Replanting a Future:

Restoring Cloud Forests, Biodiversity, and Rural Livelihoods in Andean Landscapes

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To my mom, who taught me to love trees,

my dad, who taught me their names,

and my sisters,

for climbing them with me

…in Wildness is the preservation of the World. Every tree sends its fibers forth in search of the Wild. The cities import it at any price. Men plow and sail for it. From the forest and wilderness come the tonics and barks which brace mankind…

– Henry David Thoreau, *Walking*, 1862

The next century will, I believe, be the era of restoration in ecology.

– E.O. Wilson, *The Diversity of Life*, 1992

…restoring ecosystems we regenerate the old ways or create new ones that bring us closer to natural processes and to one another. This is the power and the promise of ecological restoration.

– Eric Higgs, *Nature by Design*, 2003

Think globally, act locally.

– Jacques Ellul

ABSTRACT

Can community-based tree planting effectively restore and conserve biodiverse cloud forest in the Andes? This dissertation seeks to answer this question through a multi-site study in the Intag Valley in northwest Ecuador, a heavily deforested global biodiversity 'hotspot.' Here, working with a local NGO, communities began planting trees to restore cloud forests in the early 2000s. I visited Intag in 2010 and 2011, where, using mixed methods from the natural and social sciences, I quantified local land-use and -cover changes with satellite images from 1991, 2001, and 2010; compared tree diversity in multiple patches of primary, planted, and naturally regenerating forest; and assessed community participation in cloud forest replanting based on household interviews, focus groups and oral histories in four communities. These analyses enabled me to answer four related questions: 1) do communities reforest and deforest simultaneously?; 2) how heterogeneous are tree communities in remnant Andean cloud forests, and what strategies are needed to conserve landscape biodiversity?; 3) can community-based restoration accelerate cloud forest recovery?; and, 4) who participates in tree planting, why do they choose to do so, and does it benefit their lives and livelihoods?

Results indicate that deforestation slowed considerably between 2001 and 2010. Although people continued to clear primary forest in the highlands, forests regrew around communities, resulting in a net cover increase – a local 'forest transition.' This spatial shift in forests is partly explained by people's reasons for restoring them. Following deforestation, a decline in key ecosystem services – especially water – threatened their ability to farm, spurring people to work with a local NGO to plant trees in communal watershed reserves. Many households then applied newly acquired arboricultural knowledge and techniques on their farms, implementing innovative tree-based systems to restore soils and water availability. Tree planting accelerated forest recovery, increasing tree diversity and 'jump-starting' succession in communal reserves. But young planted forests, with their high proportion of locally useful species, are still 'novel' in this landscape, remaining ecologically distinct from the highly diverse and spatially variable primary forests.

So, can restoration be 'win-win' for cloud forests and Andean farmers? In heavily deforested regions, the answer suggested by this study is 'yes.' Restoration has limitations – results suggest that it cannot replace, nor assure, the conservation of primary cloud forests. But because restoration ultimately aided forest recovery, increased tree diversity, and had high participation rates, this case study identifies a number of important synergies between rural livelihoods and biodiversity conservation mediated through the practice of cultivating trees. Driven by local ecosystem service scarcity, this 'crisis restoration' was an integral part of a local movement to renew and sustain farming culture, and created forests for which people feel a sense of stewardship, ownership and pride. This model of restoration thus holds considerable potential to benefit rural farmers and restore biodiversity across the many heavily deforested regions of the Andes.

RÉSUMÉ

Est-ce que la plantation d'arbres communautaire peut efficacement restaurer et conserver les forêts de nuages riches en biodiversité dans les Andes? Cette thèse vise à répondre à cette question par une étude multi-sites dans la vallée d'Intag dans nord-ouest de l'Équateur, un point chaud de la biodiversité mondiale fortement déboisé. Ici, en collaboration avec une ONG locale, les communautés ont commencé à planter des arbres pour restaurer les forêts de nuages au début des années 2000. J'ai visité Intag en 2010 et 2011, où, en utilisant des méthodes mixtes des sciences naturelles et sociales, j'ai quantifié les changements d'utilisation et du couvert du sol avec des images satellites (1991, 2001 et 2010); comparé la diversité de plusieurs parcelles de forêt (primaire, plantée, et de régénération naturelle); et évalué la participation communautaire dans les replantations de forêt de nuages à l'aide d'entretiens ménagers (n = 120), de groupes de discussion et de l'histoires orales dans quatre collectivités locales. Ces analyses m'ont permis de répondre à quatre questions: 1) est-ce que les communautés reboisent et déboisent simultanément?; 2) quel est le dégrée d'hétérogénéité des communautés d'arbres dans les forêts de nuages résiduelles des Andes, et quelles sont les stratégies nécessaires pour conserver la biodiversité du paysage ?; 3) est-ce que la restauration communautaire peut accélérer la reprise des forêt de nuages ?; et, 4) qui participe à la plantation d'arbres, pourquoi choisissent-ils de le faire, et quelles en sont les répercussions sur leur vies et moyens de subsistances?

Les résultats indiquent que la déforestation a ralenti considérablement entre 2001 et 2010. Bien que les gens aient continué à défricher la forêt primaire, les forêts ont repoussées autour des communautés, entraînant une augmentation net de la couverture - une 'transition forestière' locale. Suite à la déforestation, un déclin des services écosystémiques clés - en particulier l'approvisionnement en eau – a menacé l'agriculture locale poussant les gens à travailler avec une ONG locale afin de planter des arbres dans les réserves communales de bassin versant. Plusieurs ménages ont ensuite appliqué leur nouvelles connaissances et techniques d'arboriculture sur leur ferme, mettant en place des systèmes étagés innovateurs basés sur les cultures arboricoles pour restaurer les sols et la disponibilité en eau. La plantation d'arbres a accéléré la récupération de la forêt, ce qui a augmenté la diversité des arbres et propulsé la succession dans les réserves communales. Les jeunes forêts plantées, avec leur forte proportion d'espèces utiles localement, sont toutefois encore «inédite» dans ce paysage, demeurant écologiquement distinctes des forêts primaires très diverses et spatialement variables.

En somme, est ce que la restauration peut être avantageuse autant pour les forêts de nuages que pour les agriculteurs andins? Pour les régions fortement déboisées, cette étude suggère que oui. La restauration communautaire a ses limites – les résultats suggèrent qu'elle ne peut ni remplacer, ni assurer la conservation des forêts de nuages primaires. Toutefois, la restauration a permis la récupération de la forêt, l'augmentation de la diversité des espèces; le tout avec des taux élevés de participation. D'importantes synergies ont été identifiées grâce à cette étude liant les moyens de subsistances ruraux et la conservation de la biodiversité par le biais de la culture des arbres. Face à la pénurie des services écosystémiques, cette « restauration réactive » était une partie intégrante d'un mouvement local renouvelant et maintenant la culture paysanne. Les gens ont aussi développe une intendance, une propriété et une fierté face aux forêts créés. Ce modèle de la restauration détient ainsi un potentiel considérable profitable tant aux agriculteurs ruraux et qu'à la restauration de la biodiversité dans les régions fortement déboisées des Andes.

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How lucky I am to have something that makes saying goodbye so hard.

– A.A. Milne, *Winnie the Pooh*, 1926

I have been incredibly privileged to work with a group of such talented and inspiring people over the past five years. My deepest gratitude to the many who guided, encouraged, participated in, and inspired both this research and my path as a researcher. May we continue to work, collaborate, or, at the very least, enjoy discussion over coffee together in the future.

First, thank you to my supervisor, Oliver Coomes, for providing such a constructive balance of guidance and freedom during my research. I was challenged to problem solve, innovate, 'figure stuff out', and experiment, but with a carefully mapped-out path that gave me the security of knowing that the end goals were, ultimately, attainable. I left our meetings energized, inspired, and with a clear sense of how to tackle and prepare for the next step, be it a literature review or a visit to the field. I appreciated Oliver's nuggets of experiential wisdom along the way, and only hope that I've not taken the advice that "there are perfect theses, and there are theses that get finished" too literally.

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AUTHORS' CONTRIBUTIONS

I conceived the project scope, developed research questions, and reviewed the relevant literature for this dissertation. I also located and selected a study region in Ecuador, made initial research connections there, and developed a relationship with my study communities. I designed fieldwork methodology, organized and coordinated fieldwork, hired and managed a field team, and personally collected both socioeconomic and ecological data. I performed all statistical analyses presented in this thesis, and wrote, created figures for, and assembled the manuscripts. I secured all funding for fieldwork, writing grant proposals and reports to obtain fieldwork funding from the International Development Research Centre (IDRC), the National Science and Engineering Council of Canada (NSERC), and the Institute for the Study of International Development at McGill (ISID), in addition to receiving funds from the Department of Geography.

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Camille Ouellet-Dallaire created the maps in this thesis based on my specifications and using data I collected. She also created a database of forest cover change using data from remote sensing analyses, which she used to generate the maps presented in Chapter 2. Ms. Ouellet-Dallaire is a co-author on Chapter 2.

I obtained three Landsat images and collected ground truth data for land use classifications, and Carlomagno Soto at The Geographic Information Centre (GIC) at McGill University performed standard procedures for image pre-processing and classified the images. The GIC provided an accuracy assessment of these classifications, shown in Table 3.1. Throughout the process I guided the conceptual interpretation of the images and specification of the land use classes.

CHAPTER 1: INTRODUCTION

In South America, people are clearing tropical forests at some of the highest rates in the world, primarily to grow crops and raise cattle (Fearnside, 1993; FAO, 2007; Vieira *et al.*, 2009). However, although total forest area declined in the past decade, in some regions forest cover increased – especially in dry areas and, of particular interest here, in the mountains (Aide *et al.*, 2013). Rich in endemic species, tropical montane cloud forests contain a disproportionately high number of the world's species for their extremely limited range (Bruijnzeel *et al.,* 2010; Scatena *et al*., 2010). Andean cloud forests have been extensively cleared and, until recently, have received relatively little attention from conservation agencies and researchers alike compared to moist lowland forests (Bruijnzeel *et al.*, 2010*b*; Griscom & Ashton, 2011). At first glance, then, this forest cover increase appears to be great news for global biodiversity conservation.

But there is a potential problem: returning secondary forests, both 'natural' and planted, often contain fewer or different species from cleared primary forests. The ecosystem services, forest products, and access and use rights afforded to local people also differ between forest types (Peluso, 1992; Lamb *et al.*, 2005). Transitions from primary to secondary forest are thus likely to affect both tropical biodiversity and rural people who depend on forests and forest ecosystem services for their livelihoods (Lamb *et al.*, 2005; Chazdon, 2008; Rudel, 2009*a*). Despite the prevalence of these transitions, their social and ecological drivers and outcomes are still little investigated at the local level.

In the past decade, forests regenerated both spontaneously, through natural regeneration[1], and intentionally, through planting trees for industrial timber production. This regeneration occurred on more than 340,000 hectares in South America and the Caribbean, much of it former pasture and other agricultural land (Rudel *et al.*, 2002; Rudel *et al.*, 2005; FAO, 2007, 2011;

Aide *et al.*, 2013). Plantations and young, naturally regenerated forests generally support fewer and different species, and are thus structurally and functionally distinct, from primary forests[1] (Parrotta & Knowles, 1999; Lamb *et al.*, 2005; Sampaio *et al.*, 2007; Liebsch *et al.*, 2008; Dent & Wright, 2009; Letcher & Chazdon, 2009; Bonner *et al.*, 2013). Naturally regenerating forests can accumulate biomass quickly, providing many local ecosystem services; however, they are often less biodiverse, structurally complex, and support different plant species from primary forest for decades to centuries (Lamb *et al.*, 2005; Sampaio *et al.*, 2007; Dent & Wright, 2009). In especially degraded areas, forests can be slow or unable to recover: the intensity and duration of previous land use and proximity to remnant forest determine if and how quickly forests grow back (Uhl *et al.*, 1988; Holl *et al.*, 2000; Guariguata & Ostertag, 2001; Chazdon, 2003). Fragmented and highly degraded landscapes – conditions prevalent throughout the Andes – are the least likely to regenerate unassisted (Aide & Cavelier, 1994; Metzger & Décamps, 1997; Chazdon, 2003). Large-scale industrial timber plantations – the most common planted forests promoted through government reforestation projects in South America – are often monocultures of exotic trees. Although native trees may regenerate under their canopies, plantations are typically less biodiverse than primary forests (Parrotta & Knowles, 1999; Farley, 2007, 2010), and with their high start-up costs, smallholders are often unable to invest in them (Reardon &

[1] **Forest terms used in this dissertation:**
Primary forest: Old growth forest, or forest that has not been cleared or logged in living memory (at least 80 years).
Naturally or spontaneously regenerated forest: Secondary forest that has regrown on previously cleared land with no direct planting or seeding by people.
Restored or *replanted forest*: Secondary forest where people have planted trees with the intention of restoring forest cover. Does not include industrial timber plantations.
Industrial timber plantations: Plantations of trees geared towards timber production (generally monocultures of exotic species).
Tropical montane cloud forest: Moist tropical forests occurring above 1200 masl, frequently engulfed in clouds (Scatena *et al.*, 2010)

Vosti, 1995; Sloan, 2008).

Reforesting the tropics has become a global priority. Recognizing the extent of deforestation and the limited ability of industrial plantations and, in heavily used areas, unassisted regeneration to conserve forest diversity, conservation scientists, NGOs and agencies are embracing a third type of reforestation: *landscape-level ecological restoration*, in which people take action to help forests regenerate (Lamb *et al.*, 2005; Young *et al.*, 2005; Foley, Monfreda, Ramankutty, & Zaks, 2007; Chazdon, 2008; DeFries *et al.*, 2012). In theory, the aim of restoration is to recreate tropical primary forests by 'jump-starting' succession (Higgs, 2003; Temperton *et al.*, 2004; Harris & van Diggelen, 2006; Menninger & Palmer, 2006; Palmer *et al.*, 2006; Stanturf *et al.*, 2014). In practice, this often translates to 1) establishing tree cover, and 2) reintroducing native plant species with ecological or human use value (Sarmiento *et al.*, 1995; Gandolfi *et al.*, 2007). In especially degraded areas, directly reintroducing trees by planting native seedlings is needed, and often used, to initiate this process (Chazdon *et al.*, 2003; Gandolfi *et al.*, 2007; Rodrigues., 2007; Chazdon, 2008; Aide *et al.*, 2010; Holl, 2011)**.** Despite the number of on-the-ground projects underway, to date most studies have assessed the efficacy of restoration using controlled, experimental plot studies (Holl, 2011; Pena-Domene *et al.*, 2013). The ability of these projects to restore tropical forest plant diversity and species composition in practice remains largely uninvestigated.

At the same time, development agencies commonly promote community-based tree planting as a 'win-win' for people and the environment. Tree-planting projects can benefit local communities by providing: 1) ecosystem services that enhance agriculture; 2) a source of revenue through payment for environmental services that benefit people elsewhere; and 3) food, timber, and other extractive forest products (Wunder, 2005; Chazdon, 2008; Garen *et al.*, 2009)

Participating in community-based forestry can also empower people, create or strengthen social networks, and help them self-organize (Pretty, 2003; Bray *et al.*, 2006), all important aspects of sustainable rural livelihoods in the Andes (Bebbington & Perreault, 1999).

Although attractive on paper, many conservation-development initiatives based on forest use and biodiversity fail to meet their objectives (Agrawal & Gibson, 1999; Adams *et al.*, 2004; Christensen, 2004; Coomes *et al.*, 2004; Blom *et al.*, 2010; Brooks *et al.*, 2013; Davies *et al.*, 2013). Despite limited success, they are still promoted by many foundations and NGOs (CI; WWF; Davies *et al.*, 2013). But in Andean regions, communities often face other, more pressing challenges: declining soil fertility, altered climate and precipitation patterns, and rapidly urbanizing economies are quickly changing rural Andean communities, farming cultures, and livelihoods – in the face of these challenges, restoring or conserving forests may not be a top priority. The silver lining is that restoration projects are essentially experiments – analyzing their outcomes can inform and improve future projects for both people and ecosystems (Jordan III *et al.*, 1987; Rudel, 2000; Gockel & Gray, 2009; Manzi & Coomes, 2009).

Intriguingly, given their recent recovery, dry and montane forests often regenerate more slowly than lowland rainforest. But most of what we know about tropical forest restoration comes from studies in moist lowland forests (Churchill *et al.*, 1995; Gentry *et al.*, 1995; Aide *et al.*, 2010; Bruijnzeel *et al.*, 2010*b*; Griscom & Ashton, 2011). One of the most biodiverse ecosystems on the planet, Andean cloud forests are a top global conservation priority (Myers *et al.*, 2000; Brooks *et al.*, 2006; Richter *et al.*, 2009). But conserving cloud forest biodiversity is challenging: heterogeneous in space and time, many have yet to be surveyed, and our understanding of how species are distributed in them is patchy. They also need restoring: often found in areas with a long history of human use, upwards of 50% of Andean cloud forests have been cleared, and large areas of remaining forests are fragmented (Sarmiento, 1995*a*, 1995*b*; Aide *et al.*, 2010). Despite the need for and number of projects engaged in restoring cloud forest, little is known about their efficacy to restore them.

People have high hopes for restoration. But restoring complex, diverse cloud forests to vast Andean landscapes is a big task for small rural communities, especially given the lack of ecological data in many locales. Despite a recent surge in publications on ecological restoration, we still know little about how community-based efforts affect plant diversity, how best to engage local people, and how restoring forests ultimately affects landscape-level biodiversity. This dissertation asks: What is the potential of local, community-based tree planting efforts to restore and conserve megabiodiverse cloud forests in Andean landscapes? What drives people to restore them, and what are the social and ecological outcomes of such efforts? Through a case study in northwest Andean Ecuador, I examine the trade-offs and synergies between conserving biodiversity and encouraging participation in community-based restoration projects. I also characterize regional primary forest tree biodiversity and clearing dynamics. Thus, by using a land-use and land-cover-change approach, I integrate diverse theoretical approaches and methodologies from the natural and social sciences to study projects in a holistic way that mirrors their multiple social and ecological objectives. Ultimately, I hope the results presented here will be used to improve the ecological and social outcomes of cloud forest restoration efforts throughout Latin America.

Research Questions

1) *Do communities reforest and deforest simultaneously, and does community-based restoration increase regional forest cover?* (Chapter 2)

2) *How heterogeneous are the tree communities in remnant Andean cloud forests, and what strategies are needed to conserve landscape biodiversity?* (Chapter 3)

3) *Does community-based restoration accelerate cloud forest recovery?* (Chapter 4)

4) *Who participates in tree planting, why do they choose to do so, and does it benefit people's lives and livelihoods?* (Chapter 5)

I address these questions through a multi-site study in northwest Andean Ecuador, where people in small farming communities have been planting trees to restore forests and farmland since 2003.

Conceptual Approach and Thesis Structure

In this thesis I examine the ecological and social drivers and outcomes of community-based forest restoration projects using theories and methods from both the natural and social sciences. The thesis is manuscript-based, and although each chapter is a stand-alone paper of which I am the lead author, they are thematically and theoretically linked to create a cohesive work. Manuscripts are based on field data I collected in the Intag region of northwest Ecuador in 2010 and 2011.

In this introductory chapter, I review the literature on several theoretical approaches which I apply in subsequent chapters to study the drivers, outcomes, and impacts of community-based restoration. I discuss why a land-use and land-cover-change (LUCC) approach, and specifically the forest transition model, is a useful framework to study the drivers and outcome of community-based restoration. Within that framework, I review the literature on agroforestry adoption, rural livelihoods analysis, community forestry, tropical forest ecology and succession, ecological restoration, and cloud forest ecology and biogeography, and outline why and how each can be used to examine the outcomes and drivers of community-based reforestation in the Andes (although some have not been used for this purpose in the literature to date). I then provide a general introduction to the study region, a description of fieldwork, and an overview of field methodology.

In Chapter 2 I examine forest cover changes in the Intag region over twenty years, from 1991-2001 and 2001-2010, using a LUCC approach. This chapter shows that during the period when people began planting trees deforestation rates declined but remained high, reforestation rates increased, and a local ‘forest transition’ – a net increase in forest cover – occurred. This chapter sets the stage for the following three chapters, each of which examines different aspects of the drivers that produce local forest transitions and their impact on people, forest ecology, and biodiversity in heavily deforested Andean regions.

Restoring an ecosystem requires a detailed understanding of its ecology and the species that comprise it. Restoring cloud forests is particularly challenging because their diverse communities of species vary in both space and time in ways that we are only beginning to characterize. In Chapter 3, I examine the landscape-level diversity of cloud forest trees in the Intag Valley, applying theories and methods from mountain biogeography and landscape and forest ecology. The first study to survey and compare tree communities in mid- to upper-level (1900-2250 masl) cloud forests in the region, this chapter shows that highly diverse tree communities are distinct from one ridge top to the next. I conclude with recommendations for conserving and restoring cloud forests.

Building on this 'snapshot' of the region's primary forests, Chapter 4 investigates the potential of community-based tree planting to restore them. Drawing from literature on tropical forest ecology and succession, land-use legacies, and restoration ecology, I compare the species compositions and successional processes in planted secondary forests, naturally regenerated unplanted forests, and primary cloud forests. The main findings – that planting locally 'useful' species of trees increases forest cover and tree diversity, accelerates succession, but also creates 'novel forests' with no ecological precedent – provides an optimistic yet cautionary view of the ability of restoration to conserve cloud forest diversity, and shows that restoring cloud forests is no substitute for conserving them.

Chapter 5 examines the community-level participation that produced these environmental outcomes, and their impacts on local people. It is the first study to use asset-based livelihood analysis to examine household participation in communal forest restoration, and compare it to participation in on-farm tree planting. Applying literature on community-based forestry, agroforestry and PES adoption, LUCC, and rural livelihoods, I found that people restored forests and planted on-farm trees in innovative and creative ways to restore ecosystem services and rural farming culture. Driven by a lack of key forest ecosystem services, results are context specific to heavily deforested regions. But because such regions are common throughout the Andes and to cloud forest environments around the world, results have broad applicability.

The thesis concludes by summarizing the contributions of each chapter to the literature, identifying the human-ecological synergies in tropical restoration projects, presenting the policy and project implementation recommendations arising from this work, and developing questions for future research.

Literature Review

In this thesis I integrate several academic fields, including LUCC, forest transition theory, smallholder adoption of tree planting in agroforestry, rural livelihood analysis, community forestry, tropical forest ecology and succession, landscape ecology, ecological restoration, and cloud forest ecology and biogeography. I combine theories and methods from these fields to investigate the potential of restoration projects to conserve tropical montane cloud forest biodiversity, and to analyze their impacts on rural Andean livelihoods. The relevant contributions of each are outlined below.

Land-use and land-cover change (LUCC)

Land-use and land-cover change science "seeks to understand the dynamics of land cover and land use as a coupled human-environment system to address theory, concepts, models and applications relevant to environmental and societal problems, including the intersection of the two" (Turner *et al.*, 2007). This multi-scalar approach combines research on individual and collective agency and action with the consequences of human land use on the environment (Turner *et al.*, 1997; Turner *et al.*, 2007; Turner & Robbins, 2008). Work in this field is inherently multidisciplinary, combining research techniques from the social sciences, natural sciences, and geographical information systems (GIS). LUCC has often been used to study the social and environmental causes of deforestation and, more recently, reforestation (Turner & Robbins, 2008; Brondizio *et al.,* 2012; Redo *et al.*, 2012). The forest transition theory is a subarea of this field focusing specifically on patterns and drivers of net increases in forest cover.

Forest transition theory

The term 'forest transition' describes a pattern of forest depletion followed by forest recovery over the development of a region or country. Early forest transition theorists observed that as the economies of France, the UK, the US, and other North American and European countries developed, extensive deforestation for agriculture and timber slowed and eventually reversed as forest regrew on cleared land (Mather, 1992; Grainger, 1995; Mather & Needle, 1998; Mather *et al.*, 1999) (Fig. 1.1). Rudel *et al*. (2005) summarize the two most common scenarios that lead to forest transitions: in the first, as forest resources become scarce, forests are intentionally replanted (often as industrial plantations) to supply them (the "forest scarcity" path). In the second, as countries become urbanized, rural farm labour becomes scarce, and forests regenerate naturally on abandoned marginal lands (the "economic development" path) (Mather, 1992; Mather & Needle, 1998; Rudel *et al.*, 2005). In tropical South America, which as a whole is rich in forest but relatively poor in rural labour, forests are recovering on abandoned agricultural land in many regions (Rudel *et al.*, 2002; Aide *et al.*, 2013). Reforestation projects have also been initiated by governments and agencies in response to regional and local forest scarcity to (re)establish timber trees, sequester carbon, or conserve biodiversity (e.g., (Maquipucuna Foundation; Farley, 2007; Sloan, 2008).

Mather's (1992) original forest transition model shows a U-shaped relationship between forest cover and time (Fig. 1.1). This model is essentially an inverse environmental Kuznets curve, which describes a theoretical relationship between pollution and per capita income. The theory goes that as incomes rise, the level of environmental pollution increases and then decreases again as higher-income societies develop cleaner technologies, enhanced environmental stewardship, and switch from industrial to service-based economies (Selden &

Song, 1994). This observed patterns reflects an early school of conservation-development thought which predicts that with increasing affluence people will be more willing and able to invest in forest conservation and more efficient technologies. Both the Kuznets curve and the forest transition theory operate on the assumption that there are patterns inherent in the development of a country that lead to predictable environmental outcomes.

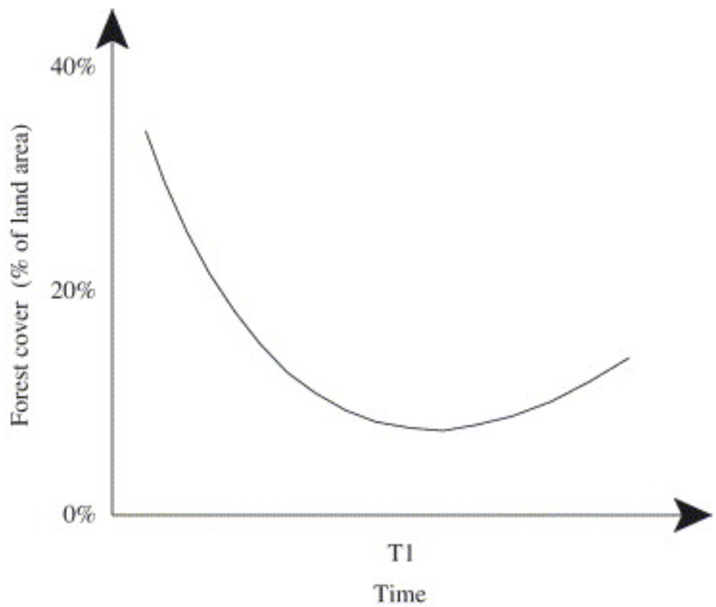


Figure 1.1: The forest transition theory (Rudel *et al.*, 2005)

Recent research has questioned how and if this rather simplistic, optimistic pattern of forest recovery applies to currently developing countries by examining the economic, political and social drivers of transitions. Lambin and Meyfroidt (2010) postulate that forest transitions can be driven by either “negative socio-ecological feedbacks” such as land degradation or forest scarcity, or “socio-economic dynamics” such as regional economic development, globalization, or other changes not directly related to local environmental conditions. Multiple drivers operating over multiple scales, from the individual to the transnational, may work together to promote or prevent forest transitions (Lambin *et al.*, 2001; Rudel *et al.*, 2002; Hecht, 2010; Redo *et al.*, 2012). Angelsen and Rudel (2013) emphasize that although forest transitions are commonly observed, the theory is not deterministic: often, conditions that lead to transitions in one locale do not produce the same result in another (Mather, 1992; Rudel *et al.*, 2005; Kull *et al.*, 2007; Redo *et al.*, 2012; Angelsen & Rudel, 2013). Forest transitions are thus scale and locale dependent, and considerably more complex than the original forest transition model would suggest.

A common assumption in the early forest transition literature is that the same factors that promote forest recovery (especially through tree planting) also halt or slow deforestation (Grainger, 1995; Mather & Needle, 1998). However, primary forest clearing and active reforestation often occur simultaneously in the same country or region – planting trees does not necessarily coincide with conserving primary forest (Rudel *et al.*, 2002; Perz & Skole, 2003; FAO, 2007; Sloan, 2008; Angelsen & Rudel, 2013). Replacing primary forest with plantations can have extremely detrimental impacts on forest biodiversity and rural people (Hecht, 1993; Healey & Gara, 2003; Rohter, 2004; Rudel, 2009*b*; Sánchez-Cuervo *et al.*, 2012; Zhai *et al.*, 2014). To evaluate and predict the social and environmental consequences of forest transitions, both the relationship between forest clearing and tree planting or regrowth, and the type(s) of secondary forests that grow back must be taken into account. Although a handful of studies have differentiated between the forest types that result from different forest transition pathways (Baptista & Rudel, 2006; Lambin & Meyfroidt, 2010; Rudel, 2010) and others on the ecological impacts of transitions (Farley, 2007, 2010), in general, comprehensive treatment of the ecological consequences of forest transitions are lacking. This thesis presents one of the first case studies to examine both the local conditions that drive forest transitions and the direct consequences of these transitions on forest biodiversity.

Forest transitions are scale-dependent: the area over which one examines forest-cover change can determine if forest appears to be increasing, decreasing, or stable. Different drivers also operate at different scales (Redo *et al.*, 2012; Aide *et al.*, 2013). Most forest transition studies have focused on large-scale patterns in forest cover changes at the continent, national or, less commonly, subnational level (Perz & Skole, 2003; Baptista & Rudel, 2006; Grau & Aide, 2008; Lambin & Meyfroidt, 2010; Redo *et al.*, 2012; Aide *et al.*, 2013), rather than on the scale of the community or the household (but see (Rudel *et al.*, 2002; Sloan, 2008). But local people make land-use decisions that affect forest cover, which can in turn have an impact on their lives and livelihoods (Jokisch & Lair, 2002; Rudel *et al.*, 2002; Schelhas & Sánchez-Azofeifa, 2006; Nelson & Chomitz, 2007; Aubad *et al.*, 2008; Hoch *et al.*, 2009; Redo *et al.*, 2012)**.** Because the forest transition theory is such a useful framework to study the combined effects of socioeconomic change and biophysical conditions on forest cover dynamics, the drivers and outcomes of local forest transitions deserve more attention.

Land-change science and livelihoods

Local people's actions often drive changes in land cover. The factors that determine why and how people decide to use land are complex and operate on multiple scales (Lambin *et al.*, 2003; Redo *et al.*, 2012), and even within seemingly homogeneous communities, people's resource use and livelihood strategies can differ substantially between households, and over the lifecycle of a given household (Ellis, 1993; Reardon & Vosti, 1995; Coomes & Burt, 1997; Ellis, 1998; Bebbington & Perreault, 1999; Chowdhury, 2010). Households within a community will thus participate in and benefit from community-based tree planting efforts to different degrees depending on their household resource endowments and livelihood strategies. To understand the effects of community-based restoration on both people and the environment, examining how different households engage in and benefit from restoration projects, and how these projects in turn affect their livelihood strategies and resource use, is key.

Agroforestry adoption

The agroforestry adoption literature focuses on why some households plant on-farm trees while others do not (Pattanayak *et al.*, 2003; Mercer, 2004). This literature is situated within a broader body of literature on adopting soil conserving measures, organic farming, or other on-farm technological innovations (Feder *et al.*, 1985; Ellis, 1993). Most of these microeconomic studies are based on the assumption that smallholders will adopt a new 'technology' or system only if they perceive it as profitable – i.e., if it will increase farm productivity or output stability by increasing overall yield, yields per unit input, or producing alternative products such as fruit or wood (Ellis, 1993; Mercer, 2004).

Adoption studies typically compare the asset portfolios and other characteristics of households that participated in on-farm tree planting with non-participant households in the same community or region (Pattanayak *et al.*, 2003). Commonly studied household variables include land holdings (farm size, land tenure security, field size, field location), other measures of wealth (assets or income) (Bannister & Nair, 2003; Pattanayak *et al.*, 2003; Sood & Mitchell, 2009), human capital (education level, family size, ages of family members) (Bellow *et al.*, 2008; Cole, 2010), physical capital (access to roads) and biophysical variables (slope of fields, soil fertility) (Sood & Mitchell, 2009). Other variables include conservation attitudes, the level of control or management given to farmers, and previous experience with tree planting and with outside development agents (Walters *et al.*, 1999; McGinty *et al.*, 2008).

In general, wealthier households with more land are more likely to plant on-farm trees, as are people who perceive they have secure land tenure (Bannister & Nair, 2003; Pattanayak *et al.*, 2003; Sood & Mitchell, 2009). People are also more likely to adopt if they have previous experience with tree planting (or development agents), or have observed others benefitting from a new technique (Walters *et al.*, 1999). People who are highly risk averse or who apply high discount rates are less likely to adopt (Bannister & Nair, 2003; Mercer, 2004). Asset-poor households are also less likely to adopt as they are more vulnerable to risk, or may lack resources to invest at all if they are below the 'investment poverty' line described by Reardon and Vosti (1995). In some cases, people who adopt one conservation measure may adopt others, or may be more inclined to support conservation initiatives (Walters *et al.*, 1999; Manzi & Coomes, 2009); however, this outcome is not guaranteed (Waylen *et al.*, 2009). Analyzing the circumstances under which conservation investments of one type lead to further conservation activities (and when they do not) can help managers foster such conditions, creating projects with pro-conservation impacts beyond their immediate objectives.

The literature on adoption has certain limitations. Because studies occur in different contexts and focus on different independent variables, household characteristics that predict adoption in one location can be insignificant or even related to non-adopters in others (Walters *et al.*, 1999; Pattanayak *et al.*, 2003; Mercer, 2004; Walters *et al.*, 2005; Doss, 2006; McGinty *et al.*, 2008). However, the local context in which adoption occurs is seldom systematically analyzed. In addition, with respect to applying this literature to restoration, most agroforestry systems are designed to increase yields or produce alternative products, providing direct, tangible economic benefits to farmers. In contrast, the benefits of community forest restoration may be fewer, less obvious (providing regulating, as opposed to provisioning ecosystem services, for example), or come from an outside source through payments for environmental services or ecotourism. Because of this, other factors – such as subsidies, prior experience with agencies, or other incentives – may have a relatively larger impact on why people participate in restoration. Chapter 5 of this thesis examines the applicability of this literature to studying community-based restoration, presenting what is, to my knowledge, the first case study of household-level participation in tree planting projects aimed at restoring forests.

Livelihood Analysis

Livelihood analysis employs a framework that integrates the asset-based analysis used in adoption studies with a more comprehensive picture of household resource use (Fig. 1.2). People's livelihood strategies will depend on their access to different assets or types of capital, such as natural, human, physical, and social capital (Reardon & Vosti, 1995; Bebbington & Perreault, 1999; Barrett *et al.*, 2001). Changes in livelihood strategies, such as obtaining off-farm employment, remittances from abroad, and shifts from extensive to intensive agriculture or from cattle to crops, can result in local increases in forest cover (Rudel *et al.*, 2002; Grau & Aide, 2008; Hecht, 2010; Redo *et al.*, 2012). Rudel *et al.* (2002) found that in the Ecuadorian Amazon when people switched from raising cattle to labour-intensive agriculture (in response to better access to produce markets) forests grew back in uncultivated areas. Kull *et al.* (2007) found that when farmers sought off-farm employment or migrated, because labour capacity was reduced on farms forests regenerated naturally on former, abandoned cropland. Government programs that provide subsidies and tax breaks for landholders to install tree plantations can drive smallholders to plant trees on agricultural land, which can also increase forest cover (Farley, 2007; Sloan, 2008). However, the effect of a given change in livelihood strategy on forest cover is context dependent: for example, in some regions, agricultural intensification and off-farm employment have led to forest clearing rather than recovery (Jokisch & Lair, 2002; Sarmiento, 2002; Schelhas & Sánchez-Azofeifa, 2006).

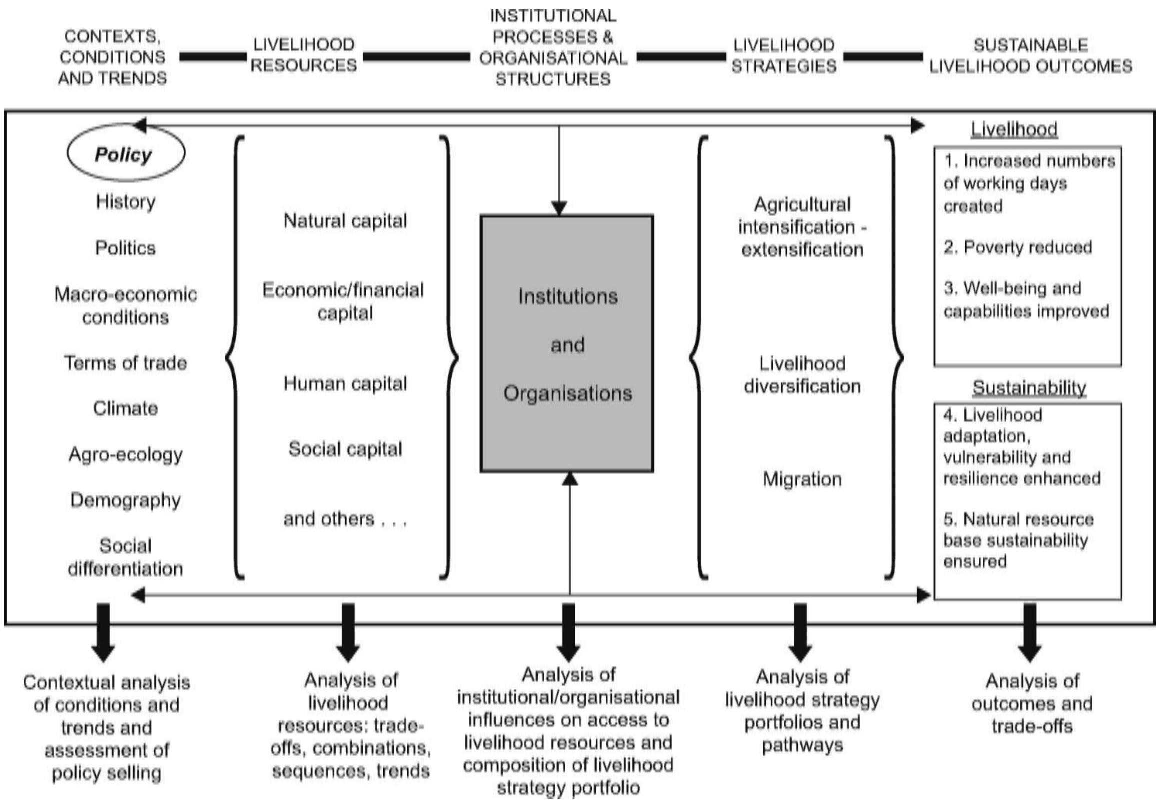


Figure 1.2: Sustainable livelihoods framework: a checklist (Scoones, 1998).

Changes in livelihood strategies often affect forest cover; and changes in forest cover, composition or protected status in turn affect rural livelihoods (Peluso, 1992; Rudel, 2009*a*; Scoones, 2009; Abram *et al.*, 2014) (Fig. 1.2). For example, creating conservation areas can limit rural people's access to land and forest resources, pushing people to use remaining land unsustainably or to pursue alternate income sources (Moore, 1993; West *et al.*, 2006*b*; Sodhi *et al.*, 2008). Changes in forest structure or species composition also affect the resources available to local people (Raffles, 1999; Robbins, 2001; Gamfeldt *et al.*, 2013). On the other hand, community-led conservation programs can open up new livelihood activities for people, including managing and harvesting timber, ecotourism, and selling non-timber forest products (Rudel, 2000; Agrawal *et al.*, 2008; Zambrano *et al.*, 2010; Bray, 2013). As livelihood strategies vary between households, the impact of changes in forest composition, abundance or protected status is also likely to differ between them. Although the poorest people in a community often depend most on forests, relatively wealthier households are better positioned to take advantage of development initiatives or to adopt new management strategies (Fischer & Vasseur, 2002; Sunderlin *et al.*, 2003; McSweeney, 2004; Grieg-Gran *et al.*, 2005; Hussain & Badola, 2010). How people adapt their livelihood strategies to cope with human-induced changes in the environment, such as deforestation and land degradation, differs depending in part on their experience with, and ability to invest in, environmental conservation measures (Walters *et al.*, 1999).

Community-based reforestation projects can change people's lives and livelihoods. Restoring forests by planting trees comes with opportunity costs: it takes time and land away from other activities. However, it can also provide people with income from forest protection payments or wages, and, in the longer term, with forest products and ecosystem services (Wunder, 2013). Working with local NGOs on projects can also provide people with connections to markets, information systems, and technical assistance (Bebbington & Perreault, 1999). Examining how households weigh these costs and benefits to choose to opt in or out of community-based restoration projects will help managers design and execute targeted projects with tangible benefits relevant to local people.

Community forest management

Because the benefits of restoring forests are often shared between members of a local community, forest restoration is potentially well suited to community-level management. Community-based forest management can be a highly successful means of managing tropical timber extraction and other products. Promoted on the basis that governments and private industries have done a poor job of managing forest resources, and that local communities, because of their knowledge of, proximity to, and reliance on forests can do better (Charnley & Poe, 2007; Agrawal *et al.*, 2008), community-managed forests have repeatedly shown lower deforestation rates than conservation areas and other forests on the landscape (Bray *et al.*, 2008; Duchelle, 2009; Persha *et al.*, 2011; Porter-Bolland *et al.*, 2012; Bray, 2013). At its best, community-based forest management employs local people and increases the benefits they receive from forests, thereby improving livelihoods, building local support for sustainable forest management, and increasing forest stewardship. It is, therefore, also said to be more sustainable than top-down government management because people stand to benefit from forests in the long term (Charnley & Poe, 2007; Bray, 2013).

Of course, not all projects produce such positive results. Problems with resource distribution can arise, and benefits are sometimes captured by 'rural elites' at the exclusion of poorer community members (Kumar, 2002; Persha & Andersson, 2014). Corruption can result in unsustainable management, and some projects fail to produce forest conservation benefits (Bray *et al.*, 2006; Bray, 2013). The most successful projects are found in well organized communities with strong local institutions and leaders (Bray *et al.*, 2006; Persha *et al.*, 2011; Bray, 2013).

Community-based forest management is only one of many types of communal land management – local people have been managing the commons for other purposes for centuries (Netting, 1972a, 1976; Gilles & Jamtgaard, 1981; Ostrom, 1985; Menzies, 2014). Netting (1976) characterizes the types of land use that people have historically managed under communal governance, and are thus potentially best suited to community-based management today (Table 1.1). Ostrom (1985) summarizes Netting's argument:

> Communal forms of land tenure are optimal when the value of production per unit of land is low, when the frequency and dependability of use or yield is low, when the possibility of improvement or intensification is low, when large areas are required for effective use, and when relatively large groups are required for capital investment activities. (Ostrom, 1985, pg. 14).

These criteria describe forest restoration well – the benefits are relatively small per unit of land (compared to crops, for example), diffuse and distributed among community members, and require relative larger areas of land to be realized. But, they can also be important for farming (e.g., pollination, water flow regulation, erosion control, and so on). Thus, although restoring forests may not be a 'worthwhile' investment for a single landholder, it can make economic sense as a community (Netting, 1972*a*, 1976).

	Land tenure type	
Attributes of land use	**Communal**	**Individual**
Value of production per unit area	Low	High
Frequency and dependability of use or yield	Low	High
Possibility of improvement or intensification	Low	High
Area required for effective use	Large	Small
Labour- and capital-investing groups	Large (voluntary association or community)	Small (individual or family)

Table 1.1: Attributes of land use patterns historically associated with communal versus private land tenure identified by Netting (1976). Table adapted from Ostrom (1985).

Restoring communal lands could also work well because degraded areas most in need of restoration will often be located in working agricultural landscapes – having the support of local people is essential for their success (Zanella *et al.*, 2014). Communal projects are attractive to donors and agencies because, compared to working with individual farmers, they have lower transaction costs, are easier to monitor, and have higher accountability (Agrawal *et al.*, 2008; Larson & Soto, 2008). Finally, having group control over land titles can also safeguard the land from being cleared, increasing the longevity of restoration projects (Agrawal *et al.*, 2008). Although restoration seems well suited to communal land tenure arrangements, to my knowledge this thesis is the first to comprehensively examine how well community-based projects work in practice to restore Andean forests: who participates in them, why they do so, and the environmental outcomes they ultimately produce.

Forest ecology: Not all forests are created equal

Land-use and -cover changes, including forest transitions, will inevitably have implications for regional biodiversity and species conservation. However, often these impacts are poorly addressed in such studies, in part because many definitions of 'forest' – including the commonly cited and applied FAO[2] definition – used in forest cover studies do not take into account forest age, vertical structure, or all but the most severe levels of degradation (Lund, 2009; FAO, 2011; Hansen *et al.*, 2014; Putz & Romero, 2014; Tropek *et al.*, 2014). This is problematic because different kinds and ages of forest support different types and quantities of species, and even naturally regenerated secondary forest often differs in plant species composition and diversity from primary forest (Pascarella *et al.*, 2000; Kanowski *et al.*, 2005; Bhagwat, 2008; Dent & Wright, 2009; Klanderud *et al.*, 2010; Chai & Tanner, 2011; Martin *et al.*, 2013; Putz & Romero, 2014). To understand the ecological impacts of changes in forest cover and forest transitions, it is essential to quantify both deforestation and reforestation rates, and to compare the ecology and biodiversity of the forests that are cleared with those that return.

[2] "Forests are lands of more than 0.5 hectares, with a tree canopy cover of more than 10 percent, which are not primarily under agricultural or urban land use." (FAO, 2000, pg. 7).

In the tropics, the three most common types of secondary forests to result in forest transitions are naturally regenerated forests, tree plantations, and trees in agroforestry systems (Rudel, 2009*a*). Now, many environmental NGOs, agencies and researchers also promote tree planting to restore forests and forest biodiversity (UNEP, 2013). But restoring forest is hard work, requiring far more labour and financial resources than natural regeneration and promising fewer financial gains than plantations or agroforestry. So, it is worth it? To identify when and where restoring forests is needed (and outperforms other secondary forests types) to conserve biodiversity requires an understanding of the processes of and barriers to forest succession in tropical landscapes.

Forest biodiversity and regeneration in tropical landscapes

Naturally regenerating tropical forest can take decades to attain the plant species diversity of primary forests, if at all (Lamb *et al.*, 2005; Dent & Wright, 2009; Klanderud *et al.*, 2010; Chai & Tanner, 2011; Martin *et al.*, 2013). Even though some secondary forests have high plant diversity, the species are often different than those in primary forests (Turner *et al.*, 1997; Pascarella *et al.*, 2000; Dent & Wright, 2009; Klanderud *et al.*, 2010; Chai & Tanner, 2011; Martin *et al.*, 2013). Landscapes with a mix of primary and secondary forest fragments can thus have very high diversity (Castillo-Campos *et al.*, 2008); however, when primary forest is replaced by secondary forest over a large area, forest composition can become homogenized, ultimately decreasing landscape-level diversity (Foster *et al.*, 1998; McKinney & Lockwood, 1999; Holl, 2002; Foster *et al.*, 2003; Lugo & Helmer, 2004; Rhemtulla *et al.*, 2007). Both landscape characteristics and land use 'legacies' – environmental conditions arising from past use – affect the plant-species composition and diversity of naturally regenerated secondary forests (Wunderle Jr, 1997; Aide *et al.*, 2000; Foster *et al.*, 2003).

Barriers to tropical forest succession

The intensity, type and duration of previous land use, distance to remnant forest patches, availability of seed dispersers, and existing vegetation in the area (both ground cover and remnant trees), all affect the rate of natural regeneration and the types and abundance of plants that regenerate (Uhl *et al.*, 1988; Parrotta *et al.*, 1997; Wunderle Jr, 1997; Aide *et al.*, 2000; Holl *et al.*, 2000; Pascarella *et al.*, 2000; Guariguata & Ostertag, 2001; Mesquita *et al.*, 2001; Chazdon *et al.*, 2003; Florentine & Westbrooke, 2004). Previous land use can affect both the rate and outcome of forest succession processes – forests can often regenerate quickly on lightly used sites, but can be slow or unable to grow back in areas with heavier or longer use (Uhl *et al.*, 1988; Styger *et al.*, 2007; Chazdon, 2008; Cramer *et al.*, 2008). Planted pasture grass can also inhibit natural regeneration (Aide & Cavelier, 1994; Holl *et al.*, 2000; Griscom *et al.*, 2009), which is often why farmers plant it.

Related to the duration and type of previous land use is the presence of remnant vegetation after clearing. Many tropical trees can coppice from stumps and roots (Uhl & Jordan, 1984; Chazdon *et al.*, 2003), and others can regenerate from seeds stored in soil (Uhl *et al.*, 1981; Whitmore, 1990; Guariguata & Ostertag, 2001; Chazdon *et al.*, 2003). Both seed banks and stumps become less productive over time and with the intensity of land use. Fire is particularly detrimental (Aide & Cavelier, 1994; Kammesheidt, 1999; Guariguata & Ostertag, 2001). The surrounding landscape also affects the types of species found in seed banks: soils in sites near primary forests have more woody species; those in cleared land, more herbaceous ones

(Guariguata & Ostertag, 2001). Remnant vegetation such as pasture trees and windbreaks also facilitate natural regeneration – even lone trees provide shade and can moderate extreme conditions, improve soil fertility, and attract seed dispersers (Guevara *et al.*, 1992; Rhoades *et al.*, 1998; Harvey, 2000; Guariguata & Ostertag, 2001; Chazdon *et al.*, 2003; Jacob, 2014). Windbreaks also increase habitat connectivity for seed dispersers (Harvey, 2000; Holl *et al.*, 2000). In a landscape context, cleared patches far from remnant forest often take longer to regenerate and have lower plant diversity than those close to forest (Guariguata & Ostertag, 2001; Günter *et al.*, 2007; Weber *et al.*, 2008; Aide *et al.*, 2010) because seed dispersal – by both wind and animals – decreases with distance (Aide & Cavelier, 1994; Wunderle Jr, 1997; Holl, 1998; Guariguata & Ostertag, 2001). In deforested tropical landscapes, forest fragments are important sources of propagules for forest recovery (Turner & Corlett, 1996; Muñiz-Castro *et al.*, 2006).

Overall, although forests can regenerate quickly in many areas, restoring tree cover and biomass, they often remain distinct from primary forests for decades, centuries, or permanently (Turner *et al.*, 1997; Foster *et al.*, 1998; Pascarella *et al.*, 2000; Lamb *et al.*, 2005; Chazdon, 2008; Dent & Joseph Wright, 2009; Ortega-Pieck *et al.*, 2011). Fragmented areas with long histories of human land use and clearing – the current conditions in many formerly cloud-forest covered Andean regions – are the least likely to recover naturally (Aide *et al.*, 2010; Mulligan, 2010; Young, 2011). Planting trees may be essential to reforest highly degraded, fragmented montane landscapes.

Planted forests and biodiversity

Plantations

Industrial plantations, typically monospecific stands of non-native species, are the most common type of planted 'forest' globally (FAO, 2007; Rudel, 2009*a*; FAO, 2011). In some cases, native trees regenerate profusely in the understories of exotic plantations (Parrotta, 1992; Lugo, 1997; Feyera *et al.*, 2002). However, managing for plant diversity often conflicts with strategies to increase growth and timber yields, and requires additional harvesting effort (Lindenmayer & Franklin, 2002). Once plantation trees are harvested, the area must be devoted to 'natural' forest, which may not be compatible with the financial goals of plantation owners. Some plantation species also do not facilitate natural regeneration – teak (*Tectona grandis*), for example, often erodes and depletes soils (Parrotta, 1995) (Healey & Gara, 2003). In general, plantation forests have low biodiversity and structural complexity, and are dissimilar to native forests (Healey & Gara, 2003; Kanowski *et al.*, 2005; Fitzherbert *et al.*, 2008; Zhai *et al.*, 2014).

Agroforestry

The term 'agroforestry' encompasses a diverse range of systems, from planted hedgerows, windbreaks and pasture trees to crops (such as coffee) grown in the understories of old-growth or old secondary forest (Schroth *et al.*, 2004). These systems can benefit farmers by preventing soil erosion, providing perennial tree crops and other wood products, increasing or stabilizing agricultural yields, and diversifying livelihoods (Jose, 2009; Power, 2010). Agroforestry systems can also aid natural succession in nearby regenerating forest by providing seed sources, habitat for seed dispersers, and favorable conditions for tree growth (Harvey, 2000; Fávero *et al.*, 2008; Vieira *et al.*, 2009; Benayas & Bullock, 2012).

Generally, the diversity of various taxonomic groups is higher in agroforestry systems than in other types of agriculture, but lower than in primary forest (McNeely & Schroth, 2006; Schroth & Harvey, 2007; Bhagwat, 2008; Power, 2010; Phalan *et al.*, 2011; Kremen & Miles, 2012). In a meta-analysis, Bhagwat *et al.* (2008) found that, on average, tree-species diversity was lower in agroforestry systems than in nearby primary forest reserves. Many agroforestry systems are also cyclical: forests are grown for a time, cleared or thinned, then replanted or allowed to regenerate (Schroth *et al.*, 2004). Thus, it cannot be assumed that the endpoint of an agroforestry system is permanent, biodiverse forest, although they do provide many forest ecosystem services and can conserve relatively high levels of biodiversity (Jose, 2009; Power, 2010). Overall, in terms of their conservation value, agroforestry systems are a potentially transient middle ground between primary forests and industrialized agricultural systems, although in a landscape context they provide a valuable refuge for many forest species (Tscharntke *et al.*, 2012).

Ecological restoration

In contrast to plantations and agroforestry systems, where trees are often planted for utilitarian purposes, the primary goal of ecological restoration is to return a 'degraded' habitat to a historical 'natural' state (Harris & van Diggelen, 2006; Palmer *et al.*, 2006). The terms 'historical' and 'natural' are problematic: humans have modified much of what we consider natural, and landscapes often pass through many historical and pre-historical eras (Denevan, 1992; Foster *et al.*, 2003). The Society for Ecological Restoration (SER) avoids these terms, defining ecological restoration "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER, 2004, pg. 4), leaving land managers to decide what ecosystem they are restoring to.

In practice, people often use restoration methods, such as planting trees, to establish tree cover and lift barriers to natural regeneration. They also reintroduce specific native species, especially ones that are endemic, endangered, keystone, or valuable for timber. Ultimately, using restoration managers aim to produce self-sustaining, ecologically functional forests (González-Espinosa *et al.*, 2007; Nave & Rodrigues, 2007, pg. 103; Griscom & Ashton, 2011). Restoring highly degraded sites may involve treating the site (i.e., by replacing soil) prior to restoration (Chazdon, 2008), whereas on sites where some natural regeneration can occur planting trees may serve primarily to reintroduce trees with limited dispersal ability. Between these two extremes lie a range of other treatments that can be applied depending on how degraded the site is and the resources available for restoration (Rodrigues., 2007; Chazdon, 2008). A common technique to reforest pasture in South America is to plant and then weed around seedlings of both fast-growing pioneer tree species and primary forest species until they are established (Leopold *et al.*, 2001; González-Espinosa *et al.*, 2007; Holl, 2011).

Planting trees is certainly more costly than natural forest regeneration (Montagnin *et al.*, 1995; Coomes *et al.*, 2008; Holl, 2011), but in degraded areas also has the potential to produce greater ecological and social benefits. People can plant species that are economically valuable or produce food (Nepstad *et al.*, 1991; Leopold *et al.*, 2001; Garen *et al.*, 2009). Many old-growth tree species have large seeds with limited dispersal capacity, and are more likely to establish if they are transported and planted directly (Whitmore, 1990; Holl *et al.*, 2000). Planting fruiting trees can attract seed dispersers such as birds (and cows) (Butterfield, 1995; Holl *et al.*, 2000; Pena-Domene *et al.*, 2013; Jacob, 2014). Forests restored with native species may, by design, be more similar to primary forests in terms of plant-species composition and diversity than naturally regenerated forests are (Nepstad *et al.*, 1991; Parrotta & Knowles, 1999; Pena-Domene *et al.*, 2013).

Restoration 'success' is often assessed by comparing the restored system both to a 'reference' site (i.e., the target ecosystem) and a 'control' site (i.e., a site that was left to regenerate naturally) (Jordan III *et al.*, 1987; Ruiz-Jaen & Mitchell Aide, 2005; Harris & van Diggelen, 2006). Many studies have measured tropical forest restoration success using growth metrics such as biomass accumulation, stem counts, or vertical structure (Young, 2000; Leopold *et al.*, 2001; Ruiz-Jaén & Aide, 2005; Ruiz-Jaen & Mitchell Aide, 2005). Surprisingly few studies compare plant diversity in restored, naturally regenerating, and primary tropical forests (Ruiz-Jaen & Aide, 2005). The few that do have shown that primary forests and older secondary forests have higher plant-species richness than either restored or young naturally regenerated forests. In some places, restored forests have higher woody species richness than naturally regenerated forests, but in others richness is similar in both (Parrotta & Knowles, 1999; Ruiz-Jaén & Aide, 2005; Ren *et al.*, 2007; Pena-Domene *et al.*, 2013). Differences in the results can be explained by the condition of the land prior to treatment. In severely degraded habitats, restoration methods can have a relatively larger impact because they are needed to reintroduce species or overcome barriers to succession (Parrotta & Knowles, 1999; Ren *et al.*, 2007) (Ruiz-Jaén & Aide, 2005), (Uhl & Jordan, 1984; Chazdon, 2003; Leopold & Salazar, 2008). To maximize restoration efficiency managers should identify the barriers to succession at a specific site and apply the minimal intervention required to overcome them (Uhl *et al.*, 1988; Chazdon, 2003; Lamb *et al.*, 2005), but because small-scale community projects often lack resources for rigorous site assessments, projects proceed by trial and error using the tools and resources available. But their outcomes are especially valuable to future restoration efforts – because they show what people can do with limited resources and common farming tools (such as machetes, axes and shovels), local restoration efforts can be analyzed to identify best practices for community-based projects in similar environments.

Restoration and conservation

Because so many of the earth's ecosystems have now been altered, degraded or destroyed, the conservation community now promotes ecological restoration to complement conservation efforts (Lamb *et al.*, 2005; Foley *et al.*, 2007; Turner *et al.*, 2007; Chazdon, 2008; DeFries *et al.*, 2012). This is a major paradigm shift from as little as two decades ago when restoring ecosystems was seen to be in conflict with conserving them – critics feared it would divert resources from conservation, become an excuse not to conserve, and ultimately result in the proliferation of 'false' natures (Eliot, 1982; Katz, 1992; Higgs, 2003). But today many recognize that although conserving ecosystems is economically and ecologically preferable to restoring them (Young, 2000; Higgs, 2003; Coomes *et al.*, 2008), restoration can complement conservation by repairing and expanding primary ecosystems. A source of ecological data, seeds and sprouts, and animal dispersers and pollinators, remnant primary forests are also essential for forest restoration efforts (Holl *et al.*, 2000; Chazdon, 2003).

Tropical montane cloud forests

Tropical montane cloud forests are just that: tropical forests typically confined to mountainous regions where they are frequently engulfed in clouds (Grubb, 1977; Scatena *et al.*, 2010). They are also wild, steep, misty places where epiphytes grow so profusely that it is difficult to tell where they stop, and trees begin. Globally, the largest pockets of tropical montane cloud forests (hereafter referred to as ‘cloud forests’) are found in Indonesia, the Congo, Brazil, Venezuela, Peru and Colombia (Mulligan, 2010). The Andean countries of Colombia, Peru, Ecuador and Bolivia collectively contain over 430,000 km^2 of cloud forest, nearly 20 % of the cloud forests on the planet (Mulligan, 2010). A type of ‘moist tropical forest’, cloud forests are characterized by moderate to high levels of precipitation, and are confined to areas with frequently cloud immersion (Grubb, 1977; Stadtmüller, 1987; Hamilton *et al.*, 1995). The elevation range in which they occur varies with latitude, precipitation, and topography (Doumenge *et al.*, 1995; Webster *et al.*, 1995), but is typically between 1200 to 3500 masl (Doumenge *et al.*, 1995; Mulligan, 2010; Scatena *et al.*, 2010). Located at high elevations and often on steep hillsides, cloud forest soils and hydrology differ from lowland forests (Bruijnzeel & Proctor, 1995; Bruijnzeel *et al.*, 2010*b*). Cooler temperatures mean that trees grow more slowly.

Cloud forests play a unique and important role in the hydrological cycle: they capture water from clouds. As wind-driven clouds contact forest vegetation, they condense and fall to the ground as droplets, or flow along the stems of trees and plants (Stadtmüller, 1987; Bruijnzeel & Proctor, 1995; Doumenge *et al.*, 1995). The absolute amount of this ‘horizontal precipitation’ depends on the amount and frequency of cloud cover (Grubb, 1977) and the direction and speed of the prevalent winds, both of which vary with topography and elevation (Stadtmüller, 1987; Doumenge *et al.*, 1995; Guswa *et al.*, 2007). The height of trees and canopy structure also affect the amount of horizontal precipitation – even an exposed solitary tree can capture a large volume of water (Stadtmüller, 1987). Horizontal precipitation increases the amount of water reaching the ground, sometimes so much so that more water reaches the ground under the forest canopy than in neighbouring unforested areas, the reverse of what occurs in many non-cloud tropical forests

(Bruijnzeel & Proctor, 1995). The volume of horizontal precipitation relative to rainfall in montane forests can be substantial, from seven to over 100 % (Bruijnzeel & Proctor, 1995; Bruijnzeel, 2004). In areas that receive less rainfall, and during the dry season, horizontal precipitation makes up a larger proportion of overall precipitation. Cloud capture thus plays a key role in sustaining forests throughout the dry season (Vogelmann, 1973; Stadtmüller, 1987; Bruijnzeel & Proctor, 1995; Guswa *et al.*, 2007).

