsiger chrysaeus*	7.55	0.04
103	*Zoothera dauma*	7.49	0.01
104	*Glaucidium brodiei*	7.46	0.02
105	*Dicaeum ignipectus*	7.46	0.02
106	*Dicaeum cruentatum*	7.44	0.02
107	*Napothera epilepidota*	7.41	0.04
108	*Carpodacus erythrinus*	7.32	0.01
109	*Sitta formosa*	7.27	0.14
110	*Brachypteryx stellata*	7.24	0.1
111	*Sitta nagaensis*	7.22	0.06
112	*Megalaima lineata*	7.16	0.05
113	*Cyornis unicolor*	7.06	0.02
114	*Treron apicauda*	7.06	0.03
115	*Dicaeum erythrorhynchos*	7.04	0.02
116	*Cutia nipalensis*	7	0.05
117	*Luscinia pectardens*	6.96	0.07
118	*Tarsiger cyanurus*	6.95	0.01
119	*Treron sphenurus*	6.93	0.02
120	*Megalaima virens*	6.88	0.02
121	*Pycnonotus striatus*	6.86	0.04
122	*Sitta castanea*	6.84	0.02
123	*Saxicola caprata*	6.83	0.01
124	*Pycnonotus cafer*	6.81	0.02
125	*Dicrurus paradiseus*	6.77	0.01
126	*Phylloscopus inornatus*	6.76	0.01
127	*Oriolus tenuirostris*	6.76	0.04
128	*Phylloscopus humei*	6.75	0.29

129	*Saxicola ferreus*	6.71	0.02
130	*Saxicola jerdoni*	6.69	0.06
131	*Cyornis rubeculoides*	6.65	0.02
132	*Pycnonotus jocosus*	6.62	0.02
133	*Cettia fortipes*	6.61	0.02
134	*Alcippe ludlowi*	6.61	0.27
135	*Ficedula sapphira*	6.55	0.04
136	*Enicurus immaculatus*	6.48	0.05
137	*Columba pulchricollis*	6.46	0.03
138	*Celeus brachyurus*	6.43	0.01
139	*Minla cyanouroptera*	6.43	0.03
140	*Orthotomus sutorius*	6.37	0.01
141	*Tragopan blythii*	6.37	0.15
142	*Cochoa purpurea*	6.34	0.03
143	*Alcippe nipalensis*	6.33	0.07
144	*Cochoa viridis*	6.33	0.07
145	*Dendrocitta formosae*	6.33	0.02
146	*Dendrocitta frontalis*	6.32	0.19
147	*Hypothymis azurea*	6.32	0.01
148	*Ducula badia*	6.32	0.02
149	*Dendrocitta vagabunda*	6.29	0.02
150	*Enicurus leschenaulti*	6.07	0.01
151	*Cyornis magnirostris*	6.06	0.21
152	*Luscinia brunnea*	6.04	0.03
153	*Picus flavinucha*	6.03	0.02
154	*Phylloscopus trochiloides*	6.03	0.01
155	*Phoenicurus ochruros*	6.03	0.01
156	*Anthus hodgsoni*	6.02	0.01
157	*Prinia crinigera*	6	0.02
158	*Prinia atrogularis*	6	0.02
159	*Turdus boulboul*	5.99	0.05
160	*Megalaima franklinii*	5.97	0.03
161	*Picumnus innominatus*	5.97	0.01
162	*Myophonus caeruleus*	5.91	0.01
163	*Nectarinia asiatica*	5.91	0.01
164	*Dicaeum concolor*	5.91	0.01
165	*Dendrocopos hyperythrus*	5.9	0.03
166	*Rhipidura albicollis*	5.87	0.02
167	*Malacocincla abbotti*	5.83	0.02
168	*Yuhina nigrimenta*	5.8	0.03
169	*Anthus godlewskii*	5.77	0.02
170	*Phylloscopus magnirostris*	5.73	0.03
171	*Prinia inornata*	5.73	0.01
172	*Cettia brunnifrons*	5.7	0.05
173	*Minla strigula*	5.68	0.03
174	*Minla ignotincta*	5.68	0.2

175	*Tesia castaneocoronata*	5.67	0.03
176	*Tesia cyaniventer*	5.67	0.04
177	*Tesia olivea*	5.6	0.05
178	*Leiothrix argentauris*	5.58	0.02
179	*Leiothrix lutea*	5.58	0.19
180	*Prinia hodgsonii*	5.56	0.01
181	*Prinia socialis*	5.56	0.02
182	*Anthus richardi*	5.55	0.01
183	*Prinia rufescens*	5.53	0.02
184	*Streptopelia orientalis*	5.48	0.01
185	*Emberiza pusilla*	5.48	0.01
186	*Chrysomma altirostre*	5.47	0.11
187	*Yuhina gularis*	5.47	0.03
188	*Pellorneum ruficeps*	5.46	0.02
189	*Chrysomma sinense*	5.45	0.01
190	*Stachyearsis nigriceps*	5.44	0.02
191	*Paradoxornis nipalensis*	5.4	0.05
192	*Paradoxornis guttaticollis*	5.39	0.03
193	*Anthus rufulus*	5.39	0.01
194	*Seicercus poliogenys*	5.38	0.04
195	*Yuhina flavicollis*	5.35	0.03
196	*Seicercus affinis*	5.32	0.06
197	*Macropygia unchall*	5.31	0.01
198	*Motacilla alba*	5.3	0.01
199	*Monticola solitarius*	5.28	0.01
200	*Ficedula superciliaris*	5.25	0.04
201	*Cuculus canorus*	5.23	0.01
202	*Cuculus saturates*	5.2	0.02
203	*Garrulax squamatus*	5.19	0.06
204	*Garrulax subunicolor*	5.19	0.06
205	*Carpodacus edwardsii*	5.14	0.06
206	*Ducula aenea*	5.04	0.01
207	*Eumyias thalassinus*	5.03	0.01
208	*Stachyearsis chrysaea*	4.97	0.02
209	*Aegithalos iouschistos*	4.94	0.07
210	*Phylloscopus reguloides*	4.88	0.02
211	*Dicrurus macrocercus*	4.86	0.01
212	*Phylloscopus cantator*	4.82	0.04
213	*Megalaima asiatica*	4.82	0.02
214	*Turdus albocinctus*	4.69	0.05
215	*Paradoxornis flavirostris*	4.68	0.3
216	*Liocichla phoenicea*	4.67	0.06
217	*Dendrocopos macei*	4.66	0.02
218	*Otus sunia*	4.64	0.01
219	*Pellorneum albiventre*	4.63	0.05
220	*Garrulax cineraceus*	4.6	0.02

221	*Alcippe vinipectus*	4.57	0.04
222	*Phylloscopus forresti*	4.53	0.07
223	*Ficedula westermanni*	4.5	0.01
224	*Cettia flavolivacea*	4.49	0.02
225	*Seicercus whistleri*	4.48	0.04
226	*Dendrocopos darjellensis*	4.44	0.05
227	*Bradypterus thoracicus*	4.43	0.03
228	*Seicercus tephrocephalus*	4.29	0.06
229	*Seicercus burkii*	4.29	0.18
230	*Ficedula albicilla*	4.28	0.01
231	*Arborophila rufogularis*	4.23	0.04
232	*Xiphirhynchus superciliaris*	4.21	0.05
233	*Phylloscopus chloronotus*	4.21	0.06
234	*Picus chlorolophus*	4.21	0.01
235	*Arborophila torqueola*	4.16	0.04
236	*Pycnonotus leucogenys*	4.11	0.02
237	*Lanius schach*	4.11	0.01
238	*Dendrocopos cathpharius*	4.03	0.03
239	*Actinodura egertoni*	4	0.06
240	*Coracina melaschistos*	4	0.01
241	*Turdus dissimilis*	3.95	0.04
242	*Spelaeornis chocolatinus*	3.87	0.63
243	*Spelaeornis troglodytoides*	3.85	0.12
244	*Spelaeornis oatesi*	3.81	0.39
245	*Psittacula finschii*	3.8	0.03
246	*Seicercus castaniceps*	3.79	0.01
247	*Terpsiphone paradisi*	3.79	0.01
248	*Spelaeornis longicaudatus*	3.78	0.26
249	*Spelaeornis caudatus*	3.78	0.14
250	*Rimator malacoptilus*	3.77	0.08
251	*Actinodura nipalensis*	3.73	0.11
252	*Hypsipetes leucocephalus*	3.72	0.01
253	*Garrulax affinis*	3.7	0.03
254	*Trichastoma tickelli*	3.68	0.02
255	*Pomatorhinus hypoleucos*	3.64	0.02
256	*Corvus macrorhynchos*	3.62	0.01
257	*Heterophasia capistrata*	3.62	0.06
258	*Pomatorhinus mcclellandi*	3.52	0.07
259	*Heterophasia picaoides*	3.39	0.02
260	*Garrulax chrysopterus*	3.39	0.08
261	*Heterophasia annectens*	3.38	0.03
262	*Garrulax nuchalis*	3.38	0.17
263	*Heterophasia gracilis*	3.37	0.09
264	*Heterophasia pulchella*	3.34	0.09
265	*Cyornis poliogenys*	3.34	0.03
266	*Garrulax virgatus*	3.3	0.13

267	*Lophura leucomelanos*	3.25	0.02
268	*Garrulax striatus*	3.24	0.05
269	*Lanius tephronotus*	3.22	0.02
270	*Garrulax gularis*	3.21	0.05
271	*Garrulax austeni*	3.19	0.17
272	*Garrulax ruficollis*	3.17	0.04
273	*Alcippe manipurensis*	3.02	0.09
274	*Garrulax caerulatus*	3.02	0.05
275	*Carduelis spinoides*	2.99	0.04
276	*Phylloscopus xanthoschistos*	2.73	0.03
277	*Turdoides earlei*	2.72	0.02
278	*Stachyearsis ruficeps*	2.6	0.01
279	*Pomatorhinus schisticeps*	2.09	0.01
280	*Pomatorhinus ruficollis*	2.09	0.01
281	*Zosterops palpebrosus*	1.69	0.01