Located on steep slopes, cloud forest soils are prone to erosion, and landslides are common especially in deforested areas (Daugherty, 1973; Baillie, 1996; Foster, 2001; Bruijnzeel, 2004). Montane soils are wet year round, relatively acidic, and have slower decomposition, mineralization, and nutrient cycling rates than lowland forests (Edwards & Grubb, 1977; Stadtmüller, 1987; Bruijnzeel & Proctor, 1995; Lieberman *et al.*, 1996; Foster, 2001). Either phosphorous or nitrogen can be a limiting nutrient (Grubb, 1977; Lieberman *et al.*, 1996; Tanner *et al.*, 1998). Slower decomposition rates result in a thicker organic matter layer and the amount of soil organic matter increases with elevation (Grubb *et al.*, 1963; Grubb & Whitmore, 1966; Baillie, 1996; Tanner *et al.*, 1998; Wilcke *et al.*, 2002; Bruijnzeel, 2004). These differences in hydrology, soils, and temperature, all driven by variation in elevation and topography, create gradients in forest structure. As elevation increases, canopies become lower and less stratified, buttresses are increasingly rare as trees become squat and twisted, and root systems are shallower. Epiphytes increase as vines decrease, and at higher elevations, are mostly mosses (Grubb *et al.*, 1963; Grubb, 1977; Stadtmüller, 1987; Lawton & Putz, 1988; Whitmore, 1990; Bruijnzeel & Proctor, 1995; Webster *et al.*, 1995; Lieberman *et al.*, 1996; Richards, 1996).

The distribution of cloud forests, and species within cloud forest, is determined by topography, aspect, elevation, slope, past geological and climatic events, and rapid radiative speciation. Topography plays a large role in where and how forests grow, with valleys and ravines proving more hospitable environments for tree growth than ridgetops (Stadtmüller, 1987; Muñiz-Castro *et al.*, 2006). Above 1500 meters, the lower boundary for cloud forests in Latin America, tree species richness decreases (Balslev, 1988; Doumenge *et al.*, 1995; Webster *et al.*, 1995). However, the diversity of epiphytes and other plant species increases with elevation and often peaks between 1500 and 2000 masl, a pattern termed the 'mid domain' or 'biodiversity bulge' effect (Gentry, 1989; Henderson *et al.*, 1991; Krömer *et al.*, 2005). In a multi-site survey of cloud forests in the Northern Andes, Gentry (1992; 1995) found major divisions in floristic composition between lowland and mid-elevation cloud forest at 1500 masl, between mid-elevation and upper-elevation cloud forest at 2500 masl, and between upper-elevation cloud forest and elfin forest above 2900 masl. Cloud forests are floristically distinct from lowland forests – in upper-elevation and elfin cloud forest, certain plant families common to lowland forest are nearly absent, while other families are much more abundant (Wolf, 1993; Gentry *et al.*, 1995; Webster *et al.*, 1995; Lieberman *et al.*, 1996).

Because of their spatially variable and dynamic species compositions, of the megabiodiverse moist tropical forests in Latin American (Gentry, 1992; Richards, 1996; Myers *et al.*, 2000; Brooks *et al.*, 2006; Griscom & Ashton, 2011) tropical montane cloud forests may be the most speciose (Bubb, 2004; Scatena *et al.*, 2010), and cloud forests are widely recognized as a top global conservation priority (Mittermeier *et al.*, 1998; Myers *et al.*, 2000; Brooks *et al.*, 2006; Scatena *et al.*, 2010). Although at a local scale, tree species richness is often lower in cloud forests (e.g., in the Andes) than in lowland forest (e.g., the Amazon), because cloud forests have higher numbers of endemic species and rapid species turnover, beta-diversity is typically higher in cloud forests (Noss, 1983; Balslev, 1988; Whitmore, 1990; Churchill *et al.*, 1995;

Gentry *et al.*, 1995; Haber, 2000; Brooks *et al.*, 2002; Küper *et al.*, 2004). Confined to narrow elevation bands (in Latin America, generally between 1500 and 3500 masl) they house a disproportionally high percentage of the earth's plant, amphibian, and bird species less than 1.4% of its land area (Doumenge *et al.*, 1995; Bruijnzeel *et al.*, 2010*b*; Mulligan, 2010). In Mexico, for example, cloud forests contain 10-12 percent of all vascular plant species in only 0.8 percent of its area (Vargas-Rodriguez *et al.*, 2010), and nearly 20% of the plant species in Peru are found in five percent of the country covered in Andean montane forest (León *et al.,* 1992; Young & León, 1995*a,b*; Young & León, 1999). Spanning the high paramos to lower montane rainforests, the Tropical Andes as a whole contain over 15% of all known plant species in less than one percent of the earth's land area (Mittermeier *et al.*, 1998; Richter *et al.*, 2009).

Most of the research on tropical forest restoration, and indeed most research on tropical forests in general, has focused on lowland moist forest such as the Amazon (Henderson *et al.*, 1991; Churchill *et al.*, 1995; Bubb, 2004; Griscom & Ashton, 2011). In contrast, cloud forests are relatively little studied yet still extremely biodiverse ecosystems. In the Northern Andes, cloud forests are now largely fragmented (Doumenge *et al.*, 1995), and an estimated 90% have been cleared (Gentry, 1989; Henderson *et al.*, 1991). Recent studies show that clearing in these forests continues today for a variety of reasons (Sarmiento, 1995*a*, 1995*b*; Jokisch & Lair, 2002). The scarcity and level of fragmentation in Andean cloud forests means that effective restoration may be a vital tool to increase forest cover in cleared areas, as these regions are both endangered and perhaps less likely to recover unassisted (Sarmiento, 1995*b*; Sarmiento *et al.*, 1995; Young & León, 1995*b*).

The location and unique flora of cloud forests mean that restoring these ecosystems comes with challenges, such as slow regeneration, highly eroded areas, steep slopes and different flora (and fauna) from lowland forests. While many small-scale restoration projects are underway in the Northern Andes, to my knowledge, no study, published or otherwise, measures how effective community-based cloud forest restoration is at restoring plant diversity or species composition. In this dissertation, I aim to fill that gap.

Restored forests are becoming more common in tropical landscapes. Although its multiple social and ecological benefits are both appealing and theoretically possible, they are by no means guaranteed. With respect to restoring forest cover, three scenarios are possible: restoring forest could 1) lessen pressure on primary forests and slow deforestation, potentially leading to a local forest transition; 2) have no effect on primary forest use or clearing; or 3) increase primary forest clearing by taking agricultural land out of production ('leakage'). In addition, reforestation has the potential to change people's lives and livelihoods because it can alter the amount of land available for other livelihood activities, and potentially creates new or enhanced revenue sources or market opportunities through payment for environmental services, government incentives, or NGO support. Finally, although the ultimate goal of ecological restoration is often to conserve diversity, the ability of communal projects to achieve this in tropical landscapes is still largely uninvestigated.

Study Region: Andean Ecuador

Hidden among the western Andean foothills of Ecuador, a few kilometers from Rio Palenque, there is a small ridge called Centinela. Its name deserves to be synonymous with the silent hemorrhaging of biological diversity. When the forest on the ridge was cut a decade ago, a large number of rare species were extinguished. They went just like that, from full healthy populations to nothing, in a few months. Around the world such anonymous extinctions – call them 'Centinelan extinctions' - are occurring, not open wounds for all to see and rush to stanch but unfelt internal events, leakages from vital tissue out of sight. Only an accident of timing led to an eyewitness account of the events on Centinela.

– E.O. Wilson on cloud forests in northwest Ecuador (*The Diversity of Life,* 1992)

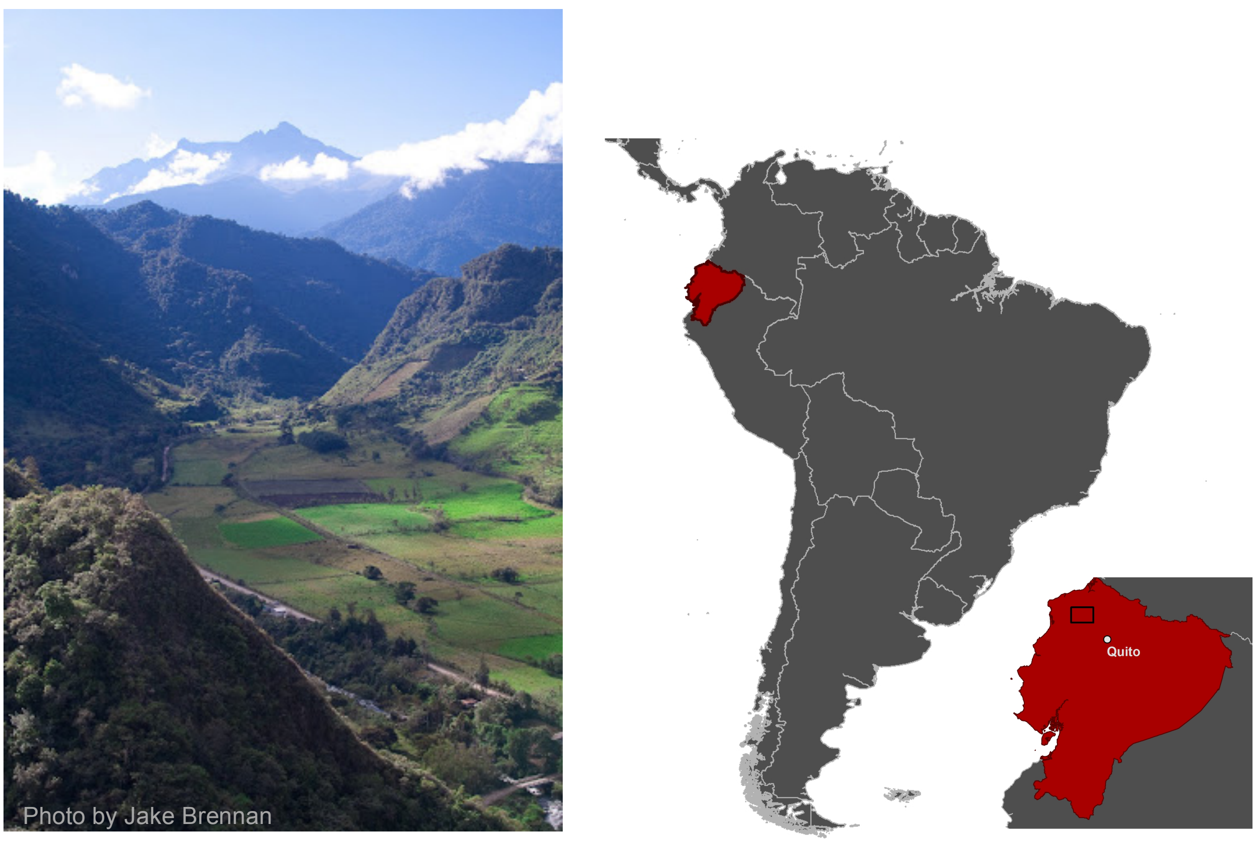


Figure 1.3: The Intag region. The image shows a view up the northern portion of the Intag Valley.

In the period from 1990 to 2005, Ecuador experienced the highest deforestation rates in South America (FAO, 2007; Mosandl *et al.*, 2008), and, over the past decade, Andean forests were cleared at a rate of about 1% per year (Portillo-Quintero *et al.*, 2012*b*). Deforestation in Andean landscapes often occurs from both 'above', as pastures in the páramo highlands are expanded using fire, and from 'below', as land is cleared for pasture and to grow crops (Jokisch & Lair, 2002; Sarmiento, 2002; Farley, 2007). However, in the midst of this deforestation, reforestation is also occurring in some regions (Aide *et al.*, 2013) through natural regeneration, timber plantations, and local reforestation projects (Maquipucuna Foundation; Jokisch & Lair, 2002; Farley, 2007). Government incentives to reforest the Andes have historically promoted, and continue to promote, plantations of non-native Eucalyptus and Pine in the Ecuadorian highlands (Gade, 1999; Farley, 2007, 2010), which have been found to increase soil erosion and reduce soil water storage capacity (Farley, 2007). As an alternative reforestation strategy, several communities in Andean Ecuador have reforested[3] land with native trees, often in partnership with local and international NGOs. NGO-supported forestry projects in Ecuador are generally clustered in areas with high conservation priority (i.e., high biodiversity, endangered species, and high numbers of endemic species) that are relatively easy to access, and where NGOs are able to work effectively with communities (Raberg & Rudel, 2007; F. Perez pers. comm. 2010).

The Northern Andes are steep, rugged, often wet, and sometimes cold. The livelihoods and agricultural practices of the people who live there are characterized by diversification on many levels, in part reflecting the diversity of microclimates in the region (Winterhalder & Thomas, 1978; Brush, 1982; Mayer, 2002). 'Traditional' agricultural practices include planting crops across a range of elevations to minimize the risk of an extreme climatic event or

[3] In this dissertation, human-induced 'reforestation' or 'forest restoration' refer to areas that are replanted with the ultimate goal being a forest, and do not include agroforestry or silvopastoral systems.

unexpected frost destroying all crops at once (Brush, 1977; Bebbington, 1993; Mayer, 2002). Cropping is staggered in time for the same reason, and people plant both diverse crops and cultivars with a single farm or community (potatoes are a well-studied example) (Brush *et al.*, 1981; Bebbington, 1990; Goland, 1993; Zimmerer, 1998; Mayer, 2002). Terracing and irrigation systems are part of this strategy in some areas (Bebbington, 1993; Trawick, 2001; Mayer, 2007). Today, in many places Andean landscapes are characterized by slow and steady land degradation, due in part to the high costs involved in sustaining these systems (Jokisch, 2002; Mayer, 2007). Bebbington (1993) links recent changes in peasant land use and management in the Ecuadorian Andes with country-wide changes in land tenure laws. Whereas under the hacienda system (large estates where peasants were granted land use in exchange for a share of their harvest) many households had access to common grazing areas, the land reforms in 1964 and 1973 partitioned estates into smaller parcels. In some cases, this eliminated access to common grazing ground for many households, and hence access to organic fertilizer, ultimately causing soil fertility to decline (Bebbington, 1993). Today, many Andean areas experience high rates of seasonal (or longer) migration to cities and abroad, which can lead to further agricultural decline (Bebbington, 1990). Under these conditions, secondary forests are returning to abandoned farms in some places, while in others extensive, low density cattle ranching (e.g., less than 1 cow per hectare) or severe land degradation have prevented forests from growing back (Sarmiento *et al.*, 1995; Jokisch & Lair, 2002). In these areas, restoration could be important role to enhance both rural livelihoods (through employment, ecosystems services, or both) and forest cover.

The Intag Valley

I studied communities in the beautiful Intag Valley, located on the western slopes of the Andes in northern Ecuador (Imbabura province) (Fig. 1.3). Many of the forests in northwest Andean Ecuador are fragmented following extensive clearing for agriculture and pasture over the past 50 years. Much of the remaining primary forest exists on privately owned land or in reserves (Sarmiento 1995*b,c*; C. Zorrilla pers. comm., 2010). The area has been prioritized as a conservation corridor to protect biodiversity and habitat for threatened species such as the spectacled bear. In recent years, a number of community forest restoration projects have been undertaken in this region with the support of local, national and international NGOs (Maquipucuna Foundation; Santa Lucía Cloud Forest Reserve; DECOIN, 2010; Rainforest Concern, 2010; Kocian *et al.*, 2011).

Cloud forests in this region are exceptionally biodiverse (Gentry, 1989). In 1976, botanists first explored the flora of Centinela, one of many cloud forest-covered ridges on the Eastern slopes of the Andes (Gentry, 1986; Dodson & Gentry, 1991). What they discovered – over 90 endemic species of epiphytes, herbs and other plants in less than 10 square kilometers – showed that small patches of cloud forest contribute disproportionately to global biodiversity. They hypothesized that this was, in large part, due to a high degree of isolation from similar habitats. Located on a ridge, Centinelian forests were separated from other forests at the same elevation – forests located only 300m lower were composed of dissimilar tree communities, and to the east, rapid increases in elevation up to the grassland *paramos* of the high Andes separated Centinelian forest from other, similar Andean forests (Gentry, 1986; Dodson & Gentry, 1991). Thus, although still connected to the surrounding landscape to some extent by the movement of animals and wind, in many ways Centinela resembled an island (MacArthur, 1967; Wilson, 1992). Rapid explosive speciation combined with relatively high geographical isolation created forests that were unique in the word both in terms of the species they supported, and the combinations in which these species were found (Gentry, 1986; Dodson & Gentry, 1991; (Wilson, 1992).

Cloud forests in the Intag region were also extensively cleared in the 1970s, 1980s, and 1990s to make way for pastures and agriculture (Sarmiento, 2002; Kocian *et al.*, 2011; C. Zorrilla pers. comm. 2010). Only 150 km from Quito, the region is still remote, due in large part to its steep and mountainous topography (several communities here became road-accessible only within the past decade). Today, land in the valley is used primarily for mixed subsistence and small-scale, market-oriented agriculture; cattle pastures; and orchards. While some people work as wage labourers or receive remittances from relatives working in Quito, opportunities to earn cash are generally scarce.

Located within a region of valuable mineral deposits, communities in Intag have been fighting off gold mining exploration for decades. As part of this effort, several local organizations formed to provide livelihood alternatives to mining and to help communities self-organize and mobilize (Bebbington *et al.*, 2008; Kocian *et al.*, 2011). These organizations include a highly successful shade coffee cooperative (AACRI), a farmers association (ACAI), and an ecotourism group (RED). In 1995, a local NGO *Defensa y Conservacion Ecologica de Intag* (DECOIN) began to help communities purchase and protect land in watershed areas serving communities in the Intag Valley. DECOIN initiated their watershed project to address the extensive loss of forest cover and biodiversity in the region, and the accompanying pollution of local water sources from agriculture in watersheds. To date they have helped 47 communities throughout the Intag Valley create small (eight to 150 ha) watershed reserves. Land in most of these reserves was fenced off and left fallow to allow forests to regenerate naturally. In six communities, DECOIN also helped communities replant forest. Reforestation efforts focused on pastures where planted, non-native pasture grass inhibited natural regeneration (Aide & Cavelier, 1994; Griscom *et al.*, 2009). DECOIN plans to include a tree-planting component in most future reserve projects, both because funding for reforestation is more readily available than funding for land acquisition alone, and because they have observed that planting trees as a community creates support for reserves (C. Zorrilla, pers. comm., 2010).

Once a project is initiated, land in the watershed is purchased from community residents. The watershed reserves are communally owned, but with use restrictions in the title – no animals, no burning, no harvesting for sale, no clearing; however, DECOIN encourages people to harvest wood, plants or fruit from the reserve for personal use, and to use the reserves for ecotourism. In communities with reforestation projects, DECOIN provides people with materials and technical expertise to build tree nurseries, and helps them apply to the municipal government for other supplies. Individual households or planting groups are contracted to grow and plant a specified number of trees, from seeds they collect from nearby forests and trees. People plant pioneer and primary forest species in combination. Seeds are sown in small bags of earth, grown in a nursery for one to a few months, and planted. DECOIN also pays some community members a daily wage to maintain trees on an 'as needed' basis. Projects have been widely accepted by community members, and DECOIN now receives more requests from additional communities wishing to establish watershed reserves than they can fill.

Methods

Field visits

I spent eight months conducting fieldwork: two in 2010, and six in 2011. In May and June 2010 I visited all community-based forest restoration projects in northwestern Andean Ecuador to learn about the region and to select sites for future study. I interviewed NGOs, community leaders, and local experts; observed people's agricultural practices, reforestation techniques, and forest use; collected preliminary data on trees in primary and secondary forests; and completed a 10-day course in tropical plant identification with a private instructor. I also interviewed field assistants, obtained research permits, and established connections and partnerships with local researchers, NGOs, and the National Herbarium of Ecuador (QNCE) in Quito.

An important goal of this visit was to make my project relevant to and accepted by local communities. After I informed NGO administrators and community leaders of the nature and objectives of my research, all were interested in having it proceed. I refined my research questions to fit the local context and established additional projects, including a tree growth monitoring program, based on community research needs.

In 2011, I returned to Intag for a total of six months to collect the bulk of the data presented in this thesis. I interviewed households, surveyed trees in primary, planted and unplanted secondary forests, conducted focus groups and oral histories, and observed and participated in daily life (methods described below). I lived in a village one hour by foot from two of my study communities, and interacted with people from them on a daily basis. I hired a total of 17 field assistants to help with interviews, plant identification, and forest surveys, including four botanists from the *Museo Ecuatoriano de Ciencias Naturales Herbario Nacional* *del Ecuador* (QNCE), three local and two Canadian field assistants to help with household interviews, and several local guides to assist with forest surveys in various locations. Ethics approval was obtained prior to beginning research.

Site selection

Out of all the forest restoration projects I visited in 2010, which included three different projects and seven different participating communities, I chose to study the Intag Valley because it had the most extensive community reforestation projects with multiple communities participating. The watershed reserves here are an ideal study site because they were restored using methods that can be easily adopted by local communities, and are commonly used in Latin America (CI; Maquipucuna Foundation; Rodrigues., 2007). The land use history in these reserves – clearing, followed by crops, and then planting exotic grasses for pasture – is common in Andean landscapes (Jokisch & Lair, 2002) and results thus apply to other Andean regions. Land was cleared between 30 and 40 years prior to reforestation, then burned, planted with crops for a few years, and subsequently planted with exotic pasture grasses (usually *Setaria sphacelata,* locally called *pasto cebolla*) and used as pasture (Table 1.2). People in Intag had also mapped the planted trees, a useful data source for ecological studies. I selected five planted reserves in which to survey trees on the basis of their accessibility, elevation (between 1900 – 2500 masl), and the presence of both planted (treatment) and unplanted (control) areas within the same reserve (the sixth reserve met none of these criteria). To study livelihoods and conservation activities in Intag, I selected four of the six communities with restoration projects based on the degree to which local people were involved in managing and executing the restoration. Each community had between 25 and 45 households.

Few primary forests remain between 1900 and 2250 masl in the Intag Valley. I selected primary forests that long-term residents identified as the most intact patches at these elevations: forest had not been cleared in living memory, nor had any significant timber extraction occurred. Botanists from QNCE confirmed that forests had structure and species composition typical of intact forests. Within each forest patch I selected sites to sample based primarily on their elevation.

Interviews

Key informants

In 2010 and 2011, I interviewed community leaders, project managers, and NGO administrative staff and employees to learn about the restoration projects – how the NGO engaged community members, restoration methods used, how species were selected, the types of training and environmental education provided, and what resources were provided. I asked them to define the project goals and if these had been met. I also interviewed local experts about agricultural practices, including seasonal crop rotations, methods to maintain soil fertility, access to markets, sharecropping arrangements, and typical wages for agricultural labour. On walks through the communities, local experts identified different types of crops, trees, land uses, and forests, and described deforestation patterns and drivers.

Focus groups

I conducted eight focus groups (two per community; men and women separately) using a semi-structured question set to learn about local tree species and use, forest use, and how community members participated in, managed, and perceived the restoration projects.

Household interviews

I interviewed 120 of the possible 134 households in participating communities. Before interviewing households, I pre-tested my questionnaire with five local residents outside the census cohort and restructured questions based on their feedback. I attempted to interview all households: in each community, one or two declined to participate, and others were unable to because they were absent, ill, or disabled. Interviews were conducted by a team of two people (an interviewer and a recorder), took between 30 and 90 minutes, and occurred in people's homes at a time of the interviewee's choosing. Questions covered household demographics, land holdings, agricultural practices, crop production, income, asset holdings, forest use, tree planting practices and preferences, and perceptions of the planting projects (Appendix A). Assets were classified as either land, productive capital, non-productive capital, or livestock (Takasaki *et al.* 2000*a*, Takasaki *et al.* 2000*b*). Interviews also covered past participation in local organizations, cooperatives and municipal governments. Households were asked if they had sold land to the communal reserves, and if so what they used the proceeds for. Questions on production and income focused on the past year, but households were also asked to identify how their land holdings, assets, yields, forest use and clearing practices, and methods for maintaining soil fertility had changed since the inception of the restoration projects (over the past eight years). Detailed data were also collected on on-farm tree planting practices: which tree species households planted, how many trees were planted, how they were planted, when, and why.

Oral histories

I interviewed 16 long-term residents (four from each community) about how communities had changed over the past 30 to 50 years. Interviews took between 60 and 120 minutes. Participants were asked to describe what the landscape looked like in their childhood, what foods they ate, how they interacted with and used trees and forests, the state of roads and market access, and how they obtained water. I then asked them to describe how each of these had changed. Interviews followed a general question set, but interviewees were encouraged to expand on these and other themes (Sommer & Quinlan, 2009).

Recruiting participants

To obtain permission to work in a community, I asked the elected community council president for permission, and also requested permission from each individual I interviewed, explaining project objectives beforehand. Because literacy rates were low I read participants a consent script introducing myself, the research goals, what their participation would entail, and that they were free to leave the study at any time. I obtained oral consent before proceeding.

Privacy and confidentiality

I identified people using a code in notes and spreadsheets, and stored data in a locked drawer in the field, and in a locked file drawer and on my password-protected computer. Individual names and information that would allow a person to be identified will not be released, and I will not share any of the raw data I have collected with other groups or organizations.

Land-use and land-cover change

I used remote sensing analysis based on satellite imagery to assess changes in forest cover, forest clearing rates, and agricultural land-use patterns pre- and post-restoration. I obtained three images from LANDSAT (taken between September and November in 1991, 2001, and 2010) and collected ground truth points in different types of fields and forests. I included questions on forest use pre- and post- restoration in the household surveys as mentioned above. I also visited the watershed reserves with previous land owners who I then asked to recount land use history starting with the inception of the restoration projects and working backwards (Coomes & Burt, 1997).

Forest surveys

Study design

I quantified tree species diversity and composition in five primary, planted, and naturally regenerated secondary forests (for a total of 15 sites). Because funds for tree planting were limited, each reserve contained both planted and unplanted areas and thus resembled a block design typically used to study forest restoration or logging treatments (Piotto *et al.*, 2004b; Weber *et al.*, 2008; Rondon *et al.*, 2009; Douterlungne *et al.*, 2010). Interviews confirmed that past land use in planted and unplanted areas was similar, and that no systematic selection bias in where trees were planted had occurred. All land in reserves was used as crops for one to four years, and then pasture for 30-35 years, prior to tree planting. All communities received comparable training and technical assistance, but selected different numbers of species to plant.

Transects

Transect surveys are commonly used in tropical forests to understand general patterns in forest species biodiversity (Gentry *et al.*, 1995; Stern, 1998), and are especially well suited to areas with steep slopes. I identified trees along transects following the sampling recommendations for biodiversity measurements in Chazdon *et al.* (1998). In each of the 15 sites, I set up four 50 × 5-m transects along slope contours (total 60 transects, 0.1 ha/site). I divided each transect into five 10 × 5-m plots, and counted, identified, and measured trees (>2.5 cm diameter-at-breast-height (DBH)), woody saplings (1-2.5 cm DBH) and seedlings (>0.5 m height, < 1cm DBH) in each. Sampling teams consisted of myself, one or two botanists from QNCE, and a local guide. Botanists identified tree species in the field and I took replicate voucher samples to botanists at QCNE to identify unknown species. A local guide provided us with common names. Plants were counted as separate individuals if the stem of the plant was not connected at or just below the soil surface (Chazdon *et al.*, 1998). At two random locations on each transect, I recorded slope, aspect, canopy density, and percent ground cover in two 1-m^2 plots. This sampling effort provided sufficient data to compare biodiversity between forest types and sites (Chazdon *et al.*, 1998; Magurran, 2004; Colwell, 2009).

Soils

I took 10 soil samples from the top 10 cm at two to three randomly located places on each transect. I mixed these to create a composite sample which was stored in a conventional refrigerator and delivered to the soil laboratory within one to five days (*Estacion Experimental "Santa Catalina", Instituto Nacional Autonomo de Investigaciones Agropecuarias,* Cutuglagua, Mejía, Pichincha). Soils were analyzed for macro and micronutrients (nitrogen (NH_4), phosphorus, potassium, calcium, magnesium), cation ratios (Ca/Mg, Mg/K, and Ca+Mg/K), organic matter content, and texture (Appendix B). Four bulk density samples were taken using a cylindrical sampler 10 cm in diameter from the top 10 cm of soil at each site (one per transect).

Summary

In the Intag region, reforestation in community watersheds was implemented with community participation and support. This research aims to understand why this was, and the effects of these projects on people and on the environment, ultimately providing information that can be used to guide and improve future community reforestation projects. In particular, understanding which households participate and what drives them to do so can make projects more accessible and relevant to local people (Chapter 5). Examining if and how participating in reforestation changes on-farm tree planting, regional forest conservation, and forest cover will illuminate potential indirect benefits (or lack thereof) of reforestation for conservation, and for the development of sustainable agriculture (Chapters 2 and 5). These analyses, combined with tree diversity surveys in primary and planted cloud forest (Chapters 3 and 4) will provide an overall picture of the direct and indirect environmental outcomes of such projects. In addition, evaluating how effectively small scale, relatively simple restoration techniques overcome barriers to forest establishment and succession will help us understand both the impact of restoration projects on landscape biodiversity and the potential of community-based forest restoration to conserve cloud forest (Chapters 3 and 4).

Finally, this work integrates research on Andean livelihoods with the literature on cloud forest ecology, two fields that are generally studied in isolation of one another. Communication between these fields is essential to inform and improve the human and environmental aspects of future conservation and development initiatives and research in the Andes.

Preface to Chapter 2

To conserve biodiversity, restoration should complement, not replace, primary forest conservation. But deforestation and reforestation are not always governed by the same processes, and can occur simultaneously.

Chapter 2 sets the stage for this thesis by comparing changes in deforestation rates, reforestation rates, and forest cover before and after restoration and examining them in the broader context of land-use change in northeast Intag. Results show that a local forest transition occurred between 2001 and 2010 amid both high deforestation and reforestation rates. Forests were redistributed from highlands to lowlands – a pattern that contradicts current forest transition theories. This transition appears to be driven by local demand for forest ecosystem services, a 'novel' path for forest transitions to occur.

Results raise questions about the social and ecological drivers and outcomes of changes in forest cover and distribution, which I address in the remaining three manuscripts of this thesis. Specifically, what is lost when primary forests are cleared (Chapter 3) and replaced by secondary forests (Chapter 4)? What drove people to reforest, and how do changes in forest cover stem from and affect local livelihoods (Chapter 5)? As a complete work, this dissertation is the first study to integrate in-depth socioeconomic and ecological data to examine the drivers and outcomes of Andean forest transitions.

Chapter 2

Local forest transitions in the Ecuadorian Andes: Forest recovery amidst deforestation, 2001-2010

Sarah Jane Wilson, Oliver T. Coomes & Camille Ouellet-Dallaire

The human impact on the environment is not simply a process of increasing change or degradation…It is instead interrupted by periods of reversal and ecological rehabilitation as cultures collapse, populations decline, wars occur, and habitats are abandoned. Impacts may be constructive, benign, or degenerative…but change is continual...
– William Denevan, *The Pristine Myth*, 1992

The advance of forests rarely takes place steadily on a broad front, but follows the principles of modern warfare; infiltrating along the valleys, surrounding successive areas of grassland, cutting them off from their supply of fire and then rapidly mopping them up.
– H. A. Osmaston, 1959

The only thing that is constant is change.
– Heraclitus, 500 B.C.

Abstract

Andean forests decreased in area over the past decade as people cleared them, mainly for farming. But amid this deforestation, forests returned to some regions, producing local 'forest transitions' – net increases in forest cover. The mechanisms that drive these local transitions – often in part the actions of residents – are still little studied. Chapter 2 investigates forest cover dynamics in Intag, an Andean region where people are actively reforesting by planting trees. I ask: do communities reforest and deforest simultaneously, and does community-based restoration increase regional forest cover? Results from remote sensing analysis using satellite imagery from 1991, 2001, and 2010 show that prior to reforestation projects (before 2001), deforestation rates in Intag were high (over 3% per year). During the subsequent period (2001-2010), forest recovery surpassed deforestation, resulting in a net forest cover increase of 3% – a local forest transition. However, although deforestation rates slowed precipitously (to less than 2%) when people began to reforest, people continued to clear forests in the highlands even as they grew back around communities. Household interviews and oral histories suggest that this spatial shift in forest cover is explained in part by people's reasons for planting trees – to restore water and other key ecosystem services to their communities. The results thus point to a new 'path' by which forest transitions occur – the ecosystem service scarcity path – in which local demand for forest ecosystem services drive forest recovery. Because biodiverse primary forests were cleared as less diverse secondary forests returned, the potential of transitions to conserve forest biodiversity is perhaps less optimistic than it might at first seem. However, given that communities throughout the Andes are experiencing environmental degradation and soil fertility loss, results are broadly applicable, and fostering the conditions that promote such transitions is of great interest to policy makers, managers, and local communities alike.

Introduction

Community-based forest-restoration projects are often implemented with the dual goals of increasing forest ecosystem services and conserving biodiversity (Silver *et al.*, 2004; Lamb *et al.*, 2005; Chazdon, 2008; Garen *et al.*, 2009). In pursuing these goals, it is generally assumed that restoring forests will increase net forest cover and biomass (Lamb *et al.*, 2005; Chazdon, 2008; Angelsen *et al.*, 2009; Pattanayak *et al.*, 2010). However, because people may continue to deforest while they plant trees elsewhere, even 'successful' tree-planting projects with high participation rates are not guaranteed to produce net gains in forest cover (Sloan, 2008; Knoke *et al.*, 2009; Groom & Palmer, 2012).

Despite high tropical deforestation rates, net increases in forest cover – known as 'forest transitions' – do commonly occur across the tropics (Perz & Skole, 2003; Nagendra, 2009; Aide *et al.*, 2013) (Fig. 2.1). The conditions that foster them are of great interest to scientists, policy makers, and environmental agencies alike. Well studied at national and continent scales, they tend to occur when: 1) economic development makes farming marginal land unattractive or unprofitable, and, as people intensify farming in accessible or fertile areas, forests recover on abandoned land; or 2) the scarcity of forest resources drives people to plant trees[4] (Mather & Needle, 1998; Rudel *et al.*, 2005, pg. 24) (Fig. 2.1). However, because the forest transition is an "empirical regularity, not a deterministic prediction" (Angelsen & Rudel, 2013, pg. 91), transitions may or may not occur even in similar environments (Mather, 1992; Rudel *et al.*, 2005; Redo *et al.*, 2012; Angelsen & Rudel, 2013).

Although Latin America as a whole experienced a net loss of forest in the past decade (FAO, 2011), in a continent-wide study Aide et al. (2013) show that forest cover is increasing in

[4] Rudel et al. (2005) define these pathways as the "economic development" path and the "forest scarcity" path, respectively (Rudel *et al.*, 2005, p. 24).

‘high and dry’ areas, including parts of the megabiodiverse Tropical Andes (Myers *et al.*, 2000; Brooks *et al.*, 2006). The authors attribute montane forest recovery to agricultural intensification in the flat lowlands and subsequent abandonment of marginal montane farmland. But the scale at which one studies transitions is important: forest has increased in some Andean regions, but not others, and the Tropical Andes still lose about 1% of their forests per year (Farley, 2010; Portillo-Quintero *et al.*, 2012*a*; Sánchez-Cuervo *et al.*, 2012; Aide *et al.*, 2013) (Fig. 2.2). Aide et al.’s (2013) agricultural intensification hypothesis does not account for these regional differences, nor for the fact that many parts of the Andes are still actively farmed and densely populated. The Andes are thus an important and compelling location to study the drivers and outcomes of local and regional forest-cover change.

Land-use choices at the community or household level often drive local and regional deforestation and reforestation patterns – thus, the biophysical and socioeconomic factors that play into smallholder decision-making can drive forest cover change (Jokisch & Lair, 2002; Rudel *et al.*, 2002; Schelhas & Sánchez-Azofeifa, 2006; Nelson & Chomitz, 2007; Aubad *et al.*, 2008; Duchelle, 2009; Hoch *et al.*, 2012; Redo *et al.*, 2012). Even within the same region under similar biophysical conditions, differences in culture, access to markets, and forest access and extent can produce very different patterns in forest clearing and recovery (Kappelle *et al.*, 2000; Rudel *et al.*, 2002; Nelson & Chomitz, 2007; Aubad *et al.*, 2008; Duchelle, 2009; Farley, 2010; Hoch *et al.*, 2012; Abram *et al.*, 2014). Examining both the exogenous drivers of transitions and the endogenous changes that occur within communities is important to understanding why local transitions occur, where they occur, and what the social and ecological outcomes will be (Perz, 2007; Farley, 2010).

Although Andean forest recovery could be a 'good news' story for cloud forests, forest transitions produce different ecological and social outcomes depending on the degree to which primary forests are depleted and the types of secondary forests that grow back (Farley, 2007, 2010; Sánchez-Cuervo *et al.*, 2012)(Chapter 4). Primary forests contain different species and often provide different levels or types of ecosystem services from secondary forests (Kanowski *et al.*, 2005; Bhagwat, 2008; Dent & Wright, 2009) – the percentage and location of primary forest remaining at the transition inflection point matters for both people and biodiversity conservation. Secondary forests (e.g., plantations, naturally regenerating forests, agroforests) also provide different ecosystem services and support different levels of biodiversity from one another (Bhagwat, 2008; Phalan *et al.*, 2011; Wilson, 2013). The types of secondary forest that return depend in part on what drives the transition: plantations are perhaps likely to result from "forest scarcity" transitions, while naturally regenerating forests are predicted in "economic development" transitions (Rudel *et al.*, 2005, pg. 24; Rudel, 2010). Unpacking what drives Andean transitions at regional and local scales, and studying the forests that these drivers ultimately produce, is thus key to understanding their effects on local people and biodiversity.

Forest transitions occur when reforestation rates surpass deforestation rates. The local drivers and environmental consequences of tropical deforestation have been well studied (Geist & Lambin, 2002; Laurance *et al.*, 2002; Meyfroidt *et al.*, 2013), but considerably less work has focused on what drives reforestation (Chazdon, 2008). In this chapter, I present the results of a study that examines forest-cover change dynamics in the Intag region in northwest Andean Ecuador, where people are actively restoring forests. Six communities began planting trees in 2002 to restore cloud forest in communal watershed reserves and enhance farming productivity on private farms. I assess if a local forest transition has occurred by quantifying changes in forest cover from 1991 to 2001 (pre-reforestation), and from 2001 to 2010 (during reforestation). I then examine the social and ecological consequences of these changes in the context of the region. I address the following questions:

1) *How have land use, reforestation, and deforestation rates changed since community restoration initiatives began?*

2) *What are the net effects of these changes on forest cover and spatial distribution?*

3) *What social and ecological conditions drove these changes?*

I conclude by discussing the social and ecological outcomes of these results in the context of the region. In particular, I examine how demand for ecosystem services can drive local reforestation, and the impacts of this particular type of transition on forest biodiversity.

Methods

Study site

This study focuses on the northeast portion of the Intag Valley, Imbabura Province, Ecuador (0.35° N, 78.5° W) (Fig 2.1). The mountains here are rugged and steep, and both agriculture and forests are found on slopes greater than 35°. The region receives 1500 to 3300 mm of rainfall per year, has an average temperature of 17 to 20°C, and a pronounced dry season from May or June to October or November (Freiberg and Freiberg 2000, Rainforest Concern 2009). Elevations within the study area range from 700 to 3700 masl.

Clearing patterns here are typical of many places in Andean Ecuador (Wunder, 1996; Jokisch & Lair 2002; Sarmiento, 2002). Originally covered in dense cloud forest, the region began to be cleared rapidly beginning in the 1970s. People cleared forest mainly for agriculture and cattle ranching. Until recently, a typical land-use sequence was to clear forest (sometimes extracting timber and/or firewood for local use), burn the cleared area, plant crops for up to four years (rotating corn and beans), and then either allow land to fallow or, more commonly, convert land to pasture. Today, education programs on the environmental problems associated with burning have made this practice less common, although it still occurs occasionally (C. Zorrilla, pers. comm., 2011). For a full description of the livelihoods and settlement patterns of people in the region, see Chapter 5.

There are several forest conservation initiatives underway in Intag. In addition to the watershed reserves with community-based tree-planting projects (described in detail in Chapters 4 and 5), the same local non-governmental environmental organization (NGO), *Defensa y Conservacion Ecologica de Intag* (DECOIN), established 35 more community-based watershed reserves (without a tree-planting component), and 41 other community-owned and -managed reserves in the Intag Valley. In response to severe declines in water quality and seasonal droughts that followed deforestation, the watershed reserves are intended to protect and restore water supply by reforesting riparian areas above local communities. Other community reserves focus on conserving primary and older secondary forests, and on protecting land in strategic locations throughout the region so that it cannot be purchased and developed[5] (DECOIN, 2010).

[5] The Intag region has been of interest to international mining companies for decades (Bebbington *et al.*, 2008; Kocian *et al.*, 2011; Buchanan, 2013). In the early 2000s, communities protested, dismantled a mining exploration camp, and ultimately prevented illegal mining exploration. Community-owned reserves create small areas of land that cannot be sold for development without communal consent, providing a safeguard against mining.

Landsat imagery and analysis

To quantify changes in forest cover, clearing and regeneration rates, and other land-use changes in the region, I obtained three Landsat images from the United States Geological Survey (USGS). Landsat imagery is commonly used in these regions to classify heterogeneous landscapes (Kintz *et al.*, 2006; Schmook *et al.*, 2011). I selected three multispectral images at approximately 10-year intervals, beginning in the early 1990s to 2010. Images were selected based on their percentage of cloud cover and the period in which they were taken (all three images were taken at the end of the dry season, between September and November in their respective years). The September 1991 image was taken by Landsat 5 Thematic Mapper, and the November 2001 and September 2010 images were taken by Landsat 7 Enhanced Thematic Mapper Plus. Landsat 5 has six optical bands and one thermal band, and Landsat 7 has six optical bands, one thermal band, and one panchromatic band (Headley, 2010). The wavelength spectrum covered a range from 0.45 micrometers to 2.35 micrometers, with a spatial resolution of 30 meters (Headley, 2010). A technical problem with the Scan-line corrector (SLC) in Landsat 7 in 2003 resulted in stripes on the image with no data, although this problem does not affect the radiometric quality of the remaining image (Chander *et al.*, 2009).

All Landsat images used were terrain corrected using the digital elevation model of the Global Land Survey 2000 (Headley, 2010). Images were downloaded from the USGS GLOVIS website (GLOVIS, 2012). To further pre-process images, digital values were converted to radiance values and then to reflectance using ENVI's Landsat Calibration tool and the parameters supplied by each image's metadata file (date of acquisition, sensor, and sensor angle) (Jokisch & Lair, 2002; Lillesand *et al.*, 2004; ENVI, 2009). Images were atmospherically corrected using the ENVI FLAASH module with a tropical atmospheric model and a rural aerosol model (ENVI, 2009).

I gathered ground-truth points using a hand-held GPS device (Garmin GPS Map 60CSx) in the field in 2010 and 2011, noting information on elevation, waypoint accuracy, and land-cover type. I defined the extent of the study area as the maximum distance people in communities with reforestation projects walked to farm or clear forests (about 2.5 hours, representing a five-kilometer buffer around each community), and used the minimum bounding rectangle around these buffer zones. Images were resized to the extent of this study area (20,600 ha). The Scan Line Corrector (SLC-off) was used to mask the lines with missing data in the 2010 image. The area of unclassified land is presented in the results. Sampling sites or regions of interest (ROIs) were selected based on photographs and GPS points collected in the field across the resized images. A total of 357 ROIs were selected for the 1991 image, 1749 ROIs for the 2001 image and 1070 ROI's for the 2010 image. ROIs were divided in half – one sub-sample was used to train the classification algorithm and the other to assess the classification accuracy.

Six land-use classes were ultimately created based on ground truth points and images: crops, forest, pasture, bare soil, indeterminate vegetation (hereafter 'vegetation') and water. An additional class, forest shadow (i.e., forests on exceptionally steep slopes that were in shadow at the time the image was taken), was initially identified but subsequently merged with the 'forest' category after ground truth points confirmed that these areas were forests. This class represented a small (less than 2%) proportion of the total forest cover in each period. Based on conversations with local residents and on Google Earth imagery, the vegetation class appears to be areas of recently cleared or heavily degraded forest that were not yet cultivated. Jokisch and Lair (2002) identified a similar class in the Ecuadorian Andes. I was unable to distinguish between secondary and primary forests, a distinction that others have also found difficult in the Ecuadorian Andes (i.e., (Keating, 1997; ENVI, 2009). Majority filter analysis was used to refine the classification and reduce the 'salt and pepper' effect (Lillesand *et al.*, 2004). In a classified image, this analysis changes isolated pixels to match the surrounding land use by evaluating the center of a kernel of n x n pixels in the classified image and altering the current class label to the class corresponding to the majority of surrounding pixels (Lillesand *et al.*, 2004). The kernel size used for this analysis was 3 x 3 pixels, and majority filter analysis was performed in ENVI. A confusion matrix – a cross-tabulation of the classified pixels against the reference pixels – was then produced to assess classification accuracy (Lillesand *et al.*, 2004) (Table 2.1). The overall accuracy was 99.8% for the 1991 image (kappa coefficient (KC) of 0.99), 99.4% for the 2001 image (KC 0.98), and 99.6% for the 2010 image (KC 0.99).

Results

Current land use and land cover in Intag

In 2010, land in Intag was primarily under forest (33%), pasture (38%), and crops (16%). Visual inspection of the land-cover map for 2010 shows that most forests are in the upper elevations of the region, far from the main market town (Apuela) and roads. Pastures tend to border the forest edge, and crops are found at the lowest elevations, close to villages and roads (Fig 2.1). Visual analysis confirms data from interviews and household surveys that crops grow best in the lower, warmer regions but pastures are well suited to the upper, colder areas.

Land-cover changes 1991-2010

Land-cover change was widespread and dynamic in Intag from 1991 to 2010 (Table 2.3).

Forest cover declined from 42% to 33% of the study area, pastures increased from 29% to 39%, and area under crops increased from 5% to 16% (Table 2.3). The 'vegetation' category, which appears to be recently cleared or heavily degraded areas, decreased from 24% to 8%, and tended to be replaced by pastures (60% of the areas under 'vegetation' in 1991 were pasture in 2010) (Table 2.2). During this time, 33% of the land area (6292 ha) changed from one land use to another in one of the two periods, and 34% (6469 ha) changed during both periods (Fig. 4).

Land cover changed slightly less from 2001-2010 than from 1991-2001, but remained near 50% during both. Between 1991 and 2001, 54% (11,208 ha) of the land in Intag was converted from one land use to another. Between 2001 and 2010, 47% of the land in Intag (8904 ha) was converted from one use to another (Fig. 2.4).

Deforestation and reforestation 1991-2010

Net forest cover declined by 22% (2030 ha) between 1991 and 2010. During this time, 4800 ha of forest was cleared, and 2770 ha regenerated in forest. During each time period, forested land was mostly converted to cattle pasture and, to a lesser extent, cropland (Table 2.3). Forest clearing rates were nearly twice as high between 1991 and 2001 (37% over 10 years) as between 2001 and 2010 (23% over nine years) (Table 4) (Fig. 2.5). While the overall trend from 1991-2010 was of forest cover loss, the ratio of deforestation to reforestation differed substantially between the first and second half of this period, which corresponded to pre-reforestation and post-reforestation periods.

<u>Forest cover 1991-2001</u>

Between 1991 and 2001, most (79%) of the 3250 ha of forests cleared were replaced by pasture. During this time, forests also grew back on an area of 1020 ha (an increase in area of 12%). Forests grew back mainly on clear-cut areas (622 ha, 61%) and pasture (300ha, 30%) (Table 2.2). Net forest loss between 1991 and 2001 was 2230 ha (25%) (Table 2.4).

Forest cover 2001-2010

Although 1550 ha (23%) of forests were cleared from 2001 to 2010, forests also grew back on 1750 ha, resulting in a net increase in forest cover of 200 ha (a 3% increase). This is a marked change from the 25% loss in the previous 10 years (Table 2.3). From 2001-2010, most of the cleared forests were replaced by pasture (986 ha, 42%) and clear-cuts (429, 18%), and forests grew back almost entirely on pasture (1479 ha, 85% of forest recovery) (Table 2.4).

Spatial location of forests

Land-cover change maps show that forests were cleared from different areas during the two time periods (Fig. 2.4). From 1991 to 2001, deforestation was widespread along the edges of forests throughout the study area. Areas of forest in close proximity to all of the communities in this study were cleared. In contrast, from 2001 to 2010, deforestation was concentrated in the northwest corner of the region. This area was close to two communities that did not participate in tree-planting initiatives, but which had community-based forest conservation reserves (DECOIN, 2010) (Fig. 2.2). Clearing along forest edges elsewhere appeared to decline (Fig. 2.5).

Discussion

Net forest cover in the Intag Valley stabilized between 2001 and 2010. Although net forest loss was extremely high in the previous decade (from 1991 to 2001), with clearing rates nearly double that of the national average for Ecuador[6] (FAO 2005), from 2001 to 2010 reforestation rates slightly surpassed deforestation rates, resulting in a small net increase in forest cover. Catching this region at a forest transition 'inflection point' – the beginning of an increase in forest cover – provides an excellent opportunity to study the social and ecological conditions and dynamics that drive small-scale forest transitions in the Andes.

Forest transitions amid dynamic local land use

From 1991 to 2001, land-use patterns in Intag were typical of those found in Andean regions (Wunder, 1996; Jokisch & Lair, 2002; Sarmiento & Frolich, 2002; Peters *et al.*, 2013). Pastures and cropland expanded almost entirely at the expense of intact and degraded forests. Land use was dynamic: over this period, 53% of the land area changed from one use to another, and the region lost 37% of its primary forest.

Although land use was also dynamic between 2001 and 2010 – almost half of the land in the region changed from one use to another – net forest cover increased. The region lost 23% of the forests present in the previous period, but reforestation rates increased dramatically from 12% in 1991-2001 to 26% in 2001-2010. Pastures continued to expand, but they replaced not only forests but cropland, too, so that the total area of cropland declined. People in Intag consume and sell crops, but cows are raised almost exclusively for sale (Chapter 5); thus, this pattern could be indicative of increased reliance on a cash economy (Jokisch & Lair, 2002; Rudel *et al.*, 2002). But farmers here also often convert cropland to pasture once soils are too nutrient-poor to produce crops, and so this transition likely, at least in part, reflects declining soil fertility. The latter hypothesis is supported by the fact that secondary forest recovered on older pastures

[6] Approximately 2.5 % per year in Intag compared to the average for Ecuador of 1.2% per year from 1990 to 2000 (FAO, 2005).

even as people cleared primary forests for new ones.

That forest cover increased while land use continued to change at high rates challenges the hypothesis that forest transitions occur when land use stabilizes, generally later in a region's development (Mather & Needle, 1998; Perz, 2007). This hypothesis assumes that forest transitions begin with a dramatic decrease in deforestation rates, which is a precursor to reforestation (Grainger, 1995). But simultaneous expansion of pasture and secondary forest has been observed in other Andean regions (Jokisch & Lair, 2002) and during local forest transitions elsewhere in Ecuador (Rudel *et al.*, 2002). More research has examined what drives local deforestation than reforestation (Geist & Lambin, 2002; Laurance *et al.*, 2002; Chazdon, 2008; Meyfroidt *et al.*, 2013). My study suggests that increased reforestation, as opposed to diminished deforestation, can also drive local forest transitions. Studying what drives reforestation is therefore key to predicting and fostering local forest transitions. Results also show that examining net forest cover alone can overlook important information about the drivers and social and environmental outcomes of local transitions. Deforestation rates, reforestation rates, and the spatial relocation of forests that result from these combined processes all deserve a closer look.

Spatial transitions in forest cover

In this dynamic landscape, simultaneous deforestation and reforestation redistributed forests in the Intag Valley. From 2001 to 2010, clearing occurred along the edges of primary forests, but forests regrew along roads and near communities (Fig. 2.2, Fig. 2.4). This pattern is a dramatic shift from the 1991-2001 period, when forest recovered mainly along the margins of primary forests far from villages (Fig. 2.2). These spatial patterns agree with observations from the field: in the 2000s, although some people reported clearing highland forests for pasture, they also began planting trees around their communities and on farms and allowing forests to regenerate unassisted in watersheds (Chapter 5). Over roughly the same period (2000-2008), the province of Imbabura as a whole lost forests at a rate of 1.2% per year, and the majority of clearing occurred in places with high forest cover (Mika Peck, pers. comm. 2014). The forest recovery observed in this study is thus a local, rather than regional, trend.

The 'ecosystem-service-scarcity' path to forest recovery

This spatial shift in forest cover runs contrary to what we would expect to observe from the agricultural intensification path that is proposed as the driver for forest transitions in the Andes (Aide *et al.,* 2013). According to this hypothesis, fertile lowland regions would remain cleared for intensive agriculture while forests regenerate on abandoned marginal land in the highlands (Rudel *et al.*, 2005; Sánchez-Cuervo *et al.*, 2012; Aide *et al.*, 2013; Pellissier *et al.*, 2013). But in Intag, people planted trees in agricultural land even as they continued to deforest distant 'marginal' highland areas. Reforestation patterns were driven by a conscious effort by smallholder farmers at household and community levels to integrate trees and forests into their farming systems and landscapes to combat steady declines in soil fertility, water quality, and dry season stream flow following deforestation in catchment areas (Chapter 5). Changes in forest cover and forest location thus reflect a fundamental shift in the way people use trees and forests in this region, and were driven, in part, by demand for forest ecosystem services (Chapter 5).

Demand for wood in heavily deforested regions has driven forest transitions at both the local and national levels by spurring governments, agencies or farmers to plant trees (Foster & Rosenzweig, 2003; Rudel *et al.*, 2005; Farley, 2007; de Jong, 2010). In Intag, farmers also promoted forest recovery in response to forest scarcity, but for forest ecosystem services rather than wood (Chapter 5). The forests they planted were not plantations geared towards timber production, as would be expected in a wood-driven, forest scarcity transition, but included both naturally regenerating and restored areas aimed at producing scarce, often regulating ecosystem services. Operating at small but no less significant scale, local ecosystem service transitions are likely to produce different ecological and social outcomes from either wood scarcity or agricultural intensification forest transition paths.

Environmental and social outcomes of the ecosystem services scarcity path

Spatial redistributions in forest cover can change the ecosystem services that people derive from forests (Mitchell *et al.*, 2013). By reforesting in and around communities, smallholders in Intag increased their access to trees and forests and the local ecosystem services they provide, including regulating services such as water purification, soil fertility, erosion control, and provisioning services such as fuel, fodder, and food (Shvidenko *et al.*, 2005, Grêt-Regamey *et al.,* 2012) (Chapter 5). However, because people continued to clear primary forests far from villages, forest ecosystem services that are often put forward as global priorities – carbon sequestration and biodiversity conservation – have likely declined over the same period. These services are generally provided to greater degrees by retaining primary or old secondary forests than by restoring or conserving young, naturally regenerating secondary forests (Barlow *et al.*, 2007; Coomes *et al.*, 2008; Dent & Wright, 2009; Gibson *et al.*, 2011; Martin *et al.*, 2013).

In particular, the continued loss of primary cloud forests can have a large impact on regional biodiversity. Net gains in forest cover do not always conserve forest biodiversity, particularly when planted forests expand at the expense of primary tracts (Fitzherbert *et al.*, 2008; Koh & Wilcove, 2008; Sloan, 2008; Bremer & Farley, 2010; Zhai *et al.*, 2014). Young secondary forests and agroforestry systems often contain lower species richness and different species from primary forests (Barlow *et al.*, 2007; Bhagwat, 2008; Chazdon *et al.*, 2008; Liebsch *et al.*, 2008; Dent & Joseph Wright, 2009; Letcher & Chazdon, 2009; Gibson *et al.*, 2011; Bonner *et al.*, 2013; Martin *et al.*, 2013). The young secondary forests in Intag are no exception – the forests that have grown back within the 2001 to 2010 period contain only a fraction of those species found in primary forests, and are less than half as species rich (Chapter 4). Moreover, because primary cloud forests have distinct communities of trees from one another (Chapter 3), but planted and naturally regenerating secondary forests are often more homogeneous at the landscape level (McKinney & Lockwood, 1999; Holl, 2002; Lugo & Helmer, 2004; Rhemtulla *et al.*, 2007), clearing primary forest is almost certain to reduce landscape biodiversity. Spatial shifts in forest cover and the replacement (in time) of primary forest by secondary can have significant impacts on both the ecosystem services people derive from forests and forest biodiversity in Andean regions and elsewhere.

This study suggests that the ecological and social outcomes of regional and local forest transitions are different depending on how forests are redistributed, the extent to which primary forests are depleted prior to a transition occurring, and the types of secondary forests that return. Local people can play an active role in engineering these changes, producing results that meet different social and environmental goals (Perz, 2007; Farley, 2010; Redo *et al.*, 2012) (Chapter 4, Chapter 5). Thus, to fully understand the drivers and outcomes of local and regional forest transitions, it is important to: 1) examine both deforestation and reforestation rates along with overall net forest cover; 2) quantify spatial shifts in forest cover; 3) distinguish between the types of forests that are cleared and those that grow back; and, 4) examine the smallholder decision-making processes that drive transitions within a given cultural, ecological, and biophysical context (Rudel *et al.*, 2002; Perz, 2007; Farley, 2010; Redo *et al.*, 2012).

Next steps: Key drivers and outcomes of local ecosystem-service-scarcity transitions

The results presented here represent the initial steps for research on the ecological and social drivers and outcomes of forest transitions at the regional and farm level. Future analyses will quantify the qualitative trends observed in Intag, including the spatial redistribution of forest cover as it relates to biophysical and human-created landscape features. The trends I observed (primary forest clearing even as secondary forest increases, and the spatial redistribution of forest cover from highlands to farmlands) raise questions about both the conditions that drive these changes and their effects on forest ecology and people's livelihoods.

First, how much of a 'good news' story are local forest transitions for conserving biodiversity? Although I observed a net increase in forest cover, between 2001 and 2010 the region still lost 2328 ha of forest, most of it likely primary or old secondary. To understand what is lost when primary forests are cleared, in Chapter 3 I characterize the tree communities in, and quantify the landscape-level biodiversity of, cloud forests in the Intag Valley. Results showing high spatial variability and species turnover in tree communities indicate that cloud forest patches are both very biodiverse and unique, and that conserving remnant patches is essential to maintain landscape-level biodiversity.

Second, given high past clearing rates and high local demand for forest ecosystem services, what potential does this community-based tree planting observed in Intag hold to restore cloud forest in the region? Chapter 4 compares the tree species composition of secondary planted forest, unplanted (naturally regenerating) forest, and primary forests in the region. Although restoration increases forest cover, accelerates forest recovery, and increases biodiversity over letting forests regenerate naturally, the species composition of restored forests remains distinct from primary forests. Restoring forests provides both environmental and social benefits, but is not a substitute for conserving them.

Third, in an area with such historically high deforestation rates, what triggered reforestation in the first place? What biophysical and socioeconomic conditions drive reforestation and deforestation during a forest transition inflection point? In Chapter 5, I examine household-level participation in community-based tree-planting projects. Using an asset-based livelihoods approach, I identify which households participated in planting trees on communal and private land, why they participated, and their perceptions (both current and historical) on the links between forests and farming. Results show that a severe decline in environmental conditions accompanied by a desire to remain on the land created ‘crisis’ conditions in which people were open to hearing about the benefits of forests for farming. With environmental education and training programs provided by a local NGO, people, especially farmers, began to restore forests and experiment with trees in farming systems. Results can be used to identify households and regions that will participate in and benefit from restoring forest.

The following chapters in this thesis examine the drivers that produced a local ‘forest transition’ in the Intag region, and the ecological and social impacts of the shift from primary to secondary forest cover and the spatial redistribution of forests in the region. These analyses will help practitioners, policy makers and agencies: 1) identify areas in which forest transitions are likely to occur; 2) provide resources and support to local communities to aid in these transitions; 3) provide appropriate incentives to maximize biodiversity conservation during reforestation projects; and, 4) ultimately maximize the many synergies and minimize the trade-offs between biodiversity conservation and rural livelihoods in heavily deforested Andean regions.

Table 2.1: Confusion matrices for classifications for the year A) 1991, B) 2001, and C) 2010. Rows are the number of test pixels used to generate a given class, and columns are the number of test pixels that were observed to be the same use, and then were used to test the classification. Numbers in the diagonal thus represent a correct classification.

	Crop test	***Forest test***	***Pasture test***	***Soil test***	***Veg. test***[1]	***Water test***	***Total***
Crop	30	0	0	0	0	0	30
Forest	0	354	0	0	0	0	354
Pasture	0	1	35	0	0	0	36
Soil	0	0	0	104	0	0	104
Veg. [1]	0	0	0	0	122	0	122
Water	0	0	0	0	0	21	21
Total	30	355	35	104	122	21	667

Table 1A

	Crop test	***Forest test***	***Pasture test***	***Soil test***	***Veg. test***[1]	***Water test***	***Total***
Crop	77	0	0	3	0	0	80
Forest	0	503	0	0	0	0	503
Pasture	0	0	37	0	0	0	37
Soil	0	1	0	102	0	0	103
Veg. [1]	0	0	0	0	74	0	74
Water	0	1	0	0	0	21	22
Total	77	505	37	105	74	21	819

Table 1B

	Crop test	***Forest test***	***Pasture test***	***Soil test***	***Veg. test***[1]	***Water test***	***Total***
Crop	34	0	0	2	0	0	36
Forest	0	258	0	0	0	0	258
Pasture	0	0	44	0	0	0	44
Soil	0	0	0	91	0	0	91
Veg. [1]	0	0	0	0	97	0	97
Water	0	0	0	0	0	0	0
Total	34	258	44	93	97	0	526

Table 1C

[1]Veg. corresponds to the 'vegetation' category: heavily degraded or recently cleared forests.

Table 2.2: Land use in the Intag region in 1991, 2001, and 2010 based on Landsat classification. Numbers are the total area in hectares under each type of land cover followed by the percentage. The remaining pixels are water.

Land Use	***1991***	***2001***	***2010***
Crop	1099 (5.3)	3827 (18.5)	3213 (16.2)
Forest	8665 (41.9)	6714 (32.4)	6493 (32.8)
Pasture	6025 (29)	7585 (36.6)	7669 (38.7)
Soil	433 (2.10)	1970 (9.5)	673 (3.4)
Vegetation	4425 (21.4)	578 (2.8)	1658 (8.4)

Table 2.3: Changes in land use from A) 1991-2001 and B) 2001-2010. The first number is the area in hectares, and the second number is the percent of pixels in the earlier year that switched to the subsequent land use, unless noted otherwise.

	Crop 1991	***Forest 1991***	***Pasture 1991***	***Soil 1991***	***Vegetation 1991***	***Class Total 1991***
Crop 2001	491 (44.7)	297 (3.6)	2056 (34)	88 (0.32)	893 (20.2)	3827
Forest 2001	25 (2.3)	5697 (50)	300 (5.0)	61 (14.2)	622 (14.1)	6713
Pasture 2001	244 (22)	2185 (39.6)	2849 (47)	179 (41.3)	2115 (47.8)	7585
Soil 2001	321 (29)	364 (5.6)	752 (12.5)	82 (19.0)	439 (9.9)	1970
Vegetation 2001	16 (1.5)	119 (1.4)	66 (1.1)	22 (5.1)	353 (8.0)	578
Class total (ha)	1099	8665	6025	434	4426	
Class changes 2001-2010 (ha, percent)	607 (55.3)	3250 (37.5)	3176 (52.7)	351 (81.0)	4071 (92.0)	
Image difference (ha)	2728	-1951	1559	1536	-3848	

	Crop 2001	***Forest 2001***	***Pasture 2001***	***Soil 2001***	***Vegetation 2001***	***Class Total 2001***
Unclassified	164 (4.3)	294 (5.1)	307 (4.05)	86 (4.4)	39 (6.8)	892
Crop 2010	1288 (33.6)	133 (5.3)	1023 (13.5)	743 (37.7)	24 (4.2)	3213
Forest 2010	107.6 (42.8)	4745 (56.6)	1479 (19.5)	128 (6.5)	32 (5.5)	6494
Pasture 2010	1861 (48.7)	986 (21.6)	3985 (52.6)	702 (35.7)	133 (23.1)	7669
Soil 2010	205 (5.4)	102 (2.1)	170 (2.25)	163 (8.3)	28 (5.0)	673
Vegetation 2010	193 (5.05)	429 (7.4)	587 (7.74)	128 (26.5)	321 (55.5)	1658
Class total	3827	6714	7585	1970	578	
Class changes 2001-2010 (ha, percent)	2539 (66.3)	1944 (28.9)	3599, 47.5	1806 (91.7)	257 (24.5)	
Image difference (ha)	-615	219	84.5	-1296	1080	

Table 2.4: Changes in forest cover from 1991-2001 and from 2001-2010.

Period	*Forest clearing*	*Forest regrowth*	*Net change*
1991 - 2001	-3250 ha, -37%	1020 ha, 12%	-2230 ha, -25%
2001-2010	-1550 ha, -23%	1750 ha, 26%,	0 to 200 ha, 0 to 3%

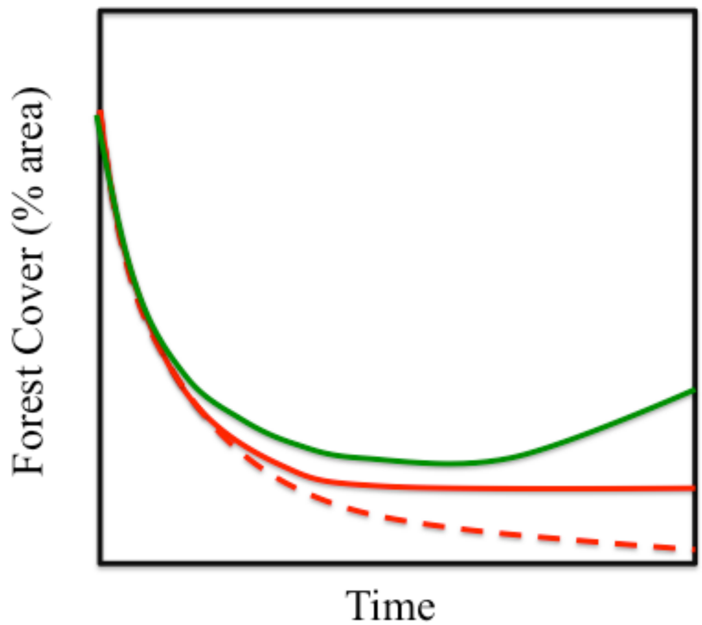


Figure 2.1: The forest transition curve: reforestation and deforestation. The green line indicates total forest cover. The solid red line indicates primary forest cover in a scenario where deforestation ceases as reforestation begins. The dotted line shows primary forest cover in a scenario where deforestation continues as reforestation begins. The area between the red and green lines indicates the relative forest cover composed of secondary forest.

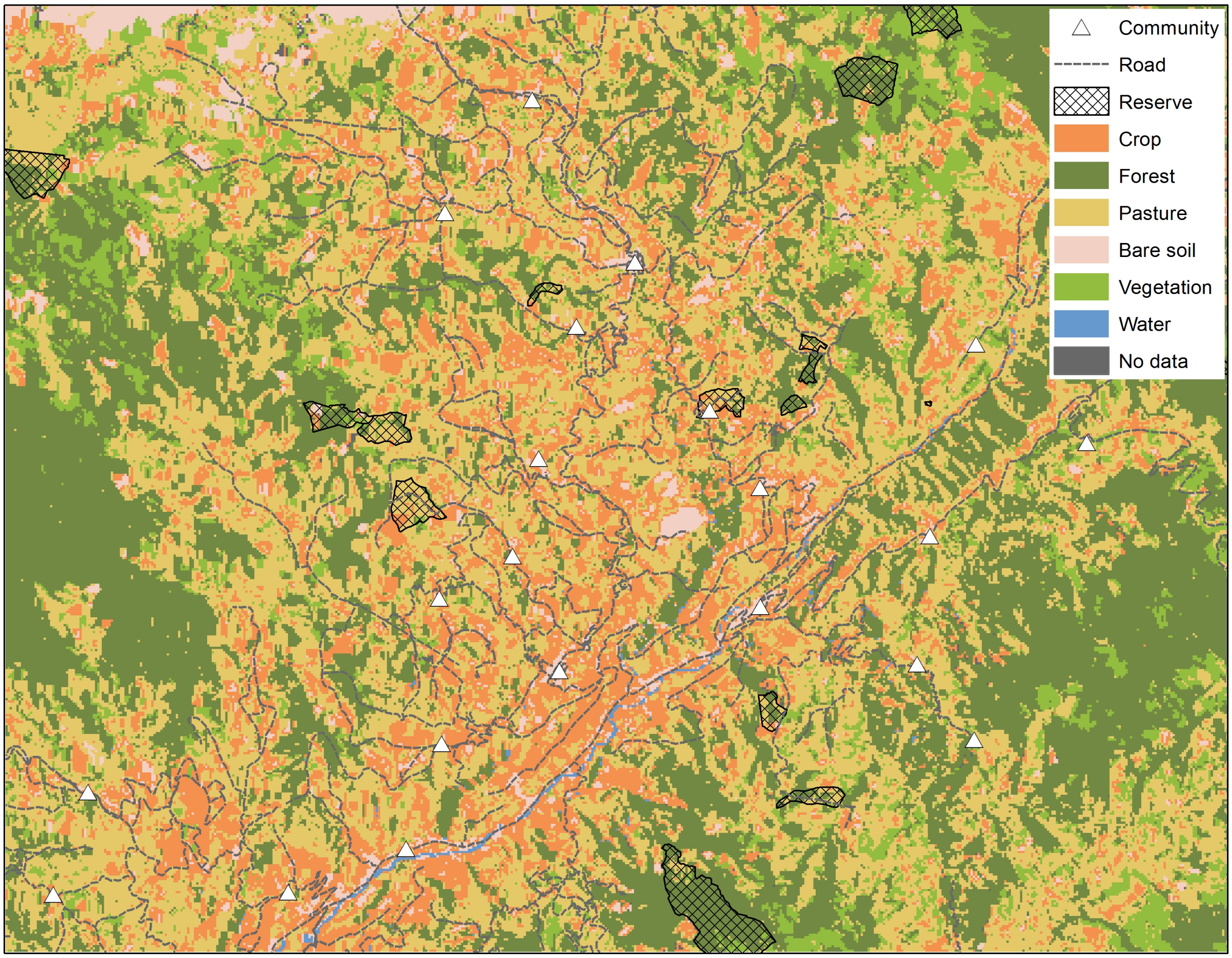


Figure 2.2: Map of the study area with land use in 2010. Land-use classification was based on Landsat imagery at the 30-meter resolution.

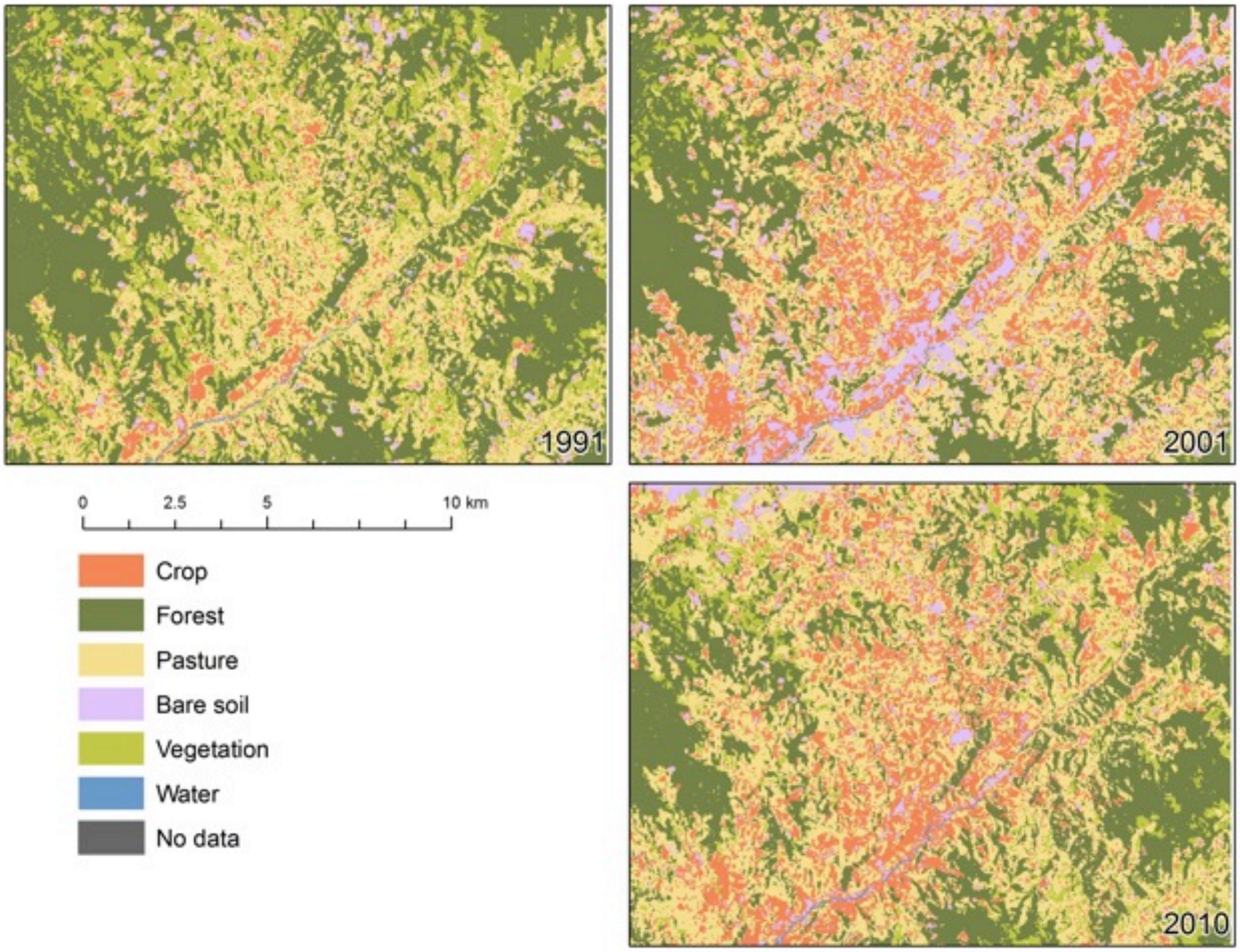


Figure 2.3: Land use in 1991, 2001, and 2010 based on an analysis of LANDSAT images at 30-meter resolution.

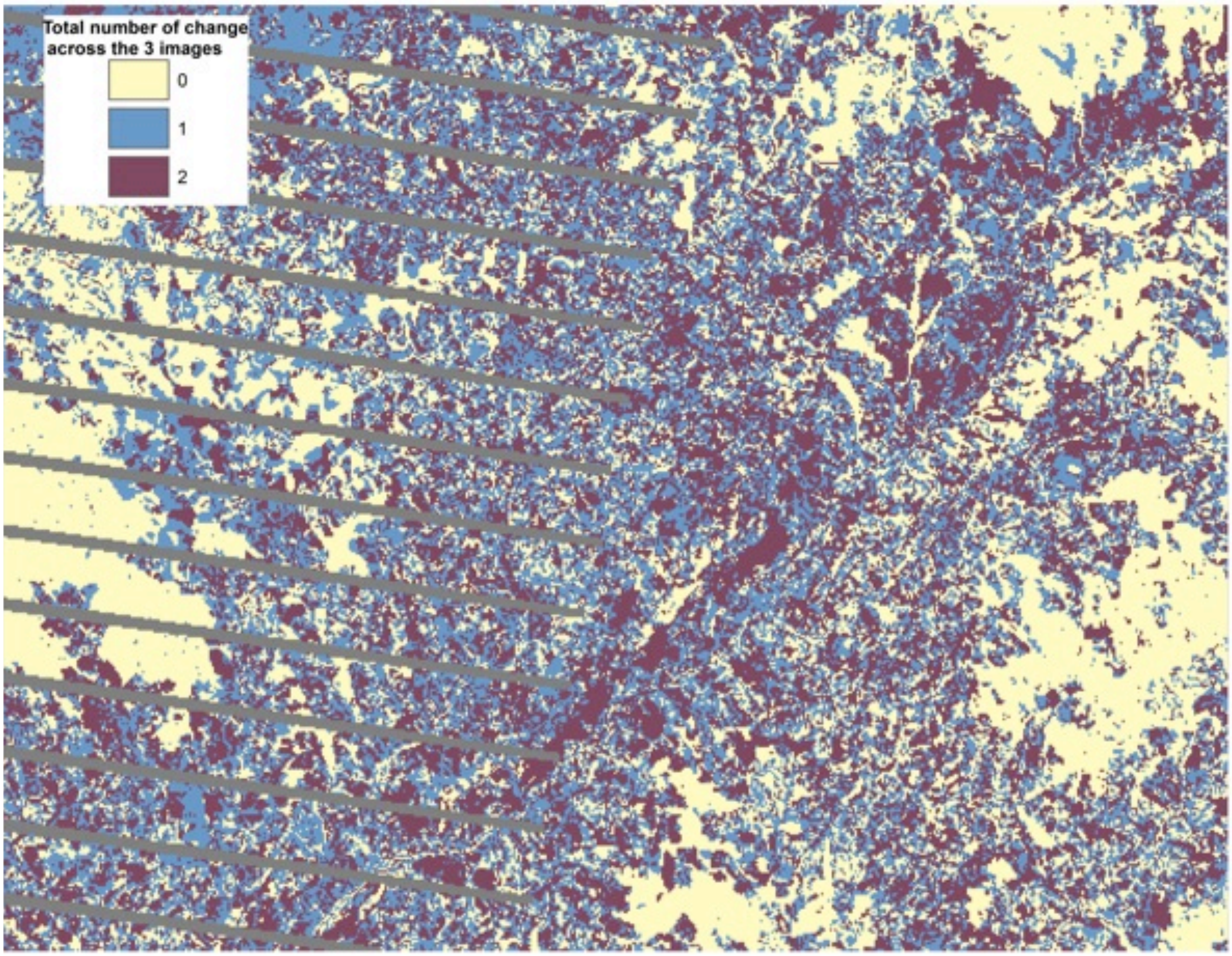


Figure 2.4: Land cover changes in the Intag region, 1991-2010. Blue areas changed from one land use to another in one period, and purple areas changed in both time periods.

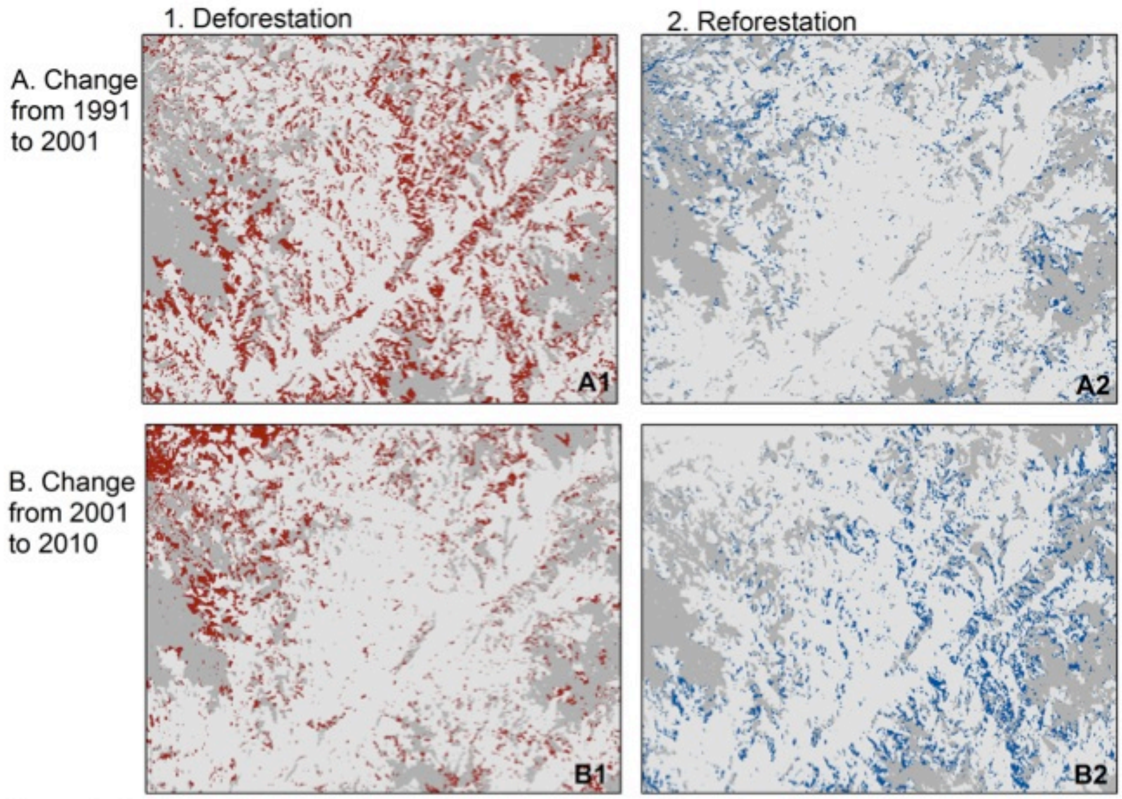


Figure 2.5: Deforestation and reforestation patterns in Intag from (A) 1991-2001 and (B) 2001-2010. Areas that were deforested over the past decade are in red, and reforested areas are in blue. Grey areas are unchanged forest cover.

Preface to Chapter 3

Chapter 2 showed that primary forests in the Intag Valley are being extensively cleared, and that although rates have declined, the process continues. To understand the impact of this clearing on forest biodiversity, create strategies to conserve remaining forests, and restore lost ones, we need to know: 1) what tree species comprise them, and 2) how species diversity is distributed across the landscape. Andean cloud forest trees change in both structure and species composition with elevation. Less is known about how they vary across patches at similar elevations.

Chapter 3 presents a study of mid-elevation forest patches in the Intag Valley. I found that forests are very species rich at the stand level, and both rich and highly variable at the landscape level. This is the first study to compare cloud forest tree communities across multiple Andean cloud forests at similar elevations.

Restoring an ecosystem requires an understanding of the nature of that ecosystem and the species that comprise it. Thus, Chapter 3 also provides baseline data for Chapter 4, which compares the diversity of forests in planted, unplanted secondary, and primary forests to assess the efficacy of restoration efforts.

Chapter 3

Seeing the cloud forest for the trees: Small reserves conserve biodiversity in fragmented Andean landscapes

Sarah Jane Wilson & Jeanine Rhemtulla

At these elevations the traveler finds himself constantly surrounded by a dense fog. This mysterious formation of water… gives the vegetation a verdant colour which is continuously renewed.

— Alexander Von Humbolt, on an Andean cloud forest in 1807

To keep every cog and wheel is the first precaution of intelligent tinkering.

— Aldo Leopold, 1949

How can you govern a country which has 246 varieties of cheese?

— Charles de Gaulle, 1962

Abstract

Montane tropical cloud forests, with their variable topography, high biodiversity, and astounding numbers of endemic species, are a top global conservation priority. However, the distribution of plant and animal species at local and landscape scales is still poorly understood, in part because many cloud forest regions have yet to be surveyed. Empirical work has focused on species distributions along elevation gradients, but spatial variation at the same elevation are less commonly investigated. In this study, I compared tree communities in a narrow elevational band at the upper end of the 'mid-elevation diversity bulge' (1900-2250 masl) in five ridge-top forest patches in the Intag Valley, a heavily deforested region of the Ecuadorian Andes. I found that tree communities were distinct in different patches, and that spatially closer patches were not necessarily more similar to one another. Although larger (1500 to 6880 ha), more intact patches contained more tree species (108-120 species/0.1ha) than smaller ones (56-87 species/0.1ha in reserves 30 to 780 ha) all contained high proportions of species not found in the others. Results suggest that conserving multiple cloud forest patches within this narrow elevational band is essential to conserve landscape-level tree diversity, and that even small forest reserves contribute significantly to biodiversity conservation. These findings can be applied to create management plans to conserve and restore cloud forests in the Andes and tropical montane forests elsewhere.

Introduction

Tropical montane cloud forests are found only in places frequently engulfed in clouds, where they span hundreds of meters in elevation, occupy nooks and crannies with different microclimates and soils, and are divided and dissected by mountain ranges, peaks and valleys. Yet they remain connected to surrounding landscapes by the movement of birds, mammals, and winds, and may be the most specious forests in the world (Gentry *et al.*, 1995; Myers *et al.*, 2000; Bruijnzeel, 2004; Brooks *et al*., 2006; Bruijnzeel *et al.*, 2010). Andean cloud forests in particular are hotspots of biodiversity and endemism: joining high *paramos* (grasslands) to lowland rainforests, the Tropical Andes contain over 15% of known plant species in less than one percent of the earth's land area (Mittermeier *et al.*, 1999), and thus house a disproportionally high percentage of the earth's plant, amphibian, and bird species[7] (Doumenge *et al.*, 1995). Although at a local level, cloud forests often have fewer tree species than lowland forests (e.g., the Andes versus the Amazon), the exceptionally high number of endemic cloud forest species make landscape-level species richness (beta-diversity) higher (Noss, 1983; Whitmore, 1990; Churchill *et al.*, 1995; Haber, 2000; Myers *et al.*, 2000; Brooks *et al.*, 2002; Küper *et al.*, 2004). This diversity arose in large part from the extremely heterogeneous topography and climate present in the Andes at regional, landscape and local scales (Gentry *et al.*, 1995; Myers *et al.*, 2000; Homeier *et al.*, 2010; Herzog *et al.*, 2011; Lippok *et al.*, 2014). But this same physical and biological heterogeneity also make it extremely challenging to inventory, and ultimately to conserve, the species that comprise them (Bruijnzeel *et al.,* 2010). This chapter characterizes the heterogeneity of cloud forest tree communities in a fragmented landscape in northwest Andean Ecuador to inform future conservation actions.

[7] For example, in Peru, it is estimated that nearly 20% of the countries' plant species are found in the five percent of the country covered in Andean montane forest (León *et al*., 1992; Young & León, 1999).

In the Andes, cloud forests occupy less than 60,000 km^2 (Mulligan 2010), and the elevation belt in which cloud forests are found (generally between 1500 and 3500 masl) coincides with relatively populated areas. People have reduced this already limited coverage even further, clearing cloud forests for pastures, agriculture and to extract resources (e.g., timber, charcoal, and minerals). Clearing in recent decades has been particularly rapid: in the 1980s, deforestation rates in Andean cloud forests were higher than in South American lowland forests (Daugherty, 1973; Sarmiento, 1995*a,b,c,*; Young & Leon, 1995*a*; Jokisch & Lair, 2002; Williams-Linera, 2002; Bubb, 2004; Echeverría *et al.*, 2007; Scatena *et al.*, 2010), although over the past decade clearing rates have fallen to one percent per year (Portillo-Quintero *et al.* 2012). To date, more than 50% of Andean cloud forests have been cleared (Mulligan, 2010), with much higher percentages in some locales (Gentry, 1988). Few large, intact patches of cloud forest remain in Latin America, with some notable exceptions in Venezuela and Colombia (Scatena *et al.* 2010). Remaining Andean cloud forests are often fragmented patches within a matrix of other land uses (Daugherty, 1973; Young & Leon, 1995*a*; Jokisch & Lair, 2002; Williams-Linera, 2002; Bubb, 2004; Mulligan & Burke, 2005*a*; Mulligan & Burke, 2005*b*; Echeverría *et al.*, 2007; Scatena *et al.*, 2010).

On a global scale, the biodiversity, rarity, and extremely high levels of endemism combined with their history of rapid clearing has made Andean cloud forests a top global conservation priority (Mittermeier *et al.*, 1999; Brooks *et al.*, 2006; Scatena *et al.*, 2010)[8]. However, within regions where cloud forests are found, deciding which stand or patch to conserve is less straightforward. This is in part because researchers and managers have limited knowledge about how species are distributed in them at the landscape scale, and also lack species

[8] Global biodiversity conservation priorities are set using different ecological indicators, including areas with high species richness (total, or of specific taxonomic groups), endemism, threat of clearing, rarity, or that represent unique environments (Prendergast *et al.*, 1999).

inventories in many cloud forests (Bruijnzeel *et al.*, 2010*b*). To effectively conserve cloud forests, we need to better understand how species are distributed within them on the landscape scale.

Currently, researchers use two main approaches to characterize landscape-scale heterogeneity in cloud forest plant communities: surveying plants at different sites to produce empirical data, and modeling the microclimates created by variation in climate, topography, and other variables important to plant growth and distribution. To date, most empirical studies on cloud forest plant distributions focus on how species composition changes with elevation (Givnish, 1998; Young & Keating, 2001; Cardelus *et al.,* 2006; Watkins *et al.*, 2006; Jankowski *et al.*, 2013; Salazar *et al.*, 2013; Williams-Linera *et al.*, 2013). Traveling upslope from foothills to peaks in the Andes, straight, towering, buttressed trees are replaced by squatter, twisted ones, laden with orchids, bromeliads, ferns and mosses (Grubb, 1977; Stadtmüller, 1987; Lawton & Putz, 1988; Bruijnzeel & Proctor, 1995; Lieberman *et al.*, 1996; Krömer *et al.*, 2005). Species turnover is high: some cloud forests do not share any species in common with the lowland forests at their base, only 1400m lower in elevation (Sanchez-Gonzalez & Lopez-Mata, 2005; Williams-Linera *et al.*, 2013). These changes are driven by steep elevation gradients which cause environmental conditions, such as temperature, edaphic conditions, fog and rainfall to change quickly (Grubb, 1977; Lieberman *et al.*, 1996; Richter *et al.*, 2009; Bruijnzeel *et al.*, 2010*b*; Homeier *et al.*, 2010; Bach & Gradstein, 2011; Martin *et al.*, 2011; Williams-Linera *et al.*, 2013). Models of microclimate heterogeneity also show that sharp gradients in precipitation and solar radiation create variation in environmental conditions that affect plant growth at different scales, from the content to the very local[9] (Beck *et al.*, 2008).

[9] Richer et al. (2009) found significant variation in topography (i.e., in "exposure, slope angle, small valleys and ridges") within an 11 km^2 area.

Tree species distribution in Andean forests is also affected by human activities, such as extracting timber and clearing forest for agriculture (Wunder, 1996). Clearing fragments landscapes such that forests at the same elevation might be disconnected or cleared to a greater extent at some elevations than others. Patches of forests tend to persist in places that are unsuitable for agriculture, like steep slopes or ridges (Sarmiento, 2002).

The value of small forest patches to conserve biodiversity has been debated for decades (Schelhas & Greenberg, 1996). Small forest patches often contain only a subset of the forest species found in intact primary forests (Turner & Corlett, 1996; Laurence *et al,*. 2006; Saura *et al*., 2014). However, they are also important for conservation because they are key sources of seeds for nearby forest regeneration, provide habitat for birds and other animals, and, where forests have been heavily cleared, may represent the last remnants of primary forest on the landscape (Kattan *et al.*, 1996; Holl, 1999; Holl *et al.*, 2000; Kattan *et al.*, 2004; Williams-Linera *et al.*, 2013). Few empirical, comparative studies have quantified how individual cloud forest patches contribute to conserving tree diversity at the landscape scale (Williams-Linera *et al.,* 2013). From the small body of literature comparing tree diversity in different patches of cloud forests, we know that protecting forests at a range of different elevations is critical to conserve biodiversity (Gentry, 1988; Williams-Linera *et al.*, 2013), but even fewer studies have quantified differences in tree species composition between patches of forest at similar elevations.

The purpose of this chapter is twofold: 1) to characterize the variability in the diversity and species composition of trees in different patches of Andean cloud forest within the same narrow elevational range (1900-2250 masl); and, 2) to determine how much of this variation can be attributed to local and landscape variables commonly measured in the field. I hypothesize that even within the same elevational range different patches of cloud forest will support distinct communities of trees. I address three research questions:

1) *What species characterize the tree communities in mid-elevation cloud forest reserves in the Intag region of northwest Andean Ecuador (1900 – 2250 masl)?*

2) *How similar or different are these tree communities from one reserve to the next?*

3) *Do variables aside from elevation (i.e., proximity to one another, reserve size, topography, and history of human disturbance) explain the variance in the community composition of trees?*

To address these questions, I sampled tree species composition in five patches (30-6880 ha) of protected primary forest in the Intag region in northwest Andean Ecuador. Cloud forests in the northwestern Andes have experienced some of the most intensive clearing in this forest type globally (Mulligan & Burke, 2005), and the landscape in found in Intag – fragments of cloud forests separated by ridges and/or other land uses – now typifies many parts of the tropical Andes (Armenteras *et al.*, 2003) (Chapter 2). This research is the first to quantitatively compare tree biodiversity in different patches of forests in the region, and to survey tree communities in the higher elevation forests in four of the five patches. I conclude with the implications of these findings for conserving and restoring tropical montane cloud forests, and in particular, the conservation potential of smaller forest fragments compared to larger, more remote sites.

Methods

Study site

Andean Ecuador is part of the 'Tropical Andes' global biodiversity hotspot, with some of the highest levels of plant endemism on the planet: 30% of tropical Andean epiphyte species in Ecuador occur nowhere else in the world (Myers *et al.*, 2000). In this region, in 1976 botanists Alwyn Gentry and Calaway Dodson found over 90 endemic species of epiphytes, herbs and other plants in a forested area (the Centinela ridge) of less than 10 km^2 (Gentry, 1986), showing that small patches of cloud forest contribute disproportionately to global biodiversity (Dodson & Gentry, 1991). Plant communities, including trees, on Centinela were distinct from the surrounding lowland forests and, because they were isolated from other forests at the same elevation, were unique in the world. When Centinelian forests were almost completely cleared less than 10 years later, this region became one of the seminal cases of 'anonymous extinctions' (Wilson 1992), where species endemic to a tiny patch of land disappear rapidly and irreversibly.

In the northwest Ecuadorian Andes, such rapid clearing over the past 50 years for agriculture and pasture has left forests fragmented. Today, less than 10% of the original cloud forest cover remains (Sarmiento, 1995*b*). For the past two decades, international and Ecuadorian environmental non-governmental organizations (NGOs) have worked to create a conservation corridor in the area to preserve forests and habitat for rare, endemic and threatened species (e.g., the spectacled bear, *Tremarctos ornatus*) (Kattan *et al.*, 2004).

Located only 100 km northeast of Centinela, my study sites occupy the next cloud forest-covered spur of the western Ecuadorian Andes. I studied forests in primary cloud forest reserves in and around the Intag Valley (hereafter Intag), located in the Imbabura and Pichincha provinces (0.35° N, 78.5° W). The region is rugged and steep (Fig. 3.1, Fig. 3.2), with an average annual temperature of 17-20° C, annual rainfall of 1500-3300 mm, and a single pronounced dry season from May/June to October (Freiberg & Freiberg, 2000).

Intag is an ideal location to study how cloud forest diversity is distributed across the landscape, and how individual patches contribute to conservation. First, cloud forests here are extremely biodiverse: previous tree surveys at lower elevations (1400 masl) found over 300 tree species in a single hectare (A. Mariscal, pers. comm., 2010), and my data indicates that almost 400 tree species are present at the elevations studied across the five reserves. Second, forests here were cleared extensively in the 1970s and 1980s mainly to provide land for pasture and agriculture (Sarmiento, 2002). Most of the remaining large, intact forest patches are located above 2300 masl because people tend to clear up mountains (Sarmiento, 2002). Local people and NGOs protect some of the few remaining mid-elevation forests on private land, in government-owned reserves, and in community-owned and -managed reserves (DECOIN). It was these forests that I studied, located at elevations between 1900 to 2250 masl. Conserving forests within this range is especially important because 1) they have been heavily cleared; 2) they are under high threat of future clearing - land at this elevation is suitable for both agriculture and pasture; 3) these elevations mark the upper end of the 'mid elevation biodiversity bulge' with high species turnover and species richness (Gentry *et al.*, 1995); and, 4) nearby communities are currently trying to restore forests at this elevation to enhance their water supply (Chapter 4, Chapter 5).

I sampled trees in five primary forests reserves (Fig. 3.2, Table 3.1) selected based on both elevations and their history of protection (none of the forests have been cleared within living memory). For at least the past 15 years forest use in all reserves has been limited to rustic ecotourism and non-invasive research. Reserves range in size from 30 ha (Nangulvi) to 6880 ha

(Los Cedros). Each reserve occupies a separate ridge. Three reserves are connected to other large patches of primary forests: Junin and Los Cedros to the higher elevation forests of the Cotocachi Cayapas reserve, and Santa Lucia to the lower elevation forests in the Maquipucuna reserve (Fig. 3.2). Pasture and small-scale agriculture border the remaining perimeter of these reserves, as well as the entire perimeter of the Bosque Intag and Nangulvi reserves. Study sites within each were located within the same narrow elevational band but separated by valleys or ridges of over 400 m vertical (Fig. 3.2).

Data collection

Forest transects

In each reserve, I surveyed trees along four 50 m x 5 m transects (total 0.1 ha/transect) at least 100 m apart perpendicular to the slope of the mountain. I placed transects near the tops of the ridges in areas of intact primary forest, avoiding gaps. Transects were typically located on a range of slopes (15 to 35°), and over an elevation range of 20-160 m within each reserve. I divided each transect into five 5 x 10 m plots to sample trees. In each plot I counted, identified, and measured trees (>2.5 cm diameter-at-breast-height (DBH)), woody saplings (1-2.5 cm DBH) and seedlings (>0.5 m height and < 1cm DBH). Trees were identified in the field to species level by Ecuadorian botanists. I took replicate voucher samples to identify unknown species by collecting leaves and fruits and flowers (when available) and preserving them in 75-80% alcohol. Botanists at the *Museo Ecuatoriano de Ciencias Naturales Herbario Nacional del Ecuador* later identified these samples. Following Chazdon *et al.* (1998), I counted plants as separate individuals if the stem of the plant was not connected at, or just below, the soil surface. I recorded elevation using a hand-held Garmin GPS Map 60 unit. At two random locations on each transect, I recorded slope (using an analog clinometer), aspect (using a hand-held compass), canopy density (using a convex spherical densitometer) (Lemmon, 1957), and percent ground cover (estimated visually in two 1 m^2 plots).

Soils

In each reserve, I took 10 samples from the top 10 cm of soil at two or three randomly located places on each transect. I made a composite soil sample by mixing these 10 samples in a plastic bucket. When transects were located more than 100 m vertical elevation apart within a single reserve, I created separate composite samples from the upper and lower transects. Composite samples were stored in a conventional refrigerator and delivered to the soil laboratory for analysis within five days of collection (*Estacion Experimental "Santa Catalina", Instituto Nacional Autonomo de Investigaciones Agropecuarias,* Cutuglagua, Mejía, Pichincha) for analysis. Soils were analyzed for macro and micronutrients (nitrogen (NH_4), phosphorus (Olson modified), potassium, calcium, magnesium), bases, organic matter content, and texture (for chemical analysis methodology, see Appendix B). I also measured the bulk density of the top 10 cm of soil using a 10-cm diameter cylindrical sampler. I collected one bulk density sample from each transect in each reserve (four per reserve). I weighed each sample both while wet and after being sun-dried in a hot attic until the dry weight had stabilized (up to four weeks).

Human influence

All five reserves have been continuously forested in living memory (i.e., have been there continuously since the oldest residents can remember) and have been designated for conservation for at least the past 15 years. Since some reserves were more accessible than others, I quantified the degree of human influence on each reserve using two locally relevant indicators: 1) the time required to walk to the transects from the nearest road; and, 2) a ranking of past disturbance based on the history of access to and extraction from each reserve (determined by interviewing landowners, managers, and long-time community residents). I noted the presence or absence of locally harvested timber species and recorded signs of human disturbance such as stumps and cut branches.

Data analysis

I used three metrics to quantify tree species diversity in each reserve: 1) *species richness*; 2) the *Chao richness estimator (Chao1)*, to estimate the number of 'unseen' species present given that sampling in this very biodiverse environment was likely incomplete(Chao, 1984; Chen *et al.*, 1995); and, 3) *species density*, the number of species per hectare. I rarefied species richness at each site using EstimateS version 8.2.0 (Colwell, 2009). Rarefaction creates smooth species accumulation curves (showing the additional number of species found as more individuals are sampled) by resampling sample data 100 times. It is not possible to statistically compare curves, but error bars representing standard deviation can be used to determine if two sites contain significantly different species counts for a given number of individuals (Chazdon *et al.*, 1998). I compared species richness rarefied to a common number of stems using ANOVA and a natural log transformation to meet assumptions of normality (SPSS IMB corp. 2011, Version 20.0). Finally, to help account for a potential under-sampling bias (common in very biodiverse environments, where samples may not represent the entire community), I compared Chao1 across sites (Chao, 1984).

I used two measures to quantify differences in the communities of trees in each reserve:

1) the *Chao estimator of shared species*, which estimates the number of shared species in each reserve while accounting for 'unseen' species (appropriate for sampling in very biodiverse habitats such as cloud forest) (Chen *et al.*, 1995); and, 2) the *Chao-Jaccard similarity estimator*, a measure of beta-diversity ranging from 0 to 1 (0 indicates completely distinct communities with no shared species and 1 indicates identical communities) which also corrects for the under-sampling bias by estimating the number of 'unseen' shared species between sites (Chao *et al.*, 2005).

Ordination was used to visualize the relative differences within and between reserves (Table 3.2). I ran a *non-metric multidimensional scaling (NMDS)* ordination using the Chao-Jaccard similarity index scores for each transect in each reserve, and used this to identify the species that characterized each site. To quantify the degree to which site level, landscape level, and spatial variables explain these differences in species compositions between sites I used *redundancy analysis* (RDA) and *variance partitioning*. RDA constrains the variation in a dataset to a specific set of variables, allowing me to compare the relative influence of the spatial and environmental variables on the species composition of tree communities (Legendre *et al.*, 2005). To determine how much of the variation within and between sites was driven by spatial location alone, and to correct for autocorrelation at our sites due to spatial proximity, I ran a partial RDA with the spatial coordinates (latitude, longitude) of each transect factored out (Legendre *et al.*, 2005). Explanatory variables in the RDA were those soil characteristics that both affect plant growth and are indicators of past disturbance (macronutrients, exchangeable bases, organic matter and bulk density), site environmental data (slope and elevation), and landscape data (accessibility and size of reserve). When soil variables were highly correlated, I used a summary metric that most represented that group of variables (for example, sum of bases to represent the highly correlated concentrations of Mg, K, and Ca in the soil). To quantify which variables explained the variation in species composition data, and the relatedness between sets of variables, I performed a variance partitioning analysis, categorizing our variables into spatial (latitude, longitude), environmental (slope, aspect, elevation, soils), human disturbance (distance to road), and reserve size. In the multivariate analysis, I only used the distance to road as this variable was highly correlated with the ranking of past disturbance. NMDS ordinations were performed in PCord Version 6 (McCune & Mefford, 2011). RDA and variance partitioning in the vegan package in R (Oksanen, 2011).

Results

General characteristics of cloud forest tree communities

I identified 3296 stems and 300 species of woody plants in the five forest reserves. The majority were trees (229 species) and woody shrubs (39 species), with some tree ferns (11 species), canopy palms (5 species), and other woody plants with unknown form (16 species). Lianas were excluded from the dataset. For simplicity, from hereon I refer to all species as trees (Norden *et al.,* 2009). Across all five reserves, the Chao 1 estimator of species richness estimates the actual combined species richness to be 395 species.

The families of trees in Intag cloud forests are typical of Andean cloud forests, with many representatives from the families Rubiaceae and Lauraceae (Gentry *et al.,* 1995). Only one species, the subcanopy tree *Palicourea demisa* (Rubiaceae), was found in all five reserves. This species was also the most abundant in the dataset, accounting for 13% of all stems. Other subcanopy trees in the family Rubiaceae were also widespread (i.e., in three or four reserves) and abundant: *Psychotria hazenii* (2.9% of total stems), *Palicourea thyrsiflora* (2.6%), and *Faramea* *calyptrate* (1.6%). Widespread and abundant (more than 1% of the total abundance of stems), canopy tree species were *Billia rosea,* (1.4%, Sapindaceae) and *Weinmannia balbisiana* (1.3%, Cunoniaceae). Although the most abundant canopy tree species, *Carapa guianensis* (Meliaceae), accounted for 3% of the overall abundance, I only found it in one reserve (Junin). Other canopy tree species that were locally abundant in one or two reserves included *Ossaea micrantha* (2.6%, Melastomataceae), *Calyptranthes maxima* (1.8%, Myrtaceae), and *Ocotea stuebelii* (1.2%, Lauraceae) (Appendix C).

Across the five reserves, 19 of the 300 species were classified as globally Endangered or Threatened, and 8 were Near Threatened (IUCN). Of the 19 Endangered or Threatened species, 12 were each only found in one reserve, six were found in only two, and only 1 was found in more than 2 reserves (Appendix C).

Tree species richness and density

Individual reserves contain different tree species and contribute to species richness at the landscape scale. I found between 58 and 120 tree species in 0.1 ha in individual reserves (Table 3.1, Fig. 3.3). Rarefaction curves showed that the two largest reserves, Junin and Los Cedros, had the most species and similar species richness. Bosque Intag had the next most species, significantly different from the other four reserves. Santa Lucia and Nangulvi had the fewest species and least similar species richness, even though Santa Lucia is nearly 20 times larger than Nangulvi (Fig. 3.3). Rarefied species richness (to 480 stems) in each reserve ranged from 58 to 94 species (Fig. 3.3, Fig. 3.4). The most specious reserve, Junin, had 1.6 times more species (90 species) than the least, Santa Lucia (58 species). The Chao 1 richness estimator showed even greater differences in species richness, with Junin (258 species) estimated to have nearly four times the species richness of Santa Lucia (81 species). The species richness rank for each reserve remained the same regardless of the metric used (Fig. 3.3): larger reserves (Junin and Los Cedros) contained the most, then Bosque Intag, with Santa Lucia and Nangulvi supporting the fewest.

Species density, which ranged from 58 to 120 trees species per 0.1 ha, followed the same pattern as richness: Junin had more than twice the species per unit area (120/0.1ha) than Santa Lucia (58/0.1ha). The other largest reserve, Los Cedros, contained 104/0.1 ha, Bosque Intag 81/0.1ha, and the smallest reserve, Nangulvi, 78/0.1ha.

Tree species composition and shared species

Each reserve had distinct tree communities (Table 3.3, Fig. 3.5, Fig. 3.6). The Chao-estimator of shared species showed that even the most similar reserves shared less than 50% of the same species, even though they were all located at the same elevation within a relatively small (875 km^2) geographic area (Table 3.3). The two most dissimilar reserves, Bosque Intag and Santa Lucia, shared only five species – an almost complete turnover in species composition over only 28 km. The Chao-Jaccard similarity index ranged from 0.10 (almost completely distinct communities) to 0.53 between different reserves (Table 3.3).

The degree to which two reserves were similar did not appear to be related to the distance between them, nor to the size of the reserve (Fig. 3.2, Fig. 3.5, Fig. 3.6). The sites that were the most similar, with an estimated 73 shared species and a Chao-Jaccard similarity index of 0.53, were the largest (Los Cedros) and smallest (Nangulvi) (Fig 3.4).

Species characterizing different reserves

Each forest is composed of distinct communities of trees. Reserves that shared more species (Nangulvi and Los Cedros; Los Cedros and Junin; and Junin and Santa Lucia) also tended to group in the NMDS ordination. Tree communities on individual transects grouped by reserve, and were more similar within each reserve than between reserves (*MRPP*, $T = -10.1$, $p < 0.0001$) (Fig. 3.6). Although transects in the same reserve varied in slope, aspect, and elevation, they did not group by these variables (*MRPP, all* $p > 0.05$).

Although each reserve supported distinct tree communities, NMDS ordination showed that three general groups of species were evident across reserves (Fig. 3.5). The trees in each of these groups are not necessarily the most abundant within each reserve; rather, they are those that are both unique to the reserve(s) that they characterize and were typically present in large numbers where they were found.

The first group of species was found in Bosque Intag. Forest here was characterized by the canopy tree *Alchornea latifolia* (Euphorbiaceae), the endangered palm *Ceroxylon alpinum* (Arecaceae), and the subcanopy trees *Hieronyma macrocarpa* (Phyllanthaceae) and *Miconia crocea* (Melastomataceae). It was also the only reserve where avocado (*Persea americana*, Lauraceae) was found. Sixty-eight percent of the species in this reserve were not found in any other reserve (Table 3.1); when 'unseen' species were estimated, Bosque Intag shared at most 20 species with any other individual reserve (Table 3.3).

The second group of species, associated with the reserves Nangulvi and, to a lesser extent, Los Cedros, was characterized by the canopy trees *Inga densiflora* (Fabaceae) and *Casearia silvestris* (Salicaceae), and the subcanopy tree *Palicourea thyrsiflora* (Rubiaceae). Although similar species characterized these two reserves, they shared only 23 species.

The third group, associated with Santa Lucia and Junin, was characterized by the canopy trees *Weinmannia balbisiana* (Cunoniaceae), *Stylogyne ambigua* (Primulaceae), and the subcanopy tree *Faramea calyptrate* (Rubiaceae). These two reserves shared 24 species.

Spatial and environmental variables associated with species composition

<u>NMDS ordination</u>

The distance between reserves was not a good predictor of similarity (Table 3.3, Fig. 3.5). However, the spatial arrangement (latitude and longitude) of the transects explained some of the variation in tree species composition. The first axis in the NMDS ordination correlated highly with longitude ($r = 0.668$), site accessibility, and several soil characteristics often associated with disturbed soils (Bautista et al. 2005): higher pH ($r = 0.588$), bulk density ($r = 0.425$), and levels of exchangeable bases ($r = 0.633$). The most eastern reserves separated from, and scored higher than, the western reserves along this axis (Fig. 3.6). The second axis correlated highly with latitude ($r = 0.698$), and negatively with soil phosphorus ($r = -0.729$) and nitrogen ($r = -0.516$) (Table 3.4).

<u>Redundancy analysis (RDA) and variance partitioning</u>

I conducted an RDA analysis to quantify the amount of variation explained by spatial arrangement versus other environmental variables, and to determine which variables significantly explained the variation in the data. The RDA explained 50.0% (adjusted r^2) of the variation in species composition (*ANOVA*, $F = 2.15$; $df = 15$; $p < 0.001$) (Fig. 3.6, Table 3.3). The geographic coordinates, elevation, soil bulk density, soil pH, soil nitrogen and soil phosphorus of each transect were all significant predictors of the variation in data (all $p < 0.05$);

however, reserve size, distance to road, slope, aspect and other soil characteristics were not (Table 3.2, Table 3.4). The RDA ordination confirmed the results of the NMDS ordination in four key ways. First, the transects at each reserve grouped tightly. Second, the same reserves came out as most similar to one another (Junin and Santa Lucia, Los Cedros and Nangulvi, with Bosque Intag the most distinct). Third, three distinct clusters of species were apparent (Fig. 3.5, Fig. 3.6). Fouth, site accessibility and several soil characteristics correlated with one another and the first axis. Latitude also correlated with the first axis. Elevation and slope correlated with each other and with the second axis (higher elevation sites score higher on the second axis).

A partial RDA with the effect of space removed explained 27.8% of the variation in our data, and was highly significant (*ANOVA; $F = 1.646$; $df = 13$; $p < 0.001$*). Transects at each site grouped, although each group of transects tended to be closer to one another in ordination space (indicating less differentiation between reserves). Elevation, soil pH and N remained significant predictors of the variation in our data, while reserve size, distance to road, slope, aspect and other soil characteristics were not (Figure not shown).

A model of variance partitioning between environmental and spatial variables explained 47% of the variance in tree community composition. Environmental variables accounted for the majority of this variance (45%), and geographic coordinates explained 21%. This means that 20% of the variation explained by environmental variables was spatially structured, and space alone explained only small portion of the variation (2%).

Taken together, these ordination and variance analyses show that: 1) each reserve supports distinct communities of tree species, and that variation in tree communities is greater among reserves than within; 2) the variation in species composition is only driven in part by their spatial proximity to one another; 3) differences in soil fertility and small-scale variation in elevation explain some variation in the data, half of which is spatially structured; and, 4) metrics of human access/disturbance (reserve size, distance to sites) do not explain a significant fraction of the variation in the data.

Discussion

This comparative study showed that the tree communities in cloud forests can vary markedly over a small area, even within the same elevation range. In Intag, forests located only 10 km apart shared as few as 10% of the same tree species overall. Some forest reserves had more species than others, but even the least specious contained a high percentage of trees not found in the others. Thus, all forest reserves contributed significantly to conserving tree species richness at the landscape scale.

Previous studies have found that elevation is the main driver of species turnover in cloud forests (Givnish, 1998; Watkins *et al.*, 2006; Jankowski *et al.*, 2013; Williams-Linera *et al.*, 2013; Lippok *et al.*, 2014). Unlike studies that examine a wide range of elevations, in my study no single variable explained the majority of the variation in species distributions. The size and accessibility of the reserve was related to species richness: larger (1500 to 6800 ha), less accessible reserves contained more species than smaller (30 to 500 ha), more accessible ones, as has been found in other tropical forests (Bierregaard Jr. *et al.*, 1992). But smaller reserves still supported unique combinations of species, including many species not found in the other reserves. This indicates that small cloud forest patches can 'punch above their weight' to conserve landscape-level tree diversity in heavily deforested areas (Schelhas & Greenberg, 1996). Because these patches represent some of the last vestiges of primary forest in this region, they also serve as important seed sources for local reforestation and restoration efforts (Schelhas

& Greenberg, 1996). In addition to protecting large areas of primary cloud forest conservation plans should protect, and even expand, small isolated forests, especially in heavily deforested regions.

Cloud forest patches hold unique combinations of tree species

Comparing forests at the same elevation, I showed that tree communities in ridge top patches of cloud forest are distinct from those on neighboring ridges – within an area of 875 km^2, only one tree species was common to all five reserves. Previous research on species distributions in the Andes has found similar patterns in epiphytes, birds, butterflies, and other organisms: patches of forests on different ridges or mountaintops contain unique combinations of species and high numbers of endemics (Gentry, 1992; Fjeldså *et al.*, 1999; Brehm *et al.*, 2008; Bruijnzeel *et al.*, 2010*b*; Herzog & Kattan, 2011; Jost, 2013). For example, forests in Centenela, a cloud-forest covered ridge only 100 km southwest of Intag, contained an estimated 90 species of epiphytes that were absent from cloud forests on neighboring ridges (Dodson & Gentry, 1991). On the eastern slopes of the Ecuadorian Andes, endemic orchids are found on some mountaintops but absent from others only a few kilometers away (Jost, 2013). Bird populations vary spatially on local scales throughout the Andes, reflecting patterns in habitat heterogeneity and historical shifts in climate (Fjeldså *et al.,* 1999). The tropical Andes have the highest rates of endemism of frogs and salamanders in the world – species are often restricted to very narrow ranges (Bruijnzeel *et al.,* 2010).

My results also agree with previous studies of tree communities in non-Andean cloud forests. In Veracruz, Mexico, tree communities in cloud forests located at similar elevations (1850 to 1950 masl), distances from one another (8 to 28 km), and in the same size study plots

(0.1ha) as the reserves in our study showed Chao-Jaccard similarity estimates comparable to the values I found (0.16, 0.18, and 0.4 for sites that were 28, 21, and 8 km apart, respectively) (Williams-Linera *et al.,* 2013) (Table 3.2). In the montane cloud forests of Venezuela at an elevation of 2550-2650 masl, two forests 30 km apart shared only 38% of the same tree species (Hetsch & Hoheisel, 1976), again comparable to the results of this study (Table 3.2). In contrast, lowland forests in Latin America tend to show greater levels of similarity across landscapes. In 1 ha plots in the Peruvian and Ecuadorian Amazon, Condit *et al*. (2002) showed that sites five to 100 km apart consistently shared 30 to 40% of the same tree species (and because this analysis did not account for 'unseen' species, the actual number of shared species may have been even higher) (Chao, 1984; Chen *et al.*, 1995). In the lowland forests of Oaxaca, Mexico, forest fragments separated by distances of 15 to 100 km had Jaccard shared species indices[10] of 0.11 and 0.57. At my sites, this index ranged between only 0.03 and 0.175 over shorter distances (10 to 35 km) (data not shown), indicating a much greater degree of dissimilarity (Gordon *et al.,* 2004). The difference in species distributions between montane and lowland forests indicates that different mechanisms drive species distributions, with topography playing an especially important role in montane regions (Gentry, 1988; Young & Leon, 1995*a*; Young and Keating, 2001; Condit *et al.*, 2002; Kessler, 2002; Knapp, 2002; Küper *et al.*, 2004) and that each require specific strategies to manage and maintain biodiversity on the landscape scale (Gentry, 1992).

Variables driving species distributions

Although the species composition of each reserve in my study was distinct, some reserves were more similar than others. Across a landscape, environmental conditions that affect the

[10] The Jaccard index varies between 0 and 1, with 0 being completely dissimilar (Chao *et al*., 2005)

distribution of plants, such as rainfall amounts or soil properties, are often spatially distributed along gradients in elevation or latitude (Richter *et al.*, 2009; Kessler *et al.*, 2011). A combination of spatially and non-spatially distributed environmental variables explained approximately half of the variation observed in my study. These are described in detail below.

1. Spatial arrangement and reserve proximity: The spatial arrangement of the reserves explained part of the variation in tree communities. Much of the variation explained by 'space' was correlated with environmental variables, but other, non-spatially distributed variables also explained variance in tree communities. Local soil conditions and microclimates, historical fluctuations in climate, and past evolutionary and extinction events are all major drivers of species distributions in Andean forests (van der Hammen, 1974; Gradstein *et al.*, 2001; Young *et al.*, 2002; Richter *et al.*, 2009; Jørgensen *et al.*, 2011; Kessler *et al.*, 2011). Over the landscape-level scale I studied, the distance between reserves was not a good proxy for the similarity (or distinctiveness) of the species they support (Aubad *et al.*, 2008). Conservation decisions based on the assumption that spatially closer sites are also, by default, more similar, a common assumption for lowland forests (Condit *et al.*, 2002), would not necessarily maximize landscape diversity in cloud forest regions.

2. Topography: I did not quantify the effect of topography on tree species distributions; however, it bears discussing in the context of my results. Unlike lowland forests, cloud forests are so distinct in part because patches at the same elevation are isolated from one another by variations in topography that create sharp changes in climate and biota (Dodson & Gentry, 1991; Young *et al*. 2002). Ridges and valleys also create boundaries that can limit plant seed dispersal and

dispersers (Graham *et al.*, 2010). Because topography creates a high degree of biotic isolation between mountains, some liken their biogeography to islands[11], where the unique communities of endemic plants and animals are driven by rapid speciation combined with geographical isolation (MacArthur, 1967; Wilson, 1992).

By comparing multiple patches of forest along the same valley, I demonstrated that this variation occurs at a very fine spatial scale. The ridge that forms the northwestern side of the Intag valley (the Cordillera Toisan) runs from the high-elevation *paramos* to the lush lowland forests. The southeastern side of the Toisan ridge divides into additional ridges separated from each other by a deep valley (Fig. 3.2). Three of the five study reserves are located on these ridges, and two more are found on the western slopes on the other side of the Intag valley. All except two are separated from each other by a ridge at least 400 m high (Fig. 3.2). Between Santa Lucia and Junin, however, there is no ridge. Although these two sites are 22 km apart and separated by a deep (700 masl) wide valley, they tended to group most closely in ordinations. Perhaps quantifying how the degree of isolation, and in particular, the differences between how ridges and valleys affect seed dispersion by wind and animals, and the composition of cloud forest tree communities overall, is a relevant question for understanding species distributions in this region and in montane cloud forests elsewhere (Ramirez-Villegas *et al.*, 2014).

<u>3. Soil properties</u>: Macro- and micronutrient levels, bulk density and organic matter content – all

[11] The aforementioned Centinelan forests provide an excellent illustration of this: located on a ridge, Centinelian cloud forests were surrounded by lowland forests with very different tree communities (Gentry, 1992). To the east, rapid increases in elevation up to the grassland *paramos* of the high Andes separated them from other Andean cloud forests at similar elevations (Dodson & Gentry, 1991). Thus, although connected to some degree to the surrounding landscape by migrating animals and wind, in many ways Centinela resembled an island (Diamond, 1975). Rapid speciation combined with geographical isolation created forests with unique combinations of species and high numbers of endemics (Wilson, 1992).

soil properties that influence what species of trees will thrive at a given site – varied among the reserves in our study (Tanner *et al.*, 1998) (Fig 3.7). The larger, less accessible reserves to the west (Los Cedros, Junin, and Santa Lucia) had higher nitrogen and phosphorus levels, organic soils, and higher levels of soil organic matter (Sommer & Quinlan, 2009). The smaller eastern reserves – Bosque Intag and Nangulvi – tended to have lower levels of nitrogen, phosphorus, and SOM; higher soil pH, bulk density, and base saturations; and texture classes of loamy or sandy loam rather than organic, all conditions sometimes found in cloud forests that have experienced some past disturbance (Bautista-Cruz & Castillo, 2005). Because these two reserves are more accessible, smaller, and also the most easterly, it was not possible to determine definitively if soil conditions were driven by underlying spatial gradients or past human activities. However, the effect of soil properties on tree species compositions is likely to be lower if changes in soil properties were driven by people. If differences were driven by an (non-human mediated) environmental variable, they would have been present longer and likely had an impact on older tree communities. On the other hand, if differences in soils were driven by people, based on regional settlement patterns (Chapter 2, Chapter 5) this disturbance likely occurred within the past 30 years, potentially having a lower impact on the older tree communities in these reserves.

4. Human Influence: The forests I studied were the most intact primary forests in the region. Botanists confirmed that they contained species and had structural characteristics typical of primary cloud forests, and long-term residents confirmed that forests had not been cleared in living history, nor had any significant timber extraction occurred. Nonetheless, people still had access to these forests in the past. Bosque Intag, Nangulvi, and Santa Lucia were all located in working agricultural landscapes where cattle may have escaped into the woods occasionally, and

in the past people likely extracted some firewood (mainly from coppicing trees). However, there were no signs of recent human intervention (i.e., stumps, clearings, cut branches) in any of the sites I studied. Furthermore, despite similarities in soil conditions, the two most accessible forest patches (Nangulvi and Bosque Intag) supported very different communities of trees from one another. The effect of past human intervention on the species composition of trees, if any, was thus not unidirectional. Even if humans have had an impact on soils, I found no evidence to suggest that people have directly altered the species composition of trees.

Overall, no single environmental factor drove differences in tree communities. My results concur with other studies that have shown that variation in Andean forests is driven by a complex interplay of landscape and local environmental variables, perhaps combined with differences in the evolutionary history of these forests (van der Hammen, 1974; Gradstein *et al.*, 2001; Wilson, 2002; Richter *et al.*, 2009; Jørgensen *et al.*, 2011; Kessler *et al.*, 2011). However, mine is one of the first empirical studies to show the degree of variation between trees in nearby patches at similar elevations.

Implications for conservation

Currently, when deciding which forests to protect and which to fell, highest conservation priority is often assigned to large, intact areas of forest; areas that are critical for connecting other habitats; areas where rapid land conversion is occurring; areas that represent rare or threatened habitats; and areas that contain high numbers of species, endemic species, or endangered species. Of these, species richness is often the default metric (Prendergast *et al.*, 1999; Raberg & Rudel, 2007). My results show that conservation schemes focusing only on species richness or the area of the forest patch are likely to miss a significant part of the conservation potential in fragmented cloud forest landscapes, where even small, isolated patches of forest make an important contribution to conserving landscape-level tree diversity.

Small reserves conserve biodiversity in fragmented montane landscapes

As conventional conservation based on species-area relationships would predict, the two largest, most remote, and most pristine reserves in this study (Junin and Los Cedros, combined area 8000 ha) also had the most species (MacArthur, 1967). Together, these two reserves contained 182 tree species in only 0.2 ha, representing 61% of the tree species in the dataset. These reserves have considerable conservation value. The Los Cedros reserve alone contains 240 bird species and more than 900 moth species (BirdLife International). Large areas of forest are also needed to conserve the range of species that occur over different elevations in cloud forests, species that must migrate, and to accommodate the range shifts predicted to accompany the rapidly changing climate (Kessler *et al.*, 2001; Lippok *et al.*, 2014). Certain species, especially those sensitive to edge effects, requiring large ranges, or large genetic populations, cannot maintain viable populations in small patches of cloud forest (Kattan *et al.*, 1994; Lippok *et al.*, 2014).

However, the three smaller study reserves (total area 1540 ha) still contained an additional 118 tree species not found in the two largest reserves[12]. Bosque Intag, the second smallest reserve, shared the fewest species with, and had tree communities that were least similar to, any of the other reserves. Even the smallest reserve, Nangulvi, contained 78 species, 35 of which were unique to that reserve, in only 30 ha of forest. Thus, smaller, less specious forest patches are also important for maintaining the landscape-level diversity in cloud forests (Kelly *et*

[12] This number may be an overestimate given that sampling was not comprehensive, although, conversely, it may also be an underestimate for the same reason; accounting for undersampling bias, the Chao index estimates that the maximum number of species shared between these smaller reserves was 30.5 species (less than 12%).

al., 1994) particularly in the highly fragmented landscapes common throughout the Andes (Daugherty, 1973; Schelhas & Greenberg, 1996; Turner & Corlett, 1996; Young, 1998; Jokisch & Lair, 2002; Williams-Linera, 2002; Echeverría *et al.*, 2007).

Conservation areas also conserve global biodiversity by maintaining populations of rare, endemic, or endangered species (Margules & Pressey, 2000). In each reserve, I found threatened species absent from other reserves. Of the 19 species of threatened or endangered trees that I found across the five sites (IUCN, 2013), 12 were found in only one of the five reserves. Although the largest, most remote reserves together contained about 50% of these species, all three smaller reserves each contained unique threatened species and thus contribute to conserving rare, endemic, and endangered species across the landscape.

Restoration and forest patches

Ecological restoration is increasingly partnered with conservation to protect, connect, and expand existing forests (Young, 2000; Higgs, 2003; Young *et al.,* 2005). A key goal of ecological restoration is to assist the "recovery of an ecosystem that has been degraded, damaged or destroyed" (SER, 2004, pg. 4), a goal that requires both knowledge of what species comprised past ecosystems, and access to seed sources from which to propagate species. Because cloud forest tree communities vary so much over small spatial scales, *local* remnant patches are extremely important as both historical reference sites and sources of propagules for active and spontaneous reforestation. Thus, to restore cloud forest landscapes, managers should conserve local remnant forests, and use them in the following ways: first, because planted forests tend to be more homogeneous to one another than primary forests (McKinney & Lockwood, 1999; Rhemtulla *et al.*, 2007) managers should use local remnant patches as seed sources to help maintain historical landscape heterogeneity. Second, because seed dispersal, a crucial component of secondary forest recovery, is severely limited by the distance of the site to sources of seeds and habitat for dispersers (Holl, 1999; Chazdon *et al.*, 2003; Aide *et al.*, 2010; Pena-Domene *et al.*, 2013), even small patches of forests near restoration or protected naturally regenerating sites should be prioritized for conservation. Third, local people should be allowed and encouraged to use remnant forest patches as sources of seeds and propagules for restoration. Having remnant forests nearby can reduce the time and effort required to gather seeds, and, by extension, allow people and agencies to use harvested seeds as opposed to commercially available species. Conserving but providing access to local forest patches is also important in community-based projects, where people may know, use, or prefer to cultivate local species (Schelhas & Greenberg, 1996; Kirby & Potvin, 2007). Finally, restoration efforts could focus on expanding existing forest patches to enhance their conservation value (Porter-Bolland *et al.,* 2012). Fortunately, unlike flat lowland areas (as found in the Brazilian Amazon) where almost all the land in a given area might be cleared and cultivated, in mountains small patches forests often remain on private land in especially steep areas, or along streams and gullies (Keating, 1997; Young, 2009). We just need policies and practices in place to conserve them, and to encourage local people to use the species found in them to restore and expand them.

Conclusion

The previously unstudied upper cloud forests in the Intag Valley show exceptional biodiversity and change over small distances at similar elevations. Comparative studies, like this one, are important to characterize and develop conservation strategies for montane tree species. I found that large patches of forest were more species rich and provided valuable habitat for forest species. However, even small, relatively isolated forest patches contained unique combinations of species. To conserve tree biodiversity in cloud forests we need to both conserve large areas of intact forests where they exist and provide incentives and resources for landholders and communities to conserve and restore remaining forest patches in heavily deforested regions, especially where they represent the last fragments of primary forests.

Table 3.1: Description of the five primary forest reserves studied in the Intag Valley, Ecuador.

ID	*Reserve name*	*Elevation (masl)*	*Lat, Long*	*Reserve size (ha)*	*Surrounding land use*[1]	*Time to road (hours*[2]*)*	*Degree of human influence*[3]	*Total no. tree spp.*	*No. unique tree species*[4]
6	Bosque Intag	1960-2060	0.1264° N, 78.5936 ° W	730	Pasture, Forest, Agriculture	1.25	3	81	55
7	Junin	2040-2150	0.3423° N 78.5641° W	1,500	Forest, Pasture, Agriculture	2.5	1	120	53
8	Los Cedros	1950-2100	0.3279° N 78.7906° W	6,880	Forest, Pasture	5	1	104	46
9	Nangulvi Alto	1980-2000	0.3118° N 78.6538° W	30	Pasture, Agriculture	0.75	3	78	35
10	Santa Lucia	2000 - 2150	0.3549° N 78.4771° W	780	Forest, Pasture, Agriculture,	3.5	2	58	25

[1] Main uses found at the perimeter of the reserve. 'Forest' indicates that the reserve is connected to another reserve. The order of the land uses corresponds to the relative amount of each surrounding each reserve.
[2] Approximate time to walk from the area sampled to the nearest car-passable road.
[3] Degree of human influence is a ranking based on interviews with long-term residents and landowners. 1= sites have not been used in the past; 2= reserves that were potentially used by people but use was either minimal or ceased more than 10 years ago; 3=sites in working agricultural landscapes.
[4] Number of tree species unique to the reserve (not found in other reserves).

Table 3.2: Explanatory variables used in multivariate analyses of primary forest tree communities.

Category	***Variable***	***Units***	***Scale***[1]	***Variable type***	***Analysis***[2]	***NMDS***[3]	***RDA (p value)***[4]
Local topography	Slope	Degrees	Transect	Continuous	NMDS, RDA, VP	0.122, 0.028	0.345
	Aspect	Degrees	Transect	Categorical	VP	NA	NA
Topography and environmental gradients	Elevation	masl	Transect	Continuous	RDA, VP	0.285, -0.166	0.005**
Geographical and spatial	Latitude	Decimal degrees	Transect, Reserve	Continuous	NMDS, RDA, VP	0.083, 0.692	0.005**
	Longitude	Decimal degrees	Transect, Reserve	Continuous	RDA, VP	0.668, -0.235	0.005**
Human influence	Distance to road	Minutes walking	Transect	Continuous	RDA, VP	-0.369, 0.266	0.090
	Historical disturbance	Categorical	Transect	Categorical	NMDS	0.593, -0.351	NA
	Number of timber species	Number species present	Reserve	Continuous	Interp.	NA	NA
Human influence, species pool size	Reserve size	Hectares	Reserve	Continuous	RDA, VP	-0.266, 0.314	0.530
Soil characteristics	Bulk density	g/cm^3	Transect	Continuous	NMDS, RDA	0.425, -0.018	0.010*
	NH_4	ppm	Reserve	Continuous	RDA	-0.279, -0.516	0.005**
	P	ppm	Reserve	Continuous	RDA	0.050, -0.729	0.045*
	K	meq/100ml	Reserve	Continuous	NMDS	0.770, 0.124	0.01
	Ca	meq/100ml	Reserve	Continuous	NMDS	0.645, -0.088	NA
	Mg	meq/100ml	Reserve	Continuous	NMDS	0.361, -0.233	NA
	Sum base	meq/100ml	Reserve	Continuous	NMDS, RDA	0.633, -0.026	0.540
	pH				NMDS, RDA	0.588, -0.192	0.015*
	SOM[5]	%	Reserve	Continuous	NMDS, RDA	-0.129, 0.265	0.190
	Texture	Categorical	Reserve	Continuous	Interp.	NA	NA
	Litter layer depth	cm	Transect	Continuous	NMDS, RDA	-0.570, -0.158	0.495

[1] Reserve - data were collected at the reserve-level; Transect - data were collected at the transect level.
[2] RDA - full and partial redundancy analysis, VP - variance partitioning, and NMDS - nonmetric multidimensional scaling.
[3] NMDS r-values for the first and second axes (first, second). Soil data were entered into NMDS to identify variables that correlated highly. Because concentrations of K, Ca, Mg all correlated highly with each other, I used the sum of the base concentration to represent this cluster of soil variables.
[4] Signficance levels are: *** p < 0.001, ** p < 0.01, *p<0.05
[5] SOM – Soil organic matter.

Table 3.3: Similarity and shared species values for primary forest reserves. The upper diagonal shows the actual number of shared species with the Chao-estimated number of shared species in parentheses. The lower diagonal shows the similarity index (Chao-Jaccard estimator).

Reserve	*6: Bosque Intag*	*7: Junin*	*8: Los Cedros*	*9: Nangulvi*	*10: Santa Lucia*
6: Bosque Intag		10 (20.4)	11 (12.6)	10 (11.9)	5 (5)
7: Junin	0.129		42 (60.7)	29 (63.5)	24 (29.0)
8: Los Cedros	0.168	0.414		23 (73.4)	24 (30.5)
9: Nangulvi	0.148	0.265	0.534		13 (16.5)
10: Santa Lucia	0.099	0.302	0.26	0.195	

Table 3.4: Physical and chemical soil properties in primary forest reserves.

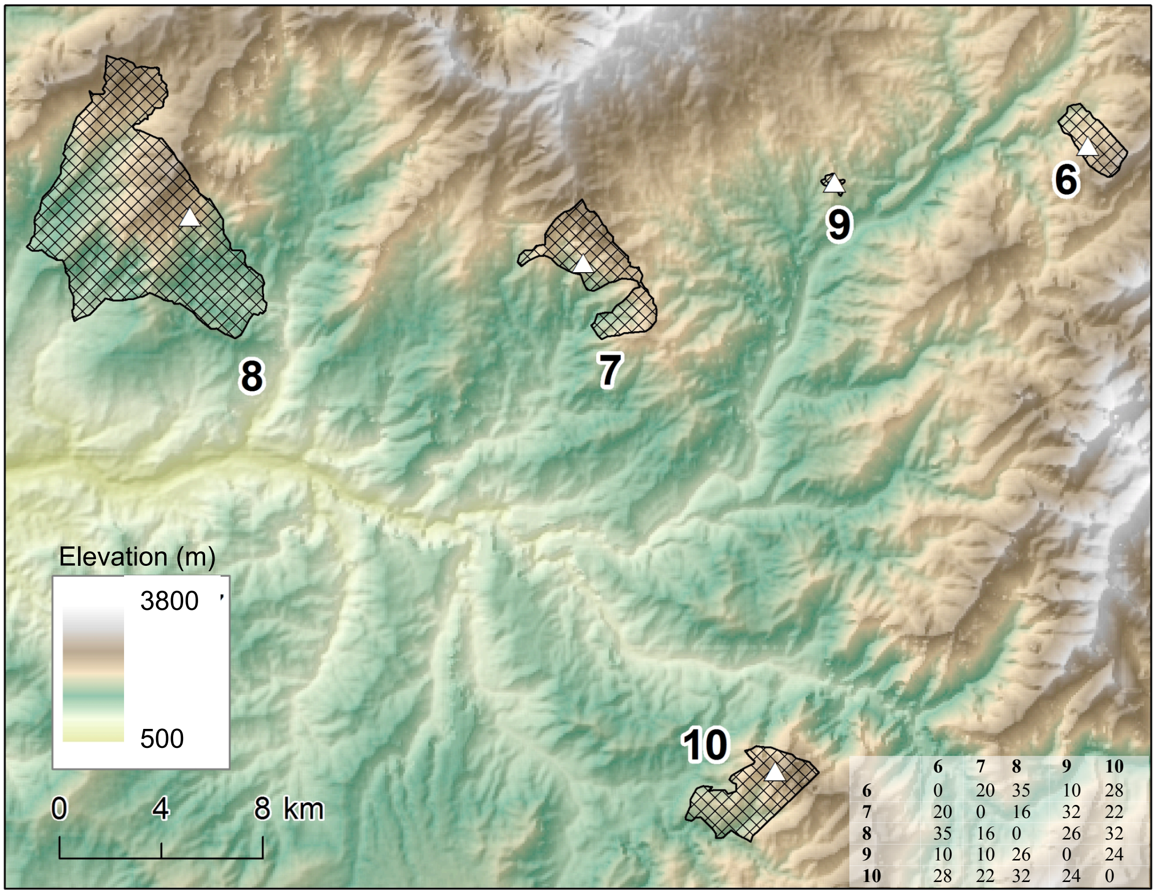
	SITE				
Soil variable	***Bosque Intag***	***Junin***	***Los Cedros***	***Nangulvi***	***Santa Lucia***
pH	6.31	4.0	4.3	6.2	4.6
Bulk density (g/cm^3)	0.615	0.364	0.372	0.580	0.456
Texture class	Organic	Organic	Organic	Franco-sandy	Organic
SOM (%)	26.1	37.2	17.7	19.1	28.4
NH_4 (mg/kg)	18.0	54.5	72.3	30.0	121.0
P (mg/kg)[1]	9.2	9.0	11.5	8.9	25
K (mg/kg)[1]	0.40	0.16	0.10	0.25	0.16
Ca (mg/kg)	17.2	2.6	2.5	14.6	4.9
Mg (mg/kg)	1.85	0.51	0.49	2.4	0.16
Sum bases (meq/100ml)	19.4	7.82	7.2	17.3	8.7

[1] Olsen extraction method

Figure 3.1: The landscape in the Intag Valley.

A) Fragments of forest, both primary and secondary, are located primarily on steep slopes and on ridgetops. Flat areas are used for agriculture, and marginally steep areas for agriculture and pasture.

B) The process of conversion from forest to pasture. Forests to the left of the image are primary or old secondary and forests to the right were likely cleared in the past 20 years. The bright green area is now a pasture (with a slope of over 30 degrees), and the brown area is a recent clear-cut, which will likely be converted to pasture within the next few years. Pastures are often located on extremely steep slopes, like this one.



	6	7	8	9	10
6	0	20	35	10	28
7	20	0	16	32	22
8	35	16	0	26	32
9	10	10	26	0	24
10	28	22	32	24	0

Figure 3.2: Map of primary forest reserves studied. Distance between sites is indicated in the table (km).

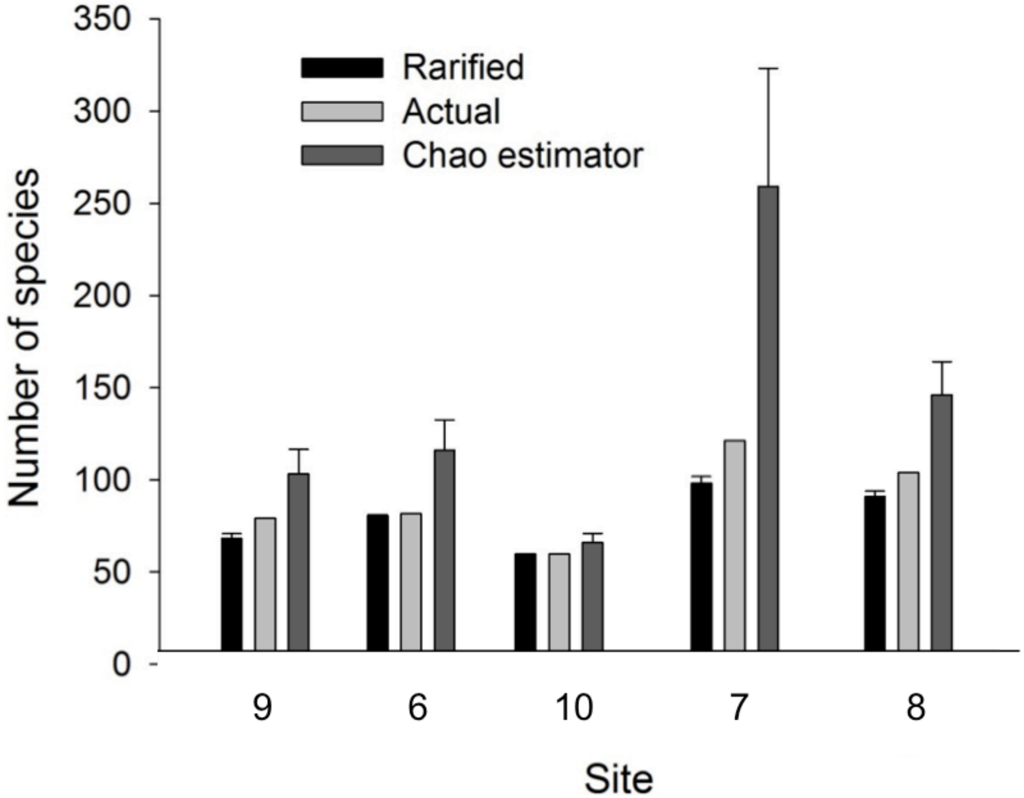


Figure 3.3: Species richness in primary forests as measured by rarefaction, the Chao estimator, and number of species encountered. Reserves are arranged by size (smallest to largest), and correspond to sites: 6=Bosque Intag, 7=Junin, 8=Los Cedros, 9=Nangulvi, 10= Santa Lucia. Error bars represent standard deviation based on 100 randomized runs.

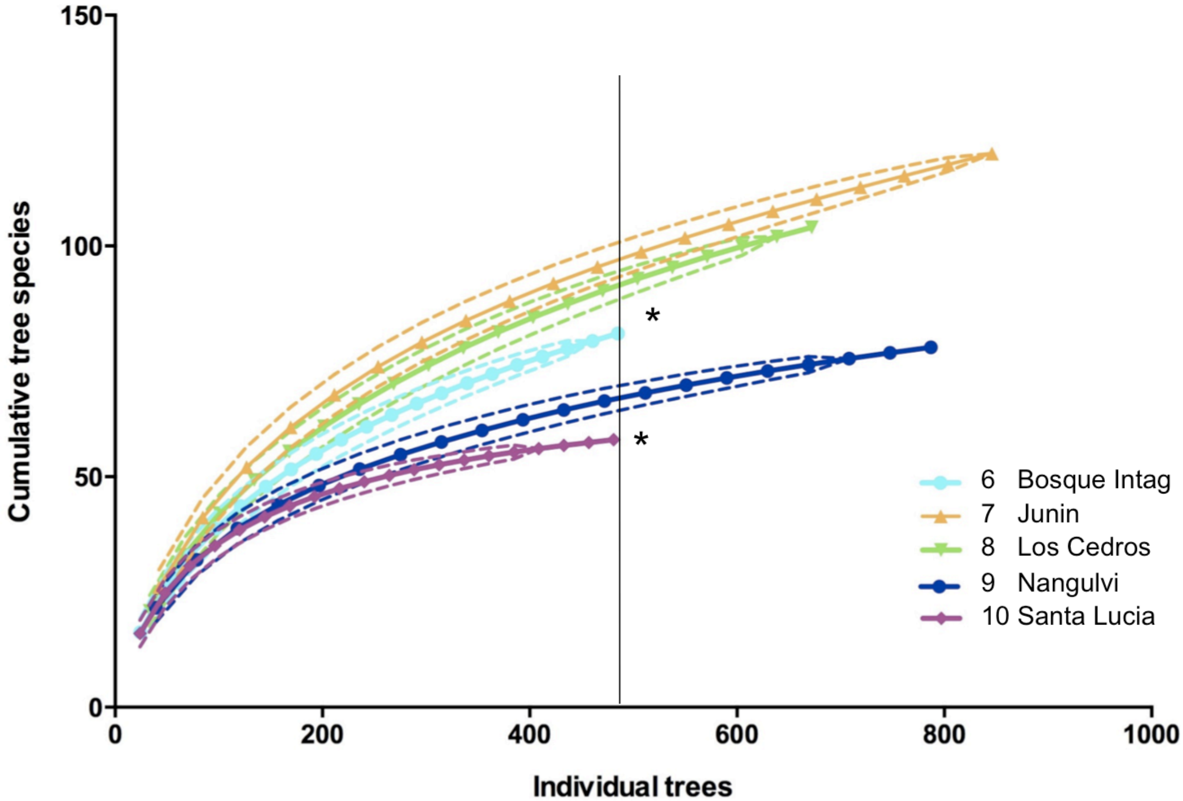


Figure 3.4: Rarified species richness in primary forests. The vertical line represents the highest common number of species, which is why it was chosen as the cut off at which I compared rarefied species richness. Error bars represent standard deviation based on 100 randomized runs.
* represents a significantly different result ($p<0.05$).

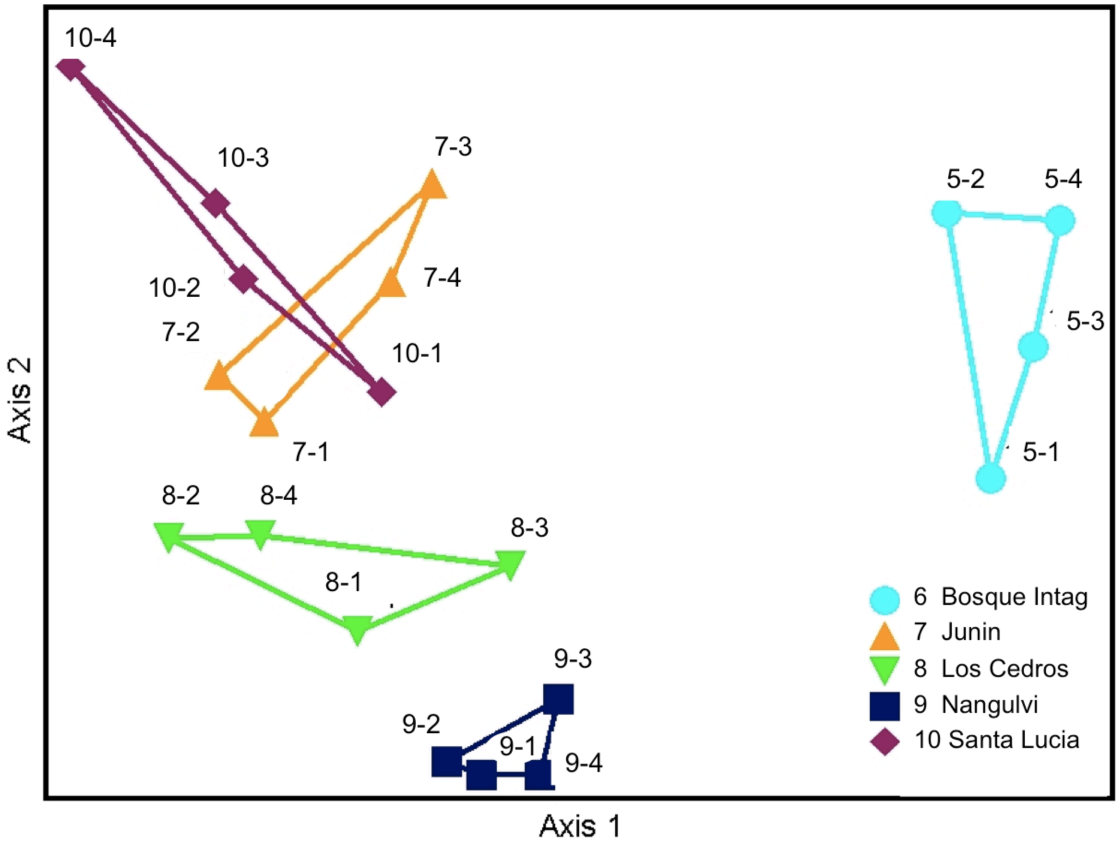


Figure 3.5: Non-metric multidimensional scaling analysis (NMDS) ordination model for primary forests. The first number indicates the site, the second is the transect number. Reserves that share more species (Nangulvi and Los Cedros; Los Cedros and Junin; and Junin and Santa Lucia) tend to group in the ordination. This two-dimensional model explains 53% of the variation in the data with a stress level of 15.08, an acceptable level for ecological studies.

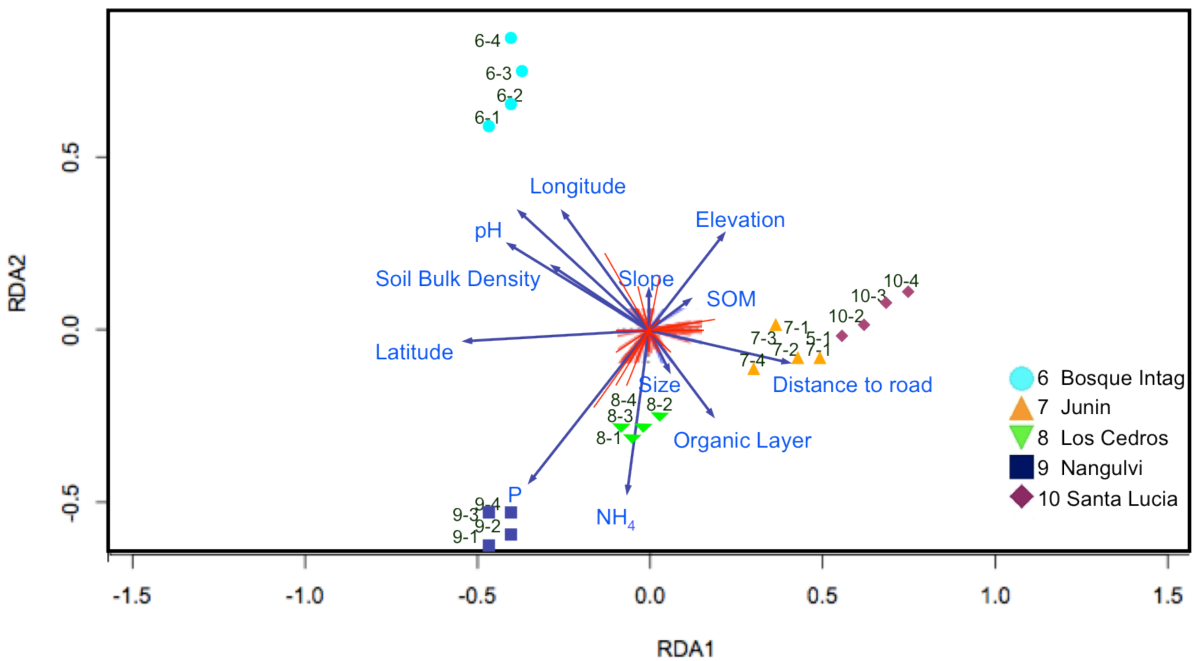


Figure 3.6: Redundancy analysis (RDA) model for primary forests explaining 50% of the variation in the data. Blue arrows represent variables entered, red lines individual species, and points are different transects within each reserve. The number on each point is the site followed by the transect. Variables entered were elevation, latitude and longitude, distance to road, reserve size, slope, and soil characteristics: bulk density, pH, NH4, P, sum bases, SOM (soil organic matter), organic layer depth.

Preface to Chapter 4

In Chapter 3, I characterized the tree communities in the remaining cloud forests in the Intag Valley, and found not only that they are extremely biodiverse, but that species in them change markedly across the landscape. Knowing what these forests are like allows us to tackle the question: Can we rebuild them?

Chapter 4 examines the ability of community-based tree planting to restore these biodiverse forests. I examine how planting trees affects secondary forest tree diversity, and its ability to 'jump-start' successional processes. I also show how the practice of reforesting with certain 'useful' species affect tree species composition and forest succession compared to the primary forests studied in Chapter 3.

The first multi-site study on efficacy of community-based efforts to restore cloud forest biodiversity, my results show that planting locally 'useful' species increased diversity and accelerated forest succession. However, the tree communities remain distinct from those in primary forests, creating 'novel' forests. Thus, restoration does not stand in for conservation, and an increase in secondary forest accompanied by continued forest clearing (Chapter 2) will have an impact on regional forest diversity (Chapter 3). However, restoration produces more diverse forests than natural regeneration, and, because clearing rates slowed as reforestation rates increased, restoring forests is ultimately an improvement over business as usual. The community participation that produced these positive outcomes is examined in Chapter 5.

Chapter 4

Community-based restoration accelerates forest recovery but creates novel forests in Andean Ecuador

Sarah Jane Wilson & Jeanine M. Rhemtulla

In reclaiming and reoccupying lands… the task of the pioneer settler… is to become a co-worker with nature in the reconstruction of the damaged fabric which the negligence or the wantonness of former lodgers has rendered untenantable. He must aid her in reclothing the mountain slopes with forests and vegetable mould, thereby restoring the fountains which she provided to water them; in checking the devastating fury of torrents, and bringing back the surface drainage to its primitive narrow channels; and in drying deadly morasses by opening the natural sluices which have been choked up, and cutting new canals for drawing off their stagnant waters.
– George Perkins Marsh, Restoration of Disturbed Harmonies, 1864

The tree is more than first a seed, then a stem, then a living trunk, and then dead timber. The tree is a slow, enduring force straining to win the sky.
– Antoine de Saint-Exupéry, The Wisdom of the Sands, 1948

Plant a new Truffula. Treat it with care.
Give it clean water. And feed it fresh air.
Grow a forest. Protect it from axes that hack.
Then the Lorax
and all of his friends
may come back.
– Dr. Seuss, The Lorax, 1971

The era of novel forests is upon us and must not be ignored.
– Ariel Lugo, 2009

Abstract

Community-based tropical forest restoration projects, often promoted as a win-win solution for both local communities and the environment, have increased dramatically in number in the past decade. Many such projects are underway in Andean cloud forests, which, given their extremely high biodiversity and history of extensive clearing, are understudied. This paper investigates the efficacy of community-based tree-planting projects to accelerate cloud forest recovery, as compared to allowing forests to regenerate without planting. This study takes place in northwest Andean Ecuador, where only 10% of the original, highly diverse cloud forests remain, in five communities that initiated tree-planting projects to restore forests in 2003. In 2011, I identified tree species along transects in five planted forests, five naturally regenerating forests, and five primary forests. I also surveyed 120 households about their restoration methods and their tree preferences and uses. I found that tree diversity was higher in planted than in unplanted secondary forest, but both were less diverse than primary forests. Ordination analysis showed that all three forests had distinct species compositions, although planted forests shared more species with primary forests than did unplanted forests. Planted forests also contained more animal-dispersed species in both the planted canopy and in the regenerating understory than unplanted forests, and contained the highest proportion of species with use value for local people. Thus, while restoring forest increased biodiversity and accelerated forest recovery, restored forests may also represent ‘novel ecosystems’ that are distinct from the region’s previous ecosystems and, given their usefulness to people, are likely to be more common in the future.

Introduction

In response to rising concerns about biodiversity and species loss in tropical forests, NGOs, governments, and other organizations increasingly encourage local communities to restore forest in degraded tropical landscapes. From an ecological perspective, the aim of tropical forest restoration is often to accelerate or 'jump-start' succession, helping forests recover lost ecosystem functions and species diversity more quickly than if left to regenerate naturally (Lamb *et al.*, 2005; Harris & van Diggelen, 2006). At the same time, development agencies promote community-based restoration to improve local environmental conditions and provide local people with employment and forest resources (UNEP, 2013). The 'win-win' potential of these projects has made community-based restoration projects, in which communities are assumed to have an active role in planning and managing restoration (Charnley & Poe, 2007; Agrawal & Angelsen, 2009), common in many tropical regions (e.g., Kumar *et al.*, 2005). But how well these projects work to achieve ecological goals, including biodiversity conservation, is still poorly understood. This paper asks: can community-based restoration projects work to restore biodiverse cloud forest, or do the trade-offs between social and ecological goals "outweigh the synergies?" (Wunder, 2001; Adams *et al.*, 2004; Chhatre & Agrawal, 2008).

Ecological forest restoration aims to overcome barriers to succession in degraded landscapes, ultimately creating forests that are similar in function, structure, or both, to the primary forests that once occupied the site (Lamb *et al.*, 2005; Chazdon, 2008). Restoration is especially important where natural regeneration is slow or impeded (Chazdon, 2008). The likelihood and speed of natural tropical forest regeneration are affected by: the intensity and duration of previous land use; proximity to mature forests; and the presence in the soil of propagules (e.g., seed banks and sprouting stumps and roots), seed dispersers, and remnant vegetation (Uhl *et al.*, 1988; Parrotta *et al.*, 1997; Harvey, 2000; Holl *et al.*, 2000; Guariguata & Ostertag, 2001; Chazdon *et al.*, 2003; Florentine & Westbrooke, 2004). Although biomass can recover quickly in secondary forests, tree composition may remain distinct for decades; in the southern Atlantic rain forests in Brazil, for example, after fifty years, secondary forests shared only 19 percent of the same species with nearby primary forest (Liebsch *et al.*, 2008; Letcher & Chazdon, 2009; Bonner *et al.*, 2013). Forest recovery is often particularly slow on pastures that have been heavily used for long periods, which now occupy large areas of Latin America (Aide & Cavelier, 1994; Ellis & Ramankutty, 2008), and those forests that do regenerate in these conditions may support different plant species than do primary forest for decades to centuries (Uhl *et al.*, 1988; Lamb *et al.*, 2005; Sampaio *et al.*, 2007; Dent & Joseph Wright, 2009). Planting native trees on such degraded lands can create conditions that allow other species to arrive and establish, creating forests that, at least in the short term, are more species rich than if they were allowed to regenerate naturally (Holl, 2011; Pena-Domene *et al.*, 2013).

Putting local people in charge of managing local resources, including restoring forests, is becoming increasingly common (Lamb *et al.*, 2005; Agrawal *et al.*, 2008). Community-based forest management and restoration is thought to increase the success of conservation projects because local people are in a better position to monitor and manage forests than outsiders (Agrawal & Angelsen, 2009) and stand to benefit from increased forest cover and local forest ecosystem services. But, because social and ecological goals may differ between ecologists and locals and even among local people within the same community (Agrawal & Gibson, 1999; Manzi & Coomes, 2009), the species that people prefer to plant might also differ from those used in ecologically oriented restoration projects. For example, local people may choose to plant exotic species common in cultural landscapes, or select species based on their use value or familiarity (Michon *et al.,* 2007; Garen *et al.*, 2009), thus potentially undermining the ecological integrity of the restoration (Hobbs *et al.*, 2009).

How effectively community-based tree planting helps to restore ecological characteristics in tropical forests is not well studied, especially for montane cloud forests (Aide *et al.*, 2010; Bruijnzeel *et al.*, 2010*b*). Cloud forests are rare, highly biodiverse, and undergoing rapid conversion (Myers *et al.*, 2000; Scatena *et al.*, 2010). Comprising only 1.1% of tropical forests in Latin America, over 50% of montane cloud forest has been cleared, largely for pasture (Gentry, 1989; Henderson *et al.*, 1991; Wassenaar *et al.*, 2007; Mulligan, 2010). These forests contain high numbers of endemic species, account for up to 10% of bird and amphibian species globally, and play a unique role in the hydrological cycle by capturing mist from passing clouds (Stattersfield *et al.*, 1998; Bruijnzeel *et al.*, 2010b). But because of their relatively thin soils, steep slopes, cool temperatures and, in many regions, long history of land use, high degree of fragmentation and conversion to pasture planted with aggressive pasture grasses, cloud forests are particularly slow to recover after clearing (Aide & Cavelier, 1994; Baillie, 1996; Rhoades *et al.*, 1998; Aide *et al.*, 2010; Ortega-Pieck *et al.*, 2011). For example, Aide and Cavelier (1994) found little natural regeneration on pastures planted with exotic grasses 15 years after abandonment. Land abandonment rates are currently high in some Andean regions, providing an opportunity for reforestation on former agricultural lands (Portillo-Quintero *et al.*, 2012). Effective restoration is thus a vital tool to increase forest cover (Sarmiento, 1995*a,b*; Young & Leon, 1995), and many community-based restoration projects are underway in the northern Andes (Maquipucuna Foundation; Rainforest Concern, 2009). Despite this, I am not aware of other multi-site studies that focus explicitly on biodiversity recovery in community-restored forests, nor that compare the diversity of restored forests in Andean montane cloud forest to spontaneous forest regeneration.

Here, I address the question: Does community tree planting accelerate cloud forest recovery? Specifically, compared to naturally regenerating secondary forests:

1) Does planting trees increase tree species richness in secondary forest?

2) Is the species composition of planted forests overall more similar to that of primary forests?

3) Are naturally regenerating (unplanted) understory seedlings and saplings more similar in composition to primary forests (i.e., is there evidence that planting trees is accelerating succession)?

4) What species do people choose to plant, and are these choices compatible with the goal of restoring biodiverse cloud forests?

I address these questions by comparing primary, planted secondary, and unplanted secondary forests in a multi-site study in the cloud forests of Andean Ecuador.

Methods

Study site

My study was conducted in five communally reforested reserves all located between 1800-2250 masl in the Intag Valley, Imbabura province, northwest Andean Ecuador (0.35° N, 78.5° W) (Fig. 4.1). The region is mountainous and steep, with an average annual temperature of 17 to 20°C, annual rainfall of 1500 to 3300 mm, and a pronounced dry season from May/June to October (Freiberg & Freiberg, 2000; Rainforest Concern, 2009).

Northwest Andean Ecuador is part of the 'Tropical Andes' global biodiversity hotspot, with exceptionally high levels of plant endemism (Myers *et al.*, 2000; Sarmiento, 2002) – primary cloud forest in Intag has over 300 species of trees per hectare, or up to 120 species in a 0.1ha plot (A. Mariscal, unpublished data; Chapter 3). These forests were extensively cleared in the 1970s and 1980s to produce timber and charcoal, and for pastures and agriculture (Sarmiento, 2002; Zorrilla, 2010). Currently, less than 10% of the original cloud forest cover remains (Sarmiento, 1995*a,b;* Zorrilla, 2010) and land is primarily used for cattle pastures and subsistence and small-scale, market-oriented agriculture (Kocian *et al.*, 2011; C. Zorrilla, pers. comm. 2011, Wilson, unpublished data). Over 80% of residents are *mestizo* (of mixed indigenous and Spanish decent), with minority populations of indigenous Otovaleños and Afro-Ecuadorians. Individual communities tend to be of mixed ethnicities (D'Amico, 2010; Kocian *et al.*, 2011; S. Wilson, unpublished data). Ninety percent of the population falls below the national poverty line (Kocian *et al.*, 2011).

In the past two decades, international and Ecuadorian environmental NGOs have worked to create a conservation corridor in the area to conserve forests. Their efforts include several community-based restoration projects. In 2001, the local NGO *Defensa y Conservacion Ecologica de Intag* (DECOIN) raised international funding to help communities purchase, reforest, and protect land in their watersheds (DECOIN). My study takes place in five of these reserves in which forests were restored by planting trees beginning in 2003 to 2007. These watershed reserves are an ideal study site because their land-use history is typical of many farmed Andean landscapes: they were cleared 30 to 40 years prior to reforestation, burned, cropped for a few years, and subsequently planted with exotic grasses (usually *Setaria sphacelata,* locally called *pasto cebolla*) and then used as pasture (Table 2). Reserves were also restored using methods that can be easily adopted by local communities and that are commonly used in Latin America (CI; Maquipucuna Foundation; Gandolfi *et al.*, 2007). Technicians trained local people to collect and propagate seeds from native trees in nearby forests, and to plant and maintain these seedlings in combination with planting some non-native species. All reserves were managed similarly. Community members cleared pasture grass around seedlings by hand every three to four months. Grazing animals, harvesting wood for sale, clearing and burning were prohibited within the reserves. In each reserve, there are planted areas and areas that have not been planted but, because of limited funds for planting, were left to regenerate naturally, creating a control site for each restored area (Table 4.1).

Forest plots

I interviewed local landholders to find areas replanted with trees ('planted forest', n=5) and areas allowed to regenerate naturally ('unplanted forest', n=5) with similar land-use history within each reserve (see above). These interviews confirmed that there was no systematic selection bias as to which pasture areas were planted and which were left to regenerate naturally (the choice to plant in only part of each reserve was due to funding constraints). I also sampled trees in primary forests ('primary forest', n=5), located in other reserves in the region (Fig. 4.1, Fig. 4.2). 'Primary' is here defined as forests that have not been cleared in living memory (80 years), are relatively intact, but may have sustained light selective logging. Few relatively undisturbed forests remain at this elevation in Intag – the plots I sampled represent the best examples of older forests in the region (A. Amieda. pers. comm., 2011, C. Zorrilla pers. comm., 2011, J. deCoux pers. comm., 2011). All sampled plots in primary and secondary forests were in the same elevation range (1900-2250 masl).

At each of the 15 sites, I ran four 50 × 5-m transects following slope contours (total 60 transects, 0.1 ha/site). I divided each transect into five 10 × 5m plots. In each plot I counted, identified, and measured trees (>2.5 cm diameter-at-breast-height (DBH)), woody saplings (1-2.5 cm DBH) and seedlings (>0.5 m height, < 1cm DBH). Trees were identified in the field by Ecuadorian botanists, and I took replicate voucher samples to botanists at the *Museo Ecuatoriano de Ciencias Naturales Herbario Nacional del Ecuador* (QCNE) in Quito, who later identified these samples. A local guide provided us with common names when possible. Plants were counted as separate individuals if the stem of the plant was not connected at or just below the soil surface (Chazdon *et al.*, 1998). At two random locations on each transect, I recorded slope, aspect, canopy density, and percent ground cover in two 1-m^2 plots.

Dispersal mechanisms and plant characteristics

Dispersal mechanisms for 'common' species (comprising at least 0.2% of the total number of stems in our dataset) were determined by botanists at QCNE. These data were verified and supplemented with other published sources. If more than one dispersal mechanism was possible, I selected animal over wind or gravity, and relied on information from local botanists over published sources as dispersal mechanisms can be different in different locales (Chazdon *et al.,* 2003). QCNE botanists also recorded the type of forest where each species is typically found (secondary or primary) based on herbarium information and their extensive field experience.

Soils

At each site, I took 10 soil samples from the top 10 cm of soil at two to three randomly located spots on each transect. I created composite soil samples and stored them in a refrigerator until I could deliver it to the soil laboratory (within 1-5 days; *Estacion Experimental "Santa Catalina", Instituto Nacional Autonomo de Investigaciones Agropecuarias,* Cutuglagua, Mejía, Pichincha). Soils were analyzed for macronutrients (nitrogen (NH_4), phosphorus, potassium), organic matter content, and texture (Appendix B). Four bulk density samples were taken using a cylindrical sampler 10 cm in diameter from the top 10 cm of soil at each site (one per transect) and weighed while wet and then periodically after being sun dried, until the dry weight had stabilized.

Anthropogenically useful species

In focus groups (n=6), interviews with local experts (n=10), and household surveys (n=120) I asked residents in four of the five communities restoring forests to identify which trees people use, or would use if available, and for what purpose (e.g., medicine, firewood, live fences, construction). I also asked people what forest products they currently harvest and for what purpose. Finally, I conducted oral histories (n=16) with older community residents about changes in forest cover and use over time.

Analysis

I made rarefaction curves using the program EstimateS (version 8.2.0) (Colwell, 2009) to compare species richness in primary, planted and unplanted forests. The technique resamples data 100 times to create smooth species accumulation curves (showing the additional number of species found as more individuals are sampled) with error bars that can be used to determine whether two sites contain significantly different species counts for a given number of individuals (Chazdon *et al.*, 1998; Colwell, 2009; Colwell *et al.*, 2012). I compared the species richness and species density (cumulative number of species per area sampled) between planted and unplanted forests in each reserve separately, and with all sites of each forest type pooled. I then rarefied species richness to a common number of stems at each site and compared sites using Analysis of Variance tests (Torres-Lezama) (SPSS IMB corp. 2011, Version 20.0). In planted forests, I assessed the correlation between the number of species initially planted and current species richness.

I used Nonmetric Multidimensional Scaling (NMDS) ordination to assess similarity in the species composition (i.e., both the types and relative abundances of species at each site) of trees in primary, planted, and unplanted forests. The Chao-Jaccard similarity estimator was used as a distance measure to help correct for the under-sampling bias often present in extremely biodiverse ecosystems (Chao *et al.*, 2005; Norden *et al.*, 2009). To compare species compositions across forest types and size classes, I used a multi-response permutation procedure (MRPP). To examine successional patterns, I separately ordinated seedlings, saplings and trees at each site (Norden *et al.*, 2009). In addition, to determine if tree species that regenerate under planted trees differ from those that regenerate on unplanted pasture, I removed all planted trees from the dataset and re-ran the diversity and species composition analyses described above.

I included soil data and ground-cover data as explanatory variables in the multivariate analyses, and statistically compared soil data between forest types using ANOVA. I also compared the absolute and relative abundance of anthropogenically useful species in the three forest types using nested ANOVAs. When necessary, variables used in ANOVAs were square-root or natural log transformed before analysis to meet assumptions of normality.

Results

Soils

Planted and unplanted sites had similar chemical and physical soil properties to one another, but primary forest soils were distinct. Soil bulk density was lower in primary forests than in either planted or unplanted secondary forests (*ANOVA*, $F_{2,14},=9.09$, $p = 0.004$, *Bonferroni post-hoc*, $p = 0.005$, $p = 0.013$), which were not significantly different from one another (*Bonferroni post-hoc*, $p = 0.861$). Primary forest soils also contained more organic matter than either planted or unplanted secondary forest (*ANOVA*, $F_{2,14},=7.39$, $p = 0.008$, *Bonferroni post-hoc*, $p = 0.013$, $p = 0.027$). Primary forest soils were generally classified as organic, and secondary forest soils as mineral (*Estacion Experimental Santa Catalina* 2011; Appendix B). Macronutrients did not differ between forest types: nitrogen (NH_4) (*ANOVA*, $F_{2,14}, = 1.003$, $p = 0. 396$) and phosphorus were both similar across forest types (*ANOVA*,

$F_{2,14}$, $=1.52$ $p = 0.254$), although in both cases levels tended to be higher in primary forest soils than in either planted or unplanted secondary soils. Potassium, however, was higher in planted forest soils than in primary forest soils (*ANOVA*, $F_{2,14} = 4.23$, $p = 0.041$, *Bonferroni post-hoc*, $p = 0.861$) (Table 3). Overall there was little evidence of soil recovery in planted sites as compared to unplanted ones.

Tree communities

I identified a total of 6936 individual trees and 416 woody plant species in all 15 sites. The majority of species were trees (345) and woody shrubs (55), with some tree fern species (11), and woody species for which the form was unknown (5). For simplicity, from here on I refer to all species as trees (Norden *et al.*, 2009). Unplanted, planted, and primary forests had a total of 58 (mean per individual site 15.4 species ± standard deviation (SD) 3.8), 129 (44.2 ± 17.4), and 300 (88.2 ± 24.1) species, respectively. In total, people planted 51 different trees species, with between 12 and 33 species planted at each site (Table 4.1).

Stem density & species richness

Mean stem density was not significantly different between primary forest (6590 stems/ha ± SD 1683) and planted forest (5690 ± 3130), but both were significantly higher than unplanted forest (1474 ± 1207) (*ANOVA*, $F_{2,14} = 10.9$, $p = 0.02$, *Boniferroni post-hoc*, $p = 0.009$, $p = 0.737$). Primary forests were more species rich than either type of secondary forest, although species richness in the least rich primary forest was comparable to that of the most rich planted forest (Fig 4.3, Fig 4.4A, 4.4B). Species richness was higher in planted secondary forest than unplanted secondary forests both when sites were compared individually (except for Site 3; Fig 4.3) and pooled (Fig 4.4B). Following the same pattern, species richness rarefied to the same number of stems was significantly different across forest types, with primary forests having the highest (mean 28.0 ± SD 2.7), planted forests the next (18.2 ± 3.8) and unplanted forests having the least (12.6 ± 2.3) number of species (*ANOVA*, $F_{2,14} = 33.2$, $p = 0.000$; *Bonferroni post-hoc, all* $p < 0.04$). Because the absolute stem density of trees was low in unplanted forest, the difference in species density between unplanted (mean 17.6 species/0.1ha ± SD 7.3) and planted forests (44.2 ± 17.4) was even greater than the difference in species richness. Primary forests had the greatest species density (88.0 ± 24.2) (Fig. 4.4C) (*ANOVA*, $F_{2,14,} = p < 0.000$; *Bonferroni post-hoc, all* $p < 0.02$). There was no relationship between the number of species planted initially and species richness ($r_5 = 0.15$, $p=0.81$).

Species composition

In the ordination, sample points (which each represent a size class – seedlings, saplings, trees – at a given site) separated by forest type on axis 1 with unplanted forests on the left, planted in the left-centre, and primary on the right (Fig. 4.5A). Forest type was a highly significant grouping variable (*MRPP*, $T = -4.397$, $p = 0.001$). Communities of seedlings, saplings and trees at the same site tended to be similar: points grouped by site, not size class (*MRPP*, $T = 2.50$, $p = 1.00$). Planted and unplanted sites tended to group by geographical location and/or age: sites 5 and 4, two geographically close, older sites, grouped together, as did the younger and also geographically close sites 2 and 3 (Fig. 4.5A, Fig. 4.1, Table 4.1). The primary forests tended to group secondarily by their geographical location in the study valley (Fig. 4.1) – primary forests sites in the northeast (sites 6 and 9) were in the upper part of the ordination, and the three SW sites (sites 7, 8, 10) in the lower part. Plots in planted forests tended to group more tightly in ordination space than unplanted and primary forests, which implies that species composition in planted forests was more homogenous (Fig. 4.5A).

Common species (≥ 0.2% of overall abundance) accounted for 83% of all trees, including 73% of trees in primary, 91.2% in unplanted, and 90.9% in planted forests. Most species were dispersed by animals (68%) (primarily birds) or wind (27%). Wind-dispersed species were strongly correlated with unplanted forests ($r = -0.873$, Fig. 4.5B), and animal-dispersed species with primary forest ($r = 0.9$, Fig. 4.5C). All planted sites contained at least some animal-dispersed species (mean 50.0%, SD 28, range 21.7-70.1%). Some older unplanted sites also contained animal-dispersed species, but younger unplanted sites contained very few (mean 24.2%, SD 20.0, range 3.1- 43.9%). The majority of species in primary forests were animal-dispersed (87.1%, SD 2.6, range 85.8-91.8%). Both planted and unplanted secondary forests lacked species with high conservation priority. In primary forests, I found 27 species that were on the UN red list for species of high conservation priority. Only two of these were present in planted secondary forests, and none in unplanted.

Secondary forests, especially unplanted sites, were strongly associated with non-native ground covers such as the exotic pasture grass *pasto cebolla* (*Setaria sphacelata*) and bracken fern (*Pteridium aquilinum*), an invasive pan-global species (Schneider, 2004). Planted secondary forests tended to have more bare soil and native grasses, and primary forests tended to have more moss, leaf litter, and native, non-grass herbaceous plants (Fig 4.6).

To further compare the secondary forest types, I ran a NMDS without the primary forest stands (Fig. 4.7). This made it possible to distinguish which species were associated with planted and unplanted secondary forests. Older, planted sites were associated with species that were mostly animal-dispersed from a range of families: *Piper aduncum* (Piperaceae)*, Urera caracasana* (Urticaceae), *Palicourea demissa* (Rubiaceae), *Carica pubescens* (Caricaceae), *Inga* sp. (Fabaceae) and *Croton mutisianus* (Euphorbiaceae, wind-dispersed). In contrast, younger, unplanted forests were mainly characterized by shrubby or small tree species with lightweight wood and wind-dispersed seeds in the Asteracaeae family. *Baccharis latifolia* (Asteraceae) characterized the youngest planted and unplanted sites on the west side of the valley. *B. trinervis* also characterized unplanted sites, especially those on the east side of the valley. Results otherwise were similar to the NMDS with primary forest sites.

Tree recruitment in unplanted and planted secondary forest (excluding planted trees)

To examine how planting trees affects tree recruitment, I compared patterns in naturally regenerating trees in planted and unplanted sites by reanalyzing the data after removing trees planted by people. Twice the number of trees, seedlings and saplings were naturally regenerating in planted sites (1580 individuals) than in unplanted sites (738). Both the density and the diversity of naturally regenerating trees was higher in planted than in unplanted sites (Fig. 4.4D). The species composition of naturally regenerating stems in planted forest was still significantly different from both unplanted and primary forests (*MRPP*, $T = -3.63$, $p = 0.005$), but it was more similar to unplanted forests than when the planted species were included (Fig. 4.5, Fig. 4.8). Three key patterns emerged: 1) *seedlings* and *saplings* in planted forest were most similar to primary forests, and dissimilar to unplanted forests; 2) *trees* in planted forest grouped with seedlings and saplings in unplanted forest; and, 3) trees in unplanted forest were the least similar to primary forests. Size class was not significant (*MRPP, $T = 2.2$, $p = 1.00$*).

In both secondary forest types, the understory also tended to have relatively more animal-dispersed species than the overstory, but planted forests had a much greater proportion of animal-dispersed trees naturally-regenerating than did unplanted forests. Of all naturally regenerating individuals in planted areas, almost half (43.3%) were animal-dispersed species (50.9% were wind-dispersed). In unplanted forests, most individuals were wind dispersed (75.6%) (23.7% of the stems were animal dispersed). Although relatively few of the naturally-regenerating *trees* in planted forests were animal dispersed (23.4%), a much higher proportion (47.6%) of the regenerating seedlings and saplings – the smaller size classes – were animal-dispersed. In unplanted forests, only 8% of the trees in unplanted forests were animal-dispersed, and 27.7% of the seedlings and saplings were animal-dispersed. Overall, this analysis suggests that both types of secondary forests are undergoing succession, but that planted forests are accumulating more animal dispersed species earlier, and at a faster rate, than unplanted forests.

Species people plant and use

Community members preferred to plant tree species: 1) that can be easily propagated and transplanted; 2) that grow quickly; 3) that have cultural significance, such as the wax palm, *Ceroxylon ventricosum,* from which people harvest fronds to celebrate the religious holiday Palm Sunday; and 4) that have local use value (Table 4.2).

People used trees mainly for firewood and timber (44% and 40% of the ‘useful’ species, respectively). Other species were used for fertilizer/nitrogen fixation (20%), food

(16%), livestock fodder (8%), and medicine (8%) (note that seven species were multi-use). Planted forests contained a significantly higher abundance of useful species than both primary and unplanted forests (*ANOVA,* $F_{(2,13)} = 11.7$, $p = 0.02$; *Bonferroni post-hoc,* $p = 0.02, 0.001$) (Fig. 4.9). The *relative* abundance of useful species also differed significantly among forest types (*ANOVA,* $F_{(2,13)} = 6.56$ $p = 0.012$) and was lower in primary forests (3% ± SD 2) than in planted secondary forests (*Bonferroni post-hoc,* $p = 0.011$), but not significantly different in unplanted (21% ± 13) and planted (40% ±18) secondary forests (*Bonferroni post-hoc,* $p = 0.36$) (Fig. 4.9). Planted forests contained proportionately more species that were used for timber, while most useful species in unplanted forests were used exclusively for firewood (*Chi-squared,* $\chi^2 = 264$, $df = 1$, $p < 0.000$).

Discussion

The results suggest that putting restoration projects under community management can fulfill a fundamental goal of ecological forest restoration – to "assist the recovery of an ecosystem that has been degraded, damaged or destroyed" (Clewell *et al.*, 2004 pg. 3). In this study, community managed tree-planting efforts quickly increased the species richness of secondary forests and also 'jump-started' ecological succession, increasing the number of tree species, and in particular animal-dispersed species, in the understory. Early forest recovery was greatly enhanced by local planting efforts.

Planting trees accelerates forest recovery over spontaneous regeneration

Restoration clearly increased forest recovery on pastures in Intag over the time frame studied – after four to seven years, forest recovery in unplanted forests was still minimal in terms of both stem density and species richness. Unplanted forests were composed mainly of wind-dispersed, shrubby trees with soft wood that commonly colonize, and sometimes dominate, Andean pastures (Zahawi & Augspurger, 1999; Posada *et al.*, 2000; Amézquita *et al.*, 2004) and that are functionally and structurally distinct from species that characterize primary cloud forests (Chazdon *et al.*, 2003). Exotic pasture grasses and ferns were abundant in the understory of unplanted sites which, along with a lack of nearby seed sources and animal dispersers, can prevent forests from regenerating (Holl *et al.*, 2000; Griscom *et al.*, 2009; Aide *et al.*, 2010; Ortega-Pieck *et al.*, 2011) and lead to a state of arrested succession (Zahawi & Augspurger, 1999; Posada *et al.*, 2000).

Young planted forests had more trees, more species of trees, and more types of tree species than unplanted forests. Planting trees helped forests recover quickly both by 1) directly reestablishing trees and canopy cover, and 2) improving site conditions to facilitate the recruitment of other tree species in the understory.

People planted a variety of trees, many with specific uses. By selecting for a diversity of specific characteristics – such as durable, fast-growing wood (timber trees), nitrogen-fixation, and fleshy, edible fruits – they created forests that were both more species rich and possessed different functional traits from unplanted forest. In particular, planted forests had more animal-dispersed species in the planted overstory.

Planting trees also facilitated the recruitment of more trees and species of trees in the understory, where fifty percent of the species were animal-dispersed (Pena-Domene *et al.*, 2013). Forests undergoing succession should, in theory, have the same species of seedlings and saplings as adult tree communities in primary forest (Terborgh & Foster, 1996; Norden *et al.*, 2009), a pattern which I observed in planted sites, especially older ones. Although communities of naturally regenerating *trees* were similar to unplanted forests, the *seedlings* and *saplings* in planted forests were distinct from unplanted forests, and more similar to primary forests. Thus, only four to seven years after planting, planted forests have accumulated more and different tree species than unplanted forests, and their species composition is relatively, and increasingly, more similar to primary forests. Other studies have shown similar results in young planted forests, particularly in those planted with animal-dispersed species (Pena-Domene *et al.*, 2013; Jacob, 2014). Previous studies have also shown that naturally regenerating secondary forests retain distinct communities of trees from primary forests for decades, and that initial differences in species compositions can persist over long time periods (Dent & Joseph Wright, 2009; Klanderud *et al.*, 2010; Chai & Tanner, 2011; Martin *et al.*, 2013). Community-based tree planting efforts have the potential to set regenerating forests on a different successional pathway than natural regeneration. Although my study examines young forests, the results indicate that planting forests could have longer-term impacts on forest species composition in this region.

It seems likely that planting trees 'jump-started' this succession in part by attracting seed dispersers. By creating an overstory with many animal-dispersed trees, planting trees could have provided birds and mammals with habitat and food (Holl, 1998; Pena-Domene *et al.*, 2013). In tropical forests, most pioneer species are wind-dispersed, but most tree species in primary tropical forests are animal-dispersed (at our sites, an average of 89%). A lack of seed dispersers is one of the largest barriers to tropical forest recovery, especially in areas where primary forest cover is low (Holl *et al.*, 2000; Chazdon *et al.*, 2003; Vieira *et al.*, 2009; Aide *et al.*, 2010), as was the case at my sites (S. Wilson unpublished data). Ensuring that animal-dispersed species regenerate is thus a priority for restoration (Holl, 1999; Holl *et* *al.*, 2000; Chazdon *et al.*, 2003; Aide *et al.*, 2010).

In addition to increasing dispersal, planting and maintaining planted trees changed local site conditions, allowing a wider range of species (both animal- and wind-dispersed) to establish. Poor site conditions can prevent many tree species from establishing even if they arrive (Holl, 1998; Holl *et al.*, 2000; Chazdon *et al.*, 2003; Aide *et al.*, 2010; Ortega-Pieck *et al.*, 2011). Exotic pasture grasses (*Setaria sphacelata*) and invasive bracken ferns (*Pteridium aquilinum*) dominated the understories of unplanted forests, but were less abundant in the understories of planted forests (Holl *et al.*, 2000). The increase in canopy cover at planted sites could have, in part, shaded out *S. sphacelata*, allowing primary forest tree species to regenerate, as has been observed at other sites (Rhoades *et al.*, 1998). In addition, although some other studies have found that removing grass decreased or did not affect tree survival (Holl, 1999), according to local people in Intag, removing exotic grass appeared to increase tree survival in the initial phases of the project. Once trees were established (2-3 years after planting), even after people stopped clearing understories contained fewer exotic plants. As has been found in past studies over a similar timeframe, planting trees did not affect soil bulk density, organic matter content, or macronutrient content in the short term (Holl & Zahawi, 2014). Soil in secondary forests retained nearly twice the bulk density of primary forest soils. Compacted soils in pastures can prevent trees from establishing and thriving, and are a major barrier to cloud forest regeneration (Kozlowski, 1999; Pedraza & Williams-Linera, 2003; Aide *et al.*, 2010), but in this case planting trees facilitated the establishment of trees without improving soils in the short term.

Planted forests have high use value

Planted forests in this study contained proportionately more species that people use, and more multipurpose and timber species, than unplanted regenerating forest and primary forest. People in Intag reported that prior to restoring forests they had already used several forest trees, but during restoration they learned of and adopted uses for several additional timber, food, fodder, and medicinal species. Planted forests contain more 'useful species,' both because people preferentially planted species that they use, and because the NGO identified uses for other trees that it recommended for restoration, including technical information on how native trees could enhance farming systems or provide timber. This type of knowledge sharing is exactly what many community-based programs aim to foster (Diemont *et al.*, 2011; Uprety *et al.*, 2012). Smallholders around the world plant trees as part of farming and forestry systems (Diemont *et al.*, 2011; Hoch *et al.*, 2012). Tapping into this local knowledge can expand the pool of species that can be found, propagated, and for which growth requirements are known (Suárez *et al.*, 2012).

Project managers and communities used both social and ecological criteria to select tree species for planting. Two exotics were planted for economic reasons: citrus trees (*Citrus* spp.), which produce a highly consumed, locally marketable food; and alders (*Alnus nepalensis*), fast-growing, nitrogen-fixing trees used for lightweight timber, firewood, fence posts, and improving soil fertility. These species were initially included to motivate people to participate because they had prior experience planting them (citrus), knew that they produced relatively quickly (alders), and were known to survive in harsh environments. However, the NGO also required people to grow native species, and maintaining a high ratio of native to exotic trees was a project priority – an example of selecting species for ecological reasons

(Zorrilla, 2010). Many species were planted for both ecological and social reasons – native timber trees were selected over exotic ones, for example. My results show that restoration projects are well suited to, and can benefit greatly from, combining local or traditional ecological and scientific knowledge to select and propagate species (Kirby & Potvin, 2007; De Koning *et al.*, 2011; Diemont *et al.*, 2011; Uprety *et al.*, 2012).

The future of community-planted forests: Restoring forests, or creating novel ones?

In my study, community-based tree-planting projects are working to restore species-rich forests that contain more animal-dispersed species and are more structurally similar to primary forests than unplanted forests are. But, these forests are still not guaranteed to become the forests seen in the past: the species composition of planted and primary forests remained distinct in ways that may persist over time.

My primary forest sites contained 27 tree species on the IUCN Red List (IUCN, 2013); only two of these were found in my planted sites, (intentionally planted because of their conservation status), and none in unplanted forest. This finding may be symptomatic of another pattern that I observed: that planted forests in different sites were more homogenous than either unplanted or primary forests. Homogenization has been observed in secondary forests that have spontaneously regenerated (McKinney & Lockwood, 1999; Holl, 2002; Lugo & Helmer, 2004; Rhemtulla *et al.*, 2007). Rare or endemic species may thus be lost, which is of particular concern in tropical montane cloud forests characterized by high numbers of endemic species and high species turnover on a landscape scale (Bruijnzeel *et al.*, 2010b; Chai & Tanner, 2011; Jost, 2013; Martin *et al.*, 2013). Indeed, my primary forest sites exhibited distinct species compositions, with communities of seedlings, saplings and trees grouping by site, rather than by size class as has been observed in some lowland tropical forests where the composition of tree communities varies less in space (Condit *et al.*, 2002; Norden *et al.*, 2009).

People in these restoration projects planted a total of 50 species, with 12 to 26 species planted at each site. Fifty species is a relatively large number for tropical restoration projects to find and propagate (past projects have typically planted between 5 and 25) (Fang & Peng, 1997; Ruiz-Jaen & Mitchell Aide, 2005; Leopold & Salazar, 2008; Pena-Domene *et al.*, 2013), but is still a small subset of the more than 300 species found in the primary forests in this study. We currently lack knowledge about the many cloud forest tree species (physiology, growth requirements, reproductive cycles, etc.) that would allow us to propagate them. It is thus inevitable that a selection process in which 'useful' species or other species that work well in restoration will be widely used and propagated across the landscape. This species bias could mean that rather than recreating primary forests, restoration may be creating 'novel forests' (Lugo, 2009) that may persist for decades if not centuries to come (Lamb *et al.*, 2005; Dent & Joseph Wright, 2009; Klanderud *et al.*, 2010; Chai & Tanner, 2011; Martin *et al.*, 2013).

The potential to create novel forests through tree planting is not unique to community-based forest restoration. Currently, *any* tropical restoration project is unlikely to have the resources to replant the entire suite of tropical forest species. The use of exotic timber plantations to 'jump-start' forest recovery, for example, is a contentious issue: although trees can often regenerate in plantation canopies, we still do not know the long-term effects of introducing these exotics into native forests (Lugo, 1997, 2009). How acceptable these changes in species composition are will vary with the social and ecological goals of a given project (Higgs, 2003; Holl, 2011). Community-based projects have an advantage here: at least local people can tailor these inevitable differences in biotic communities to meet their needs.

In sum, in this study people are planting species-rich, self-assembling, functional forests, but that, in the short term, are distinct from historical forests. Long-term studies are needed to determine if forests are moving in the direction of a primary forest, or if the presence of planted trees and higher abundance of particular species will alter the species composition of secondary forests permanently, resulting in novel forests.

Implications for restoration management and policy

Policy should encourage projects to propagate and plant species that fulfill dual ecological and social goals. In particular, project managers could promote locally used species that also attract seed-dispersers, such as food trees with large, fleshy fruits or hardwood timber species which are also often animal-dispersed.

Making local communities central to planning, maintaining and protecting planted forests can make projects more relevant to local people and create local stewardship (Agrawal & Angelsen, 2009; Wilson, 2013). I found that the time and effort that local people invested in reforestation – 75 households working up to 50 days/year for one to three years (S.Wilson, unpublished data) – also made people more intent on conserving these sites. When asked if they planned to clear planted forests, people unanimously replied no, often explaining that they were planting trees to create a permanent forest. Local people commonly referred to planted forests as *buen bosque* (good forest), while naturally regenerating areas were not called forests but *chaparro*, or scrubby areas that are commonly cut for agriculture (some farmers even mentioned plans to clear *chaparro* to replace it with planted forests).

Practitioners should also not assume that local people make choices that are optimal for specific environmental conservation goals (Robbins, 2001). If certain threatened or endangered species need to be conserved, this needs to be made an explicit goal of restoration. Projects should also provide funding for enrichment planting with rare or threatened species, because there is no guarantee that these species will reach the sites without human intervention.

Finally, as I have demonstrated here, studying community restoration sites can provide valuable insights on the ecological and social outcomes of restoration that can be applied to improve future restoration projects. In addition to experimental plots studies, researchers can study the field-based 'experiments' inherent in so many restoration projects. Projects provide opportunities for researchers and local people to collect data at several stages: 1) how to propagate different local species in nurseries, 2) which local species grow best under what conditions, and 3) how plant communities reassemble once species are planted in the field (Menninger & Palmer, 2006; Lovett *et al.*, 2007). Indeed, in the course of this research, I encountered local research and monitoring projects in more than a dozen communities, which, were they to be published, would provide valuable data to scholars and practitioners alike from previously unstudied tropical forest regions.

Conclusion

Community-based projects that involve planting trees on abandoned or degraded pastures can help to conserve tropical forest biodiversity. Planting trees rapidly increased the tree species richness and density of secondary forests, and the future of these forests looks bright: in all sites, planting different combinations of useful species facilitated the establishment of animal-dispersed species found in primary forest, ‘jump-starting’ succession. Involving local communities inevitably leads to social-ecological trade-offs, such as planting some exotic species. But there are also powerful synergies: because local people implemented and stand to benefit from the projects, they are also dedicated to conserving planted forests. This research also raises additional questions about the long-term successional trajectories in planted forests stocked with proportionately high numbers of planted species, ‘useful’ or otherwise. Will these forests continue to evolve in the direction of a primary forest, or do these planted areas represent ‘novel forests’ on the landscape? Where on this continuum these forests will eventually lie is a key issue: because of the prevalence of these projects now, the myriad of policies in place that encourage tree planting, and because the need for restoration in cloud forests is certain to increase in the future, we can expect that these forests will only become more common in decades to come.

Table 4.1: Description of planted, unplanted and primary forest sites studied.

ID[1]	*Site*	*Forest type*[2]	*Elev (masl)*	*Land use history*	*Reserve size (ha)*	*Surrounding land use*[3]	*Year planted*	*No. spp. planted*	*No. spp. IDed*[4]
1 UP	Apuela, UP	Unplanted sec.	2000-2050	Cleared 1970s Crops (4 yrs), Pasture (30 yrs)	19	Pasture, secondary forest.	2004	0	10
1 PL	Apuela, PL	Planted sec.	1880-1920	Cleared 1970s Crops (4 yrs), Pasture (30 yrs)	19	Pasture, secondary forest.	2004	12	34
2 UP	El Crystal, UP	Unplanted sec.	2140-2200	Cleared 1970s Crops (4 yrs), Pasture (30 yrs)	25.5	Pasture, crops, sec. forest	2004	0	13
2 PL	El Crystal, PL	Planted sec.	2100-2150	Cleared 1970s Crops (4 yrs), Pasture (30 yrs)	25.5	Pasture, crops, sec. forest	2004	27	35
3 UP	El Paraiso UP	Unplanted, sec.	2000-2100	Cleared 1970s Crops (4 yrs), Pasture (25 yrs)	40	Pasture, crops	2006	0	17
3 PL	El Paraiso PL	Planted sec.	2000-2100	Cleared 1970s Crops (4 yrs), Pasture (25-30 yrs)	40	Pasture, crops	2006	33	37
4 UP	La Esperanza UP	Unplanted, sec.	2170-2190	Cleared 1970s Crops (4 yrs), Pasture rotated with crops (30 yrs)	8.5	Sec. forest, pasture, crops	2003	0	19
4 PL	La Esperanza PL	Planted, sec.	2140-2165	Cleared 1970s Crops (4 yrs), Pasture rotated with crops (30 yrs)	8.5	Sec. forest, pasture, crops	2003	19	40
5 UP	Pueblo Viejo, UP	Unplanted, sec.	2050-2080	Cleared 1970s Crops (4 yrs), Pasture rotated with crops (30 yrs)	13	Sec. forest, pasture, crops	2003	0	29
5 PL	Pueblo Viejo PL	Planted, sec.	2040	Cleared 1970s Crops (4 yrs), Pasture rotated with crops (30 yrs)	13	Sec. forest, pasture, crops	2002	18	75
6	BI	Primary	1960-2060	Primary forest	730	Pasture, sec. and primary forest	na	0	81
7	Junin	Primary	2040-2150	Primary forest	5709	Sec. forest, primary forest, pasture	na	0	120
8	Los Cedros	Primary	1950-2100	Primary forest	6880	Sec. forest, primary forest, pasture	na	0	104
9	Nangulbi Alto	Primary	1980-2000	Primary forest	30	Sec. forest, crops, pasture	na	0	78
10	Santa Lucia	Primary	2000 - 2150	Primary forest	730	Primary forest, sec. forest, pasture	na	0	58

[1]ID is the number of each site used in figures 2 to 6.

[2]Primary forest refers to forest that has not been cleared in the past 80 years or more; sec. is secondary forest.

[3]Surrounding land use refers to areas bordering the reserve, based on personal observation of SJW.

[4]Number of species IDed is the number of species found at each site.

Table 4.2: Species that people use as defined by community members.

Species	***Local name***	***Uses[1]***	***Forest type[2]***	***Notes***
Alnus accuminata	*Aliso*	Timber - soft (fences, furniture, framing); silvopastoral; soil	PL	Non-native, fast growing, easy to propagate, seeds harvested from farms
Amasonia sp.1	Atambo	Firewood	PL, UP	Very soft wood
Baccharis nitida	Chilca	Firewood	PL, UP	Very soft wood
Baccharis latifolia	Chilca	Firewood	PL, UP	Very soft wood
Baccharis sp.	Pichulan	Firewood	PL, UP	Very soft wood
Brunellia cf. acostae	Fresno	Timber	P, PL	Produces seeds often, easy to propagate
Calliandra pittieri	Tura	Firewood, fertilizer (leaves), silvopasture, soil	PL	
Casearia sp.	Pilche, Pilche blanco	Timber		
Cedrilla odorata	Cedro	Timber – hard (furniture, houses)	PL	Seeds brought from outside community
Cinchona sp.	Cascarillo	Medicinal	P, PL	
Clusia alata	Guandera	Firewood	P, PL	Food for birds
Clusia crenata	Guandera	Firewood	P, PL	Food for birds
Ocotea glaucosericea	Laurel	Timber	PL	Food for birds
Croton floccosus	Drago	Timber, medicinal	PL	
Delostoma integrifolium	Dialoman	Firewood, silvopastoral	P, PL, UP	
Erythrina edulis	Poroton	Firewood, food for guinea pigs, fences, edible seeds	PL	
Guaiacum sp.	Guayacan	Timber – hard (used for houses)	PL	
Hyeronima scabrida	Motilon	Timber	PL	
Inga densiflora	Guaba	Food, soil	P, PL	
Juglans neotropica	Nogal	Timber	PL	
Leucaena sp.	Leucaena	Feed for guinea pigs, soil live fences	NA	Non-native, used on farms
Myrcianthes cf. orthostemon	Cungla	Firewood	PL, UP	
Nectandra purpurea	Canelo	Timber - hard	P	
Persea americana	Agucate	Food, firewood	P, PL, UP	Food for birds
Saurauia brachybotrys	Moco	Firewood, edible fruit	P, PL, UP	

Table 4.3: Physical and chemical soil properties in planted, unplanted and primary forests.

Soil variable	**Mean ±SD**			**F (p)**
	Primary	***Planted***	***Unplanted***	
pH	5.05 ± 0.97	6.02 ± 0.16	5.85 ± 0.22	0.44 (0.65)
Bulk density (g/cm^3)	0.47 ± 0.12	0.81 ± 0.22	0.78 ± 0.17	9.09 (0.004)
SOM	25.6 ± 10.7	12.06 ± 3.02	13.66 ± 6.4	7.4 (0.008)
NH_4 (mg/kg)	59.9 ± 38.7	33.6 ± 3.2	39.6 ± 19.1	1.003 (0.40)
P (Olson modified, mg/kg)	12.8 ± 7.0	7.44 ± 3.6	9.82±3.8	1.54 (0.25)
K (Olson modified, mg/kg)	210 ± 110*	488 ± 226	324 ± 129	4.23 (0.41)

N = 5 sites in each treatment. The values given are the mean ± standard deviation. SOM = soil organic matter.
* indicates a significant difference between sites ($p<0.05$)

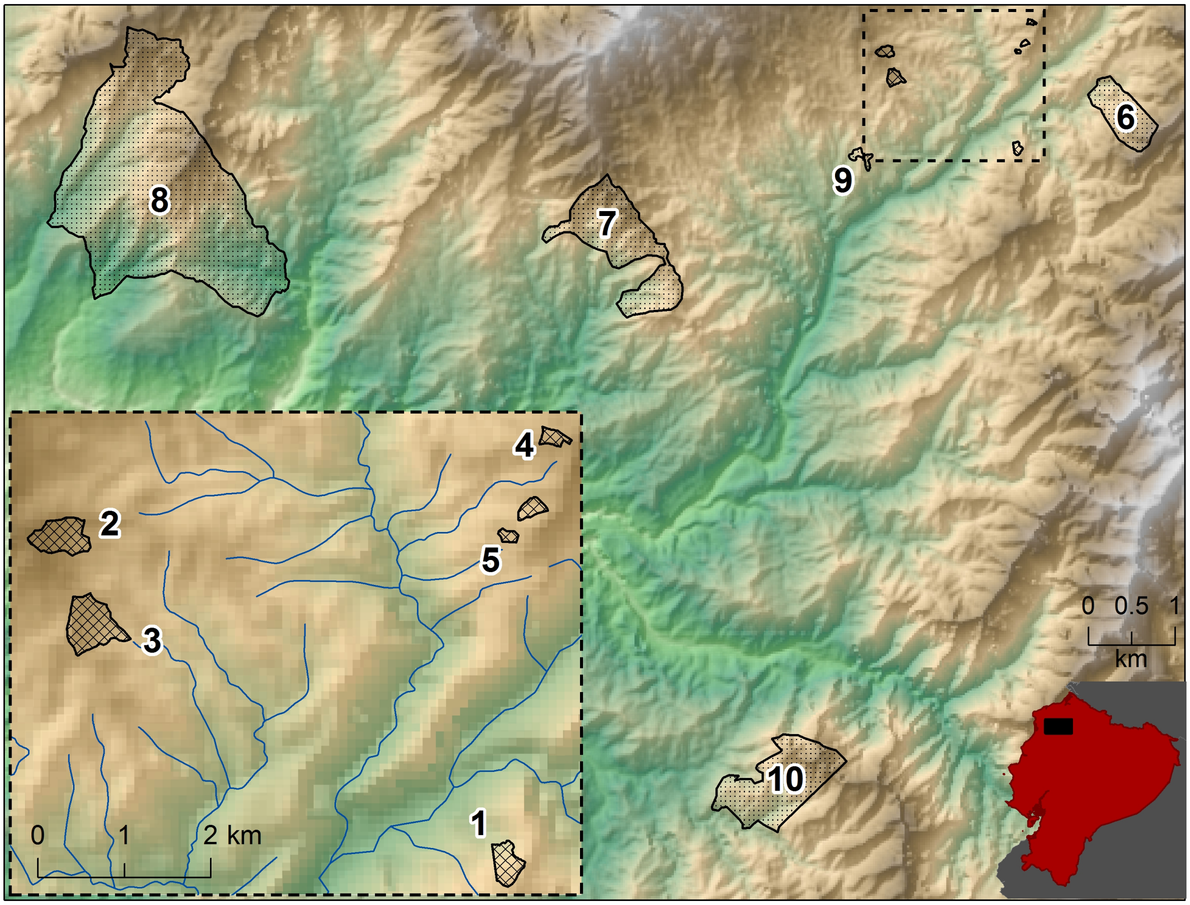


Figure 4.1: Map of study reserves in Intag. Numbers 1-5 are watershed reserves, each of which contains an area of planted forest and an area of unplanted forest. Numbers 6-10 are primary forest reserves.

Figure 4.2: Planted, unplanted and primary forest in Intag.

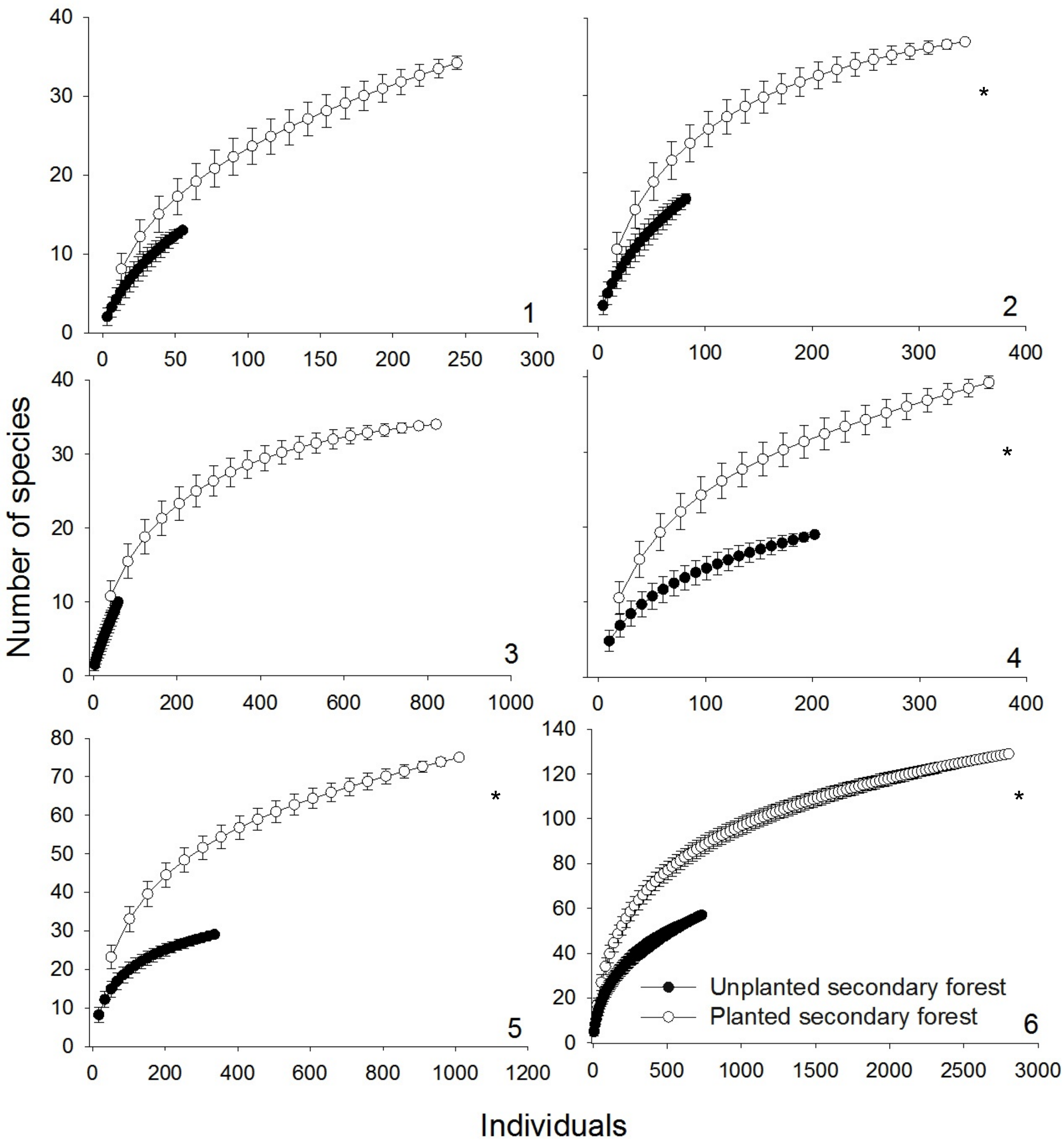


Figure 4.3: Species richness in planted secondary (open circles) and unplanted secondary (filled circles) forest. Numbers 1-5 refers to individual reserves. In 6, all sites are pooled. Error bars represent standard deviation based on 100 randomized runs. Note the different scale on the y and x-axes.
* represents a significantly different result ($p < 0.05$).

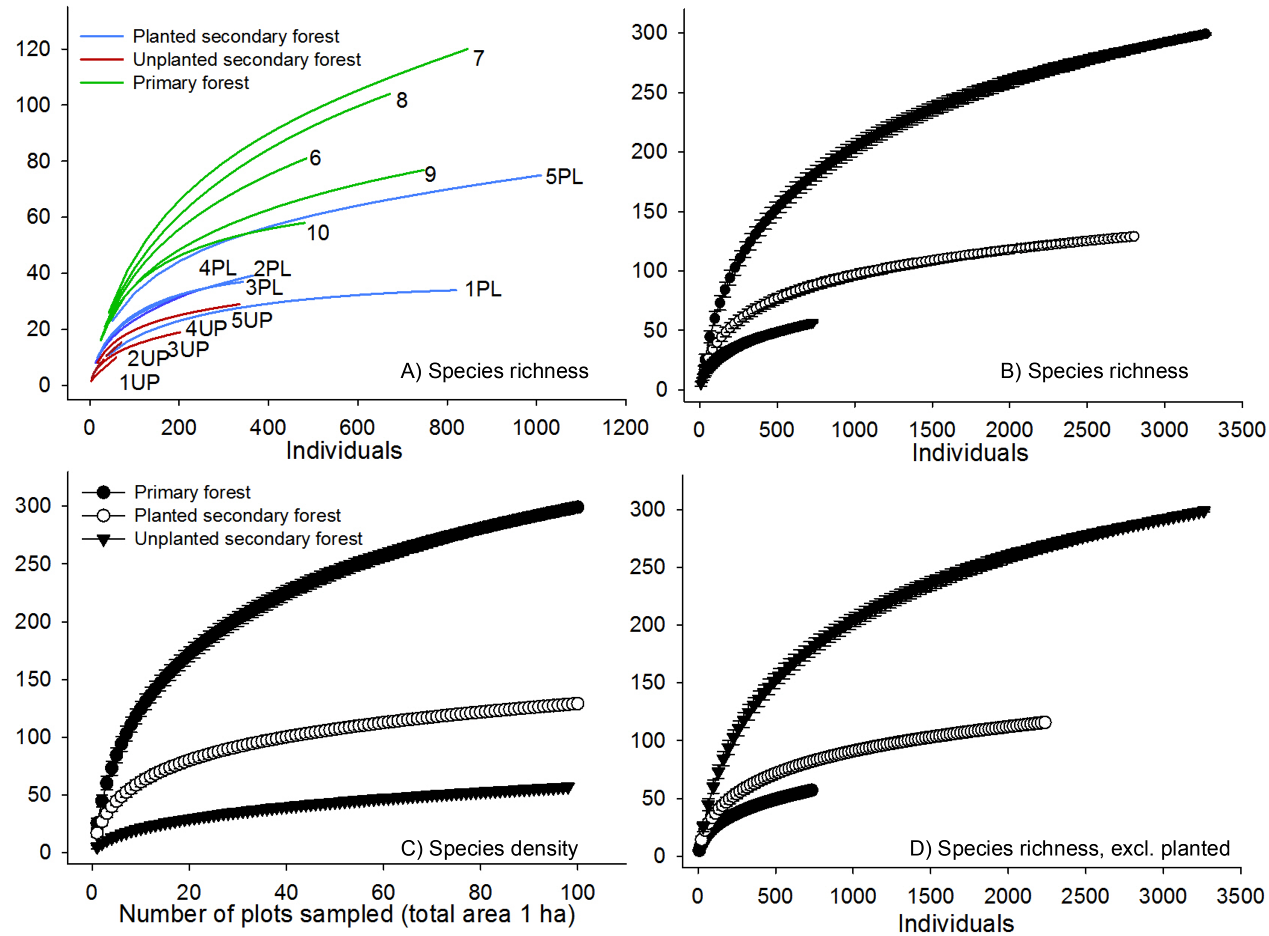


Figure 4.4: Species richness and species density. A) Species richness in individual sites: primary (green), planted secondary (blue, PL), and unplanted secondary (orange, UP). Numbers (1-10) represent different sites. B) Species richness pooled in primary (filled circles), planted secondary (open circles), and unplanted secondary (filled triangles) forest. C) Species density in pooled primary, planted, and unplanted forest. D) Species richness in primary, planted and unplanted forest, with the planted trees excluded. In B), C), and D), error bars represent standard deviation based on 100 randomized runs. Note the different values on the x and y axes.

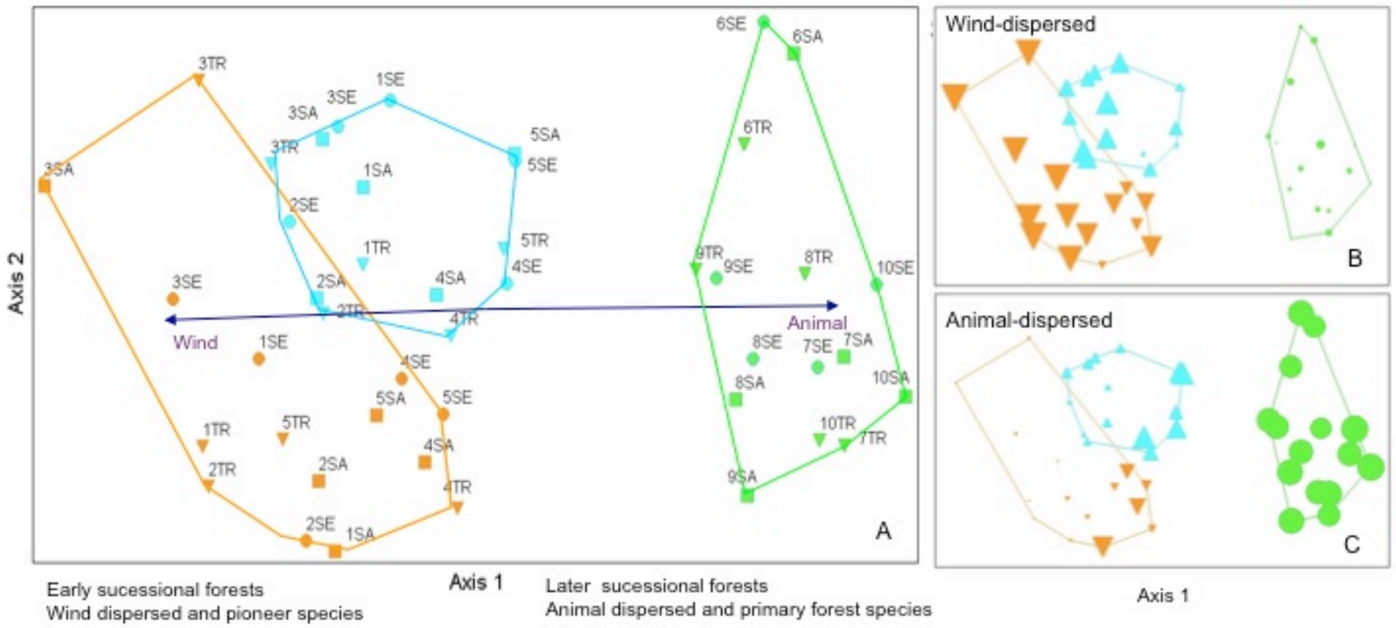


Figure 4.5: NMDS ordination for primary (green symbols), planted (turquoise symbols), and unplanted (orange symbols) forests. Numbers 1-5 are secondary forest sites, 6-10 are primary forest sites. At each site trees were divided by size class. In part A) seedlings are SE, circles, saplings SA, squares and trees TR, triangles. Part A) shows that wind and animal dispersed species clearly separate along the x axis. Part B) shows wind dispersed species and their relative influence in different sites and size classes, as indicated by the size of the symbol for each, and C) shows the same for animal-dispersed species. This two-dimensional model explains 52% of the variation in the data with a stress level of 14.32, an acceptable level for ecological studies.

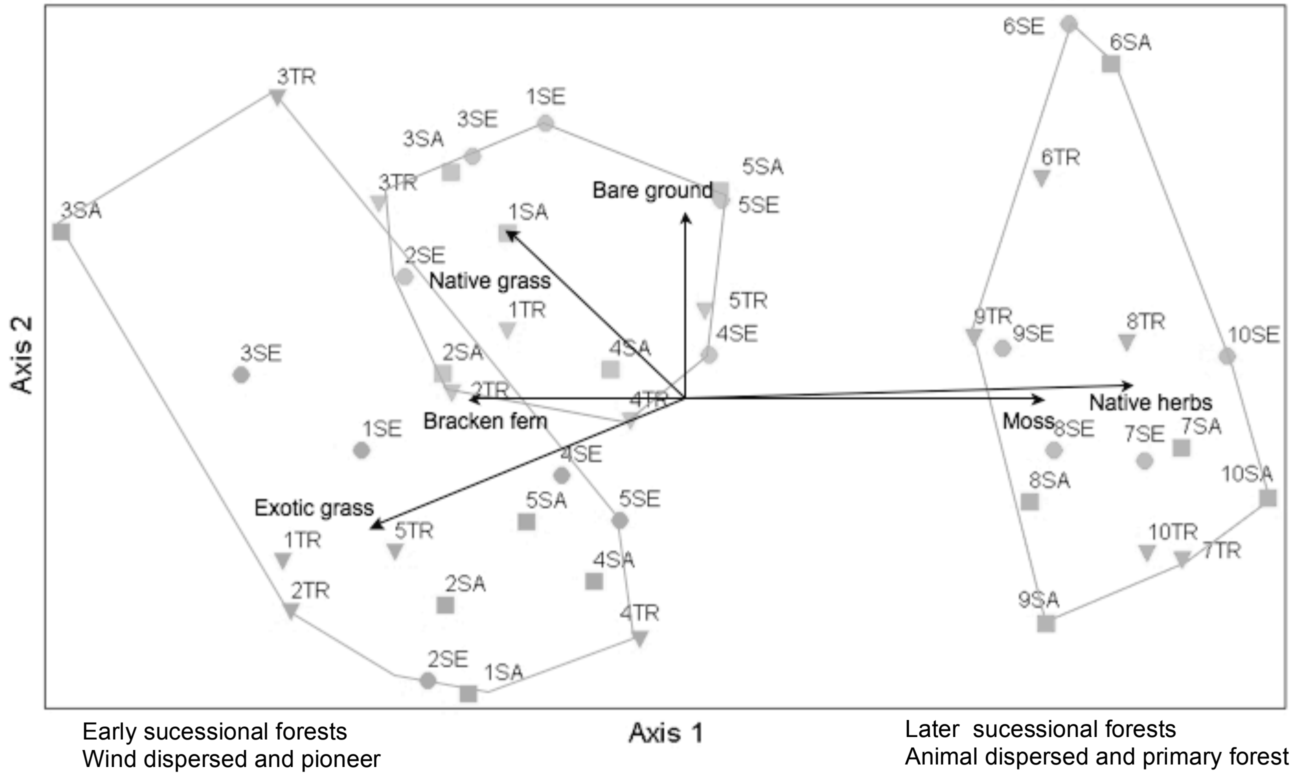


Figure 4.6: NMDS ordination of tree communities in primary forest, unplanted forest, and planted forest with ground cover variables. Numbers 1-5 correspond to different secondary forest reserves (light grey = planted, dark grey unplanted), and 6-10 to different primary forest sites. At each site trees were divided by size class: seedlings are SE, circles, saplings SA, squares and trees TR, triangles. Arrows represent ground cover characteristics that explained more than 25 % of the variation in the data ($r^2 > 0.25$).

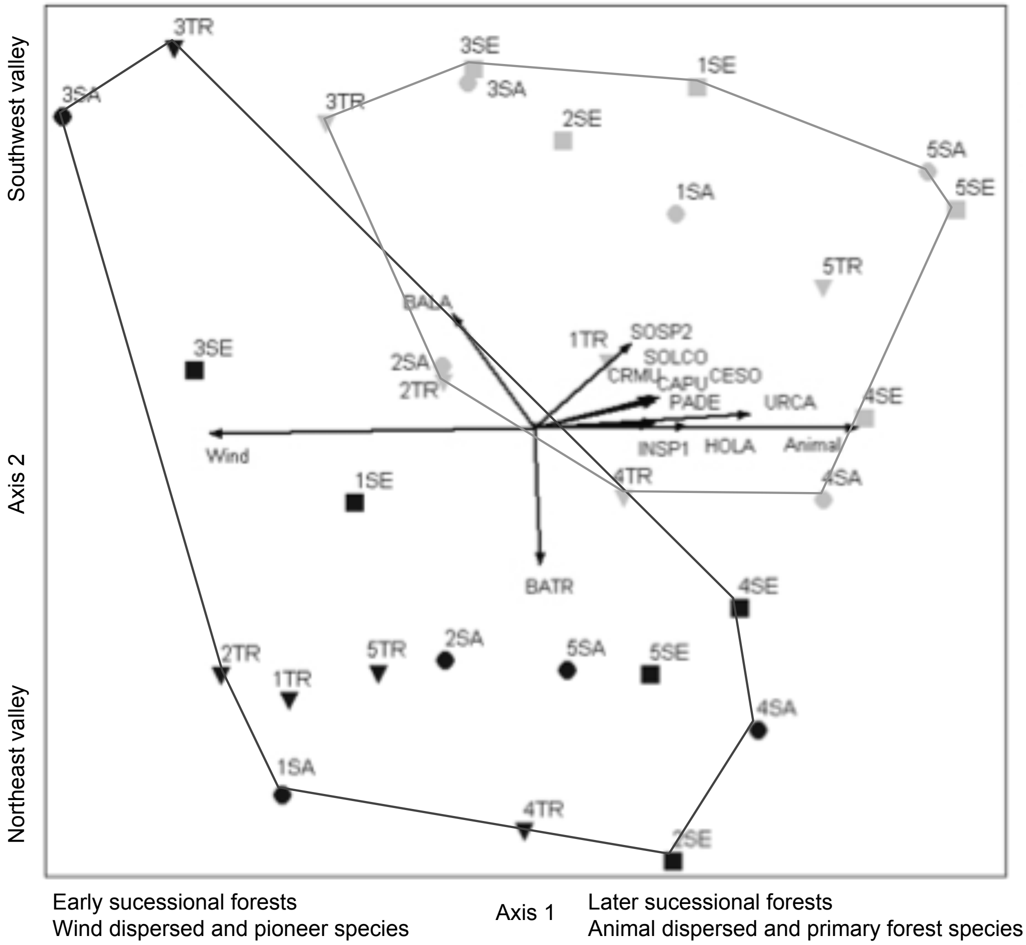


Figure 4.7: NMDS ordination with only secondary forests (planted, grey symbols, and unplanted, black symbols). Numbers 1-5 are secondary forest reserves. At each site trees were divided by size class: seedlings (SE, circles), saplings (SA, squares) and trees (TR, triangles). Species that explain more than 25 % of the variation along either axis (r^2> .25) are also shown. We selected a two dimensional model which explained 64% of the data with a stress level of 18.2. Species codes are: BALA = *Baccharis latifolia*, BATR = *B. trinervis*, CAPU = *Carica pubescens,* CESO = *Centropogon solanifolius,* CRMU = *Croton mutisianus,* HOLA = *Hoffmannia latifolia,* INSP1 = *Inga sp.*, PADE = *Palicourea demissa*, SOLCO = *Solanum confertiseriatum,* SOSP2 = *Solanum sp*, and URCA = *Urera caracasana.*

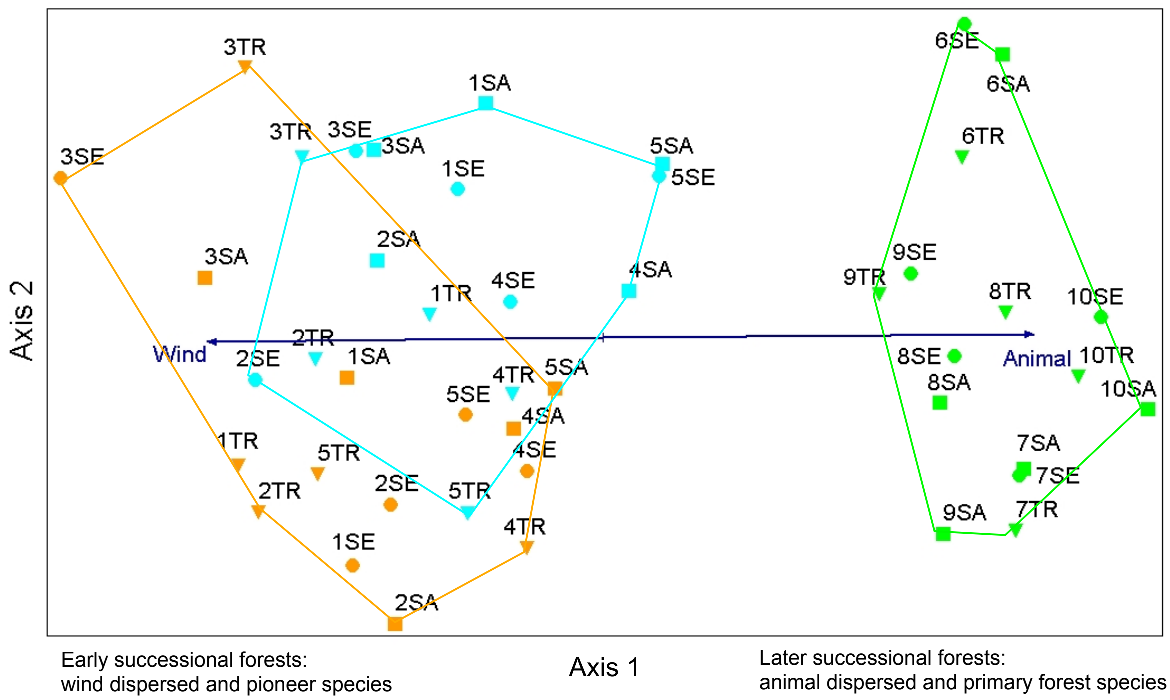


Figure 4.8: NMDS ordination for primary forests (green symbols) unplanted forests (orange symbols), and planted forests with planted trees removed (turquoise symbols). Numbers 1-5 are secondary forest sites, 6-10 are primary forest sites. At each site trees were divided by size class: seedlings (SE, circles), saplings (SA, squares) and trees (TR, triangles). This is a two dimensional model that explains 50% of the variation in the data, with a stress level of 13.43.

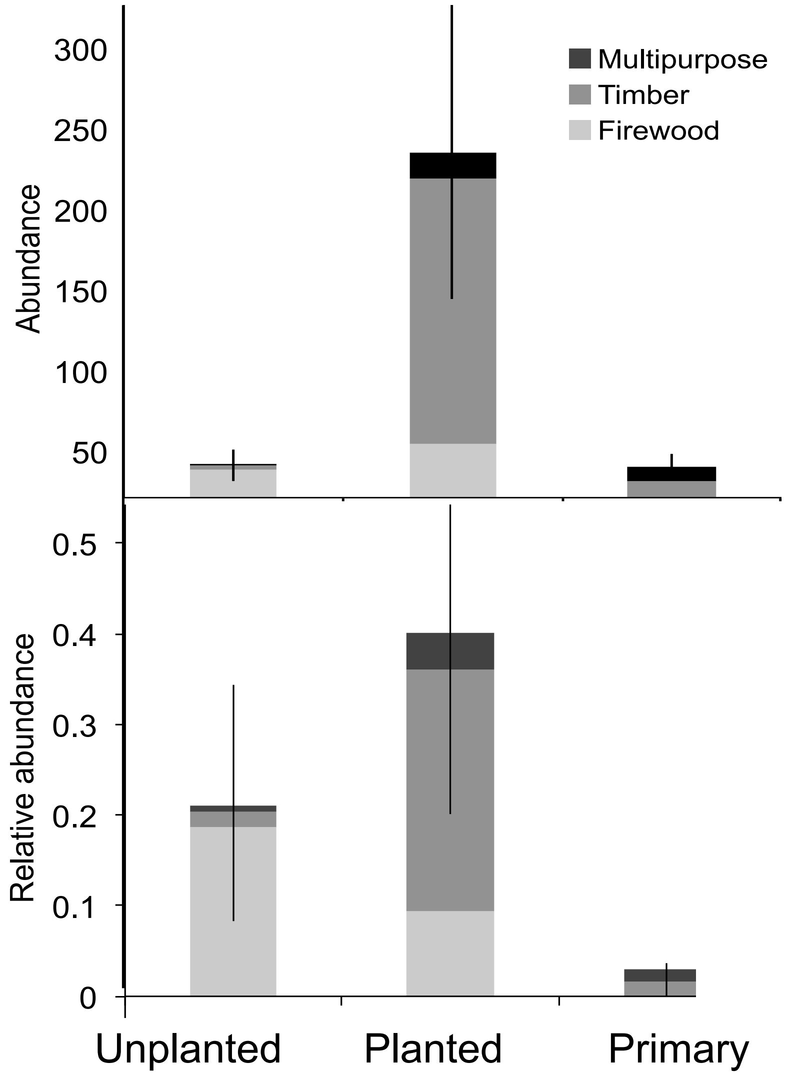


Figure 4.9: The A) abundance and B) relative abundance (with standard error) of species that people use in unplanted, planted and primary forests, divided by use.
* indicates a significant difference between sites ($p < 0.05$)

Preface to Chapter 5

The results from Chapter 4 are promising, showing that community-based tree planting with locally 'useful' species can increase forest diversity and jump-start successional processes. But why do people replant in the first place? In Chapter 5, I study the community-level participation that led to these outcomes, with an eye to encouraging and facilitating community-based restoration efforts elsewhere.

Chapter 5 examines why people participated in tree-planting projects both to restore forests in communal reserves and on their own farms. To my knowledge, it is the first multi-site study to employ a livelihoods approach to examine household-level participation in communal cloud forest restoration, and to compare tree-planting practices on private and communal land. Results concur with the 'ecosystem-service scarcity path' proposed in Chapter 2: people planted trees in response to a local 'environmental crisis' in the region, in which the decline of key ecosystem services (i.e., water) threatened their ability to farm. From an ecological perspective, these heavily deforested Andean regions can also benefit greatly from restoring (Chapter 4). Local actions are also important to conserve biodiversity that varies so greatly across the landscape (Chapter 3). Taken together, findings from Chapters 3, 4, and 5 suggest that community-based tree planting holds great potential as a 'win-win' for forests and people in heavily cleared regions, now prevalent in the Andes.

Chapter 5

Crisis restoration in post-frontier tropical environments: Replanting cloud forests in the Ecuadorian Andes

Sarah Jane Wilson & Oliver T. Coomes

Unless someone like you cares a whole awful lot,
nothing is going to get better. It's not.
– Dr. Seuss, *The Lorax*, 1971

If you want to build a ship,
don't drum up people to collect wood,
and don't assign them tasks and work,
but rather teach them to long for the endless immensity of the sea.
– Antoine de Saint-Exupéry, 1943

Necessity is the mother of invention.
– Old English Proverb

Abstract

Seen as a silver-bullet solution to conserve biodiversity and improve rural livelihoods, community-based tree planting is promoted by conservation and development organizations alike to restore forests. But for such participatory projects to succeed, people must, logically, participate. Given that low participation rates are a major reason integrated conservation-development projects often fall short of their objectives, studying successful examples can provide much-needed insights for future projects. In Chapter 5 I examine why people participate in tree-planting projects to restore forests both in communal reserves and on their own farms. Specifically, who participates, why, and how do their lives and livelihoods benefit? This is the first multi-site study to examine household-level participation in communal forest restoration using an asset-based livelihood approach, and the first to compare tree-planting practices on private and communal land. I interviewed 120 households, conducted oral histories and focus groups, and observed communal workdays. Results concur with the 'ecosystem-service scarcity path' proposed in Chapter 2: people planted trees in response to a local 'environmental crisis' in the region, in which the decline of key ecosystem services (i.e., water) threatened their ability to farm. People planted trees on communal land to restore forests (and ultimately water supply), and on private land to replenish soil fertility and provide other on-farm services. Those who were most involved in their communities, and households with ample household labour, tended to participate more in both types of planting. In particular, farmers (as opposed to wage earners) planted more trees, more kinds of trees, and more native trees, integrating them into farming systems in innovative ways. Although communal restoration projects appear to hold greater potential to restore landscape biodiversity than on-farm planting, people report undertaking both to provide ecosystem services essential to continuing farming. Findings show that communities

experiencing environmental crisis may be most in need of, and thus willing to support and participate in, local tree-planting efforts. Ecologically, these degraded environments are also most in need of restoring – a win-win for forests and people in heavily cleared regions throughout the Andes.

Introduction

Over the past two decades, the conservation community has increasingly recognized that ecological restoration is needed to conserve biodiversity and provide people with vital ecosystem services (Lamb *et al.*, 2005; Young *et al.*, 2005; Chazdon, 2008; DeFries *et al.*, 2012). In particular, restoring tropical *forests* is a global priority because although they house more than 50% of the earth's terrestrial species (Gardner *et al.*, 2009) and more than 800 million people depend on them directly for their livelihoods (Wright, 2010), currently, just over 50% have been cleared (FAO, 2011). Spurred also by efforts to mitigate climate change by sequestering carbon, many governments, conservation and development agencies now promote forest restoration and tree planting to meet both environmental and social goals (Cao, 2011; Ennenbach, 2013; Lall, 2013; UNEP, 2013).

Community-based forest management can be a highly successful means of managing tropical timber extraction and other products (Bray *et al.*, 2006; Persha *et al.*, 2011; Bray, 2013) and could well work for restoration. Rural smallholder communities are especially well situated to restore tropical forests. First, unlike conservation areas which can be (and often are) tucked away in relatively isolated or uninhabited regions (Joppa & Pfaff, 2009), active reforestation[13] will be most needed in agricultural landscapes with a long history of land use (Chazdon, 2003; Lamb *et al.*, 2005), such as the degraded pastures that now occupy large areas of Latin America (Aide & Cavelier, 1994; Ellis & Ramankutty, 2008; Aide *et al.*, 2010). Second, because restoring forests by planting trees is labour intensive and requires locally specific knowledge, it works especially well at small scales (Chokkalingam *et al.*, 2005; Vieira *et al.*, 2009; Hoch *et al.*, 2012). Indeed, some of the most successful restoration projects are carried out by and for local

[13] In this chapter, the terms 'active reforestation' and restoration imply that people are altering the site to promote tree growth by planting and/or tending to trees (Chazdon, 2008).

communities (Higgs, 2003; Egan *et al.*, 2011) which can be 'scaled up' by coordinating efforts at the landscape scale (Chazdon, 2008; Harvey *et al.*, 2008). Third, long-term residents can identify degraded areas most in need of active reforestation (Fairhead & Leach, 1994; Chazdon, 2008), and smallholders may also be able to identify suitable species if they have past experience planting or tending native trees (Altieri, 2004; Hoch *et al.*, 2012; Suarez *et al.*, 2012). Finally, local residents are ideally placed to monitor and protect restored areas and provide day-to-day maintenance (Charnley & Poe, 2007).

Involving local people in planning and executing restoration projects can also foster long-term stewardship (Leopold, 1949; Higgs, 2003; Egan *et al.*, 2011). Contributing the intensive labour required to plant and maintain trees can connect people to restored environments, creating a sense of collective responsibility (Higgs, 2003; Chokkalingam *et al.*, 2005; Vieira *et al.*, 2009; Hoch *et al.*, 2012; Holmes & Potvin, 2014).

However, for community-based restoration projects to succeed, smallholders need to participate in the first place. Often, participation rates in on-farm tree planting projects are low (e.g., many studies report rates of between 20 and 45%) (Fujisaka & White, 1998; Mercer, 2004; Piotto *et al.*, 2004*a*; Cochran & Bonnell, 2006; McGinty *et al.*, 2008; Gamboa *et al.*, 2010; Aguilar-Støen *et al.*, 2011; Frey *et al.*, 2012; Ramírez *et al.*, 2012). A key challenge for smallholder-oriented restoration programs, as with other integrated conservation-development projects, is making projects both attractive and accessible to people in a given context (Current *et al.*, 1995; Mercer, 2004; Pagiola *et al.*, 2005; Zanella *et al.*, 2014). Designing successful projects thus requires in-depth knowledge about which households participate in restoration and why they choose to do (Mercer, 2004; Manzi & Coomes, 2009; Brandt *et al.*, 2013).

Governments, agencies, and donors often try to engage smallholders in tree planting by promising them benefits, now or later. In so doing, they assume people need to be enticed to plant trees, even though in many places smallholders do plant trees of their own accord (Smith *et al.*, 1996; Sears *et al.*, 2007; Hoch *et al.*, 2009, 2012). Two widely used strategies are: 1) paying households to plant trees through payment for environmental service (PES) schemes (including for carbon sequestration) (Grieg-Gran *et al.*, 2005; Wunder, 2005; Zanella *et al.*, 2014); and 2) encouraging farmers to adopt agroforestry systems that promise to improve, expand, or diversify production, providing direct economic benefits (Mercer, 2004). PES schemes focus on what planting trees can give people *outside* the farm, compensating smallholders for the opportunity cost of using land for other productive activities and paying them for the positive externalities gained from planting trees[14] (Wunder, 2005, 2013). On the other hand, agroforestry systems are often presented as 'new' technologies that will improve on-farm production. Studies on why farmers adopt new farming technologies (such as seed varieties, tilling techniques, or fertilizers) assume that farmers will only use the new technology if it increases farming efficiency (i.e., decreases the inputs needed to produce the same yield, or increases yields for the same inputs) (Ellis, 1993). Thus, they assume that households will only adopt agroforestry if they see it as profitable. Agroforestry is thus geared toward providing direct benefits to smallholder farmers by increasing provisioning ecosystem services: higher yields or new products (Current *et al.*, 1995; Mercer, 2004).

Ecological restoration is a somewhat different endeavor. Like agroforestry, it can provide smallholders with ecosystem services (Harvey *et al.*, 2008; Vieira *et al.*, 2009; Mansourian & Vallauri, 2014), but because the focus on restoring forest ecosystems these are more likely to be

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[14] Positive externalities are environmental outcomes of on-farm actions that benefit others besides the smallholder, such as the increased carbon sequestration or the water-purification capacity of reforested lands (Wunder, 2005, 2013).

shared, regulating services, such as moderating stream flow or preventing soil erosion (Myers, 1997; Chazdon, 2008; Mansourian & Vallauri, 2014). Because these benefits are dispersed, harder to quantify, and perhaps less obvious than those from PES or agroforestry schemes, where incentives are clear and benefits flow directly to landholders, restoration is, in theory, especially well-suited to communal land (Netting, 1972*a*, 1976; Bodin *et al.*, 2014), although it is also practiced on private farms (Rodrigues *et al*., 2007). Despite these differences, their shared focus on tree planting and smallholder decision-making at the household level means that insights from the agroforestry and PES adoption literature could help inform restoration efforts in tropical regions. We may not need to reinvent the wheel to identify households likely to participate in restoration-oriented tree planting.

In this study, I use a livelihoods approach to examine why smallholder farmers in the Andes choose to participate in community-based tree planting to restore forests. Although restoration can occur through different interventions (including fire suppression, soil remediation, etc.)(Chazdon, 2008), I focus on planting trees, a widespread, effective restoration method commonly promoted by agencies and donors (UNEP, 2013). I first provide a brief overview of the agroforestry adoption literature, and discuss these findings in the context of restoring forests on communal and private land. Working with four communities in the Intag region in northwest Andean Ecuador involved in a locally initiated, internationally funded restoration project since 2002, I use a livelihoods framework to characterize and compare the smallholder households planting trees in communal reserves and on private farmland. I examine the ecological outcomes of different types of planting, the context in which restoration occurs, and the potential for community-based projects to act as demonstration sites to spur tree planting on private land. Specifically, I address the following questions:

1) Which households participate in planting and maintaining trees in communal restoration projects, and which households plant trees on private land?

2) Do the same household characteristics predict participation in both restoration and on-farm planting?

3) Does participating in restoration or on-farm planting make households more likely to participate in the other?

4) Can we identify 'conservationist households'? Do people who participate in tree planting also adopt other green innovations on their farms?

Household-level participation in planting trees

To design restoration projects that maximize the synergies and manage the trade-offs between conserving and restoring biodiversity and enhancing, improving, or at least maintaining, people's livelihoods, examining who plants, how they plant, why they plant, and how these practices vary depending on land tenure and local context is key. Below, I present a summary of what we know so far from the agroforestry and PES literature, and the relevance of this information to restoration.

Findings from agroforestry adoption studies are often context-specific, with landholdings, education levels, wealth, and on-farm biophysical conditions showing various degrees of significance in different regions and contexts (Pattanayak *et al.*, 2003; Mercer, 2004). But quite often, it is wealthier farmers with more land and secure land tenure who plant on-farm trees

(Bannister & Nair, 2003; Pattanayak *et al.*, 2003; Sood & Mitchell, 2009; Blinn *et al.*, 2013). The same is generally true of participation in PES schemes (Zbinden & Lee, 2005) unless targeting poorer households is made an explicit goal of such projects (Grieg-Gran *et al.*, 2005). Farm size is especially important because larger landholders may not have the labour or need to cultivate all of their land, and can devote some to tree plantations or forests (Grieg-Gran *et al.*, 2005; Zbinden & Lee, 2005; Duke *et al.*, 2014). Working with larger landholders can also mean lower transaction costs: projects can access more land with fewer administrative costs. People who apply high discount rates in resource-use decisions are also less likely to adopt, as are asset-poor households: more vulnerable to risk, they may also simply lack resources to invest at all (Reardon & Vosti, 1995; Bannister & Nair, 2003; Mercer, 2004).

Experience and education also matter. In some cases, better-educated people are more likely to plant trees, as are people with access to extension agents, training, or who are members of forest management groups (Pattanayak *et al.*, 2003; McGinty *et al.*, 2008; Cole, 2010; Pompeu *et al.*, 2012(Frey et al. 2011, Ramirez *et al.* 2011). Farmers with prior experience working with trees, adopting other novel on-farm technologies, and working with extension agents are also sometimes more likely to participate (Walters, 1999). Engaging stakeholders in project design and management, and giving them tree planting choice and options, can also increase participation by creating clear objectives and building trust (Zanella *et al.*, 2014).

The agroforestry and PES adoption literature focus on smallholder planting on-farms. However, restoration can, in theory, be undertaken anywhere that biophysical conditions will permit. Unlike forest conservation, which must happen in and around existing forests, people can decide where to restore forests – on farms, or in communally managed areas. On-farm restoration is attractive because the incentives are clear – benefits flow directly to land owners, avoiding the problems of collective ownership and management (Engel *et al.*, 2008; Andersson & Agrawal, 2011), and people can adapt it to best suits their needs, choosing species and integrating trees into farming systems. On-farm restoration also eliminates the problem of displacing people to create communal reserves (West *et al.*, 2006*b*). On the other hand, restoration is well suited to communal lands: because benefits are shared between members of the community, are higher with more land dedicated to forest, and have low potential gains per unit area relative to crops, restoration may not be a worthwhile investment for a single landholder, but makes economic sense as a community (Netting, 1972*a*). Lands can be strategically selected in specific locations to maximize benefits for people or ecosystems (e.g., in watersheds). Pooling resources such as labour, experience, knowledge and funds in communal projects can also create lower-risk conditions in which people can experiment with different kinds of trees (Bunce & West 1994; Sveiby, 2001; Collins & Smith, 2006; Francisco, 2010). Communal projects are also attractive to donors and agencies because, compared to working with individual farmers, they have lower transaction costs, are easier to monitor, and have higher accountability (Agrawal *et al.*, 2008; Larson & Soto, 2008), and having group control over land titles can also safeguard the land from being sold and cleared. Finally, communal restoration can build community. Planting trees to build forests requires collective decision-making and trust, and can connect people to place (Higgs, 2003; Egan *et al.*, 2011; Zanella *et al.*, 2014).

So, who restores forests on communal lands, and why? Do the same people also choose to plant on private land? Despite the potential of restoration to achieve these objectives on both communal and private land, these questions remain largely uninvestigated. Their answers will help managers and practitioners target and design restoration projects that are appealing and beneficial to smallholder communities in deforested Andean landscapes.

Study area: *The Intag Valley*

The Intag Valley is a rural farming region in the *canton* (county) of Cotacahi in the province of Imbabura, Ecuador. Located on the western flanks of the Andes, the Intag region is mountainous and steep, ranging in elevation from 650 to nearly 4000 masl (Kocian *et al.*, 2011). Despite its proximity to the capital, Quito (only 50 km north as the crow flies), Intag is also quite remote – the 64-kilometre trip linking Apuela, Intag's main market town, with Otavalo, the closest major market town in the central Andeans, takes between 2.5 and 4.5 hours by bus, and is sometimes impassible during the rainy season (November through April). Annual rainfall varies between 1500 and 3300 mm, with a pronounced dry season from May through October.

The roughly 1600 inhabitants of the Intag region live in 76 communities, located in different seven parishes (Kocian *et al.*, 2011). Most people in the region (about 90%) own some land that they farm, and the average farm size is approximately eight hectares (Kocian *et al.*, 2011). About 17% of the residents are illiterate, by far the highest levels in the province (INEC, 2010). Although children now often attend high school, 89% of the household heads in my study communities had only attended elementary school (typically for three to six years). Apuela is the main market town in the region – crops and agricultural goods are either sold here, or to intermediaries who pass through communities, transporting them to larger market towns (Otavalo and Ibarra).

Historically, Intag was almost completely covered in dense cloud forest. Forests here are extremely high in endemic plant and animal species, making them one of the most biodiverse ecosystems on the planet (Gentry, 1992; Myers *et al.*, 2000). In pre-Columbian times, the area was occupied by the pre-Incan Yumbos people (Costales Samaniego & Costales Peñaherrera, 2002). Following centuries of sparse to no habitation, the area was most recently settled about

150 years ago (Kocian *et al.*, 2011). Deforestation rates accelerated in the 1970s and remained high into the 1980s and early 1990s (Sierra & Stallings, 1998; Kocian *et al.*, 2011; C. Zorrilla pers. comm. 2010; Chapter 2). Although people harvest forest trees to build houses and for firewood, the primary reason for clearing forests is and was historically small-scale farming and beef production. Today, approximately 90% of the forests in the region have been cleared.

In the 1990s, people began to experience environmental problems associated with deforestation in their watersheds (Knee & Encalada, 2012). Residents reported that water was contaminated, making children sick. Communities experienced increased flooding and reduced summer streamflow. Local residents recalled having to rise as early as 1 a.m. to draw water from the stream before it ran dry for the day. These realized environmental problems were compounded by impending ones from outside the region. In the 1990s, the gold mining company, Bishi Metals, began exploration operations in the region. Local resistance built against mining operations until, in 1997, a group of grassroots organizations and leaders rallied citizen together to protest, burning down the mining exploration camp and ultimately forcing the company to leave (Bebbington *et al.*, 2008). Combined with declining environmental conditions, mining was seen as a threat to farming in the region because many smallholders would have to sell their land, and some were also aware of how mining pollutes the environment. In response to these concerns, in 2000 local NGO *Defensa y Conservacion Ecologica de Intag* (DECOIN, 2010) helped the communities purchase, protect, and reforest land in their watersheds (Bebbington *et al.*, 2008; Kocian *et al.*, 2011; C. Zorrilla pers. comm., 2010). An environmental organization, DECOIN was founded in 1995 by long-term resident Carlos Zorrilla (originally from Cuba, Zorrilla has lived in Intag since the 1970s) with the mission to protect cloud forests in the Intag region. Projects were funded through private donations and through partnerships with other environmental NGOs. The goals of the watershed reforestation projects were to: 1) improve the quality of water resources in communities and maintain summer streamflow; 2) restore and conserve forest biodiversity in the region; and, 3) increase environmental awareness about the value of forests and promote environmental stewardship (C. Zorrilla, pers. comm., 2010, 2011; DECOIN, 2010).

To create each reserve, DECOIN purchased land in watersheds from local farmers and signed the title over to the community. In most cases forests were allowed to regenerate naturally, but in six DECOIN initiated tree-planting efforts to restore forest. For these projects, which were much more labour intensive than sites with natural regeneration and thus required a substantial commitment from people in the community, DECOIN solicited interest from communities who were experiencing severe declines in water and who were on-board to engage in the projects as a community (e.g., had come to a community-level agreement to participate). Restoration involved planting (mostly native) trees in former pastures where planted, non-native pasture grass inhibited natural regeneration (Aide & Cavelier, 1994; Griscom *et al.*, 2009). DECOIN helped each community establish a tree nursery, and taught them to harvest seeds from nearby forests, grow seedlings, and plant and care for trees. People in communities also learned to cultivate some native tree species by trial and error. Maintenance involved clearing grass from around seedlings at least every three months for two years after planting. In two communities, DECOIN paid people a daily wage to plant and maintain trees, funding that came from a specific donor with an interest in reforesting these two communities. In others, community members were unpaid, but agreed to participate as a community. People often worked together in *mingas* (communal workdays), after which they typically shared a communally prepared meal. DECOIN did not provide support or trees for on-farm tree planting. But, in addition to training people to cultivate tree species that they already knew and used, DECOIN introduced people to new tree species with additional uses (Chapter 4). They also provided a non-monetary incentive to plant trees – the belief that tree planting would restore much-needed clean water to communities.

I worked with residents in four villages in the northeast end of the valley (Fig. 5.1). Of the 47 communities which DECOIN helped establish watershed reserves, I selected these four because they had planted trees rather than relying on spontaneous regeneration. Of the six communities had tree planting projects, I chose these four because 1) tree planting was carried out by community members (and not foreign volunteers, the case the Apuela reserve); and, 2) residents were willing to participate (the sixth community was still divided over the mining conflict and surveying households would have been problematic). The four study villages are located within 3 km (as the crow flies) of one another, and village centres (i.e., the location of the elementary school) are located between 1850 and 1960 masl. All four are accessible by road (two, El Paraíso and La Esperanza, only since 2005). A deep valley bisects the study region, separating the two northeast villages (El Paraíso and El Cristal) from the two southwest ones (La Esperanza and Pueblo Viejo) (Fig. 5.1). As of 2011, no public transport existed between the southeast and northwest communities, but residents from all four communities travel to Apuela, especially on market day.

People in the four villages work primarily as farmers, and almost all households surveyed produce some subsistence crops (corn, climbing beans, and root vegetables such as yuca and camote). In addition, some (78%) produce cash crops (corn, bush beans, and fruit: *tomate de árbol* and *naranjilla,* two fruiting shrubs in the *Solanaceae* family), and/or cattle for beef production (39%). Many households also own chickens, pigs, and dairy cows to produce both subsistence and for-sale products (middle- to upper-level income earners generally own at least one cow). In the past decade, with the construction of roads, households have replaced traditional subsistence staples with purchased rice and pasta. Over a third of the households also earn income from either wage labour (these households tend to have lower incomes) or skilled, off-farm work. Unlike in some other Andean regions (Bebbington & Perreault, 1999; Flora & Flora, 2003; Solimano, 2003), remittances from relatives working abroad or in Ecuadorian cities do not contribute significantly to household incomes. However, 77% of households receive the *bono* – a federal government subsidy for people 65 and older, disabled people, or households with children (in 2011, $35 per month, per eligible person).

Each of the four communities is comprised of between 23 and 45 households. Residents are mainly *mestizo*, with minority populations of *Otavaleños* (indigenous people from the Central Valley) and Afro-Ecuadorians (Kocian *et al.*, 2011). In 2011, the average household in these communities owned 12.8 hectares (range from 0 to 135), $830 US in productive assets, and had an income of $6900 US (including subsistence production). The four communities are similar with respect to their average landholdings, income, and other key demographic indicators (Table 5.1). Within a community, land use tends to vary with elevation. The highlands (above approximately 2100 masl) are used primarily for cattle ranching, and lower areas near the village used mainly for agricultural production (Chapter 2). A given household might own land in the highlands, around the community, or both. Sharecropping on small areas of land (less than one hectare) is also common.

In addition to DECOIN's reforestation initiatives, other farmers associations and cooperatives operate in the region. However, DECOIN is unique in that they engage participants at the community level, turning operations over to communal management. They did not work directly with individual households, and their focus was exclusively on tree planting in watershed reserves. Moreover, DECOIN did not work through existing farmers associations, but disseminated information at community meetings and through the elementary schools. All other associations in the regions work with individual households. Other organizations exist to help households: 1) improve their agricultural productivity through new farming methods and inputs (*Asociación de Campesinos Agroecológicos de Intag*, ACAI); 2) increase pasture productivity (PRODERNA); 3) implement and manage shade-grown, fair trade coffee production systems (*Asociación Artesanal de Caficultores Río Intag*, AACRI); and, 4) with microfinance (*Ministerio de Agricultura, Ganaderia, Acuacultura y Pesca*, MAGAP). Other local community groups include a women's cooperative (producing medicinal plants and guinea pigs for sale), and a chicken-farming cooperative.

Methods

Data collection

Data were collected during eight months of fieldwork over two field stays: seven weeks in May and June 2010, and six months from March to September 2011. Ethics approval for field research activities was obtained through McGill prior to commencing the first field season, and all participants gave oral consent prior to participating in research (Appendix A). I interviewed households, conducted focus groups and oral histories, observed participants, and interviewed NGO personnelle and local experts. I surveyed 120 of the 134 households in the four communities (Table 5.1, Appendix A). Household interviews each lasted 30 to 90 minutes. Each interview team consisted of two people: a local field assistant who would generally ask questions, and a recorder. Questions focused on land holdings, demographics, agricultural production, assets, forest use, tree-planting activities, community engagement, agricultural practices, and people's perceptions of the restoration projects and life in the community. Before interviewing households, I pre-tested my questionnaire with five local residents outside my census cohort, and restructured the survey instrument based on their feedback. In each community, one or two households declined to participate, and others were unable to for health reasons. Incomplete questionnaires from two other households were dropped.

To refine my questionnaire and provide context for interpreting their results, I conducted key informant interviews with NGOs, extension agents, and long-term residents; ran focus groups (n=8) with men and women in each community; recorded oral histories with long-term residents (n=16); participated in town meetings and planting projects; and helped people on their farms. Planted tree species were identified by botanists from the National Herbarium of Ecuador in Quito (QNCE) (scientific names) and local residents (local names) who also explained their use. I collected extensive ecological data on the forests that were planted in the communal watershed reserves (Chapter 4) which I used to inform the analysis.

Analysis

Harvests and income

I estimated annual household income by asking people to recall crop harvests from Easter week of the past year to the current year. Crops that were consumed throughout the year or seasonally were estimated based on weekly harvests. I systematically asked about cash income from other sources, including wage earnings, salaried earnings, government subsidies, and remittances. To calculated a monetary value ($US) for subsistence production I asked farmers in each community at what price they could sell subsistence crops, small livestock, dairy products or firewood. People could give prices for most crops because surplus is often sold to intermediaries – "men with trucks" – who pass through communities weekly. Other crops were sold or exchanged between neighbors. Subsistence production was included in total household income. The value of household non-land assets (productive and non-productive) was determined by visiting the marketplace, and by asking residents for a 'typical' price that someone would pay (many items are purchased used). The value of land was not estimated but I determined household total holdings (hectares) through household surveys.

Probit and tobit model selection

I developed a series of probit models to assess which households participated in tree planting and adopting on-farm, conservation-oriented (green) farming techniques. Probit models are a type of generalized linear model (GLiM) appropriate for binary response variables that model the probability of a given response base on predictor variables (UCLA 2007, Long and Scott 1997). Models were developed to analyze participation in tree planting: in communal watershed reserves (models one and two); on farms (model three); in agroforestry systems on farms (model four); to restore forests on farms (model five); and for participation in two related, conservation-oriented on-farm practices – green technologies (model six) and organic agriculture (model seven). I selected independent variables for this analysis that were theoretically important according to the literature on micro development and agricultural economics and relevant to the question posed, based on past studies on agroforestry adoption and smallholder decision-making (Reardon & Vosti, 1995; Pattanayak *et al.*, 2003; Manzi & Coomes, 2009). To minimize multicolinearity, I only included variables with a correlation of less than 0.4 with any other variable in the dataset, and a variance inflation factor of less than 10 (Stevens, 2012).

In an ordinary least squares (OLS) regression, the estimated coefficients represent the marginal effect of adding one more unit of a given independent variable (i.e., one year of education) on the dependent variable because they are additive. Unlike OLS regression, probit coefficients are multiplicative, and so do not represent the marginal effects of the independent variables (Fernihough, 2011). To interpret these coefficients I computed the marginal effects separately in R (Fernihough, 2011). These marginal effects indicate how much the probability of participating changes with an additional unit – for example, a year of education – above or below the mean for each of the independent variables in the model.

To assess the degree of participation in on-farm planting and on-farm adoption of green farming techniques, I developed several tobit models. Tobit models are a probit/multiple regression hybrids used when the response variable takes on a limiting value (e.g., 0, for non-participants) for many of the respondents, but a wide range of values above or below this limit for other respondents (Tobin, 1958). In the current study, the limit is 0 (non-participants), with participants having positive values (Tobin, 1958; Fisher *et al.*, 2005). I developed Tobit models to predict: 1) the number of species planted; 2) the number of trees planted; 3) the percentage of native trees planted; 4) the number of trees planted to restore forests on-farm; 5) the number of trees planted in on-farm agroforestry systems; 6) the number of types of tree systems in which trees were planted; and 7) the number of green technologies adopted. I performed the same variable selection process as above, ultimately selecting the same variables.

A partial correlation controlling for key variables was used to test if people who planted in the watershed reserves were also more likely to plant on private land. ANOVAS, Chi-squared, and probit models were conducted using SPSS version 20 (IBM Corp. 2011). Tobits and marginal analyses were conducted using the VGAM package (Yee, 2010) in R (version 2.14.2). Marginal analyses used methods described by Fernihough (2011) in R (version 2.14.2).

Results

Tree planting the Intag Valley

Over the past decade, people in the four study communities planted approximately 21,000 trees on private land, and more than 75,000 in communal watershed reserves. They planted a total of 49 different species (44 of which were native) in communal reserves, with between 12 and 23 species in each reserve, and 80% of the planted trees were native species (Fig. 5.2) found in both regenerating and primary forests, including a number of species people use for a specific purpose (Chapter 4). Most exotic trees were either soil-improving (*Alnus nepalensis*) or food-producing species (e.g., lemon trees, *Citrus C. limon*.). All trees were planted as 'forests', densely spaced plantations (2 to 2.5 m apart) in which pasture grass and ferns were regularly cleared so that other, non-planted trees could regenerate (Fig. 5.3).

People planted 34 different species (26 of them native) on private farms, with an average of 1.9 species per farm. On farms, people planted more exotic species and a higher proportion of exotic trees – 83% of the trees planted were exotic species, and 17% native (Fig. 5.2). In particular, people preferred to plant *aliso* (*A. nepalensis*), a fast-growing, nitrogen-fixing alder used to restore soil fertility in pastures and fields that accounted for more than 60% of the trees planted on farmland (Fig. 5.3). Native species planted on farmland were a subset of those planted in communal reserves (i.e., all native species planted on-farm were also planted in reserves). Species planted on farms but not in reserves were thus all exotics, and included several fruit trees (other *Citrus* spp.); leucaena (*Leucaena leucocephala*), a nitrogen-fixing shrub commonly used in silvopastoral systems (Murgueitio et al. 2011); and several other exotic trees commonly planted in this region: pines (*Pinus* sp.), cypress (*Cupressus* sp.), African tulip tree (*Spathodea campanulata*), fresno (*Fraxinus* sp.), and cujaco (*Solanum* sp.).

Households planted trees on their farms in a variety of different ways: as hedgerows along fields, interspersed with other crops (e.g., in shade coffee systems and in home gardens), in pastures, around the house for shade, in small orchards and plantations, and to create small forests along streams and elsewhere (Fig. 5.4). Planting trees for on-farm 'forests' means that, unlike in production systems, people allowed some natural regrowth in the understory and, unlike in plantations, were not planning to harvest planted trees.

Overall, tree-planting activities on private farms sought different ecological outcomes than those in communal watershed reserves. Planting in communal reserves was focused on conserving and restoring forests, and people planted more trees, more species of trees, and more native trees. They also planted many species – especially native ones – with which they had little prior experience cultivating or maintaining. On farmland, people planted more exotic trees, fewer species of trees, and integrated trees into a variety of different farming systems. On-farm planting was also production-oriented, and households used mainly species known to produce specific outcomes (fruit production, timber production, soil enhancement), and that were commercially available (e.g., fruit trees, pines and cypress, leucaena) (Fig. 5.4). Tree-planting activities, including the number of species planted and the types of systems that they were planted in, also varied from farm to farm. The results presented below examine the household-level participation that produced these diverse outcomes, beginning with a description of how people make a living in the region.

Livelihoods

Based on household survey data, expert and NGO interviews, and in-the-field observations, I identified five major livelihood strategies in the region: subsistence farming, market-oriented farming, cattle ranching, off-farm skilled work, and day labour (Table 5.2).

About 40% of the households in Intag rely primarily on *subsistence farming.* Subsistence farmers produce crops, small livestock (chickens, guinea pigs), and sometimes pigs and cow's milk, primarily for household consumption. If they own cattle, they are usually dairy cows. They have a mean annual income of $5657 US, own an average of 7.1 ha land, and $674 dollars worth of productive assets. They have an average of 4.8 years of education (Table 5.2), and two adults per household.

Market-oriented farmers, cattle ranchers, and off-farm skilled workers occupy the upper half of the income distribution. *Market-oriented farmers* typically produce beans, corn, fruit (especially tree tomatoes) and/or coffee for sale. A household will usually produce either fruit, or corn and beans (which are grown in rotation). Coffee production is small-scale but becoming more common. They have a mean annual income of $8135 US, own an average of 16.5 ha of land, and productive assets worth $507 US. The mean number of years of education for the household head is 5.8 years.

Cattle ranchers raise cows primarily for beef production, and households very rarely consume the beef they produce. They have a mean annual income of $9218 US, own an average of 28 ha land, and $1477 dollars worth of productive assets. They have an average of 5.4 years of education (Table 5.2), and two adults per household.

Off-farm skilled workers include people with a trade (e.g., carpentry), people who manage their own business, or people whose work requires a certain level of education (e.g., agricultural extension workers, teachers). They have a mean annual income of $9977 US, own an average of 13.8 ha land, and $1580 dollars worth of productive assets, and have, on average, 7.9 years of education (Table 5.2).

Day labourers work for a daily wage on other people's farms, generally in the same community in which they live, or in neighboring communities (within an hour or two walk). Many also practice sharecropping on other people's land. Households tend to have few adults (and household heads are often single men), and more residents over age 64, many of whom still work (Table 5.2). They have a mean annual income of $4354 US, own an average of 3.7 ha land, and $87 US worth of productive assets. They have an average of 4.0 years of education (Table 5.2).

Subsistence farmers and day labourers together make up the lower half of the income and landholding distributions in Intag, but, on average, subsistence farmers have higher incomes, more landholdings, and own more productive assets. They also sharecrop less. Market-oriented farmers tend to have more cattle and twice as much land as subsistence farmers. Along with cattle ranchers, market-oriented farmers also have higher incomes (significantly higher than day labourers (*Tukey post-hoc*, $p < 0.05$) (Table 5.2). Cattle ranchers own the most land; many also produce some cash crops. Not surprisingly, they also own significantly more cows than any other group (on average, 11.9 per household, *Tukey post-hoc*, $p < 0.05$) (Table 5.2). Off-farm skilled labourers have more formal education (significantly more than both subsistence farmers and wage labourers (*Tukey post-hoc*, $p < 0.05$) (Table 5.2). Almost all off-farm skilled households also maintain a farm and rear cattle. Many also own a motorcycle or truck. Day labourers tend to have the least land, fewest productive assets, lowest incomes, and own fewer cows.

Participation in tree planting to restore communal forests

Overall, people from 69 households (58.5%) planted trees to restore forests in communal watershed reserves. The main reason reported for participating (75% of households) was to improve the quality and quantity of the community's water supply. Twenty percent of households planted trees to earn money (in communities where people were paid to plant); another 20% participated because they felt obligated to do so by the community (in unpaid communities) (Fig. 5.3). Non-participants stated that they were either not invited (mainly people from the community with the lowest participation rates, El Cristal) or physically unable. Six percent also declined to participate because they have access to a separate well, stream or spring and did not receive water from the community.

A probit model predicting tree planting in community reserves captured a relatively large amount of the variation in the data (*pseudo* $r^2 = 0.46$[15]) and shows that households with more labour, members of farmer's organizations, and residents of El Paraiso, La Esperanza and Pueblo Viejo are more likely to plant trees (Table 5.3). Members of farmer's associations are 14.8% more likely to participate than those who are not (Table 5.3, Table 5.4), and households with an additional adult member are 8.6% more likely to participate. Households in El Cristal, a community where people were paid to plant, are 38.9% less likely to participate than households in other communities. Land, the value of non-land productive assets, age, and pursuing a land-based livelihood are all non-significant.

Participation rates were higher in communities where the work was not monetarily compensated: 77.8% of the households in unpaid communities participated, compared to 42.7% in communities that were paid a daily wage. A second probit model which includes whether or

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[15] Because true r^2 values cannot be computed for probit regressions, a pseudo r^2, although not the percentage, is computed as a measure of the degree to which the variation in y is explained by the model. Here, we show the Nagelkerke's r^2 following Sood and Mitchell (2011) and Kiptot et al. (2007).

not people were paid to participate captured slightly less of the variation in the data (*pseudo* $r^2 = 0.40$), and shows that people in unpaid communities are 27.6% more likely to participate than people in paid communities (Table 5.4). Otherwise, the same variables were significant as in the first model (Table 5.3, Table 5.4).

Participation in tree planting on private land

Most households (66%; 79 households) planted trees on their farms. People planted trees to produce wood (lumber or firewood, 46%), increase the quality or quantity of water in their community (37%), or improve soil by increasing fertility or reducing erosion (19%). Twenty-eight percent of households gave sustainability as a reason for planting – so that future generations would have trees and forests.

A probit model predicting participation in on-farm tree planting captures a high degree of the variation in my data (*pseudo* $r^2 = 0.55$), and shows that households with higher levels of community involvement, labour, education, and reliance on the land are more likely to plant trees on their farms. Community involvement was especially important: those elected to local government are 23.6% more likely to participate, and members of farmer's associations are 16.7% more likely. As with the community reserves, households from different communities participate in on-farm planting to different degrees – households in El Paraíso are 20.4% more likely to plant. Households with land-based livelihoods (ranching, farming) are also 20.2% more likely to plant on-farm trees. For additional each year that the household head attends school, likelihood of tree planting increases by 3.4%, and an additional adult in the household increased the likelihood of planting by 5.5%. The area of land owned, value of productive assets, and the age of the household head did not significantly predict participation in on-farm tree planting.

In community reserves, trees were planted with the ultimate goal of restoring forest: increasing forest cover and stimulating natural regeneration in the understory. Within each community, the time invested in tree planting was similar for most participants, although a few people were also more involved in the managerial aspects of planting and growing trees. In contrast, people participated in on-farm planting to different degrees, planting trees in a variety of ways (Fig. 5.3) to achieve diverse goals (Fig. 5.4). Many of the ways that people planted on-farm were new to them, as was the practice of cultivating and planting native trees for most (only 11% of households had planted trees on farms before the communal projects).

<u>Diversity in on-farm tree planting systems</u>

On farms, trees were planted in hedgerows, in agroforestry and silvopastoral systems, as orchards, in small plantations, to create forests, and 'just around the house'. Some households planted trees in only one type of system, others in many. A tobit model capturing 40% of the variation in the data shows that households who participate in farmers' associations and local government, who earn a living from the land, and who are more educated and have more available labour plant trees in a greater number of systems on their farms (Table 5.6). The amount of land owned, productive assets, age of the household head, and the community do not significantly predict the number of systems in which a household will plant. Of particular interest in this study are households who plant trees to restore forests, and who plant as part of agroforestry systems.

<u>Restoring forests on private land</u>

Twenty-eight households planted trees in an effort to restore forests on their farms. Probit model four (*pseudo* $r^2 = 0.34$) indicates that households with more land, more adults, and who are members of farmers associations are more likely to restore on-farm forests. Of these, membership in a farmer's association is most important, increasing the likelihood by 17.3%, whereas each additional adult household member increases the likelihood by 5.2%. Planting on-farm trees to restore forest is the only planting outcome significantly predicted by landholdings, and owning an additional hectare makes restoration 0.27% more likely (Table 5.3). Productive assets, education, age of the household head, being elected to local government, and the community a household is from do not significantly predict on-farm restoration.

<u>Planting trees in agroforestry systems</u>

Fifty-nine households planted trees in agroforestry systems. A probit predicting planting agroforestry systems again captures a high degree of the variation in the data (*pseudo* $r^2 = 0.45$), and tells a different story from participation in on-farm restoration. Community involvement is important – people who have been elected to local government and/or are members of farmers associations are more likely to plant trees in agroforestry systems (29.8% and 20.0% more likely, respectively) (Table 5.4). Labour is also a significant factor (an 8.5% increase in participation likelihood with each additional adult) as is education (an additional year of education increases the likelihood of participation by 2.9%). People from El Paraíso are 20.0% more likely to plant in agroforestry systems, and people who earn a living from the land are 10.7% more likely. Land holdings, productive assets and age do not significantly predict on-farm agroforestry adoption.

Extent of on-farm tree planting: number of trees and species planted

In community reserves, the time invested in tree planting was similar for most participants within each community. But on farms, people participated to very different degrees, planting anywhere from three to 1200 trees. Tobit regression models examining the extent of participation in on-farm tree planting capture between 19 and 44% of the variation in the data and show that better educated household heads tend to plant more trees and more species of trees, as do members of farmers associations (Table 5.5), who also plant more native species. The results presented below show which household characteristics significantly predict the extent of tree planting in terms of the number of trees and species planted in different on-farm systems.

<u>(i) Education</u>: Households with more education plant a greater number of species per farm. Although as a group people planted 34 species on farms, the average per farm was only 1.9 (± 2.0) with a range from one to seven. Households plant one additional species for every four years of education (about the difference between finishing elementary school and high school, or between high school and a post-secondary degree or certificate). Better educated households also plant more trees on farms overall: on average, households planted 263 trees (± 359) per farm, and an additional year of education means 28 more trees above this mean. In particular, better educated households plant more trees in agroforestry systems – 34 more trees for each additional year (Table 5.5).

<u>(ii) Participation in civic life (farmers associations and local government)</u>: Members of farmers associations participate to a greater degree than non-members. Members are predicted to plant 0.45 additional species, 344 trees, and 32% more native species than non-members. They also

plant more trees in on-farm restoration (192 more, on average), and in agroforestry systems (310 more, on average) (Table 5.5).

Although households who participated in local government did not plant more trees per farm, they planted more species of trees (0.55 more than non-elected community members), more trees in agroforestry systems (145 more trees), and a higher percentage of native trees (32%).

(iii) Land-based livelihoods: Farmers and ranchers plant more species of trees on their farms – approximately one more species, on average, than people who earn wages or salaries. They also plant a higher proportion of native species (an additional 18%), but do not plant more trees or more trees in restoration or agroforestry systems.

(iv) Geographical location: In general, the community to which a household belongs does not affect the extent to which they participate in tree planting, with the exception the number of species planted on their farm. People in El Paraíso plant far more species than other communities: 1.4 species per farm above the mean.

(v) Non-significant characteristics: Surprisingly, household wealth – as measured by land holdings and productive assets – does not predict the number of trees or species that a household plants. The age of the household head is also non-significant. Additional household labour increases the number of species that a household will plant – an additional species for every three adults above the mean – but households with more labour are not predicted to plant more trees or more native trees.

In sum, education and membership in farmers associations are the strongest predictors of how many trees a household plants. The tobit model for number of *species* per farm, however, captures an especially high percentage of the variation in the data (49%) and shows that farmers and ranchers, households with more labour, households involved in local government, and households in El Paraíso all plant more species of trees per farm, as do better educated households and members of farmer's associations.

Does communal planting spur on-farm tree planting?

Many households began planting trees on their farms soon after the communal tree-planting projects began. Before the communal tree-planting projects were initiated, 13 households (11% of our survey interviewees) had planted trees on private farms. After the communal projects were initiated (accounting for the year that planting activities began in each community), 64 more households planted trees on farms – an additional 54% (Fig. 5.5).

However, not all of the households that began planting trees on their farm participated in the communal projects. Of the 64 households that began planting on their farms after the communal projects were initiated, 34 of them had participated in communal projects, and 20 had not. A total of 55 households participated in both communal and on-farm tree planting. Of these, 12 planted on their own farms before the community projects, and 43 of them began on-farm planting after they participated in communal restoration (all of the initial 12 also continued to plant trees on their farms after participating). A partial correlation between planting trees in communal reserves and on-farm planting, controlling for key variables[16], showed that planting in reserves was significantly correlated with on-farm planting afterwards ($r = 0.21, p < 0.05$), but

[16] I controlled for the following theoretically important variables, and variables identified as important in the probit regression models: land owned, education of the household head, productive assets, number of adults, household age, community, and civic life.

not with on-farm planting before ($r = 0.0, p > 0.05$). Although we cannot conclude that participating in communal tree-planting projects will lead to tree planting on private land, it seems likely that the training and education provided by the NGO are, at least in part, responsible for the dramatic increase in planting rates on private farms that followed.

Do people who adopt 'green' farming practices plant more trees?

I examined which households adopted two related, conservation-oriented on-farm practices – green technologies and organic agriculture. I defined three levels of adoption: 1) 'green adopters' employ more than two kinds of green technologies on their farms (composters, systems used to create liquid fertilizer (*biol*), bio-gas-producing systems from pig manure (*biodigestores*), wire systems for growing beans, or various types of green composts on their farms); 2) 'some green' households employ one or two types of green fertilizers or have adopted one of the above technologies; and, 3) 'no green' households do not use green technologies. Because some households employ green technologies but continue to use chemical fertilizers, pesticides, herbicides, or fungicides on certain crops, I also created an 'organic adopter' category. Organic farmers intentionally practice organic agriculture. This means that: 1) they employ at least one 'green technology' or compost, as defined above; 2) using this technology represents a change in the way they practice agriculture in the past 10 years; and, 3) they do not use any synthesized chemicals on their farms.

<u>Adopting green technologies on-farms</u>

A probit model captures a large degree of the variation in the data ($pseudo\ r^2 = 0.65$), and shows that many of the same household characteristics predict on-farm tree planting and adoption of green technologies. In particular, households that belong to farmer's associations are 13% more likely to adopt green technology, as are households with more education (but the marginal increase per year of education is small – only 1.2%). Households in La Esperanza are 7% less likely to adopt green technologies (Table 5.7). Land owned, the value of productive assets, age, number of adults, participating in local government, or having a land-based livelihood do not predict if a household will adopt green technologies.

Adopting organic agriculture

A probit model predicting participation in organic agriculture again captures a large amount of the variation in the data (*pseudo* $= r^2$ *0.56*) and shows that membership in a farmers association and more education both increase the likelihood of participating (by 16% and 2.5% per additional year, respectively). Households with more adults are also more likely to adopt (each adult increases the likelihood by 5.4%). Households with more productive assets and with an older household head were also more likely to adopt (for each additional $100 in productive assets owned, the likelihood increased by 0.13%, and by 0.4% for each additional year of age). Adoption varied by community: members of El Paraíso and El Cristal, the two communities on the southwest side of the valley, are 21 and 19% more likely to adopt, respectively). Land holdings, participation in local government, and being a farmer or rancher did not predict participation.

Households that participated in on-farm tree planting and reserve tree planting also adopted more green technologies on their farms. Participating in on-farm tree planting and adopting environmentally oriented farming techniques and technologies are both predicted by some of the same household characteristics. 'Green adopter' households, which employ a variety of new, environmentally oriented technologies on their farms: 1) participated more in both on-farm and communal tree-planting projects; 2) planted more trees; 3) planted more species of trees; and, 4) planted a higher proportion of native species on their farms (Table 5.7). Households that employ only one or two green technologies participated in tree planting to a lesser extent than the green adopter households, but more than those households that did not use any. Green adopters also plant trees in agroforestry systems and restore forests on their farms more than other groups (75% of green adopters planted trees in agroforestry systems, and 39% practice on-farm restoration, compared to means of 50% and 22% for the population, respectively). Although none of the 'green technologies' identified here involve planting trees, it should be noted that some use tree products, such as leaves, to make fertilizer.

Livelihoods and tree planting: farmers and ranchers plant more trees

In the Intag Valley, households that depended on their land to earn a living (i.e., farmers, both subsistence and market-oriented, and cattle ranchers) tended to participate more, and more extensively, in tree planting than those who worked for wages (Table 5.8). Farmers and ranchers also participated more in farmers associations and in local government, and were more likely to have adopted green technologies (such as organic composts or biogas production systems) on their farms. In particular, both subsistence and market-oriented farmers tended to plant more species of trees, more native trees, and also tended to adopt more different kinds of green technologies. Interestingly, these two groups engaged in tree-planting activities in similar ways, even though market-oriented farmers were substantially wealthier, with, on average, twice as much land, twice as many cows, and 30% higher incomes than subsistence farmers. Day-labourer households participated the least in any type of tree-planting activity. The subset of day labourers who did participate tended to have more productive assets, more education, more adults in the household, and higher incomes (*T-test*, all $p < 0.05$).

Discussion

Planting trees is part of a ‘back to the land’ green farming movement in Intag. Pursuing land-reliant livelihood strategies did not conflict with planting trees on private or communal land. On the contrary, farmers tended to engage more frequently and extensively in the most conservation-oriented types of on-farm tree planting, integrating forest restoration and trees into farming systems and developing innovative ways of using trees. Results suggest that perceived environmental degradation – especially water availability – following watershed deforestation, combined with timely environmental education, motivated these high participation rates. People tended to planted trees to ameliorate conditions that threatened their ability to farm to confront what they perceived as a ‘crisis’.

How do we engage households to restore forests in ‘crisis’ conditions?

My results show that three main factors explain which households chose to participate in restoration on communal land: 1) access to channels of information about the projects; 2) strong community-level governance of the projects, and high community engagement; and 3) the availability of labour.

Farmers associations served as important channels of information. Although DECOIN did not work through farmers associations, instead hiring their own local employees to train farmers and disseminate information at meetings and through elementary schools, farmer’s association meetings served as a platform for people to discuss communal projects (Walters *et* *al.*, 1999; Isaac *et al.*, 2007; Bodin & Crona, 2009). Membership also served as an indicator of a household's engagement and investment in the community. It makes intuitive sense that households more invested in their communities would participate in communal restoration, given that tree-planting activities were coordinated and managed by communities with the main goal of sustaining and restoring a shared communal good (Netting, 1972*a*; Bodin & Crona, 2009). It follows that 'united' communities with better-managed projects would have higher participation rates (Agrawal & Gibson, 1999; Bray *et al.*, 2006; Persha *et al.*, 2011; Bray, 2013; Wunder, 2013), which was indeed the case. Participation ranged from 24 to 84% of households in each community. Residents of El Cristal, the community with the lowest participation rates, complained that projects had been mismanaged in the past (rumors of embezzlement were rife). Non-participants here often said they were not interested in participating, or not invited to. In the other three communities, where participation rates were high (67 to 84%), there were few complaints of mismanagement and people generally spoke highly of the projects and project leaders.

Strong local governance and community-level associations are key components of organized communities (Bray *et al.*, 2006; Persha *et al.*, 2011; Bray, 2013). Self-organization at the community level is essential for success in other innovative community-based forest management arrangements, including those geared toward commercial timber production (Bray *et al.*, 2006; Bray, 2013), firewood and other non-timber forest products (Persha *et al.*, 2011), and other environmental conservation initiatives related to forest and watershed management (Becker, 2003; Wunder, 2013). My results show that such organization is also key to attracting participants in restoration projects oriented toward producing non-extractive, shared ecosystem services.

Does compensation increase participation in communal restoration? Paying people can allow or entice them to plant trees by compensating them for the opportunity cost of investing time and effort in other profit-generating activities (Wunder, 2005; Engel *et al.*, 2008). However, in my study, unpaid communities had higher participation rates than paid ones. Paying people introduced two related factors that explain this result: First, the potential for corruption, realized or perceived; and second, because monetary resources are finite, either more people can participate for fewer hours each, or fewer people can participate more. In El Cristal, participation rates started high, then fell as financial gains became concentrated in fewer households (Persha & Andersson, 2014). In contrast, El Paraíso, the other paid community, had similar participation rates to unpaid communities and employed the same *minga* model – people worked together as a community. These results suggest that if people are so motivated to conserve forests that they will commit to doing it as a community, compensating the participants does not necessarily lead to higher participation rates – without payment, it may in fact be easier for more people to participate. Limited resources might be better spent on materials for restoration, on creating or supporting community-level associations that provide resources or training for tree planting, or ensuring that people in communities have access to services such as basic health care and education that allow or encourage people to participate in such projects in the first place (Reardon & Vosti, 1995; Franzel, 1999; Scherr, 2000; Barbier, 2012).

Tree-planting strategies on private land

After restoring forests on communal land, 55 households (79% of the participants) planted trees on private farms. Twenty-two households that had not participated in the projects also began planting on-farm trees at that time. Compared to non-adopters, households who planted on farms tended to: 1) be more involved in the community; 2) be more reliant on the land for their livelihoods; and, 3) have greater human capital in terms of education and labour. Being invested and involved in the community was key – participating in farmers associations, involvement in local government, and earning a livelihood from the land were all strong predictors of on-farm planting. As with communal restoration, membership in farmers associations would have provided a direct link to information about tree planting, both by interacting with other farmers and through programs run by the associations. Households with more invested in their communities, as demonstrated through participation in government and farmers associations, may also stand to gain more from planting trees on farms – tree planting is a long-term investment, the benefits of which are often not realized for several years (Current *et al.*, 1995).

Households also tailored tree planting to meet their individual goals, needs and preferences – the types of tree systems and number and types of species planted varied considerably. Two tree-planting strategies stand out: 1) innovative planting to sustain and improve agriculture, and 2) planting as an investment in future harvests.

Farmers are tree innovators. In Intag, farmers, both subsistence and market-oriented, planted many species of native trees which were not commercially available or widely cultivated. They planted these 'new' species, along with better known ones, to produce a range of different services, incorporating trees into green farming systems (i.e., to produce compost and green mulch) and a variety of different on-farm agroforestry systems (including silvopastoral, shade coffee, intercropping, and hedgerows and windbreaks) to a greater degree than wage earners or ranchers. In short, they took tree planting to a whole new level. Because they planted more species, and more native species, the systems and forests they produced have potentially higher conservation value.

Stress can spur innovation (Netting, 1993; Bunce & West, 1994; West, 2002; Verma *et al.*, 2004; West *et al.*, 2006*a*; West *et al.*, 2006*b*). In the context of needing to increase food production under pressure from population growth, Boserup writes: “additional labour is likely to be used as a means to undertake a radical change of the system of cultivation in part of the area, while no change is made in other parts of the area” (Boserup, 1965). ‘Radical change’ refers to agricultural intensification that can require the use of new tools and farming techniques. Water stress can also spur farmers to adopt and develop new technologies. In India, for example, rural communities experiencing severe droughts adapted and refined a cost-effective drip irrigation system that was then widely adopted by others (Verma *et al.*, 2004). In Intag, the switch from felling on-farm trees to planting them was a similarly radical change.

Compared to farmers, households engaged in off-farm, skilled work tended to plant conservatively. They often planted trees either in small orchards or ‘around the house’ rather than integrating them into other production systems. They also planted more exotic species, and in particular, exotic fruit trees. Commercially available, with relatively well-known methods for care and propagation, fruit trees are a safer bet: their growth and survival is more predictable, as are the products they will produce. Less reliant on the land for their livelihoods, their planting practices are more typical of those in regions under less severe environmental stress. This tree-planting strategy is an investment in fruit production, not an overhaul of farming systems and strategies (Smith *et al.*, 1996).

Past work has shown that the poorest of the rural poor are often less likely to participate in conservation initiatives, including tree planting (Sunderlin *et al.*, 2003; Grieg-Gran *et al.*, 2005). In this study, wage labourers participated the least in planting, both on-farm and in communal reserves. Those households that did participate tended to have higher levels of education, own more productive assets, and had an average income more than triple that of non-participants. Thus, the poorest households in these communities generally did not participate in tree planting. Some stress might be a good motivation for planting – and planting in new and different ways – but, as with other conservation initiatives, ultimately people's basic needs must be met before these types of innovations and applications of tree planting become a priority (Barrett *et al.*, 2001; Lewis *et al.*, 2011).

Communal restoration and on-farm tree planting

A greater range of households participated in communal restoration projects than in on-farm tree planting: households participating in communal projects had a larger range in both income and asset holdings, and fewer household characteristics predicted participation. However, many households participated in both. The surge in on-farm tree planting directly following communal projects suggests that restoring forests on communal land spurred farmers to adopt these practices on their farms.

Communal restoration projects served as classrooms. Participating in them, people learned how to collect tree seeds and seedlings, and to propagate, plant, and care for young trees. They were also introduced to new uses for native tree species. The reserves thus also served as laboratories where people could experiment with new and different native species in a low-risk environment. Planting on communal land made the opportunity costs low, as compared to private land, where the failure of a tree species could have higher consequence for production. And because they planted such a high diversity of trees in the reserves, even if some species did not thrive, the risk of having the planting fail altogether was reduced. Participants could also see first-hand which species grew fastest and had the highest survival rates. Finally, communally restored sites served as accessible and public demonstration sites where other community members could observe tree-planting outcomes. Community restoration projects thus provided as a low-risk learning environment for people to hone their tree-planting skills and preferences.

The fact that the availability of labour, not land, affects participation in both communal and on-farm tree planting is indicative of the labour-intensive nature of both of these systems. Initially, each household contributed up to 50 days a year restoring forests in communal reserves, a significant investment. And although planting trees in plantations is often seen as a low-labour alternative to farming (labour intensive to start, less so to maintain), incorporating trees into farming systems can require long-term maintenance. Trees in agroforestry systems require pruning and tending to produce the desired effects (Schroth *et al.*, 2004; McGinty *et al.*, 2008), and as they grow, the rest of the system must be adapted (Schroth *et al.*, 2004; Benayas *et al.*, 2008). Given this, households with more people may be better positioned to adopt and manage on-farm tree planting. Smallholders are more efficient producers of other labour-intensive products, such as vanilla and rubber, than industrial operations (Dove, 2011; Osterhoudt, 2014), and tend to experiment more with planting mixed crops and diverse agricultural systems than larger landholders (Smith *et al.*, 1996). Restored forests with a wide diversity of native species can also be seen as such a product, requiring specialized knowledge to find and propagate species, and sustained maintenance and care to sustain planted trees. Although larger landholders are often better able to handle the higher start-up costs (including taking land out of production initially) (Zbinden & Lee, 2005), restoration and agroforestry projects that use native species might target smaller farms with more available labour (Smith *et al.*, 1996).

Thus, in communities experiencing environmental crisis, targeting *smallholder farmers* is a good choice to optimize participation rates, ecological outcomes (such as high numbers of native species), and to promote sustainable livelihoods. Although the ecosystem services that people hoped to obtain from communal and private planting were different, the channels through which information was distributed (i.e., farmer's associations) and the conditions that facilitated participation (i.e., households with more labour) are similar. Lessons learned from tree planting on private land in 'crisis' situations may well inform the literature on restoring forests at the community level.

From deforestation to reforestation in Intag

Visiting Intag today, it is hard to believe that in the 1990s the region was losing more than 3.5% of its forests per year (Chapter 2). Today, many of the green-technology-adopting, tree-planting residents of Intag self-define as '*ecologistas*' – stewards of the land (Buchanan, 2013). Although some (17, or 14%) households have continued to harvest trees and clear forests since restoration projects began, asking people (*ecologistas* or not) if they intend to cut the trees planted in watershed reserves elicits puzzled or even indignant 'no' responses. A farmer in El Paraíso summarized the sentiments of many: "*Por que plantamos árboles? Porque NOS DAN AGUA!* (Why do we plant trees? Because THEY GIVE US WATER!)." Explaining that in the past, they did not know trees were important for water, *ecologistas* describe how trees prevent soil erosion, leaves capture clouds and roots retain moisture. In two communities with especially high on-farm tree-planting participation rates (La Esperanza and El Paraíso), people commonly state "*Somos unidos* (We are united)" in their commitment to remain on the land. As in other regions, planting trees is part of this communal commitment (Higgs, 2003; Menzies, 2014).

This attitude of stewardship toward trees and forests in Intag represents a complete switch from past decades. One long-term resident described her experience as a young farmer 30 years ago: "*Todo el mundo fue trabajando en los montes. Cada propietario trabajó. Botaron los montes al suelo. Eso es lo que pasó.* (Everyone cleared forests. Every landholder worked. They cut forests to the ground. That's what happened)." As in other areas of Ecuador, forests were a vast resource to be exploited – quickly – mainly for fertile land (Wunder, 1996; Jokisch & Lair, 2002; Sarmiento, 2002)[17]. A few decades after clearing rates accelerated, people began to notice a severe decline in environmental conditions, especially water availability. People in two communities had to travel hours for drinking water, and rise in the middle of the night to beat their neighbours to it. All communities experienced severe problems with water quality from cattle pasture runoff. Changes in both daily and seasonal precipitation and cloud formation patterns (brought on by global climate change, local deforestation, or a combination (Bruijnzeel *et al.*, 2010*a*) increased the occurrence of *la lancha* – midday rain followed by sun – which burns crop leaves. People reported declines in soil fertility and pasture production. To combat these challenges, some farmers began using synthetic fertilizer and pesticides (39.8% and 61.9% of households, respectively), but with mixed results. Although some reported an increase in production, others reported that synthetic fertilizers 'burn the soil', were too expensive, and were a poor investment as they tend to run off the steep slopes.

Responding to this dire situation, DECOIN's mantra (as relayed by one resident): *"No boten el monte. Dejen para que haga sombra, no se acabe el agua* (Do not clear forest. Leave it to provide shade so that the water won't run out)*"* was welcomed by residents of several

[17] Although harvesting lumber for houses and collecting firewood were common, markets were not in place to sell these goods. Some hunted, but many people also complained that bears and other animals from the forest would raid crops.

communities, particularly those experiencing the greatest water shortages. So far, many agree that it's working – 61% of households report an increase in either water quality (57.6%) and/or quantity (34.0%) since the inception of the projects, with others commenting that changes have not occurred yet because the trees are not large enough. After working with over 40 communities to create watershed reserves[18], DECOIN currently receives more requests than they can fill from communities hoping to establish their own watershed reserves.

Crisis restoration – restoring for the future

My results suggest that the environmental context influences people's motivation to plant trees, both on farms and on communal land to restore forests. In Intag, people chose to restore forests because they faced a dire situation: their future as farmers was uncertain in the face of environmental change. Diminishing water supplies, declining soil fertility, and changes in daily and seasonal rainfall patterns presented major challenges to their traditional agrarian practices. By framing forest restoration as a way to alleviate real and urgent environmental problems, the NGO DECOIN initiated restoration projects with high participation rates, even when work was unpaid.

Households who participated in communal restoration projects were those who were most engaged in the community, independent of their wealth or income. In contrast to the findings of other studies that identify land and wealth as major factors influencing on-farm tree planting adoption (Pattanayak *et al.*, 2003; Grieg-Gran *et al.*, 2005; Zbinden & Lee, 2005; Duke *et al.*, 2014), community engagement and reliance on the land (along with labour availability and education) were also the defining features of households that planted trees on private farms,

[18] Six of which had a tree-planting component.

independent of the amount of land owned and other indicators of household wealth. Other studies have found similar results – people's experience with planting agencies and access to information are as important, if not more so, than landholdings or assets (Walters *et al.*, 1999; Pattanayak *et al.*, 2003; McGinty *et al.*, 2008; Cole, 2010; Pompeu *et al.*, 2012; Zanella *et al.*, 2014). The degree of variability in which household characteristics predict participation shows that asset portfolios and livelihood strategies of households that engage in tree planting are different in different contexts (Pattanayak *et al.*, 2003; Mercer, 2004). My results suggest that the degree of environmental degradation and the environmental problems that people are experiencing should be taken into account to target which households are most likely to plant trees in either communal restoration or on farms (Hoch *et al.*, 2012; Abram *et al.*, 2014).

Although households in Intag planted trees in communal reserves and on farms to obtain different ecosystem services, the ultimate goal of both was the same – to maintain and sustain rural farming culture just as conditions nearly forced them to leave their land. This '*crisis restoration'* – in which people chose to reforest to combat changes in environmental conditions that threaten their ability to sustain themselves and their communities through farming – requires that people look backward to move forward. Recalling a past when forests provided vital ecosystem services, people can envision and work to build a future where they can sustain their farming practices and rural livelihoods. In this process, trees and forests, which have been harvested and cleared for decades in these farming communities, are re-envisioned as a means to help sustain farm productivity. Ultimately, crisis restoration is an endogenous change or transition from exploiting forest to protecting it – exactly the type of change that many conservation and development agencies aim to foster, but still often fail to produce (Angelsen & Wunder, 2003; Blom *et al.*, 2010; Hoch *et al.*, 2012).

In Intag, people engaged in 'crisis restoration' because they identified strongly as farmers and desired to continue farming their land, experienced land degradation that threatened their ability to farm, and believed that forests and tree planting are an integral part of achieving their desired goal. Similar scenarios could occur in other regions where smallholders are experiencing unprecedented forms of environmental change extreme enough to threaten the viability of their farming systems, such as in post-frontier regions and in regions experiencing the effects of climate change (Buytaert *et al.*, 2011; Grêt-Regamey *et al.*, 2012; Abram *et al.*, 2014).

Reforestation in post-frontier regions

The way people perceive and interact with forests can depend on how much of it they have left (Perz & Skole, 2003; Satake & Rudel, 2007; Abram *et al.*, 2014). In some post-frontier regions, as landscapes have transitioned from forests to farms, there has been a parallel, delayed shift in peoples' attitudes towards forests from exploitation to stewardship. In the USA, for example, during the 1800s European settlers cleared the continent at historically unprecedented rates, seeing forest as a vast resource to be exploited, an impediment to farming and, ultimately, as standing in the way of developing a civil society (Williams, 1992, 2011). However:

> [i]t was only a few generations after Europeans began altering this landscape in earnest that an alternative view of the forest and its inhabitants began to emerge: salvation, claimed the Romantic philosophers and writers, lay not in a tame and planted landscape, but in the raw wilderness. But the proponents of these views came from settled areas (Vaillant, 2006, pg. 86).

Realizing that forests and forest resources are finite after seeing the rapid pace of clearing and its environmental impacts led to the relatively early establishment of the national park system in the US (Williams, 1992). Similar changes in the way people perceive forests and forest conservation over the trajectory of the forest also occur at local scales in the tropics (Schelhas & Sánchez-

Azofeifa, 2006; Abram *et al.*, 2014). In post-frontier regions of Costa Rica, for example, cattle ranchers began to see the value of forest conservation to their farming only once forests were largely cleared (Schelhas & Sánchez-Azofeifa, 2006). Similar shifts in forest conservation attitudes were observed in heavily deforested communities in the Bayano Lake district in Panama, where residents began adopting agroforestry after deforesting for years (O. Coomes pers. comm., 2014). People's attitudes towards forests change in these cases because 1) the resources (e.g., timber, game) that people obtain from forests become scarce, or people come to realize that they are finite, or, 2) people discover that they have lost services they were unaware forests provided (e.g., water, erosion control, local climate regulation) (Rudel *et al.*, 2005; Satake & Rudel, 2007; Barbier *et al.*, 2010; Rudel, 2010; Abram *et al.*, 2014). Of course, seeing the value in conserving forests and actually conserving them are two different things. The farmers in Costa Rica, for example, did little to conserve the forest fragments on their land even as they claimed it was important (Schelhas & Sánchez-Azofeifa, 2006). However, these conditions can still provide a starting point for governments, NGOs, or communities to discuss and plan further conservation actions.

As deforestation continues, 'post frontier' environments will become even more common across the tropics. In the Andes alone, more than 55% of the cloud forests have been cleared, and others fragmented (Mulligan, 2010). People throughout the Andes are thus experiencing environmental conditions similar to those in Intag – declining soil fertility and water shortages (Henderson *et al.*, 1991; Bebbington, 1993; Cavelier & Etter, 1995; Young & Leon, 1995*a*; Zimmerer, 1998; Jokisch & Lair, 2002; Williams-Linera, 2002; Mulligan & Burke, 2005*b*; Echeverría *et al.*, 2007; Mulligan, 2010; Scatena *et al.*, 2010). When the environmental impacts of deforestation threaten farmers' ability to farm (e.g., flooding, drought, topsoil loss, or declining soil fertility (Bruijnzeel, 2004; Grêt-Regamey *et al.*, 2012), they face a tough choice: abandon or decrease agricultural production and seek economic alternatives (which often involves migration, especially to cities), or adapt to changing conditions by modifying or intensifying farming practices (Bebbington, 1993; Grainger, 1995; Mather & Needle, 1998; Grau & Aide, 2008). In theory, the first option should increase forest cover because forests can regenerate on abandoned land, as has been observed in many tropical and temperate regions (Rudel, 2000; Rudel *et al.*, 2005; Grau & Aide, 2008). However, forest recovery is not guaranteed. Sometimes, instead of abandoning land, farmers who pursue off-farm employment switch to low-labour, extensive alternatives, such as raising cattle (Jokisch & Lair, 2002; Rudel *et al.*, 2002). My results suggest the opposite: in post-frontier regions, scarce ecosystem services may drive farmers – with help – to plant trees. Remaining on the land can promote forest recovery if farmers believe that forests and trees can provide ecosystem services that make farming viable and sustainable.

Where forest cover is high, or where people are less reliant on the land, restoring forests to increase ecosystem services may be a less powerful incentive to plant trees[19] (Abram *et al.*, 2014). In these situations, other incentives, such as monetary compensation, can gain relative importance. Different motivating factors mean that households with different asset and livelihood portfolios will participate in different contexts.

Implications for future community-based restoration projects

The conclusions from the Intag example suggest that to create successful reforestation

[19] The farmers living in post-frontier regions regions of Costa Rica, for example, opted for cattle over forest, in part because even though they stated that forest conservation is important, many of them had other income sources from the city. Cattle ranching was a low labour option to use otherwise unproductive land, not a core livelihood activity (Schelhas & Sánchez-Azofeifa, 2006).

projects in areas experiencing severe environmental degradation, we should target farmers. Unlike absentee landholders, people who rely on the land may have a vested interest in improving local environmental conditions. Farmers may both be motivated and have the necessary experience to adopt tree planting and adapt it to meet their needs regardless of farm size.

To encourage households to participate in restoration and on-farm tree planting, providing smallholders with motivation that respond to their needs in a given context is key (Current *et al.*, 1995; Coomes *et al.*, 2004; Verma *et al.*, 2004; Abram *et al.*, 2014). DECOIN's project in Intag worked well because they focused reforestation efforts on specific communities in a small region where people were experiencing a severe decline in environmental conditions that threatened their ability to farm, and that could be improved by restoring forests. They presented forest restoration as a solution to these conditions, implementing education programs with both adults and children (through schools) about the importance of cloud forests for water. An unintended benefit of using a community-based model to implement restoration was that it created communal spaces where people could practice and learn about tree planting in a relatively low-cost environment, knowledge which people later applied on their own farms. Several elements of this model may be used to guide agencies, organizations and governments seeking to promote community-based restoration in cloud forests and elsewhere in the tropics. Based on these findings, I propose several strategies for implementing community-based restoration projects in the tropics.

1. *Focus restoration efforts on areas where people are experiencing an environmental crisis* – and where planting trees can improve the situation within a reasonable timeframe for the

severity of the situation. In such cases, there is a clear motivation for people to participate, and restoration efforts can also have a larger impact on people's lives and livelihoods – a 'win-win' for people and the environment. Mountainous cloud forest regions are often heavily deforested, populated by smallholder, subsistence farmers, and are experiencing negative effects (soil erosion, flooding, droughts, etc.) of deforestation. Planting trees can increase the infiltration capacity of compacted soils often found in pasture, increasing their ability to store water and regulate stream flow. Because cloud forests capture fog, they can also increase throughflow (the amount of water reaching the ground) in forests (Bruijnzeel *et al.* 2005). This function is especially important in the dry season, when cloud capture makes up a relatively greater proportion of the overall precipitation (Bruijnzeel & Proctor, 1995; Bruijnzeel *et al.,* 2005; Guswa *et al*., 2007) – exactly the time when communities are most likely to experience water shortages. Cloud forest regions are thus good candidates for restoration projects aimed at restoring water or soil resources.

2. *Present projects as a solution* to problems that people are experiencing – provided that restoration can improve the situation. Local organizations and NGOs are often well positioned to both identify local problems and disseminate information to the people about how to fix them (Becker *et al.* 2003; Baral & Stern, 2011). Provide training in the types of tree planting best suited to meet these needs (Current *et al.*, 1995).

3. *Target communities with farmers associations and strong local governments*. (Bray *et al.*, 2006; Persha *et al.*, 2011). Projects could also support and expand, or help communities to establish, farmers associations.

4. *To run projects, engage and train local leaders* who are well-respected, trusted, and liked in communities (Becker, 2003).

5. *Target farmers and other households with strong ties to the land.* Farmers will have a vested interest in improving environmental conditions. They are also well positioned to care for and cultivate such systems, provided they have enough people to do the work. Farmers possess practical, hands-on skills working with plants, knowledge that transfers well to caring for trees (Vieira *et al.*, 2009; Hoch *et al.*, 2012). The most innovative individuals often possess a combination of education, knowledge, physical skills, aptitudes, and real-life experience (Francisco, 2010). Providing farmers with training or education to complement their intimate knowledge of tools, farm craft, and the land could not only encourage participation, but foster innovation in on-farm tree planting.

6. *Provide the space, flexibility and resources for smallholders to innovate*. Stressful situations and commitment to a goal (in this case, of staying on the land) can provide fertile ground for innovation and creativity (Boserup, 1965; Bunce & West, 1994; West, 2002; West *et al.*, 2006*a*). Conditions that foster innovation include: 1) spaces to experiment and interact with others; 2) channels of communication, both within communities and with other people and organizations who have specific expertise and knowledge; 3) access to relevant information and training; 4) travel; and, 5) education (Bunce & West, 1994; Sveiby, 2001; West, 2002; Collins & Smith, 2006; West *et al.*, 2006*a*; Francisco, 2010). Projects could create community spaces (a community tree nursery, a communal reserve, or

building) where people can meet, share information, and plan (Walters *et al.*, 1999; Isaac *et al.*, 2007; Bodin & Crona, 2009; Shiferaw *et al.*, 2009). Project managers could organize community exchanges where members of one community travel to locations where others have undertaken such projects to learn from their experience. Fostering innovation is essentially an exercise in engineering chance – the same conditions that work in one place fail in others. Thus, it's not only creating the conditions, but also the quality of the interactions that occur. Outside donors and regional, national, or international NGOs and agencies could play a key role by providing access to resources and research on tree planting and restoration directly to the local people who will apply it on the ground.

Conclusion

Crisis restoration is the restoration of the future. As more communities and regions reach a 'post frontier' state, more people will experience the effects of deforestation. And, as has been observed in both temperate and tropical regions, rather than migrating to cities, some of those people will choose to remain on the land (Desmarais, 2002). Reframing restoration as a forward-looking solution to current environmental problems, rather than an attempt to recreate lost forests, can make such projects relevant, useful, and desired by local communities. The Intag example shows that communities experiencing severe environmental problems are willing to participate in community-based restoration projects if they believe such projects will ameliorate current conditions. Focusing restoration efforts on the communities and households who are both able to participate in it and stand to benefit most from restoration can lead to high participation rates, high levels of community and on-farm engagement with the projects, and can foster new and innovate ways of using trees in rural farming systems. Part of a grassroots, sustainable farming movement, tree-planting projects provide hope for a farming future in these communities.

This model of restoration will become increasingly important as more people experience negative effects of past deforestation, or seek solutions to adapt to a changing climate. Because of the unique role that cloud forests play in the hydrological cycle (Bruijnzeel *et al.*, 2010*b*), Andean communities, and communities in cloud forest regions elsewhere experiencing water shortages and other problems related to deforestation stand to benefit greatly from restoration projects. Targeting restoration projects to meet the concerns of communities, however, is a principle that can and should be applied in other regions and contexts. If we assume that people want to plant trees, maybe they will.

Table 5.1: Community and household characteristics in each community. Values are the average for households in each community followed by the standard deviation (unless stated otherwise).

Community	*EP*	*EC*	*LE*	*PV*	*Total*
Number of households surveyed	27	37	23	31	118
Number of households, total	31	44	25	34	134
Participation rate in surveys (%)	87	84	92	92	88
Elevation (of school, masl)	1850	1960	1860	1910	
Income ($US)	7714 ± 6645	5640 ± 4938	6065 ± 3409	8638 ± 7987	6985 ± 6119
Land owned (ha)	10 ± 19	10 ± 11	19 ± 25	14 ± 27	13 ± 21
Land sharecropped or rented (ha)	1 ± 3	0 ± 1	1 ± 3	1 ± 2	1 ± 2
Primary forest (% of total land)	9 ± 16	9 ± 19	13 ± 22	13 ± 23	11 ± 20
Pasture (% of total land)	30 ± 29	37 ± 31	28 ± 29	25 ± 30	30 ± 30
Number of cows owned	4 ± 6	4 ± 6	5 ± 5	6 ± 12	5 ± 8
Vehicle (% households who own one)	7	19	17	23	17
Productive assets ($US)	255 ± 310	209 ± 241	1300 ± 4155	1728 ± 5210	831 ± 3271
Non-productive assets ($US)	1156 ± 798	1166 ± 803	928 ± 882	1201 ± 978	1126 ± 861
Education (years in school, household head)***	7 ± 4	5 ± 3	4 ± 3	5 ± 3	5 ± 3
Age household head (years)	49 ± 16	56 ± 18	57 ± 19	51 ± 17	53 ± 18
Number of adults (16-64)	2 ± 1	2 ± 1	2 ± 1	2 ± 1	2 ± 1
Land-based livelihood (% households)	78	62	77	70	71
Member of farmer's association (% households)***	52	14	30	38	32
Elected to local government (% households)	11	10	17	29	17

Community codes: EP – El Paraíso, EC – El Cristal, LE – La Esperanza, PV – Pueblo Viejo.
Levels of significance: * $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$
P-values were determined using ANOVA for continuous variables and Chi-squared for counts (i.e., % participation).

Table 5.2: Characteristics of households across livelihood strategies. Numbers are the mean and standard deviation unless stated otherwise.

	Subsistence farmers	*Market-oriented farmers*	*Cattle ranchers*	*Off-farm skilled workers*	*Day labourers*	*Overall average*
Number of households	47	14	23	15	19	118
Income ($US)***	5657 ± 3545	8135 ± 4094	9218 ± 8197	9977 ± 6191	4354 ± 7773	6984 ± 6199
Land owned (ha)***	7.1 ± 9.4	16.5 ± 15.5	28.8 ± 36	13.85 ± 16.2	3.7 ± 7.1	12.74 ± 21
Land rented or sharecropped (ha)	0.7 ± 1.1	1.5 ± 2.7	1.2 ± 3.2	0.5 ± 1.1	1.6 ± 3.2	1
Primary forest (% area of landholdings)**	6 ± 12	27 ± 33	18 ± 20	12 ± 22	2 ± 6	11 ± 19
Pasture (% area of landholdings)	34 ± 30	27 ± 22	41 ± 30	28 ± 31	15 ± 29	30 ± 30
Cows owned***	2.64 ± 3.0	4.64 ± 3.1	11.91 ± 13.3	5.33 ± 8.2	0.74 ± 1.2	4.72 ± 7.8
Vehicle (% households)**	6.0	21	22	40	16	19
Productive assets ($US)	674 ± 2980	507 ± 410	1477 ± 4450	1580 ± 5170	83.7	831 ± 3270
Non-productive assets ($US)**	971 ± 650	1340 ± 843	1228 ± 924	1606 ± 995	849 ± 1012	1126 ± 860
Age household head (years)	56.2 ± 16.4	47.9 ± 17.4	55.6 ± 17.2	46.4 ± 17.3	52.2 ± 21.1	53.2 ±17.7
# Adults (15-64 yrs)	2.0 ± 1.3	2.2 ± 1.1	2.1 ± 1.2	2.3 ± 1.1	1.7 ± 1.3	2.0 ± 1.2
# Dependents (<15, >65 yrs)***	1.7 ± 1.5	3.8 ± 2.4	2.0 ± 1.5	2.5 ± 1.4	2.6 ± 2.2	2.2 ± 1.8
Education (of household head, years)***	4.8 ± 2.5	5.8 ± 3.1	5.4 ± 4.0	7.9 ± 4.9	4.0 ± 2.1	5.3 ± 3.4
Farmer's associations (% households)	34.0	42.9	43.5	20.0	15.8	32.2
Elect. to local government (% households)	12.8	28.6	8.7	33.3	15.8	17.0
Trade (% households)	15	0	17	73	0	19

Levels of significance: * $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$
P-values were determined using ANOVA for continuous variables and Chi-squared for counts (i.e., % participation). Post-hoc tests for those variables that were significantly different: 1) *Income* Market-oriented farmers and cattle ranchers had marginally higher incomes than day labourers (Tukey post hoc, $p < 0.06$). 2) *Land owned* Cattle ranchers owned significantly more land than day labourers and subsistence farmers (Tukey post hoc, $p < 0.000$). 3) *Number of cows* Cattle ranchers had more cows than any other group (Tukey post hoc, $p < 0.05$). 4) *Education* Off-farm skilled workers had more years of education than day labourers and subsistence farmers (Tukey post hoc, $p < 0.01$). 5) *Trade* More off-farm skilled workers participated in a trade than any other group (Tukey post hoc, $p < 0.01$). 6) *Dependants* Market-oriented farmers had more dependants per household than cattle ranchers and subsistence farmers.

Table 5.3: Probit regressions for participation in tree planting

Variable		***Plant trees in reserves***		***Plant trees on farmland***	***Restore forest on farmland***	***Plant agroforestry systems on farmland***
Constant		1.50 (1.33)	0.051 (0.97)	-0.91 (1.50)	- 0.23 (1.40)	-0.37 (1.29)
Land owned (ha)		0.003 (0.009)	0.002 (0.009)	-0.018 (0.012)	0.012 (0.008)*	0.001 (0.008)
Productive assets ($100US)		0.007 (0.01)	0.006 (0.010)	0.12 (0.07)*	0.005 (0.005)	-0.001 (0.005)
Education (of household head, years)		0.022 (0.050)	0.041 (0.049)	0.16 (0.07)**	0.007 (0.054)	0.10 (0.053)**
Age household head (years)		0.003 (0.01)	0.001 (0.009)	0.01 (0.332)	-0.002 (0.01)	0.013 (0.01)
# Adults (15-64 yrs)		0.327 (0.142)**	0.331 (0.137)**	0.25 (0.16)*	0.24 (0.14)*	0.31 (0.13)**
Participate in civic life	Farmer's associations	0.562 (0.33)*	0.68 (0.32)**	0.76 (0.42)*	0.78 (0.32)**	0.73 (0.32)**
	Elected to government	0.64 (0.436)	0.69 (0.32)	1.08 (0.51) ***	0.50 (0.39)	1.08 (0.40) ***
Land-based livelihood[3]		0.38 (0.32)	0.50 (0.31)*	0.93 (0.35)***	0.44 (0.38)	0.56 (0.32)*
Community 1 (El Paraiso)		-0.66 (0.44)	n.a.	0.93 (0.57)*	0.48 (0.45)	0.69 (0.43)*
Community 2 (El Crystal)		-1.46 (0.39)***	n.a.	-0.36 (0.40)	0.42 (0.45)	-0.04 (0.38)
Community 3 (La Esperanza)		-0.38 (0.43)	n.a.	0.23 (0.45)	0.06 (0.47)	0.20 (0.40)
Paid [4]		n.a.	-0.99 (0.30)***	n.a.	n.a.	n.a.
Number observations		118	118	118	118	118
Pseudo r^2 [5]		0.46	0.40	0.55	0.34	0.45
Prob > Chi2		0.000	0.000	0.000	0.002	0.000
Log likelihood		111.2	118.7	91.6	94.7	115.2

*p < 0.10, ** p < 0.05, *** p < 0.01

[1] 'Green adopters' adopted at least two conservation-oriented technologies or techniques on their farms in the past 10 years.

[2] 'Organic farmers' do not used any synthetic fertilizers, pesticides, or herbicides, and have adopted the use of at least one type of organic fertilizer on their farms.

[3] Land-based livelihood indicates a farming (subsistence or market-oriented) or cattle ranching household.

[4] Paid indicates a community that was paid a daily wage to plant.

[5] Pseudo r^2: Nagelkerke

Table 5.4: The marginal effects for probit regressions (presented in Tables 5.3 and 5.6) for participating in tree planting and on-farm green farming techniques. The numbers given are the marginal effects of the regressions, with standard error in parentheses.
*p < 0.10, ** p < 0.05, *** p < 0.01

Variable		*Plant trees in reserves*		*Plant trees on farmland*	*Restore forest on farmland*	*Plant agroforestry systems on farmland*	*Green adopter*[1]	*Organic farmer*
Land owned (ha)		0.0.001 (0.003)	0.0.001 (0.002)	-0.004 (0.004)	0.003 (0.003)*	0.0003 (0.002)	0.003 (0.004)	-0.003 (0.006)
Productive assets ($100US)		0.0018 (0.003)	0.0017 (0.003)	0.026 (0.024)*	0.001 (0.001)	-0.0003 (0.001)	0.015 (0.021)	0.001 (0.001)*
Education household head (years)		0.006(0.014)	0.011 (0.015)	0.035 (0.029)**	0.0015 (0.013)	0.029 (0.019)**	0.012 (0.019)**	0.026 (0.034)***
Age household head (years)		0.0008 (0.002)	0.0002 (0.002)	0.0012 (0.055)	-0.0005(0.04)	0.003 (0.01)	-0.001 (0.002)	0.003 (0.006)*
# Adults (15-64 yrs)		0.090 (0.055)**	0.092 (0.056)**	0.055 (0.055)*	0.052 (0.044)*	0.085 (0.049)**	0.038 (0.051)	0.054 (0.072)**
Participate in civic life	Farmer's associations	0.15 (0.11)*	0.19 (0.12)**	0.17 (0.16)*	0.173 (0.122)**	0.190 (0.116)**	0.130 (0.085)***	0.164 (0.211)***
	Elected to government	0.17 (0.14)	0.16 (0.14)	0.23 (0.20)***	0.111 (0.111)	0.299 (0.156)***	-0.132 (0.170)	-0.110 (0.180)
Land-based livelihood[3]		0.10 (0.10)	0.14 (0.10)*	0.93 (0.35)***	0.098 (0.110)	0.153 (0.107)*	0.08 (0.112)	0.094 (0.141)
Community 1 (El Paraiso)		-0.17 (0.15)	n.a.	0.20 (0.20)*	0.106 (0.123)	0.170 (0.200)*	0.167 (0.200)	0.213 (0.291)*
Community 2 (El Crystal)		-0.39 (0.20)***	n.a.	-0.07 (0.13)	0.092 (0.131)	-0.051 (0.127)	0.051 (0.126)	0.195 (0.279)
Community 3 (La Esperanza)		-0.10 (0.13)	n.a.	0.013 (0.119)	0.06 (0.47)	0.073 (0.140)*	-0.073 (0.136)*	0.050 (0.181)*
Paid[4]		n.a.	-0.27 (0.14)***	n.a.	n.a.	n.a.	n.a.	n.a.

[1] 'Green adopters' adopted at least two conservation-oriented technologies or techniques on their farms in the past 10 years.
[2] 'Organic farmers' do not used any synthetic fertilizers, pesticides, or herbicides, and have adopted the use of at least one type of organic fertilizer on their farms.
[3] Land-based livelihood indicates a farming (subsistence or market-oriented) or cattle ranching household.
[4] Paid indicates a community that was paid a daily wage to plant

Table 5.5: Tobit models showing the extent of planting trees and adopting green technologies on private land.

Variable		*Diversity of tree systems*[1]	*Number of species planted*	***Number of trees planted***			
				Total	*% native*	*On-farm restoration*	*Agroforestry*
Constant		0.079 (0.82)	0.167 (1.316)	-44.4 (242.2)	-0.008 (0.278)	-200.014 (200.53)	-272.014 (241.365)
Land owned (ha)		0.001 (0.006)	-0.006 (0.010)	1.41 (1.79)	-0.001 (0.002)	1.068 (1.680)	1.122 (1.700)
Productive assets ($100US)		0.002 (0.004)	-0.004 (0.006)	-0.70 (1.10)	-0.006 (0.006)	0.248 (0.754)	-0.882 (1.060)
Education household head (years)		0.106 (0.044)**	0.246 (0.072)***	28.04 (13.02)**	0.015 (0.015)	1.181 (10.633)	33.738 (12.713)***
Age household head (years)		0.007 (0.009)	0.012 (0.014)	2.35 (2.67)	-0.002 (0.003)	0.4 (2.10)	4.21 (2.60)*
# Adults (15-64 yrs)		0.226 (0.114)**	0.361 (0.184)**	35.3 (33.41)	0.056 (0.038)	40.814 (27.744)	44.910 (32.258)
Participate in civic life	Farmer's associations	1.203 (0.278)***	1.907 (0.450)***	344.86 (80.77)***	195.948 (67.408)***	0.325 (0.093)***	310.220 (77.81)***
	Elected to government	0.729 (0.328)**	1.370 (0.529)***	145.97 (95.47)	130.618 (75.042)*	-0.105 (0.111)	146.89 (91.44)*
Land-based livelihood[3]		0.781 (0.289)***	0.961 (0.465)**	120.51 (85.91)	0.180 (0.106)*	63.348 (76.080)	62.70 (84.05)
Community 1 (El Paraiso)		0.497 (0.366)	1.437 (0.590)**	118.50 (106.23)	0.107 (0.117)	117.241 (88.444)	128.023 (102.531)
Community 2 (El Crystal)		-0.202 (0.349)	-0.119 (0.567)	-99.70 (105.6)	-0.082 (0.121)	55.456 (93.149)	-101.50 (105.893)
Community 3 (La Esperanza)		0.093 (0.363)	0.065 (0.598)	54.77 (106.62)	-0.146 (0.133)	73.526 (90.520)	74.83 (103.95)
Number observations		118	118	118	118	118	118
r^2		0.40	0.486	0.287	0.22	0.182	0.285
Prob > Chi2		0.000	0.000	0.000	0.000	0.002	0.001
Log likelihood		151.03	186.45	571.84	42.703	203.277	450.094

*p < 0.10, ** p < 0.05, *** p < 0.01

[1] Diversity of tree systems refers to the number of different ways that farmers planted trees on their land (in agroforestry systems, along roads, in pastures, to restore forests, around the house for shade, in orchards).

[3] Land-based livelihood indicates a farming (subsistence or market-oriented) or cattle ranching household.

Table 5.6: Regression models predicting participation in on-farm green farming techniques. The first two models (Green adopter and Organic farmer) are probits, and model three (Green technologies) is a tobit.

Variable		*1. Green adopter*[1]	*2. Organic farmer*[2]	*3. Green technologies*[3]
Number of households who adopted		28	18	na
Constant		3.2 (1.9)*	2.94 (2.51)	-0.37 (1.29)
Land owned (ha)		0.004 (0.009)	-0.025 (0.022)	0.001 (0.008)
Productive assets ($100US)		0.008 (0.006)	0.011 (0.006)*	-0.001 (0.005)
Education (of household head, years)		0.16 (0.07)**	0.22 (0.08)***	0.10 (0.053)**
Age household head (years)		0.018 (0.015)	0.034 (0.019)*	0.013 (0.01)
# Adults (15-64 yrs)		0.21 (0.17)	0.46 (0.20)**	0.31 (0.13)**
Participate in civic life	Farmer's associations	1.98 (0.40)***	0.73 (0.32)**	1.39 (0.51)***
	Elected to government	-0.47 (0.51)	1.08 (0.40) ***	-1.81 (0.85)
Land-based livelihood[3]		0.024 (0.45)	-0.80 (0.55)	0.56 (0.32)*
Community 1 (El Paraiso)		0.40 (0.49)	1.81 (0.86)**	0.69 (0.43)*
Community 2 (El Crystal)		0.06 (0.54)	1.65 (0.90)*	-0.04 (0.38)
Community 3 (La Esperanza)		-1.10 (0.67)*	0.41 (0.95)	0.20 (0.40)
Paid [4]		n.a.	n.a.	n.a.
Number observations		118	118	118
Pseudo r^2 [4]		0.65	0.56	0.45
Prob > Chi2		0.000	0.000	0.000
Log likelihood		62.4	50.3	115.2

[1] 'Green adopters' adopted at least two conservation-oriented technologies or techniques on their farms in the past 10 years.
[2] 'Organic farmers' do not used any synthetic fertilizers, pesticides, or herbicides, and have adopted the use of at least one type of organic fertilizer on their farms.
[3] 'Green technologies: refers to the number of green technologies that people employ on their farms, adopted within the past 10 years.
[4] Pseudo r^2 reported for models one and two is Nagelkerke. The actual r^2 is presented for model three.

Table 5.7: Participation in tree planting across different levels of participation in green technologies.

	Green adopter [1]	***Some green*** [2]	***No green*** [3]	***Total***
Number of households	28	40	50	118
Households planting trees in in community reserves (69 households)***	79%	63%	48%	60%
Households planting trees on private land (79 households) ***	93%	73%	43%	
Households planting in on-farm agroforestry (59 households)***	75%	57.5%	30%	50%
Households restoring forests on-farm (26 households)***	39%	27.5%	8%	22%
Number of trees planted on-farm (average)†**	22.6%	15.8%	141 ± 260	263 ± 358
% native trees planted on-farm† (average)**	3.9 ± 2.0	2.8 ± 1.7	7.7%	15.6%
Number of species planted on-farm†*			2.0 ± 1.2	2.9 ± 1.8

* $p < 0.1$, ** $p < 0.05$, *** $p<0.01$

P-values were determined using ANOVA for continuous variables and Chi-squared for counts (i.e., % participation). Tukey post-hoc tests were used to determine which groups were significantly different: 1) the *number of trees* planted differs significantly between no green and green adopters. 2) The *number of species planted* differs significantly between green adopters and both some green and no green

[1] 'green adopters' employ more than two kinds of green technologies on their farms (composters, systems used to create liquid fertilizer (*biol*), bio-gas-producing systems with pigs (*biodigestores*), wire systems for growing beans, or various types of green composts on their farms); [2] 'some green' households employ one or two types of green fertilizers or have adopted one of the above technologies; [3] 'no green' households do not use green technologies.

†Includes only results for households that planted trees on their farms (N=79).

Table 5.8: Participation in tree planting and on-farm conservation technologies across livelihood groups.

	Day labourers	*Subsistence farmers*	*Market-oriented farmers*	*Cattle ranchers*	*Off-farm skilled workers*	*Overall average*
Number of households	19	47	14	23	15	118
Plant trees on farms (% households)***	32	79	71	65	67	66
Agroforestry**	21	53	71	52	53	50
Restoration	0.5	28	21	26	20	23
Plant trees in reserves (% hh)	42	68	64	65	47	0.60
Number of trees planted (average)†*	184 ± 203	211 ± 296	382 ± 326	318 ± 335	301 ± 641	263 ± 359
Number of native trees (average)†*	11 ± 12	73 ± 123	100 ± 150	32 ± 68	25 ± 63	57 ± 109
% native trees (average)†	16 ± 19	20 ± 26	23 ± 27	7 ± 11	5 ± 13	16 ± 22
% fruit trees (average)†	1 ± 3	17 ± 35	2 ± 6	6 ± 18	30 ± 43	13 ± 30
% *aliso* (*Alnus nepalensis*) (average)	50 ± 31	58 ± 36	62 ± 33	69 ± 34	60 ± 41	60 ± 35
Number of species planted (average)†	2.3 ± 1.2	3.1 ± 1.9	2.9 ± 1.9	2.3 ± 1.2	2.6 ± 2.3	2.9 ± 1.8
Number of kinds of tree systems (average)[1]	0.8 ± 1.0	1.6 ± 1.6	1.4 ± 1.4	1.5 ± 1.7	0.9 ± 1.3	1.4 ± 1.5
Adopt 'green' technology (% households)	37.8	63.8	71.4	60.9	46.7	58
some green[2] (% households)	26	34	43	39	27	34
green adopter[3](% households)	11	30	29	22	20	24
Organic farmer[4] (% households)	11	17	7	14	20	

* p < 0.1, **p < 0.05, *** p<0.01
P-values were determined using ANOVA for continuous variables and Chi-squared for counts (i.e., % participation). Tukey post-hoc tests were used to determine which groups were significantly different.
[1]Diversity of tree systems refers to the number of different ways that farmers planted trees on their land (in agroforestry systems, along roads, in pastures, to restore forests, around the house for shade, in orchards). [2] 'some green' households employ one or two types of green fertilizers or have adopted one of the below technologies; [3]'green adopters' employ more than two kinds of green technologies on their farms (composters, systems used to create liquid fertilizer (*biol*), bio-gas-producing systems with pigs (*biodigestores*), wire systems for growing beans, or various types of green composts on their farms). [4] 'Organic farmers' do not used any synthetic fertilizers, pesticides, or herbicides, and have adopted the use of at least one type of organic fertilizer on their farms
†Includes only results for households that planted trees on their farms (N=79).

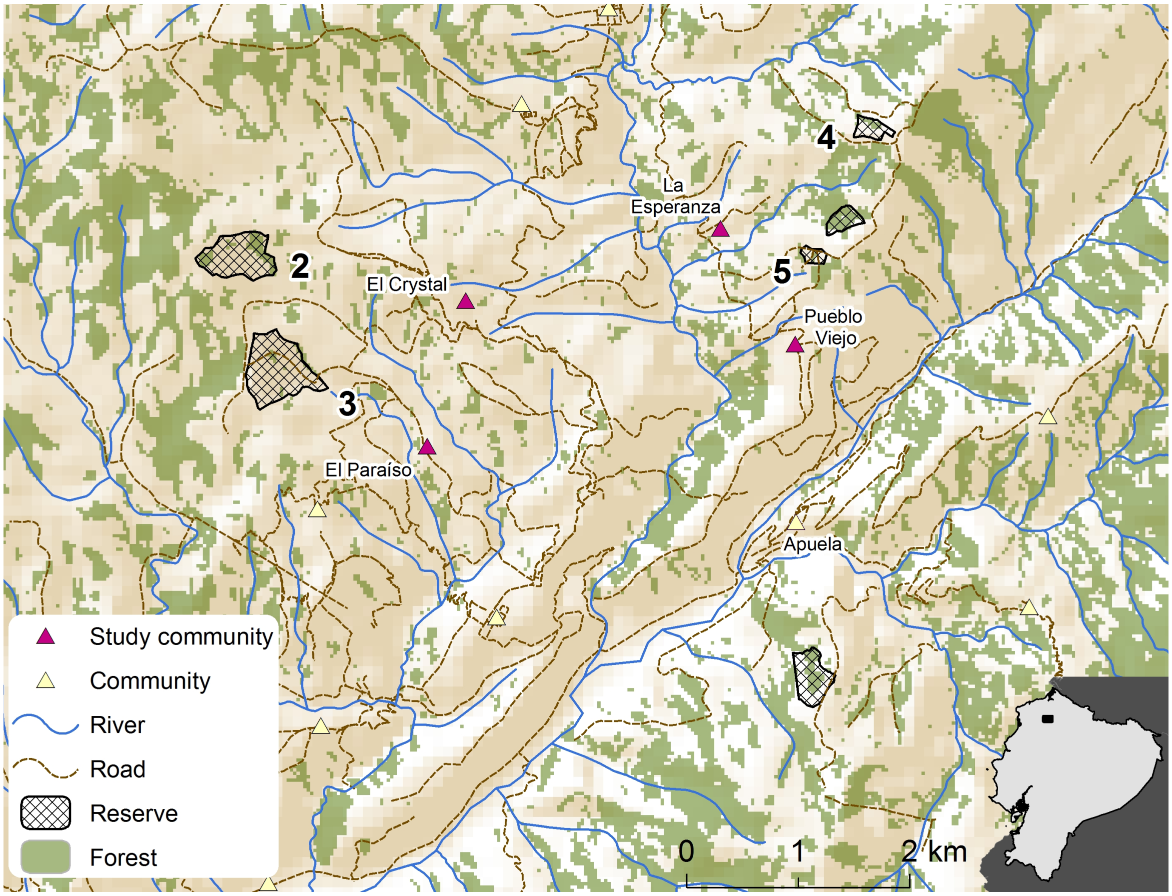

Figure 5.1: Map of the four study communities in the Intag Valley.

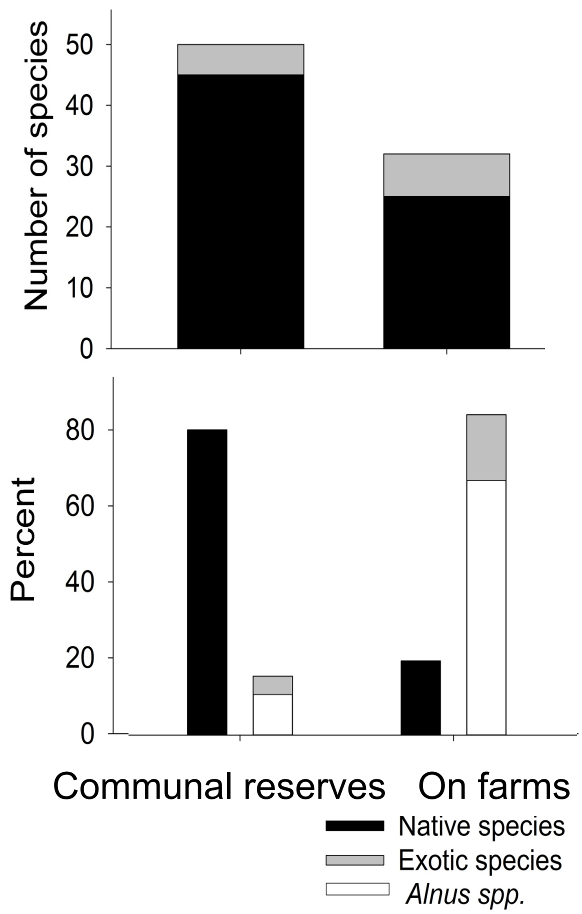

Figure 5.2: Native and exotic trees planted in communal watershed reserves and on private farms.

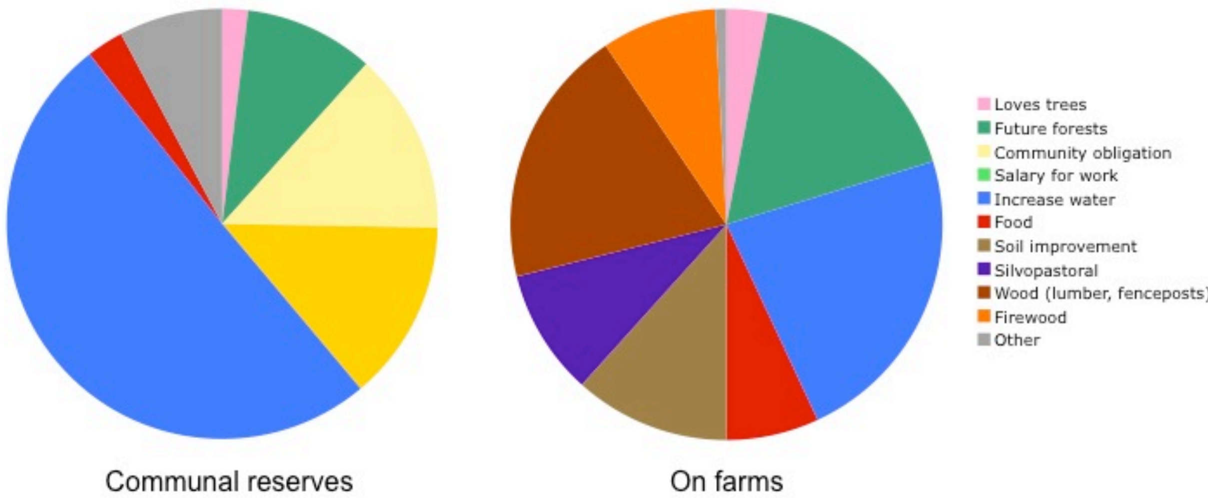

Figure 5.3: Household reasons and production goals for planting trees in communal watershed reserves and on farms.

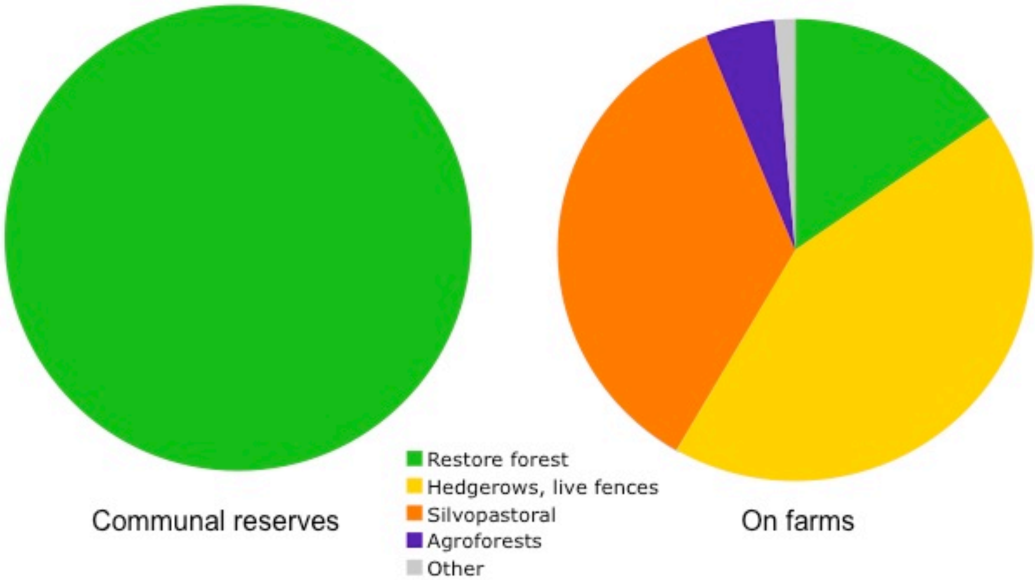

Figure 5.4: The percentage of trees planted in different systems in communal watershed reserves and on farms.

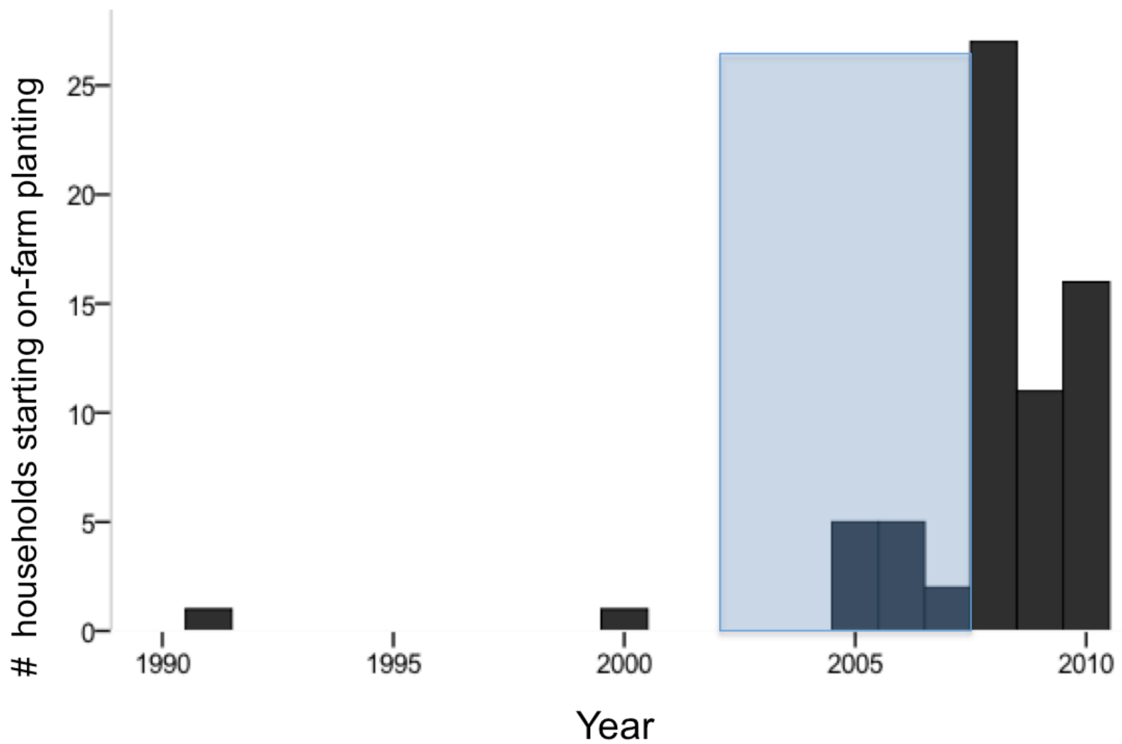

Figure 5.5: Number of households initiating tree planting on private land each year. Note that the communal projects were started between 2003 and 2007, as indicated by the blue box.

Some things you need to build a cloud forest

Imagination	Seeds
Curiosity	Soil
Practice	Stakes
Hands-on experience	Plastic bags
Intuition	Tarps
Sun	Shovels
Rain	Machetes
Wind	Wire
Clouds	Baskets
Botanical knowledge	Fence posts
Cultural knowledge	Lunch
Patience	Land
Community	People

CHAPTER 6: SYNTHESIS AND CONCLUSIONS

Community cloud forest restoration - Can biodiversity and human needs be reconciled?

When I started my doctorate degree in 2009, my primary goal was to answer the question: Can heavily cleared, yet very biodiverse, cloud forests be restored with the participation and support of local people? Fresh from a 15-month trip to some of the world's most threatened forests and landscapes and armed with a solid background in ecology, I was eager for solutions. But I also had the firm belief that conflict between human needs and ecological goals would be inevitable – restoring ecosystems would, at best, involve compromise at the expense of either human or ecological needs.[20] Through the course of this study I have come to believe that this is not necessarily the case – that synergies exist between people and forest conservation that managers can promote and build upon. I have also learned that identifying these synergies, which at times seem few and far between, requires optimism and hope to complement theory and rigorous methodology. By employing scientific methods rooted in the extensive literature on rural livelihoods, tree planting, forest and landscape ecology, and biogeography, this dissertation presents the results from a case study from Intag where a number of these synergies exist.

Restoration is an inherently hopeful endeavor. Building an ecosystem is hard work, requiring patience and optimism alongside fences and seeds. But although people can make a big difference by planting and maintaining trees, they only have so much control – at a certain point, they have to rely on natural processes to take over.

The restoration projects in Intag are far from perfect. They suffer from many of the same problems identified in community-based projects elsewhere, including resource capture by rural

[20] My initial dissertation title was "*Community cloud forest restoration: can biodiversity and human needs be reconciled?*", the word 'reconciled' indicating this inherent compromise.

elites (in one community) and a lack of resources for monitoring (Kumar, 2002; Persha & Andersson, 2014). By planting a selection of those species found in cloud forest alongside exotic species, they have also changed the species composition of cloud forests in the region. Primary forests in Intag are still being cleared while people reforest, although at much lower rates. However, because they ultimately 1) aided forest recovery; 2) increased forest tree diversity; 3) were widely accepted and participated in by local people; and, 4) were an integral part of a grassroots sustainable farming movement in the region, this case study illustrates a number of synergies between livelihoods and biodiversity conservation that are mediated through the practice and craft of growing, planting and cultivating trees. The Intag model of restoration holds considerate potential to benefit smallholder farmers and help conserve and restore local biodiversity in heavily deforested regions throughout the Andes. Studying why and how this model works and how it might be applied elsewhere was a primary goal of this thesis. I hope my analysis of this model will be used to guide best practices for other restoration projects in Intag, and in tropical montane forests in the Andes and elsewhere.

Through this research I aimed to ask and answer questions that are grounded in theory, but based on the local context in Intag. I first visited the Intag region on a reconnaissance trip in 2010. According to people there, I was the first researcher to survey households, collect diversity data in planted forests, and survey trees in upper-elevation cloud forests in the region. I met with community leaders and long-term residents to learn their research needs. Based on these conversations and my own observations, I modified my research questions and, in 2011, helped them set up a tree-growth monitoring program.

In addition to providing locally and regionally relevant data, each chapter of this thesis answers questions that have been put forward as top research questions and priorities in the fields of biodiversity conservation and restoration (Sunderlin *et al.*, 2003; Chazdon *et al.*, 2008; Bruijnzeel *et al.*, 2010b). Because people have altered the Earth's ecosystems to such a great extent (Foley *et al.*, 2007; Turner *et al.*, 2007; DeFries *et al.*, 2012), the literature on restoring ecosystems has expanded rapidly in the last decade. However, there is still much to learn about how on-the-ground projects affect both people and the environment, particularly in cloud forest regions (Lamb *et al.*, 2005; Chazdon *et al.*, 2008; Aide *et al.*, 2010; Bruijnzeel *et al.*, 2010b). Key governance and managerial challenges include how to engage local people and communities in restoration, and how to promote long-term stewardship of restored sites and secondary forests (Wunder, 2013; Pirard *et al.*, 2014; Vieira *et al.*, 2014). On the other hand, although many researchers and policies emphasize that restoration projects should be carried out with and by local communities, by integrating ecological restoration methods with local knowledge, tools and skills, empirical data on how well this works to restore tropical biodiversity is still scarce (Chokkalingam *et al.*, 2005; Chazdon, 2008; Chazdon *et al.*, 2008; Parrotta, 2010; Uprety *et al.*, 2012). Chapters 2, 3 and 4 tackle these questions by examining the ecological outcomes and synergies of *community-based* restoration: why people restore forests, how restoration fits (or does not fit) into people's livelihoods, and how their practices affect the ecology and diversity of the forests they restore. Of course, the first step to conserving and restoring ecosystems is developing a detailed understanding of their ecology (SER, 2004). Andean cloud forests hold a disproportionate percentage of the world's plant and animal species, are heavily cleared, and are still relatively poorly understood in terms of the spatial distribution of biodiversity within them (Bruijnzeel *et al.*, 2010*a*). The comparative study presented in Chapter 3 shows that because the tree species in mid-elevation cloud forests vary so much from one ridge to the next, protecting even small forest fragments is important for local restoration efforts and for conserving landscape-level diversity.

How best to 'scale up' local projects to the landscape scale is another key challenge to restoring and conserving biodiverse forests (Holl *et al.*, 2000; Parrotta, 2010; Calmon *et al.*, 2011; Melo *et al.*, 2013), particularly in very biodiverse regions such as cloud forests (Aide *et al.*, 2010). Applying the forest-transition framework to examine the drivers and outcomes of forest-change dynamics, in Chapter 2, I identified a novel pathway by which regional transitions occur: the ecosystem-scarcity path, in which local needs for ecosystem services resulted in a net increase in forest cover. The drivers and ecological outcomes of ecosystem-service scarcity transitions are different from other types of transitions commonly discussed and applied in the literature (Rudel *et al.*, 2005; Farley, 2007; Rudel, 2010; Aide *et al.*, 2013). The resulting landscapes could well benefit local communities, but biodiversity conservation benefits of such transitions could be enhanced by providing additional incentives to conserve primary forests.

To conclude this dissertation, below I summarize the main findings and original contributions of each chapter, how the chapters relate to one another, and how the findings from this case study can be applied to improve cloud forest conservation efforts in Intag, in the Andes, and globally. I then provide ideas for future research directions in the field.

Context is key

As a whole, this dissertation shows that conserving and restoring forest in heavily degraded Andean landscapes requires context-specific action. The main findings from each chapter – from the importance of conserving small forest fragments to the idea that environmental crises can motivate tree planting and drive forest transitions – apply specifically to heavily deforested landscapes. But this does not mean their relevance and utility are limited to the local context in Intag. Today, most Andean cloud forests have been either cleared or fragmented (Daugherty, 1973; Young & Leon, 1995*a*; Jokisch & Lair, 2002; Williams-Linera, 2002; Bubb, 2004; Mulligan & Burke, 2005*a*; Mulligan & Burke, 2005*b*; Echeverría *et al.*, 2007; Scatena *et al.*, 2010). Environmental problems associated with deforestation, such as soil erosion and reduced or redistributed water supply, likely threaten people's ability to farm and thrive in many of these regions (Bebbington, 1993; Hecht, 2010). Most of these problems are predicted to worsen in coming years with continued deforestation and global climate change (Bruijnzeel, 2004; Buytaert *et al.*, 2011; Grêt-Regamey *et al.*, 2012). Thus, results may well apply to farming communities throughout the Andes. In this context, community-based restoration can be a powerful tool to restore forest cover and tree diversity more quickly than natural regeneration, with the added benefit of teaching people the technique and on-farm benefits of arboriculture. With the current global policy emphasis on tree planting, many resources are currently available for tree-planting projects; identifying ways to use these resources constructively is a major research and policy challenge (Agrawal & Angelsen, 2009; Wilson, 2013; Holmes & Potvin, 2014). The results presented here can help address that challenge in deforested Andean landscapes.

The ecosystem service scarcity forest transition path

Forest transitions occur through multiple paths and drivers, are scale dependent, and have different ecological and social outcomes depending on when, where and how they occur (Nanni & Grau; Rudel *et al.*, 2005; Satake & Rudel, 2007; Rudel, 2010; Redo *et al.*, 2012; Angelsen & Rudel, 2013). Chapter 2 of this dissertation makes a novel contribution to the literature by identifying a new mechanism for the forest transition: the ecosystem-service scarcity path. To date, two main paths have been identified – the 'economic development' path, and the 'forest (wood) scarcity' path (Rudel *et al.*, 2005). Like the wood-scarcity path, in which governments, industries or smallholders plant trees to obtain timber or firewood when wood becomes scarce (Rudel *et al.*, 2005), ecosystem-service scarcity forest transitions occur when: 1) forest cover around communities is depleted to such low levels that people are affected by a decline in forest ecosystem services; 2) people make the link between forest cover and declining ecosystem services; 3) people have or are taught techniques to reforest degraded lands, and have the resources to do so; and, 4) ultimately, these actions result in a net increase in forest cover. It should be noted that although studied at the local scale in Intag, such transitions could theoretically occur at the global scale through demand for ecosystem services such as carbon sequestration.

Satake and Rudel (2007) hypothesize that as ecosystem services decline during deforestation, local people will benefit less from forests and so will value and protect them less, resulting in a 'poverty trap' of deforestation and land degradation. My results suggest the opposite is possible – that the lack of services compared to historical conditions can motivate people to restore forests. Making the link between forests and the services they provide is thus a critical component to spur ecosystem-service transitions. Local environmental NGOs and farmers associations are well positioned to help farmers make this link because connecting forest ecosystem services to local needs and livelihoods requires an intimate knowledge of the environmental problems people are facing.

The ecosystem service transition in Intag led to a spatial redistribution of forest cover contrary to what would be predicted by other forest transition paths. Although forests continued to be cleared in 'marginal' highland areas, they returned in and around people's farms and villages. This result suggests that depopulation of the Andes by rural to urban migration is certainly not the only path for reforestation (Aide *et al.*, 2010; Hecht, 2010; Aide *et al.*, 2013) because farming and trees are not necessarily in opposition, as is often postulated. Chapters 4 and 5 highlight a number of synergies between restoration, on-farm tree planting, and forest conservation. Nor will Andean transitions necessarily rely on government-funded tree plantations (Farley, 2007), which may be beyond the reach of smaller landholders. Grassroots tree-planting initiatives can lead to increases in forest cover, and arguably provide the most benefits to local people and thus the greatest incentives to maintain and protect forests.

Ecosystem service-driven transitions will produce different types of forests from the economic-development and wood-scarcity paths. Transitions result from economic development when farmers abandon marginal agricultural land on which forests regenerate spontaneously (Rudel *et al.*, 2005). Although woody biomass can recover quickly and sequester carbon in naturally regenerating secondary forests, they often contain different species from primary forests (Chazdon *et al.*, 2008; Liebsch *et al.*, 2008; Letcher & Chazdon, 2009; Bonner *et al.*, 2013). Wood-scarcity-driven transitions occur when people or governments plant trees to produce wood. At the national level, this typically results in an increase in low-diversity, high-carbon industrial timber plantations (Farley, 2007; Sloan, 2008; Rudel, 2010). In an ecosystem service transition, the types of forests that return will likely depend on the ecosystem services people lack. In Intag, the demand for water prompted people to *restore* forests – natural regeneration was too slow, and monoculture plantations were too low in biodiversity (and thus vulnerable to pests and disease, as well as unattractive to donors) to be self-sustaining (C. Zorrilla, pers. comm., 2010). On the other hand, declining soil fertility from overuse and erosion, combined with the expense, perceived inutility, and negative health effects of chemical fertilizers in steep, wet environments, led people to adopt agroforestry systems and plant trees along hedges and fences, a very different approach to integrating trees into the landscape.

Although overall forest cover in the region increased and clearing rates decreased precipitously from the previous period, deforestation continued in the uplands. Thus, primary forests decreased in area while secondary forests increased. Chapters 3 and 4 of this thesis examine the impacts of this type of forest transition on the ecology and biodiversity of forests in the region.

Forest clearing declined from 37% in the 1990s to 23% in the 2000s. Based on historical patterns, in the 2000s over 1000 hectares of forest were thus 'spared' from deforestation. But selecting for local ecosystem services also appears to have redistributed forest cover in a way that implies a parallel redistribution of forest ecosystem services. Because older forests continued to be cleared, the services they provide to a much greater extent than secondary forests (e.g., forest biodiversity, Chapters 3 and 4) will continue to decline, although at slower rates. At the same time, people report that the secondary forests around communities provide ecosystem services essential for farming in the region (Chapter 3, Chapter 5), perceived benefits which appear to have increased forest stewardship. Although ultimately a 'win-win' for people and forests, incentives to conserve primary forests as people replant could potentially make the outcomes even better. Chapters 3 and 4 of this thesis examine the impacts of this type of forest transition on the ecology and biodiversity of forests in the region.

How do 'ecosystem-scarcity-driven' transitions affect regional forest biodiversity?

As a whole this dissertation contributes to the forest transition literature by providing detailed ecological and socioeconomic data on the drivers and outcomes of this local transition.

Chapters 3 and 4 show that, at both the stand and landscape levels, primary forests in Intag contain far more tree species than secondary forests. Thus, even though a transition occurred, it is likely that forest tree species richness declined over this period. Ultimately, expanding secondary forests and declining primary forests will likely result in forests that are more homogeneous across the region (Williams-Linera *et al.*, 2013; Holl & Zahawi, 2014), because primary forests appear more variable in space (Chapter 3) than secondary forests (Chapter 4). However, when compared to land-use choices from the previous decade, forest recovery has increased dramatically, forest clearing has slowed, and people seem intent on continuing to reforest for the foreseeable future (Chapter 2, Chapter 5). Despite its limitations, this forest transition is indeed good news for cloud forest recovery and biodiversity. In the following sections, I summarize the results of Chapters 3 and 4 and the implications of these findings for conserving and restoring Andean cloud forests.

Tree communities in small cloud forest patches are unique

Because the tropical Andes have highly heterogeneous topography and a history of rapid geological upheaval and climatic shifts, they contain a disproportionate number of the world's species in a tiny fraction of its area (Doumenge *et al.*, 1995; Gentry *et al.*, 1995; Myers *et al.*, 2000; Bruijnzeel, 2004; Brooks *et al.*, 2006; Bruijnzeel *et al.*, 2010*a*; Homeier *et al.*, 2010; Mulligan, 2010; Herzog *et al.*, 2011; Lippok *et al.*, 2014). Past work has demonstrated that communities of plants and animals in Andean forests change rapidly over elevation gradients (Givnish, 1998 Cardelus et al. 2006, Jankowski et al. 2013, Salazar et al. 2013, Williams-Linera et al. 2013; Watkins *et al.*, 2006; Jankowski *et al.*, 2013; Salazar *et al.*, 2013; Williams-Linera *et al.*, 2013). In Chapter 3, by comparing cloud forest patches in the Intag Valley, I contribute to this literature by demonstrating that tree communities at the upper limit of the Andean 'biodiversity bulge' are distinct from one another over a small spatial scale, even within the same narrow elevation range. Because these forests are so variable and so extensively cleared, even small forest fragments contribute substantially to landscape-level forest diversity.

In 1992, E.O. Wilson coined the term "Centinelian extinctions" – mass extinctions that happen frequently but so silently as to be unobservable, unless you happen to be in the right place at the right time (pg. 244, Wilson 1992) – based on a cloud-forest-covered ridge only 100 km southeast of Intag. Eight years after botanists Alwyn Gentry and Calaway Dodson first surveyed Centinela, the forests had been almost completely clear cut, taking with them an estimated 90 species of endemic plants (Dodson & Gentry, 1991). Although historical plant surveys of Intag's forests are lacking, given their extensive clearing and highly variable species compositions, it is certain that this region has lost many species unknown to science. But Chapter 3 shows that despite this, remnant forest patches with high species richness provide an opportunity to restore and conserve remaining biodiversity. In addition to large areas of forest, future conservation efforts in Intag should protect small patches wherever possible.

Findings in Chapter 3 also have implications for restoring cloud forest. Restoring a cloud forest to a 'historical reference state' – an often stated, but somewhat controversial, goal of ecological restoration (Harris & van Diggelen, 2006; Higgs, 2003; Temperton *et al.*, 2004; Menninger & Palmer, 2006; Palmer *et al.*, 2006; Stanturf *et al.*, 2014) – is an unrealistic goal with the knowledge and technology we possess now. First, historical data is lacking because many Andean cloud forests were cleared before they were surveyed (Wilson, 1992). Second, because plant communities vary so much and are affected by multiple environmental and spatial variables operating on different scales, it is difficult to deduce past species compositions based on current ones (Grubb, 1977 Richter et al. 2009, Bruijnzeel et al. 2010, Homeier et al. 2010, Martin et al. 2010, Bach and Gradstein 2011, Williams-Linera et al. 2013; Lieberman *et al.*, 1996; Beck *et al.*, 2008; Richter *et al.*, 2009; Bruijnzeel *et al.*, 2010b; Homeier *et al.*, 2010; Bach & Gradstein, 2011; Martin *et al.*, 2011; Williams-Linera *et al.*, 2013). Third, even if we could deduce past species compositions from existing forests, cloud forests contain such a high proportion of endemic species that many of their original species have been lost (Dodson & Gentry, 1991; Wilson, 1992). Once a cloud forest is gone, it's gone – restoration cannot replicate it, and should not be used as a substitute for conservation.

However, even if replicating a historical reference state is an unrealistic end goal, it can still be used as a guide. Rather than taking a photo-realistic approach to restoration, managers can maximize the conservation potential of restoration by striving to create forests composed of at least a subset of the same, locally-adapted species that were likely there in the past. Local remnant forests are an important source of both information and propagules for restoration efforts. To maintain species heterogeneity on the landscape, restoration projects require local seed sources for both human-mediated and passive regeneration (Schelhas & Greenberg, 1996; Holl, 1999; McKinney & Lockwood, 1999; Holl, 2002; Lugo & Helmer, 2004; Rhemtulla *et al.*, 2007; Jacob, 2014). Following planting, succession in recovering forests occurs more quickly (and perhaps more overall) the closer they are to remnant primary forest patches (Aide & Cavelier, 1994; Holl, 1999; Chazdon, 2003). Conserving primary forest patches is thus essential to both maintain and restore cloud forest diversity.

In otherwise cleared areas, small forest patches also provide local farming communities with a myriad of ecosystem services (e.g., reducing flooding, soil retention, firewood, food, and fodder) (Schelhas & Greenburg 1996, Wunder 1996, Bruijnzeel 2004, Grêt-Regamey *et al.*

2012). Some of these services, such as erosion control and seasonal flow regulation, will become more important as communities there adapt to climate change.

Community-based restoration and biodiversity conservation

Recent experimental plot studies have shown that planting trees can aid short-term tropical forest recovery on degraded land by changing environmental conditions and enhancing seed dispersal (Parrotta & Knowles, 1999; Ruiz-Jaén & Aide, 2005; Ren *et al*., 2007; Pena-Domene *et al.*, 2013; Zahawi *et al.*, 2013). The study presented in Chapter 4 is the first, to my knowledge, to show through a replicated design that similar results can be achieved through community-based efforts incorporating local knowledge and species preferences. It is also the first multi-site study to compare the diversity of restored forests to spontaneously regenerating forests in Andean landscapes. I found that planting locally 'useful' species of trees increased both tree diversity and the number of animal-dispersed species, jump-starting succession in young secondary forests on degraded pastures. Community-based tree planting efforts have great potential to aid forest recovery and increase forest biodiversity in degraded pastures, which are common in Andean landscapes (Aide & Cavelier, 1994; Sarmiento *et al.*, 1995; Young & León, 1995*b*; Baillie, 1996; Rhoades *et al.*, 1998; Jokisch & Lair, 2002; Sarmiento, 2002; Aide *et al.*, 2010; Ortega-Pieck *et al.*, 2011).

At the same time, these young restored forests are 'novel' additions to the landscape (Lugo, 2009)**.** Even though planting trees can increase forest diversity and accelerate succession over natural regeneration, because people's actions prior to restoration (e.g., planting exotic pasture grass) altered ecosystems to such a great extent, and because they planted different species and proportions of species from those found in primary cloud forests, young restored forests still support distinct combinations of species (Chapter 4). Longer-term studies are needed to determine if restored forests will, by accumulating more species over time, become more similar to neighbouring primary forests, or if these human legacies will lead to self-perpetuating novel forests with distinct species communities (Hobbs *et al.*, 2009; Lugo, 2009).

Restoring forest function and ecosystem services

Engaging rural people in community-based conservation projects is a major challenge for environmental and development agencies alike. Although ecological restoration seems especially well suited for smallholder participation – requiring place-specific knowledge, land-based skills and tools, and local stewardship – why and how local people participate in tropical forest restoration is still little investigated. Chapter 5 is the first study to use an asset-based livelihood approach to examine household-level participation in Andean forest restoration. It shows that households experiencing environmental 'crisis' will participate in restoration regardless of their landholdings or wealth. The environmental and social context into which a restoration project is introduced will affect which households participate, and why.

In Intag, a wide range of households participated in community-based restoration projects. Rather than immediate financial gain, the primary motivation was to restore and sustain ecosystem services essential for farming. Although different ecosystem services are derived from communal and on-farm planting, similar households participated in both. Because people planted trees to invest in their future ability to farm, those households most involved (and thus perhaps most dedicated to remaining) in their communities were most likely to restore forests and plant on-farm trees. Additional household labour also increased a household's ability to participate. Households that planted on-farm trees were also more land-reliant and well-educated, but land holdings and other measures of wealth did not predict either the extent of tree planting nor a household's adoption of the practice.

When compared to the substantial number of studies that show a relationship between farm size and planting, my results suggest that the environmental and social context in which restoration and tree-planting projects are introduced affects which households will participate in them. In Intag the environment was degraded to the point that peoples' ability to farm was compromised. People were motivated to participate because of the local ecosystem services that forests provide. In particular, farmers and ranchers restored forests and planted on-farm trees to improve water supply, restore soil fertility, and create forests for future use and generations.

Restoring forests is especially well suited to communal land: with benefits that are dispersed, less obvious, and require relatively large areas of land to be realized, they fit the typology of land uses that have been managed communally for centuries in different environments (including mountains and highlands) (Netting, 1972*b*, 1976; Ostrom, 1985; Menzies, 2014). In addition, my findings suggest that because many people contribute resources and knowledge to restoration, and because the risk of failure is both shared and diminished, communal projects can provide a low-risk, low-cost environment to learn about and experiment with tree planting. This can lead to restored forests that are both larger and more diverse than those installed on private land, and can give farmers the knowledge and tools to plant trees on their own private farms. Chapter 5 also thus shows that communal restoration projects can facilitate and motivate on-farm tree planting, extending the benefits of restoration beyond the locally restored site and into the landscape – a finding which, I believe, is a novel contribution to this literature. Restoring forest under small-scale communal land-tenure arrangements can be an effective way to increase both local forest cover and awareness about what services forests provide, thereby providing fertile ground for learning about and innovating with trees.

Chapter 5 also shows that peasant farming practices and forests are not necessarily in conflict. Farmers, not less land-reliant wage and salary earners, tended to participate most extensively in tree planting, experimenting with ‘new’ tree species and installing trees in a variety of different on-farm tree systems. They have the most to gain from having tree planting succeed, and stand to lose the most if it does not.

Implications for restoring and conserving forests with and for communities

Taken together, the four chapters of this dissertation illustrate several powerful synergies between sustaining rural livelihoods and conserving biodiversity through ecological restoration in deforested Andean landscapes. First, planting locally useful species – especially animal-dispersed ones – can both jump-start forest succession and increase forest biodiversity more quickly, and perhaps more overall, than letting forests regenerate naturally. Faster regeneration also means local people will benefit from more forest ecosystem services more quickly.

Second, because of the work invested in this process, and the motivation to restore them for the ecosystem services they provide, people tend to value these forests highly. Planting trees is seen as an investment in the future. In Intag, each household spent up to 50 days a year restoring forests at the start of the projects – a considerable amount of time spent off their own farms. People often reported being surprised at how hard it is to bring back the forests they originally had for free. Because of this effort, people reported that planted forests are not to be cleared.[21] In this sense, restoration cannot only complement conservation ecologically, by expanding and repairing forest habitat, but also socially, by having people realize how hard it can be to rebuild forests. The switch from clearing to planting in and around communities represents a fundamental shift in the way people view and interact with forests in the region, and likely contributed to the local forest transition that occurred over the 10 years in which the projects were implemented (Chapter 2).

Below I present four recommendations to maximize these and other synergies in community-based restoration projects that stem from the findings presented in this thesis.

1) Target restoration to degraded environments

To maximize the ecological and social benefits of restoration, agencies and NGOs should focus restoration activities on heavily degraded areas. Although this might seem obvious, in temperate environments managers will employ the opposite tactic – they often prioritize environments with the greatest potential to be returned to a pristine state. But implementing restoration projects with communities experiencing environmental crisis can maximize the benefits for both people and forests. Because forest ecosystem services are scarce, smallholder farmers have a built-in motivation for restoring forests – to bring back the ecosystem services forests used to provide and sustain rural farming culture into the future. Heavily degraded tropical environments are also most in need of restoration because forests will be slow or unable to recover unassisted. Thus, money specifically designated for tree-planting activities (through carbon sales, tree campaigns, and PES schemes, for example) could have the greatest impact in these environments. This approach is especially important in heavily deforested regions like the

[21] This attitude is apparent in the nomenclature: although people refer to young secondary forests as *chaparro* (which in Intag means scrubby brushland), both restored and primary forests are *bosque* (forest).

Andes, where conserving and restoring remnant forests is critically important to conserving global biodiversity.

2) Conserve even small patches of primary cloud forest

Chapter 3 shows that protecting small forest fragments is invaluable to conserving and restoring landscape-level cloud forest biodiversity. In Intag, as in other heavily deforested regions in Latin America, remnant forest patches are often located on private land (Schelhas & Greenberg, 1996; Agrawal *et al.*, 2008). To conserve cloud forests, conservation programs and policies that are accessible and attractive to smallholders[22] should be specifically targeted to promote conserving small forest fragments in strategic places – i.e., heavily deforested areas, or areas with currently high clearing rates (Myers *et al.*, 2000). These incentives are an important complement to 'crisis' restoration efforts, which, as was demonstrated in Chapter 2, are vulnerable to clearing even in the midst of reforestation efforts. If cleared forests are far from communities, people may be less likely to conserve them because the benefits they provide may be fewer, less obvious, or less immediately important.

3) Restoration is not a substitute for conservation

Protecting remnant forests is also important because restoration cannot substitute for conserving primary forests (Chapter 3, Chapter 4). My results, combined with other studies across the tropics, cast doubts on environmental policies being planned or implemented in Latin American countries (including Colombia, home to a significant portion of remaining Andean

[22] In Ecuador, some of these policies already exist – the Sociobosque program compensates private landholders for leaving forests on their lands, and there are numerous examples of other Payment for Environmental Service projects aimed at conserving forest for water on a small scale (Becker, 2003; Wunder & Albán, 2008; De Koning *et al.*, 2011; Wunder, 2011; MAE, 2014).

cloud forest (Mulligan, 2010) that promise ‘zero biodiversity loss’ when implementing extractive or industrial activities (Maron *et al.*, 2012; Gardner *et al.*, 2013; Sarmiento, 2013). Although recognizing the need to conserve biodiversity is certainly a positive step for conservation, it is unrealistic to promise that no species will be lost during industrial activities, especially in cloud forest areas where ecosystems vary greatly from one mountaintop to the next. These policies imply an inflated degree of confidence in our ability to manipulate forests (Maron *et al.*, 2012; Gardner *et al.*, 2013). At this point, neither ecologists nor engineers have the baseline knowledge about the distributions and environmental tolerances of many tropical forest species to completely restore cloud forests (Aide *et al.*, 2010; Bruijnzeel *et al.*, 2010*a*; Herzog *et al.*, 2011; Maron *et al.*, 2012; Gardner *et al.*, 2013). Because mining is a major threat to cloud forest conservation throughout the Andes, including in the Intag region, there is a very real concern that such policies will provide governments with an excuse for clearing irreplaceable cloud forests (Bubb, 2004; Bebbington *et al.*, 2008; Bruijnzeel *et al.*, 2010*a*; Kocian *et al.*, 2011; Buchanan, 2013).

4) Empower and work with local organizations

Local people emphasized restoring forest function rather than forest biodiversity. But the practice of restoring ecosystems emphasizes biodiversity as a major restoration goal. The local environmental NGO DECOIN, whose mandate is environmental protection, played a major (and impressively successful) role in reconciling these different endpoints between international donors (who emphasized biodiversity conservation) and local people (who needed water). Providing the initial impetus to plant by communicating to local people that ‘forests’ provide water, DECOIN encouraged them to select species to plant. But they also emphasized that most of these species had to be native, because exotic species could introduce plagues, lower genetic and species diversity, and would ultimately produce less resilient forests than native species adapted to local environmental conditions. In their reforestation manual, a document that consists of detailed technical methods alongside locally targeted information on why forests and native species are important, DECOIN states that “ *Es indispensable estar consiente que una plantación de arboles no es un bosque.* (It is important to remember that a tree plantation is not a forest.)”, referring to plantations of non-native trees (Zorrilla, 2010). Their message: forests, not just trees, were required to bring back water. Local agencies can thus facilitate restoration efforts by integrating local knowledge with ecological restoration best practices to present projects in a way that entices local people to participate.

Potential restoration pitfalls and future work

Although ‘crisis restoration’ holds great promise to conserve both forest biodiversity and rural livelihoods in degraded Andean regions, a few problems still need to be worked out. In particular, future work could investigate the reasons for primary forest clearing during reforestation, the longevity and sustainability of ecosystem service-driven forest transitions, and the successional trajectories that occur in community-restored novel forests.

Restoration, conservation, and deforestation – Who’s responsible for each?

In Intag, most people I surveyed reported a decrease in their forest clearing practices in the past decade. Although regional deforestation rates decreased, the question remains: What drives primary forest clearing during reforestation projects? Is the clearing in Intag due to ‘leakage’ – the displacement of deforestation elsewhere on the landscape by the same households who are reforesting – or are different households, communities, and factors driving this deforestation? I hypothesize that forest clearing in the region is driven less by 'leakage' than by declining productivity in long-used pastures. A study comparing reforestation and deforestation at the household and community levels could identify which households and communities should be targeted in forest conservation interventions and could provide appropriate incentives to maximize biodiversity conservation during reforestation projects.

Restoration longevity and sustainability – What happens if promised benefits are not delivered?

If people are motivated to restore forest for the ecosystem services they provide, what will happen to these forests if the promised services are not delivered (Brauman *et al.*, 2007; Farley & Costanza, 2010; Grêt-Regamey *et al.*, 2012), or if the effects of global climate change mask the positive impacts that reforestation provides (Buytaert *et al.*, 2011)? Will people continue to conserve restored forests, or clear them? Future research could focus on the longevity of existing restoration projects in relation to their initially stated ecological and social goals and the actual outcomes they produce. Fortunately, there is ample opportunity to study this: each restoration project is an experiment, and the many restoration projects currently underway could be systematically surveyed to answer these questions.

This question could also be answered by investigating the trajectories of forest use and ecosystem services that are derived from restored forests over time. How do the ecosystem services that local people obtain from on-farm trees and local forests change with the age of those forests? How does the way they use and manage these forests evolve with the services they provide? Ultimately, the answers to these questions would show the long-term effects of tree planting on people's ability and desire to remain on their farms, and help managers understand

and improve the sustainability and longevity of community-restored forest ecosystems.

The future of community-planted forests: Restoring forests, or creating novel ones?

My research also raises questions about the long-term successional trajectories in planted forests stocked with proportionately high numbers of planted species, ‘useful’ or otherwise. Will these forests continue to evolve in the direction of a primary forest, or do these planted areas represent self-perpetuating, ‘novel forests’ on the landscape (Lugo, 2009; Hobbs *et al.*, 2013)? Research on successional trajectories in community-restored forests is needed to guide future restoration efforts and to manage current restoration projects. Specifically, longitudinal studies in community-restored plots could focus on how biomass, species diversity, and functional diversity of tree communities accumulate and change in restored forests over time. This information could help managers predict and guide future outcomes and would shed further light on the ability of restoration projects to meet their implicit goal – restore forests. Where on the continuum between ‘historical’ and ‘novel’ restored forests eventually lie is a key question. Because of the prevalence of these projects now, the myriad policies in place that encourage tree planting, and because the need for restoration in cloud forests is only likely to increase in the future, we can expect that these forests will become more common in decades to come.

Conclusion: So, can biodiversity and human needs be reconciled?

Targeting restoration projects to communities experiencing environmental crisis can be a win-win for biodiversity conservation and development.

Taken together, the chapters in this thesis show that community-based projects that involve planting trees on abandoned or degraded pastures can help conserve tropical forest biodiversity. The absence of forest ecosystem services can, if presented in the right way, make clear to people just how important forests are for farming, and change their attitudes and practices around trees and forests. Involving local communities inevitably leads to social-ecological trade-offs, such as planting some exotic species and higher proportions of certain native ones. But there are also powerful synergies: planting ‘useful’ tree species not only rapidly increased the tree-species richness and density of secondary forests, but also facilitated the establishment of animal-dispersed species found in primary forest, thereby ‘jump-starting’ succession. Because local people implemented and stand to benefit from the projects, they are also dedicated to conserving planted forests. Future work is needed to determine the typical longevity of such projects and their ultimate ability to conserve diversity and deliver on the ecosystem services they promise. In the meantime, restoring forests with and for rural farming communities in heavily-cleared cloud forest landscapes is about as close to a livelihood-conservation win-win story as they come. My hope is that the results from this thesis can be used to motivate and improve future projects that take into account local contexts, needs, and biodiversity.

LITERATURE CITED

Abram, N. K., E. Meijaard, M. Ancrenaz, R. K. Runting, J. A. Wells, D. Gaveau, . . . K. Mengersen. (2014). Spatially explicit perceptions of ecosystem services and land cover change in forested regions of Borneo. *Ecosystem Services, 7*, 116-127.

Adams, W. M., R. Aveling, D. Brockington, B. Dickson, J. Elliott, J. Hutton, . . . W. Wolmer. (2004). Biodiversity conservation and the eradication of poverty. *Science, 306*, 1146-1149.

Agrawal, A., & A. Angelsen. (2009). Using community forest management to acheive REDD+ goals. In A. Angelsen (ed), *Realising REDD+: National Strategy and Policy Options* (pp. 201-212). Denmark: Center for International Forestry Research.

Agrawal, A., A. Chhatre, & R. Hardin. (2008). Changing governance of the world's forests. *Science, 320*, 1460-1462.

Agrawal, A., & C. C. Gibson. (1999). Enchantment and disenchantment: the role of community in natural resource conservation. *World development, 27*, 629-649.

Aguilar-Støen, M., A. Angelsen, & S. R. Moe. (2011). Back to the forest: Exploring forest transitions in Candelaria Loxicha, Mexico. *Latin American Research Review, 46*, 194-216.

Aide, T. M., & J. Cavelier. (1994). Barriers to lowland tropical forest restoration in the Sierra Nevada de Santa Marta, Colombia. *Restoration Ecology, 2*, 219-229.

Aide, T. M., M. L. Clark, H. R. Grau, D. Lopez-Carr, M. A. Levy, D. Redo, . . . M. Muniz. (2013). Deforestation and Reforestation of Latin America and the Caribbean (2001–2010). *Biotropica, 45*, 262–271.

Aide, T. M., M. Ruiz-Jaen, & H. Grau. (2010). What is the state of tropical montane cloud forest restoration. In L. A. Bruijnzeel, F. N. Scatena, L. S. Hamilton (*eds*), *Tropical Montane Cloud Forests: Science for Conservation and Management.* Cambridge: Cambridge University Press. 101-110.

Aide, T. M., J. K. Zimmerman, J. B. Pascarella, L. Rivera, & H. Marcano-Vega. (2000). Forest regeneration in a chronosequence of tropical abandoned pastures: implications for restoration ecology. *Restoration Ecology, 8*, 328-338.

Altieri, M. A. (2004). Linking ecologists and traditional farmers in the search for sustainable agriculture. *Frontiers in Ecology and the Environment, 2*, 35-42.

Amézquita, M. C., M. Ibrahim, T. Llanderal, P. Buurman, & E. Amézquita. (2004). Carbon sequestration in pastures, silvo-pastoral systems and forests in four regions of the Latin American tropics. *Journal of Sustainable Forestry, 21*, 31-49.

Andersson, K., & A. Agrawal. (2011). Inequalities, institutions, and forest commons. *Global Environmental Change, 21*, 866-875.

Angelsen, A., S. Brown, & C. Loisel. (2009). Reducing emissions from deforestation and forest degradation (REDD): an options assessment report.

Angelsen, A., & T. K. Rudel. (2013). Designing and implementing effective REDD+ policies: A forest transition approach. *Review of Environmental Economics and Policy, 7*, 91-113.

Angelsen, A., & S. Wunder. (2003). Exploring the Forest—Poverty Link. *CIFOR occasional paper, 40*, 1-20.

Armenteras, D., F. Gast, & H. Villareal. (2003). Andean forest fragmentation and the representativeness of protected natural areas in the eastern Andes, Colombia. *Biological Conservation, 113*, 245-256.

Aubad, J., P. Aragón, M. Á. Olalla-Tárraga, & M. Á. Rodríguez. (2008). Illegal logging, landscape structure and the variation of tree species richness across North Andean forest remnants. *Forest Ecology and Management, 255*, 1892-1899.

Bach, K., & S. R. Gradstein. (2011). A comparison of six methods to detect altitudinal belts of vegetation in tropical mountains. *Ecotropica, 17*, 1-13.

Baillie, I. (1996). Soils of the humid tropics. In P. W. Richards (Ed.), *The Tropical Rainforest: An Ecological Study* (pp. 256-258). Cambridge, MA: Cambridge University Press.

Balslev, H. (1988). Distribution patterns of Ecuadorean plant species. *Taxon*, 567-577.

Bannister, M., & P. Nair. (2003). Agroforestry adoption in Haiti: the importance of household and farm characteristics. *Agroforestry Systems, 57*, 149-157.

Baptista, S. R., & T. K. Rudel. (2006). A re-emerging Atlantic forest? Urbanization, industrialization and the forest transition in Santa Catarina, southern Brazil. *Environmental Conservation, 33*, 195-202.

Baral, N., & M. J. Stern. (2011). A comparative study of two community-based conservation models in Nepal. *Biodiversity and Conservation, 20*, 2407-2426.

Barbier, E. B. (2012). Natural capital, ecological scarcity and rural poverty.

Barbier, E. B., J. C. Burgess, & A. Grainger. (2010). The forest transition: Towards a more comprehensive theoretical framework. *Land Use Policy, 27*, 98-107.

Barlow, J., T. A. Gardner, I. S. Araujo, T. C. Ávila-Pires, A. B. Bonaldo, J. E. Costa, . . . M. I. Hernandez. (2007). Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences, 104*, 18555-18560.

Barrett, C. B., T. Reardon, & P. Webb. (2001). Nonfarm income diversification and household livelihood strategies in rural Africa: concepts, dynamics, and policy implications. *Food policy, 26*, 315-331.

Bautista-Cruz, A., & R. F. D. Castillo. (2005). Soil Changes During Secondary Succession in a Tropical Montane Cloud Forest Area. *Soil Science Society of America Journal, 69*, 906-914.

Bebbington, A. (1990). Farmer knowledge, institutional resources and sustainable agricultural strategies: a case study from the eastern slopes of the Peruvian Andes. *Bulletin of Latin American Research*, *9,* 203-228.

Bebbington, A. (1993). Sustainable livelihood development in the Andes: local institutions and regional resource use in Ecuador. *Development Policy Review, 11*, 5-30.

Bebbington, A., D. Humphreys Bebbington, J. Bury, J. Lingan, J. P. Muñoz, & M. Scurrah. (2008). Mining and social movements: struggles over livelihood and rural territorial development in the Andes. *World Development, 36*, 2888-2905.

Bebbington, A., & T. Perreault. (1999). Social capital, development, and access to resources in highland ecuador. *Economic Geography, 75*, 395-418.

Beck, E., R. Mosandl, M. Richter, & I. Kottke. (2008). The Investigated Gradients. In E. Beck, J. Bendix, I. Kottke, F. Makeschin, R. Mosandl (*eds*), *Gradients in a Tropical Mountain Ecosystem of Ecuador*. Springer: Ecological Studies. *198*, 55-62.

Becker, C. (2003). Grassroots to Grassroots: Why Forest Preservation was Rapid at Loma Alta, Ecuador. *World Development, 31*, 163-176.

Bellow, J., R. Hudson, & P. Nair. (2008). Adoption potential of fruit-tree-based agroforestry on small farms in the subtropical highlands. *Agroforestry Systems, 73*, 23-36.

Benayas, J. M. R., & J. M. Bullock. (2012). Restoration of biodiversity and ecosystem services on agricultural land. *Ecosystems, 15*, 883-899.

Benayas, J. M. R., J. M. Bullock, & A. C. Newton. (2008). Creating woodland islets to reconcile ecological restoration, conservation, and agricultural land use. *Frontiers in Ecology and the Environment, 6*, 329-336.

Bhagwat, S. A., K. J. Willis, H. J. B. Birks, R. J. Whittaker. (2008). Agroforestry: a refuge for tropical biodiversity? *Trends in Ecology and Evolution, 23*, 261-267.

Bierregaard Jr, R. O., T. E. Lovejoy, V. Kapos, A. A. dos Santos, & R. W. Hutchings. (1992). The biological dynamics of tropical rainforest fragments. *BioScience*, *42 (11),* 859-866.

BirdLife International. (2014). Important Bird Areas factsheet: Bosque Protector Los Cedros Retrieved 09/06, 2014, from http://www.birdlife.org

Blinn, C. E., J. O. Browder, M. A. Pedlowski, & R. H. Wynne. (2013). Rebuilding the Brazilian rainforest: Agroforestry strategies for secondary forest succession. *Applied Geography, 43*, 171-181.

Blom, B., T. Sunderland, & D. Murdiyarso. (2010). Getting REDD to work locally: lessons learned from integrated conservation and development projects. *Environmental Science & Policy, 13*, 164-172.

Bodin, Ö., B. Crona, M. Thyresson, A.-L. Golz, & M. Tengo. (2014). Conservation success as a function of good alignment of social and ecological structures and processes. *Conservation Biology*. *28*, 1371-1379.

Bodin, Ö., & B. I. Crona. (2009). The role of social networks in natural resource governance: What relational patterns make a difference? *Global Environmental Change, 19*, 366-374.

Bonner, M. T., S. Schmidt, & L. P. Shoo. (2013). A meta-analytical global comparison of aboveground biomass accumulation between tropical secondary forests and monoculture plantations. *Forest Ecology and Management, 291*, 73-86.

Boserup, E. (1965). *The conditions of agricultural growth: The economics of agriculture under population pressure*. London & New York: Faber.

Brandt, R., S.-L. Mathez-Stiefel, S. Lachmuth, I. Hensen, & S. Rist. (2013). Knowledge and valuation of Andean agroforestry species: the role of sex, age, and migration among members of a rural community in Bolivia. *Journal of Ethnobiology and Ethnomedicine, 9*, 83.

Brauman, K. A., G. Daily, T. Duarte, & H. Mooney. (2007). The Nature and Value of Ecosystem Services: An Overview Highlighting Hydrologic Services. *Annual Review of Environment and Resources, 32*, 67-98.

Bray, D. B. (2013). From Mexico, Global Lessons for Forest Governance. *Solutions*, *4(3)*, 51-59.

Bray, D. B., C. Antinori, & J. M. Torres-Rojo. (2006). The Mexican model of community forest management: The role of agrarian policy, forest policy and entrepreneurial organization. *Forest Policy and Economics, 8*, 470-484.

Bray, D. B., E. Duran, V. H. Ramos, J.-F. Mas, A. Velazques, R. B. McNab, . . . J. Radachowsky. (2008). Tropical Deforestation, Community Forests, and Protected Areas in the Maya Forest. *Ecology and Society, 13*, 56-56.

Brehm, G., J. Homeier, K. Fiedler, I. Kottke, J. Illig, N. Nöske, . . . S. Breckle. (2008). Mountain rain forests in southern Ecuador as a hotspot of biodiversity–limited knowledge and diverging patterns. In E. Beck, J. Bendix, I. Kottke, F. Makeschin, R. Mosandl (*eds*), *Gradients in a Tropical Mountain Ecosystem of Ecuador*. Ecological Studies: Springer. *198*, 15-23.

Bremer, L. L., & K. A. Farley. (2010). Does plantation forestry restore biodiversity or create green deserts? A synthesis of the effects of land-use transitions on plant species richness. *Biodiversity and Conservation, 19*, 3893-3915.

Brooks, J., K. A. Waylen, & M. B. Mulder. (2013). Assessing community-based conservation projects: A systematic review and multilevel analysis of attitudinal, behavioral, ecological, and economic outcomes. *Environmental Evidence, 2*, 2.

Brooks, T. M., R. A. Mittermeier, G. A. da Fonseca, J. Gerlach, M. Hoffmann, J. F. Lamoreux, . . . A. S. Rodrigues. (2006). Global biodiversity conservation priorities. *Science, 313*, 58-61.

Brooks, T. M., R. A. Mittermeier, C. G. Mittermeier, G. A. Da Fonseca, A. B. Rylands, W. R. Konstant, . . . G. Magin. (2002). Habitat loss and extinction in the hotspots of biodiversity. *Conservation Biology, 16*, 909-923.

Bruijnzeel, L. A. (2004). Hydrological functions of tropical forests: not seeing the soil for the trees? *Agriculture, Ecosystems & Environment, 104*, 185-228.

Bruijnzeel, L. A., & J. Proctor. (1995). Hydrology and biogeochemistry of tropical montane cloud forests: what do we really know? In L. A. Bruijnzeel, F. N. Scatena & L. S. Hamilton (eds). Tropical *montane cloud forests*. Springer. 38-78.

Bruijnzeel, L., F. Scatena, & L. Hamilton (eds). (2010*a*). *Tropical montane cloud forests: Science for conservation and management.* International Hydrology Series. Cambridge: Cambridge Press.

Bruijnzeel, L. A., M. Kappelle, M. Mulligan, and F. N. Scatena. (2010*b*). Tropical montane cloud forests: state of knowledge and sustainability perspectives in a changing world. In L.A. Bruijnzeel, F. N. Scatena, & L. S. Hamilton *(eds)*. *Tropical montane cloud forests: science for conservation and management*, International Hydrology Series: Cambridge University Press. 691.

Brush, S. B. (1977). *Mountain, field, and family: the economy and human ecology of an Andean valley*: University of Pennsylvania Press Philadelphia.

Brush, S. B. (1982). The natural and human environment of the central Andes. *Mountain Research and Development*, 19-38.

Brush, S. B., H. J. Carney, & Z. Humán. (1981). Dynamics of Andean potato agriculture. *Economic Botany, 35*, 70-88.

Bubb, P., May, I., Miles, L., Sayer, J. (2004). Cloud Forest Agenda. Cambridge, UK: UNEP-WCMC.

Buchanan, K. S. (2013). Contested discourses, knowledge, and socio-environmental conflict in Ecuador. *Environmental Science & Policy, 30*, 19-25.

Bunce, D., & M. West. (1994). Changing work environments: Innovative coping responses to occupational stress. *Work & Stress, 8*, 319-331.

Butterfield, R. P. (1995). Promoting biodiversity: advances in evaluating native species for reforestation. *Forest Ecology and Management, 75*(1), 111-121.

Buytaert, W., F. Cuesta-Camacho, & C. Tobón. (2011). Potential impacts of climate change on the environmental services of humid tropical alpine regions. *Global Ecology and Biogeography, 20*, 19-33.

Calmon, M., P. H. Brancalion, A. Paese, J. Aronson, P. Castro, S. C. da Silva, & R. R. Rodrigues. (2011). Emerging Threats and Opportunities for Large-Scale Ecological Restoration in the Atlantic Forest of Brazil. *Restoration Ecology, 19*, 154-158.

Cao, S., L. Chem, D. Shankman, C. Wang, X. Wang, H. Zhang. (2011). Excessive reliance on afforestation in China's arid and semi-arid regions: Lessons in ecological restoration. *Earth-Science Reviews, 104*, 240-245.

Castillo-Campos, G., G. Halffter, & C. E. Moreno. (2008). Primary and secondary vegetation patches as contributors to floristic diversity in a tropical deciduous forest landscape. *Biodiversity and Conservation, 17*, 1701-1714.

Cavelier, J., & A. Etter. (1995). Deforestation of montane forests in Colombia as a result of illegal plantations of opium (Papaver somniferum). In P. Churchill, H. Balslev, E. Forero, J. L. Luteyn (*eds*). *Biodiversity and conservation of Neotropical montane forests. Proceedings of a symposium,* New York Botanical Garden, 21-26 June 1993. 541-549.

Chai, S.-L., & E. Tanner. (2011). 150-year legacy of land use on tree species composition in old-secondary forests of Jamaica. *Journal of Ecology, 99*, 113-121.

Chander, G., B. L. Markham, & D. L. Helder. (2009). Summary of current radiometric calibration coefficients for Landsat MSS, TM, ETM+, and EO-1 ALI sensors. *Remote sensing of environment, 113*, 893-903.

Chao, A. (1984). Nonparametric estimation of the number of classes in a population. *Scandinavian Journal of Statistics*, *11,* 265-270.

Chao, A., R. L. Chazdon, R. K. Colwell, & T. J. Shen. (2005). A new statistical approach for assessing similarity of species composition with incidence and abundance data. *Ecology Letters, 8*, 148-159.

Charnley, S., & M. R. Poe. (2007). Community Forestry in Theory and Practice: Where Are We Now? *Annual Review of Anthropology, 36*, 301.

Chazdon, R. L. (2003). Tropical forest recovery: legacies of human impact and natural disturbances. *Perspectives in Plant Ecology, Evolution and Systematics, 6*, 51-71.

Chazdon, R. L. (2008). Beyond deforestation: restoring forests and ecosystem services on degraded lands. *Science, 320*, 1458-1460.

Chazdon, R. L., S. Careaga, C. Webb, & O. Vargas. (2003). Community and phylogenetic structure of reproductive traits of woody species in wet tropical forests. *Ecological Monographs, 73*, 331-348.

Chazdon, R. L., R. K. Colwell, J. S. Denslow, & M. R. Guariguata. (1998). Statistical methods for estimating species richness of woody regeneration in primary and secondary rain forests of northeastern Costa Rica. In F. Dallmeier & J. A. Comiskey (*eds*). *Forest biodiversity research, monitoring and modeling: conceptual background and old world case studies. Man and the Biosphere Series. 20,* 285-309.

Chazdon, R. L., C. A. Harvey, O. Komar, D. M. Griffith, B. G. Ferguson, M. Martınez-Ramos, . . . M. van Breugel. (2008). Beyond Reserves: A Research Agenda for Conserving Biodiversity in Human-modified Tropical Landscapes. *Biotropica, 41*, 142-153.

Chen, Y. C., W.-H. Hwang, A. Chao, & C.-Y. Kuo. (1995). Estimating the number of common species. Analysis of the number of common bird species in Ke-Yar Stream and Chung-Kang Stream. *Journal of the Chinese Statistical Association, 33*, 373-393.

Chhatre, A., & A. Agrawal. (2008). Forest commons and local enforcement. *Proceedings of the National Academy of Sciences, 105*, 13286-13291.

Chokkalingam, U., C. Sabogal, E. Almeida, A. P. Carandang, T. Gumartini, W. de Jong, . . . L. R. Wibowo. (2005). Local participation, livelihood needs, and institutional arrangements: three keys to sustainable rehabilitation of degraded tropical forest lands *Forest restoration in landscapes*. Springer. 405-414.

Chowdhury, R. R. (2010). Differentiation and concordance in smallholder land use strategies in southern Mexico's conservation frontier. *Proceedings of the National Academy of Sciences, 107*, 5780-5785.

Christensen, J. (2004). Win-Win Illusions. *Conservation in Practice, 5*(1), 12-19.

Churchill, S. P., H. Balslev, E. Forero, & J. Luteyn. (1995). *Biodiversity and conservation of neotropical montane forests. Proceedings of the symposium.* the New York Botanical Garden, 21-26 June 1993.

Churchill, S. P., H. Balslev, E. Forero, & J. L. Luteyn. (1995). Introduction. In P. Churchill, H. Balslev, E. Forero, J. L. Luteyn (*eds*). *Biodiversity and conservation of Neotropical montane forests. Proceedings of a symposium.* New York Botanical Garden, 21-26 June 1993. 541-549.

CI. Conservation International website Retrieved 15/04, 2010, from http://www.conservation.org/discover/Pages/about_us.aspx

Clewell, A., J. Aronson, & K. Winterhalder. (2004). The SER International Primer on Ecological Restoration, from https://http://www.ser.org/resources.

Cochran, J. B., & R. Bonnell. (2006). Patterns of sustainable agriculture adoption/non-adoption in Panamá. *Journal of Sustainable Agriculture, 27*, 147-162.

Cole, R. J. (2010). Social and environmental impacts of payments for environmental services for agroforestry on small-scale farms in southern Costa Rica. *International Journal of Sustainable Development & World Ecology, 17*, 208-216.

Collins, C. J., & K. G. Smith. (2006). Knowledge exchange and combination: The role of human resource practices in the performance of high-technology firms. *Academy of Management Journal, 49*, 544-560.

Colwell, R. K. (2009). Estimates: Statistical estimation of species richness and shared species from samples. Version 8.2 and earlier. User's Guide and application. *Published at:* http://purl.oclc.org/estimates.

Colwell, R. K., A. Chao, N. J. Gotelli, S.-Y. Lin, C. X. Mao, R. L. Chazdon, & J. T. Longino. (2012). Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. *Journal of Plant Ecology, 5*, 3-21.

Condit, R., N. Pitman, E. G. Leigh, J. Chave, J. Terborgh, R. B. Foster, . . . G. Villa. (2002). Beta-diversity in tropical forest trees. *Science, 295*, 666-669.

Coomes, O. T., B. L. Barham, & Y. Takasaki. (2004). Targeting conservation–development initiatives in tropical forests: insights from analyses of rain forest use and economic reliance among Amazonian peasants. *Ecological Economics, 51*, 47-64.

Coomes, O. T., & G. J. Burt. (1997). Indigenous market-oriented agroforestry: dissecting local diversity in western Amazonia. *Agroforestry Systems, 37*, 27-44.

Coomes, O. T., F. Grimard, C. Potvin, & P. Sima. (2008). The fate of the tropical forest: Carbon or cattle? *Ecological Economics, 65*, 207-212.

Costales Samaniego, A., & D. Costales Peñaherrera. (2002). *Etnografía, lingüística e historia antigua de los caras o yumbos colorados (1534-1978)*. Quito: Abya-Yala, IEAG.

Cramer, V. A., R. J. Hobbs, & R. J. Standish. (2008). What's new about old fields? Land abandonment and ecosystem assembly. *Trends in Ecology & Evolution, 23*, 104-112.

Current, D., E. Lutz, & S. J. Scherr. (1995). *Costs, Benefits, and Farmer Adoption of Agroforestry: Project Experience in Central America and the Caribbean*. World Bank.

D'Amico, L. (2010). Grassroots Environmentalism in Intag, Ecuador. Prepared for the 2nd UNITAR-Yale Conference on Environmental Governance and Democracy, 17-19

September, New Haven, USA.
Daugherty, H. E. (1973). The Montecristo cloud-forest of El Salvador-A chance for protection. *Biological Conservation, 5*, 227-230.
Davies, T., I. Fazey, W. Cresswell, & N. Pettorelli. (2013). Missing the trees for the wood: Why we are failing to see success in pro-poor conservation. *Animal Conservation*, *17*, 303-312.
de Jong, W. (2010). Forest rehabilitation and its implication for forest transition theory. *Biotropica, 42*, 3-9.
De Koning, F., M. Aguiñaga, M. Bravo, M. Chiu, M. Lascano, T. Lozada, & L. Suarez. (2011). Bridging the gap between forest conservation and poverty alleviation: the Ecuadorian Socio Bosque program. *Environmental Science & Policy, 14*, 531-542.
DECOIN. (2010). Defensa y Conservation Ecologica de Intag Retrieved 20/12, 2013, from http://www.decoin.org
DeFries, R. S., E. C. Ellis, F. S. Chapin, P. A. Matson, B. Turner, A. Agrawal, . . . P. M. Kareiva. (2012). Planetary opportunities: a social contract for global change science to contribute to a sustainable future. *BioScience, 62*, 603-606.
Denevan, W. M. (1992). The pristine myth: the landscape of the Americas in 1492. *Annals of the Association of American Geographers, 82*, 369-385.
Dent, D. H., & S. Joseph Wright. (2009). The future of tropical species in secondary forests: a quantitative review. *Biological Conservation, 142*, 2833-2843.
Desmarais, A.-A. (2002). Peasants speak-The Vía Campesina: Consolidating an international peasant and farm movement. *The Journal of Peasant Studies, 29*, 91-124.
Diamond, J. M. (1975). The island dilemma: lessons of modern biogeographic studies for the design of natural reserves. *Biological Conservation, 7*, 129-146.
Diemont, S. A., J. L. Bohn, D. D. Rayome, S. J. Kelsen, & K. Cheng. (2011). Comparisons of Mayan forest management, restoration, and conservation. *Forest Ecology and Management, 261*, 1696-1705.
Dodson, C. H., & A. H. Gentry. (1991). Biological extinction in western Ecuador. *Annals of the Missouri Botanical Garden*, 78, 273-295.
Doss, C. R. (2006). Analyzing technology adoption using microstudies: limitations, challenges, and opportunities for improvement. *Agricultural Economics, 34*, 207-219.
Doumenge, C., D. Gilmour, M. R. Pérez, & J. Blockhus. (1995). Tropical montane cloud forests: conservation status and management issues. In L. Hamilton, J. Juvik, F. Scatena (*eds*). *Tropical montane cloud forests.* Ecological Studies: Springer, *110,* 24-37.
Douterlungne, D., S. I. Levy‑Tacher, D. J. Golicher, & F. R. Dañobeytia. (2010). Applying indigenous knowledge to the restoration of degraded tropical rain forest clearings dominated by bracken fern. *Restoration Ecology, 18*, 322-329.
Dove, M. (2011). *The Banana Tree at the Gate: The History of Marginal Peoples and Global Markets in Borneo*. New Haven: Yale University Press.
Duchelle, A. E. (2009). *Conservation and livelihood development in Brazil-nut producing communities in a tri-national Amazonian frontier.* (Dissertation), University of Florida, Gainesville, USA.
Duke, E. A., J. H. Goldstein, T. L. Teel, R. Finchum, H. Huber-Stearns, J. Pitty, . . . L. O. Sánchez. (2014). Payments for ecosystem services and landowner interest: Informing program design trade-offs in Western Panama. *Ecological Economics, 103*, 44-55.

Echeverría, C., L. Cayuela, R. Manson, D. Coomes, A. Lara, J. Rey-Benayas, & A. Newton. (2007). Spatial and temporal patterns of forest loss and fragmentation in Mexico and Chile. In A. Newton (*ed*) *Biodiversity loss and conservation in fragmented forest landscapes. The forests of montane Mexico and temperate South America.* Wallingford: CABI. 14-42.

Edwards, P., & P. Grubb. (1977). Studies of mineral cycling in a montane rain forest in New Guinea: I. The distribution of organic matter in the vegetation and soil. *The Journal of Ecology*, 943-969.

Egan, D., E. Higgs, E. E. Hjerpe, & J. Abrams. (2011). *Human dimensions of ecological restoration: Integrating science, nature, and culture*. Springer.

Eliot, R. (1982). Faking Nature. *Inquiry, 25*, 81-93.

Ellis, E. C., & N. Ramankutty. (2008). Putting people in the map: anthropogenic biomes of the world. *Frontiers in Ecology and the Environment, 6*, 439-447.

Ellis, F. (1993). *Peasant economics: Farm households in agrarian development* (Vol. 23). Cambridge: Cambridge University Press.

Ellis, F. (1998). Household strategies and rural livelihood diversification. *The journal of development studies, 35*, 1-38.

Engel, S., S. Pagiola, & S. Wunder. (2008). Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological economics, 65*(4), 663-674.

Ennenbach, M. (2013). Assessing the viability of China's reforestation program, *Columbia Political Review*. Retrieved from http://cpreview.org/2013/03/the-great-green-wall-of-china/

ENVI. (2009). Atmospheric Correction Module: QUAC and FLAASH User's Guide. *ITT Visual Information Solutions, Boulder, CO*.

Fairhead, J., & M. Leach. (1994). Contested forests: Modern conservation and historical land use in Guinea's Ziama Reserve. *African Affairs*, *93,* 481-512.

Fang, W., & S. Peng. (1997). Development of species diversity in the restoration process of establishing a tropical man-made forest ecosystem in China. *Forest Ecology and Management, 99*, 185-196.

FAO. (2000). On definitions of forest and forest cover change. *FRA Programme*. Rome: FRA, Food and Agriculture Organization of the United Nations (FAO).

FAO. (2005). *Global Forest Resources Assessment 2005*. Rome: Food and Agriculture Organization of the United Nations (FAO).

FAO. (2007). *State of The World's Forests 2007*. Rome: Food and Agriculture Organization of the United Nations (FAO).

FAO. (2011). *State of The World's Forests 2011.* Rome: Food and Agriculture Organization of the United Nations (FAO).

Farley, J., & R. Costanza. (2010). Payments for ecosystem services: from local to global. *Ecological Economics, 69*, 2060-2068.

Farley, K. A. (2007). Grasslands to tree plantations: forest transition in the Andes of Ecuador. *Annals of the association of American Geographers, 97*, 755-771.

Farley, K. A. (2010). Pathways to forest transition: Local case studies from the Ecuadorian Andes. *Journal of Latin American Geography, 9*, 7-26.

Fávero, C., I. C. Lovo, & E. d. S. Mendonça. (2008). Recovery of degraded areas using agroforestry systems in Vale do Rio Doce, Minas Gerais. *Revista Árvore, 32*, 861-868.

Fearnside, P. M. (1993). Deforestation in Brazilian Amazonia: the effect of population and land tenure. *Ambio-Journal of Human Environment Research and Management, 22*, 537-545.
Feder, G., R. E. Just, & D. Zilberman. (1985). Adoption of agricultural innovations in developing countries: A survey. *Economic Development and Cultural Change*. Chicago: University of Chicago Press. 255-298.
Fernihough, A. (2011). Simple logit and probit marginal effects in R. No 201122, Working Papers, School Of Economics, University College Dublin.
Feyera, S., E. Beck, & U. Lüttge. (2002). Exotic trees as nurse-trees for the regeneration of natural tropical forests. *Trees, 16*, 245-249.
Fischer, A., & L. Vasseur. (2002). Smallholder perceptions of agroforestry projects in Panama. *Agroforestry Systems, 54*, 103-113.
Fisher, M., G. E. Shively, & S. Buccola. (2005). Activity choice, labor allocation, and forest use in Malawi. *Land Economics, 81*, 503-517.
Fitzherbert, E. B., M. J. Struebig, A. Morel, F. Danielsen, C. A. Brühl, P. F. Donald, & B. Phalan. (2008). How will oil palm expansion affect biodiversity? *Trends in Ecology & Evolution, 23*, 538-545.
Fjeldså, J., E. Lambin, & B. Mertens. (1999). Correlation between endemism and local ecoclimatic stability documented by comparing Andean bird distributions and remotely sensed land surface data. *Ecography, 22*, 63-78.
Flora, J. L., & C. B. Flora. (2003). Desarrollo comunitario en las zonas rurales de los Andes. *Capital social y reducción de la pobreza en América Latina y el Caribe: en busca de un nuevo*, 555.
Florentine, S. K., & M. E. Westbrooke. (2004). Restoration on abandoned tropical pasturelands—do we know enough? *Journal for Nature Conservation, 12*, 85-94.
Foley, J. A., C. Monfreda, N. Ramankutty, & D. Zaks. (2007). Our share of the planetary pie. *Proceedings of the National Academy of Sciences, 104*, 12585-12586.
Foster, A. D., & M. R. Rosenzweig. (2003). Economic growth and the rise of forests. *The Quarterly Journal of Economics*, *118*, 601-637.
Foster, D., F. Swanson, J. Aber, I. Burke, N. Brokaw, D. Tilman, & A. Knapp. (2003). The importance of land-use legacies to ecology and conservation. *BioScience, 53*, 77-88.
Foster, D. R., G. Motzkin, & B. Slater. (1998). Land-use history as long-term broad-scale disturbance: regional forest dynamics in central New England. *Ecosystems, 1*, 96-119.
Foster, P. (2001). The potential negative impacts of global climate change on tropical montane cloud forests. *Earth-Science Reviews, 55*, 73-106.
Francisco, S. (2010). The innovation paradox: How innovation products threaten the innovation process. *Reconstruction: Studies in Contemporary Culture, 10*.
Franzel, S. (1999). Socioeconomic factors affecting the adoption potential of improved tree fallows in Africa. *Agroforestry Systems, 47*, 305-321.
Freiberg, M., & E. Freiberg. (2000). Epiphyte diversity and biomass in the canopy of lowland and montane forests in Ecuador. *Journal of Tropical Ecology, 16*, 673-688.
Frey, G. E., H. E. Fassola, A. Pachas, L. Colcombet, S. M. Lacorte, O. Pérez, . . . F. W. Cubbage. (2012). Perceptions of silvopasture systems among adopters in northeast Argentina. *Agricultural Systems, 105*, 21-32.
Fujisaka, S., & D. White. (1998). Pasture or permanent crops after slash-and-burn cultivation? Land-use choice in three Amazon colonies. *Agroforestry Systems, 42*, 45-59.
Gade, D. W. (1999). *Nature and Culture in the Andes*. Madison: University of Wisconsin Press.

Gamboa, V. G., J. Barkmann, & R. Marggraf. (2010). Social network effects on the adoption of agroforestry species: preliminary results of a study on differences on adoption patterns in Southern Ecuador. *Procedia-Social and Behavioral Sciences, 4*, 71-82.
Gamfeldt, L., T. Snäll, R. Bagchi, M. Jonsson, L. Gustafsson, P. Kjellander, & J. Bengtsson. (2013). Higher levels of multiple ecosystem services are found in forests with more tree species. *Nature communication, 4*, 1340.
Gandolfi, S., S. V. Martins, & R. R. Rodrigues. (2007). Forest restoration: Many views and objectives. In R. R. Rodrigues, S. V. Martin & S. Gandolfi (*eds*). *High Diversity Forest Areas: Methods and Projects in Brazil*. New York: Nova Science Publishers. 2-26.
Gardner, T. A., J. Barlow, R. Chazdon, R. M. Ewers, C. A. Harvey, C. A. Peres, & N. S. Sodhi. (2009). Prospects for tropical forest biodiversity in a human-modified world. *Ecology Letters, 12*, 561-582.
Gardner, T. A., A. Von Hase, S. Brownlie, J. M. M. Ekstrom, J. D. Pilgrim, C. E. Savy, . . . K. Ten Kate. (2013). Biodiversity Offsets and the Challenge of Achieving No Net Loss. *Conservation Biology, 27*, 1254–1264.
Garen, E. J., K. Saltonstall, J. L. Slusser, S. Mathias, M. S. Ashton, & J. S. Hall. (2009). An evaluation of farmers' experiences planting native trees in rural Panama: implications for reforestation with native species in agricultural landscapes. *Agroforestry systems, 76*, 219-236.
Geist, H. J., & E. F. Lambin. (2002). Proximate Causes and Underlying Driving Forces of Tropical Deforestation Tropical forests are disappearing as the result of many pressures, both local and regional, acting in various combinations in different geographical locations. *BioScience, 52*, 143-150.
Gentry, A. H. (1986). Species richness and floristic composition of Chocó region plant communities. *Caldasia, 15*, 71-91.
Gentry, A. H. (1988). Changes in plant community diversity and floristic composition on environmental and geographical gradients. *Annals of the Missouri Botanical Garden, 75*, 1-34.
Gentry, A. H. (1989). Northwest South America (Colombia, Ecuador and Peru). *Floristic inventories of tropical countries.* New York: The New York Botanical Garden. 393-400.
Gentry, A. H. (1992). Diversity and floristic composition of Andean forests of Peru and adjacent countries: implications for their conservation. *Memorias del Museo de Historia Natural, UNMSM (Lima), 21*, 11-29.
Gentry, A. H., S. Churchill, H. Balslev, E. Forero, & J. Luteyn. (1995). *Patterns of diversity and floristic composition in Neotropical montane forests.* In P. Churchill, H. Balslev, E. Forero, J. L. Luteyn *(eds)*. *Biodiversity and conservation of Neotropical montane forests. Proceedings of a symposium,* New York Botanical Garden, 21-26 June 1993.
Gibson, L., T. M. Lee, L. P. Koh, B. W. Brook, T. A. Gardner, J. Barlow, . . . T. E. Lovejoy. (2011). Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature, 478*, 378-381.
Gilles, J. L., & K. Jamtgaard. (1981). Overgrazing in Pastoral Areas. The Commons Reconsidered. *Sociologia Ruralos, 21*, 129-141.
Givnish, T. J. (1998). Altitudinal gradients in tropical forest composition, structure, and diversity in the Sierra de Manantlán. *Journal of Ecology, 86*, 999-1020.
GLOVIS. (2012) Retrieved 12, 2012, from http://glovis.usgs.gov/

Gockel, C. K., & L. C. Gray. (2009). Integrating conservation and development in the Peruvian Amazon. *Ecology and Society, 14*, 11.
Goland, C. (1993). Field scattering as agricultural risk management: A case study from Cuyo Cuyo, Department of Puno, Peru. *Mountain Research and Development*, *13,* 317-338.
González-Espinosa, M., N. Ramírez-Marcial, A. Newton, J. Rey-Benayas, A. Camacho-Cruz, J. Armesto, . . . A. Altamirano. (2007). Restoration of forest ecosystems in fragmented landscapes of temperate and montane tropical Latin America. In A. C. Newton (ed), *Biodiversity Loss and Conservation in Fragmented Forest Landscapes: The Forests of Montane Mexico and Temperate South America*. Cambridge: CABI. 335-369.
Gordon, J. E., W. D. Hawthorne, A. Reyes-Garcia, G. Sandoval, & A. J. Barrance. (2004). Assessing landscapes: a case study of tree and shrub diversity in the seasonally dry tropical forests of Oaxaca, Mexico and southern Honduras. *Biological Conservation, 117*, 429-442.
Gradstein, S., D. Griffin, M. Morales, & N. Nadkarni. (2001). Diversity and habitat differentiation of mosses and liverworts in the cloud forest of Monteverde, Costa Rica. *Caldasia, 23*, 203-212.
Graham, C. H., N. Silva, & J. Velásquez-Tibatá. (2010). Evaluating the potential causes of range limits of birds of the Colombian Andes. *Journal of Biogeography, 37*, 1863-1875.
Grainger, A. (1995). The forest transition: an alternative approach. *Area, 27*, 242-251.
Grau, H. R., & T. M. Aide. (2008). Globalization and land-use transitions in Latin America. *Ecology and Society, 13*, 16.
Grêt-Regamey, A., S. H. Brunner, & F. Kienast. (2012). Mountain ecosystem services: Who cares? *Mountain Research and Development, 32*, S23-S34.
Grieg-Gran, M., I. Porras, & S. Wunder. (2005). How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. *World Development, 33*, 1511-1527.
Griscom, H. P., & M. S. Ashton. (2011). Restoration of dry tropical forests in Central America: a review of pattern and process. *Forest Ecology and Management, 261*(10), 1564-1579.
Griscom, H. P., B. W. Griscom, & M. S. Ashton. (2009). Forest regeneration from pasture in the dry tropics of Panama: effects of cattle, exotic grass, and forested riparia. *Restoration Ecology, 17*, 117-126.
Groom, B., & C. Palmer. (2012). REDD+ and rural livelihoods. *Biological Conservation, 154*, 42-52.
Grubb, P. (1977). Control of forest growth and distribution on wet tropical mountains: with special reference to mineral nutrition. *Annual Review of Ecology and Systematics*, *8,* 83-107.
Grubb, P., J. Lloyd, T. Pennington, & T. Whitmore. (1963). A comparison of montane and lowland rain forest in Ecuador I. The forest structure, physiognomy, and floristics. *The Journal of Ecolog*y, 51, 567-601.
Grubb, P., & T. Whitmore. (1966). A comparison of montane and lowland rain forest in Ecuador: II. The climate and its effects on the distribution and physiognomy of the forests. *The Journal of Ecology*, *54,* 303-333.
Guariguata, M. R., & R. Ostertag. (2001). Neotropical secondary forest succession: changes in structural and functional characteristics. *Forest Ecology and Management, 148*, 185-206.

Guevara, S., J. M. d. Castillo, P. Moreno-Casasola, & J. Laborde. (1992). Floristic competition and vegetation structure under isolated trees in neotropical pastures. *Journal of Vegetation Science, 3*, 655-664.

Günter, S., M. Weber, R. Erreis, & N. Aguirre. (2007). Influence of distance to forest edges on natural regeneration of abandoned pastures: a case study in the tropical mountain rain forest of Southern Ecuador. *European Journal of Forest Research, 126*, 67-75.

Guswa, A. J., A. L. Rhodes, & S. E. Newell. (2007). Importance of orographic precipitation to the water resources of Monteverde, Costa Rica. *Advances in Water Resources, 30*, 2098-2112.

Haber, W. A. (2000). Plants and vegetation. In N. M. Nadkarni & N. T. Wheelwright (*eds*). *Monteverde: Ecology and conservation of a tropical cloud forest*, New York: Oxford University Press, 39-70.

Hamilton, L. S., J. O. Juvik, & F. N. Scatena. (1995). The Puerto Rico tropical cloud forest symposium: introduction and workshop synthesis. In L. Hamilton, J. Juvik, F. Scatena (*eds*). *Tropical montane cloud forests*. Ecological Studies: Springer, *110,* 1-18.

Hansen, M., P. V. Potapov, R. Moore, M. Hancher, S. A. Turubanova, A. Tyukavina, & D. Thau. (2014). Response to comment on "High-resolution global maps of 21st-century forest cover change". *Science, 344*, 981.

Harris, J. A., & R. van Diggelen. (2006). Ecological restoration as a project for global society. In J. Van Andel & J. Aronson (*eds*). *Restoration Ecology*. Malden: Blackwell publishing. 3-15.

Harvey, C. A. (2000). Windbreaks enhance seed dispersal into agricultural landscapes in Monteverde, Costa Rica. *Ecological Applications, 10*, 155-173.

Harvey, C. A., O. Komar, R. Chazdon, B. G. Ferguson, B. Finegan, D. M. Griffith, . . . L. Soto-Pinto. (2008). Integrating agricultural landscapes with biodiversity conservation in the Mesoamerican hotspot. *Conservation Biology, 22*, 8-15.

Headley, R. (2010). Landsat—A global land-imaging project: U.S. Geological Survey Fact Sheet 2010–3026 Retrieved 08/22, 2014, from http://pubs.usgs.gov/fs/2010/3026/

Healey, S. P., & R. I. Gara. (2003). The effect of a teak Tectona grandis plantation on the establishment of native species in an abandoned pasture in Costa Rica. *Forest Ecology and Management, 176*, 497-507.

Hecht, S. B. (1993). Land speculation and pasture-led deforestation in Brazil. In M. Colchester & L. Lohman (*eds*). *The Struggle for Land and the Fate of the Forests*. London and New Jersey: Zed Books. 164-178.

Hecht, S. B. (2010). The new rurality: Globalization, peasants and the paradoxes of landscapes. *Land Use Policy, 27*, 161-169.

Henderson, A., S. P. Churchill, & J. L. Luteyn. (1991). Neotropical plant diversity. *Nature, 351*, 21-22.

Herzog, S. K., & G. H. Kattan. (2011). Patterns of Diversity and Endemism in the Birds of the Tropical Andes. In S. K. Herzog, R. Martínez, P. M. Jørgensen, H. Tiessen (*eds*). *Climate change and biodiversity in the Tropical Andes*, Inter-American Institute for Global Change Research, Climate Change and Biodiversity in the Tropical Andes, 245-259.

Herzog, S. K., R. Martinez, P. M. Jorgensen, & H. Tiessen. (2011). *Climate Change and Biodiversity in the Tropical Andes*. Inter-American Institute for Global Change Research (IAI) and Scientific Committee on Problems of the Environment (SCOPE).

Hetsch, W., & H. Hoheisel. (1976). Standorts- und vegetations- gliederung in einem tropischen

nebelwald. *Allg. Forst Jagdz, 147*, 200-209.
Higgs, E. (2003). *Nature by design: people, natural process, and ecological restoration*. Cambridge: MIT Press.
Hobbs, R. J., E. Higgs, & J. A. Harris. (2009). Novel ecosystems: implications for conservation and restoration. *Trends in ecology & evolution, 24*, 599-605.
Hobbs, R. J., E. S. Higgs, & C. Hall. (2013). *Novel ecosystems: intervening in the new ecological world order*. Oxford: John Wiley & Sons.
Hoch, L., B. Pokorny, & W. De Jong. (2009). How successful is tree growing for smallholders in the Amazon? *International Forestry Review, 11*, 299-310.
Hoch, L., B. Pokorny, & W. De Jong. (2012). Financial attractiveness of smallholder tree plantations in the Amazon: bridging external expectations and local realities. *Agroforestry systems, 84*, 361-375.
Holl, K. D. (1998). The role of bird perching structures in accelerating tropical forest recovery. *Restoration Ecology, 6*, 253-261.
Holl, K. D. (1999). Factors Limiting Tropical Rain Forest Regeneration in Abandoned Pasture: Seed Rain, Seed Germination, Microclimate, and Soil1. *Biotropica, 31*, 229-242.
Holl, K. D. (2002). Long-term vegetation recovery on reclaimed coal surface mines in the eastern USA. *Journal of Applied Ecology, 39*, 960-970.
Holl, K. D., Aide, T. M. (2011). When and where to actively restore ecosystems? *Forest Ecology and Management, 261,* 1558–1563.
Holl, K. D., M. E. Loik, E. H. Lin, & I. A. Samuels. (2000). Tropical montane forest restoration in Costa Rica: overcoming barriers to dispersal and establishment. *Restoration Ecology, 8*, 339-349.
Holl, K. D., & R. A. Zahawi. (2014). Factors explaining variability in woody above-ground biomass accumulation in restored tropical forest. *Forest Ecology and Management, 319*, 36-43.
Holmes, I., & C. Potvin. (2014). Avoiding Re-Inventing the Wheel in a People-Centered Approach to REDD+. *Conservation Biology, 28,* 1380-1393.
Homeier, J., S. W. Breckle, S. Günter, R. T. Rollenbeck, & C. Leuschner. (2010). Tree Diversity, Forest Structure and Productivity along Altitudinal and Topographical Gradients in a Species-Rich Ecuadorian Montane Rain Forest. *Biotropica, 42*, 140-148.
Hussain, S. A., & R. Badola. (2010). Valuing mangrove benefits: contribution of mangrove forests to local livelihoods in Bhitarkanika Conservation Area, East Coast of India. *Wetlands Ecology and Management, 18*, 321-331.
INEC. (2010). 2010 Census: Instituto Nacional de Estadística y Censos.
Isaac, M. E., B. H. Erickson, S. J. Quashie-Sam, & V. R. Timmer. (2007). Transfer of knowledge on agroforestry management practices: the structure of farmer advice networks. *Ecology and Society, 12*, 32.
IUCN. (2013). The IUCN Red List of Threatened Species Version 2013.2. Retrieved 21/11, 2013, from http://www.iucnredlist.org.
Jacob, A. (2014). *Conserving and restoring biodiversity and ecosystem services in African tropical rainforest.* (PhD), McGill University.
Jankowski, J. E., C. L. Merkord, W. F. Rios, K. G. Cabrera, N. S. Revilla, & M. R. Silman. (2013). The relationship of tropical bird communities to tree species composition and vegetation structure along an Andean elevational gradient. *Journal of Biogeography, 40*, 950-962.

Jokisch, B. D. (2002). Migration and agricultural change: The case of smallholder agriculture in highland Ecuador. *Human Ecology, 30*, 523-550.
Jokisch, B. D., & B. M. Lair. (2002). One last stand? Forests and change on Ecuador's eastern cordillera. *The Geographical Review 92*, 135-156.
Joppa, L. N., & A. Pfaff. (2009). High and far: biases in the location of protected areas. *PLoS One, 4*, e8273.
Jordan III, W. R., M. E. Gilpin, & J. D. Aber. (1987). *Restoration ecology: Ecological restoration as a technique for basic research.* Cambridge: Cambridge University Press.
Jørgensen, P. M., C. Ulloa Ulloa, B. León, S. León-Yánez, S. G. Beck, M. Nee, . . . R. Gradstein. (2011). Regional patterns of vascular plant diversity and endemism. In S. K. Herzog, R. Martínez, P. M. Jørgensen, H. Tiessen (*eds*). *Climate change and biodiversity in the Tropical Andes*, Inter-American Institute for Global Change Research, Climate Change and Biodiversity in the Tropical Andes, 192-203.
Jose, S. (2009). Agroforestry for ecosystem services and environmental benefits: an overview. *Agroforestry Systems, 76*, 1-10.
Jost, L. (2013). Biogeography of the Pastaza watershed, from http://www.loujost.com/Biogeography of the Pastaza Watershed/Pastaza.htm
Kammesheidt, L. (1999). Forest recovery by root suckers and above-ground sprouts after slash-and-burn agriculture, fire and logging in Paraguay and Venezuela. *Journal of Tropical Ecology, 15*, 143-157.
Kanowski, J., C. P. Catterall, & G. W. Wardell-Johnson. (2005). Consequences of broadscale timber plantations for biodiversity in cleared rainforest landscapes of tropical and subtropical Australia. *Forest Ecology and Management, 208*, 359-372.
Kappelle, M., G. Avertin, M. E. Juárez, & N. Zamora. (2000). Useful plants within a campesino community in a Costa Rican montane cloud forest. *Mountain Research and Development, 20*, 162-171.
Kattan, G. H., H. Alvarez-Lopez, & M. Giraldo. (1994). Forest fragmentation and bird extinctions: San Antonio eighty years later. *Conservation Biology, 8*, 138-146.
Kattan, G. H., H. Alvarez-Lopez, J. Schelhas, & R. Greenberg. (1996). Preservation and management of biodiversity in fragmented landscapes in the Colombian Andes. In J. Schelhas & R. S. Greenberg (*eds*). *Forest patches in tropical landscapes*. Washingotn, DC: Island Press, 3-18.
Kattan, G. H., P. Franco, V. Rojas, & G. n. Morales. (2004). Biological diversification in a complex region: a spatial analysis of faunistic diversity and biogeography of the Andes of Colombia. *Journal of Biogeography, 31*, 1829-1839.
Katz, E. (1992). The big lie: human restoration of nature. *Reseach in Philosophy and Technology, 12*, 231-241.
Keating, P. L. (1997). Mapping vegetation and anthropogenic disturbances in southern Ecuador with remote sensing techniques: implications for park management. Paper presented at the Yearbook. Conference of Latin Americanist Geographers.
Kelly, D. L., E. V. J. Tanner, E. M. Nic Lughadha, & V. Kapos. (1994). Floristics and Biogeography of a Rain Forest in the Venezuelan Andes. *Journal of Biogeography, 21*, 421-440.
Kessler, M. (2002). The elevational gradient of Andean plant endemism: varying influences of taxon-specific traits and topography at different taxonomic levels. *Journal of Biogeography, 29*, 1159-1165.

Kessler, M., S. K. Herzog, J. Fjeldså, & K. Bach. (2001). Species richness and endemism of plant and bird communities along two gradients of elevation, humidity and land use in the Bolivian Andes. *Diversity and Distributions, 7*, 61-77.

Kessler, M., J. Kluge, A. Hemp, & R. Ohlemüller. (2011). A global comparative analysis of elevational species richness patterns of ferns. *Global Ecology and Biogeography, 20*, 868-880.

Kintz, D. B., K. R. Young, & K. A. Crews-Meyer. (2006). Implications of land use/land cover change in the buffer zone of a national park in the tropical Andes. *Environmental Management, 38*, 238-252.

Kiptot, E., P. Hebinck, S. Franzel, & P. Richards. (2007). Adopters, testers or pseudo-adopters? Dynamics of the use of improved tree fallows by farmers in western Kenya. *Agricultural Systems, 94*, 509-519.

Kirby, K. R., & C. Potvin. (2007). Variation in carbon storage among tree species: implications for the management of a small-scale carbon sink project. *Forest Ecology and Management, 246*, 208-221.

Klanderud, K., H. Z. H. Mbolatiana, M. N. Vololomboahangy, M. A. Radimbison, E. Roger, ò. Totland, & C. Rajeriarison. (2010). Recovery of plant species richness and composition after slash-and-burn agriculture in a tropical rainforest in Madagascar. *Biodiversity and Conservation, 19*, 187-204.

Knapp, S. (2002). Assessing patterns of plant endemism in Neotropical uplands. *The Botanical Review, 68*, 22-37.

Knee, K. L., & A. E. Encalada. (2012). La calidad del agua en la zona de Intag (Imbabura) y su relacion con el uso del suelo. *Informe publicado por Karen L. Knee con la colaboracion de Defensa y Conservacion Ecologica de Intag*.

Knoke, T., M. Weber, J. Barkmann, P. Pohle, B. Calvas, C. Medina, . . . R. Mosandl. (2009). Effectiveness and distributional impacts of payments for reduced carbon emissions from deforestation. *Erdkunde, 63*, 365-384.

Kocian, M., D. Batker, & J. Harrison-Cox. (2011). An Ecological Study of Ecuador's Intag Region: The Environmental Impacts and Potential Rewards of Mining. Tacoma, WA: Earth Economics.

Koh, L. P. & D. S. Wilcove. (2008). Is oil palm agriculture really destroying tropical biodiversity? *Conservation Letters, 1*, 60-64.

Kozlowski, T. (1999). Soil compaction and growth of woody plants. *Scandinavian Journal of Forest Research, 14*, 596-619.

Kremen, C., & A. Miles. (2012). Ecosystem services in biologically diversified versus conventional farming systems: benefits, externalities, and trade-offs. *Ecology and Society, 17*, 40.

Krömer, T., M. Kessler, S. R. Gradstein, & A. Acebey. (2005). Diversity patterns of vascular epiphytes along an elevational gradient in the Andes. *Journal of Biogeography, 32*, 1799-1809.

Kull, C. A., C. K. Ibrahim, & T. C. Meredith. (2007). Tropical forest transitions and globalization: Neo-liberalism, migration, tourism, and international conservation agendas. *Society and Natural Resources, 20*, 723-737.

Kumar, N. A. V., B. Pozzoni, & G. G. Woodall. (2005). The Effectiveness of World Bank Support for Community-Based and -Driven Development: An OED Evaluation. World Bank.

Kumar, S. (2002). Does “participation” in common pool resource management help the poor? A social cost–benefit analysis of joint forest management in Jharkhand, India. *World Development, 30*, 763-782.
Küper, W., H. Kreft, J. Nieder, N. Köster, & W. Barthlott. (2004). Large-scale diversity patterns of vascular epiphytes in Neotropical montane rain forests. *Journal of Biogeography, 31*, 1477-1487.
La Torre, M. D. L. Å., S. Herrando-Pérez, & K. R. Young. (2007). Diversity and structural patterns for tropical montane and premontane forests of central Peru, with an assessment of the use of higher-taxon surrogacy. *Biodiversity and Conservation, 16*, 2965-2988.
Lall, R. R. (2013). Haiti to plant millions of trees to boost forests and help tackle poverty: Government-backed campaign aims to double Caribbean country's forest cover by 2016, *The Guardian*. 05/28/2013.
Lamb, D., P. D. Erskine, & J. A. Parrotta. (2005). Restoration of degraded tropical forest landscapes. *Science, 310*, 1628-1632.
Lambin, E. F., H. J. Geist, & E. Lepers. (2003). Dynamics of land-use and land-cover change in tropical regions. *Annual Review of Environment and Resources, 28*, 205-241.
Lambin, E. F., & P. Meyfroidt. (2010). Land use transitions: Socio-ecological feedback versus socio-economic change. *Land Use Policy, 27*, 108-118.
Lambin, E. F., B. L. Turner, H. J. Geist, S. B. Agbola, A. Angelsen, J. W. Bruce, . . . C. Folke. (2001). The causes of land-use and land-cover change: moving beyond the myths. *Global Environmental Change, 11*, 261-269.
Larson, A. M., & F. Soto. (2008). Decentralization of natural resource governance regimes. *Annual Review of Environment and Resources, 33*, 213.
Laurance, W. F., A. K. M. Albernaz, G. Schroth, P. M. Fearnside, S. Bergen, E. M. Venticinque, & C. Da Costa. (2002). Predictors of deforestation in the Brazilian Amazon. *Journal of Biogeography, 29*, 737-748.
Lawton, R. O., & F. E. Putz. (1988). Natural disturbance and gap-phase regeneration in a wind-exposed tropical cloud forest. *Ecology*, 69, 764-777.
Legendre, P., D. Borcard, & P. R. Peres-Neto. (2005). Analyzing beta diversity: partitioning the spatial variation of community composition data. *Ecological Monographs, 75*, 435-450.
Lemmon, P. E. (1957). A new instrument for measuring forest overstory density. *Journal of Forestry, 55*, 667-668.
León, B., K. R. Young, & L. Brako. (1992). Análisis de la composición **X**orística del bosque montano oriental del Perú. In K.R. Young & N. Valencia (*eds*). *Biogeografía, Ecología y Conservación del Bosque Montano en el Perú.* Memorias del Museo de Historia Natural 21. Universidad Nacional Mayor de San Marcos, Lima, Perú. 141-154.
Leopold, A. (1949). *A Sand County Almanac*. New York: Ballantine Books.
Leopold, A. C., R. Andrus, A. Finkeldey, & D. Knowles. (2001). Attempting restoration of wet tropical forests in Costa Rica. *Forest Ecology and Management, 142*, 243-249.
Leopold, A. C., & J. Salazar. (2008). Understory species richness during restoration of wet tropical forest in Costa Rica. *Ecological Restoration, 26*, 22-26.
Letcher, S. G., & R. L. Chazdon. (2009). Rapid recovery of biomass, species richness, and species composition in a forest chronosequence in northeastern Costa Rica. *Biotropica, 41*, 608-617.
Lewis, D., S. D. Bell, J. Fay, K. L. Bothi, L. Gatere, M. Kabila, . . . C. I. Moraru. (2011). Community Markets for Conservation (COMACO) links biodiversity conservation with

sustainable improvements in livelihoods and food production. *Proceedings of the National Academy of Sciences, 108*, 13957-13962.
Lieberman, D., M. Lieberman, R. Peralta, & G. S. Hartshorn. (1996). Tropical forest structure and composition on a large-scale altitudinal gradient in Costa Rica. *Journal of Ecology, 84*, 137-152.
Liebsch, D., M. Marques, & R. Goldenberg. (2008). How long does the Atlantic Rain Forest take to recover after a disturbance? Changes in species composition and ecological features during secondary succession. *Biological Conservation, 141*, 1717-1725.
Lillesand, T. M., R. W. Kiefer, & J. W. Chipman. (2004). *Remote Sensing and Image Interpretation*, Hoboken: John Wiley & Sons Ltd.
Lindenmayer, D. B., & J. F. Franklin. (2002). *Conserving forest biodiversity: a comprehensive multiscaled approach*. Washington, DC: Island Press.
Lippok, D., S. G. Beck, D. Renison, I. Hensen, A. E. Apaza, & M. Schleuning. (2014). Topography and edge effects are more important than elevation as drivers of vegetation patterns in a neotropical montane forest. *Journal of Vegetation Science, 25*, 724-733.
Lovett, G. M., D. A. Burns, C. T. Driscoll, J. C. Jenkins, M. J. Mitchell, L. Rustad, . . . R. Haeuber. (2007). Who needs environmental monitoring? *Frontiers in Ecology and the Environment, 5*, 253-260.
Lugo, A. E. (1997). The apparent paradox of reestablishing species richness on degraded lands with tree monocultures. *Forest Ecology and Management, 99*, 9-19.
Lugo, A. E. (2009). The emerging era of novel tropical forests. *Biotropica, 41*(5), 589-591.
Lugo, A. E., & E. Helmer. (2004). Emerging forests on abandoned land: Puerto Rico's new forests. *Forest Ecology and Management, 190*, 145-161.
Lund, H. G. (2009). What is a degraded forest? White paper prepared for FAO. Gainesville, Virginia: Forest Information Services.
MacArthur, R. H. (1967). *The theory of island biogeography* (Vol. 1), Princeton: Princeton University Press.
MAE. (2014). SocioBosque: Programa de proteccion de bosques. Ministerio del Ambiente, Ecuador. http://www.ambiente.gob.ec/programa-socio-bosque/
Magurran, A. E. (2004). *Measuring biological diversity*. Malden: Blackwell Science Ltd., Cambridge.
Mansourian, S., & D. Vallauri. (2014). Restoring Forest Landscapes: Important Lessons Learnt. *Environmental Management, 53*, 241-251.
Manzi, M., & O. T. Coomes. (2009). Managing Amazonian palms for community use: A case of aguaje palm (Mauritia flexuosa) in Peru. *Forest Ecology and Management, 257*, 510-517.
Maquipucuna Foundation. Retrieved 15/04, 2013, from http://www.maqui.org/
Margules, C. R., & R. L. Pressey. (2000). Systematic conservation planning. *Nature, 405*, 243-253.
Maron, M., R. J. Hobbs, A. Moilanen, J. W. Matthews, K. Christie, T. A. Gardner, . . . C. A. McAlpine. (2012). Faustian bargains? Restoration realities in the context of biodiversity offset policies. *Biological Conservation, 155*, 141-148.
Marsh, G.P. (1864). *Man and nature, or Physical geography as modified by human action.* New York: Charles Schribner.
Martin, P. A., A. C. Newton, & J. M. Bullock. (2013). Carbon pools recover more quickly than plant biodiversity in tropical secondary forests. *Proceedings of the Royal Society B: Biological Sciences, 280*, 2013-2236.

Martin, P. H., T. J. Fahey, & R. E. Sherman. (2011). Vegetation zonation in a neotropical montane forest: environment, disturbance and ecotones. *Biotropica, 43*, 533-543.

Mather, A., J. Fairbairn, & C. Needle. (1999). The course and drivers of the forest transition: the case of France. *Journal of Rural Studies, 15*, 65-90.

Mather, A. S. (1992). The forest transition. *Area*, *24*, 367-379.

Mather, A. S., & C. Needle. (1998). The forest transition: a theoretical basis. *Area, 30*, 117-124.

Mayer, E. (2002). *The articulated peasant: household economies in the Andes*. Boulder: Westview Press.

McCune, B., & M. J. Mefford. (2011). PC-ORD for windows. *Multivariate analysis of ecological data. Version, 6.0*.

McGinty, M. M., M. E. Swisher, & J. Alavalapati. (2008). Agroforestry adoption and maintenance: self-efficacy, attitudes and socio-economic factors. *Agroforestry Systems, 73*, 99-108.

McKinney, M. L., & J. L. Lockwood. (1999). Biotic homogenization: a few winners replacing many losers in the next mass extinction. *Trends in Ecology & Evolution, 14*, 450-453.

McNeely, J. A., & G. Schroth. (2006). Agroforestry and biodiversity conservation–traditional practices, present dynamics, and lessons for the future. *Biodiversity & Conservation, 15*, 549-554.

McSweeney, K. (2004). Forest product sale as natural insurance: the effects of household characteristics and the nature of shock in eastern Honduras. *Society and Natural Resources, 17*, 39-56.

Melo, F. P., V. C. Arroyo-Rodriguez, L. Fahrig, M. Martinez-Ramos, & M. Tabarelli. (2013). On the hope for biodiversity-friendly tropical landscapes. *Trends in Ecology & Evolution, 28*, 462-468.

Menninger, H. L., & M. A. Palmer. (2006). Restoring ecological communities: from theory to practice. In D.A. Falk, M. A. Plamer, and J. B. Zedler (*eds*). *Foundations of restoration ecology,* SER. Washington, DC: Island Press, 88-112.

Menzies, H. (2014). *Reclaiming the Commons for the Common Good*. Gabriola Island: New Society Publishers.

Mercer, D. E. (2004). Adoption of agroforestry innovations in the tropics: a review. *Agroforestry Systems, 61*, 311-328.

Mesquita, R. C., K. Ickes, G. Ganade, & G. B. Williamson. (2001). Alternative successional pathways in the Amazon Basin. *Journal of Ecology, 89*, 528-537.

Metzger, J.-P., & H. Décamps. (1997). The structural connectivity threshold: an hypothesis in conservation biology at the landscape scale. *Acta Oecologica, 18*, 1-12.

Meyfroidt, P., E. F. Lambin, K.-H. Erb, & T. W. Hertel. (2013). Globalization of land use: distant drivers of land change and geographic displacement of land use. *Current Opinion in Environmental Sustainability, 5*, 438-444.

Michon, G., De Foresta, H., Levang, P., & Verdeaux, F. (2007). Domestic forests: a new paradigm for integrating local communities' forestry into tropical forest science. *Ecology and Society*, *12*, 1.

Mitchell, M. G., E. M. Bennett, & A. Gonzalez. (2013). Linking landscape connectivity and ecosystem service provision: current knowledge and research gaps. *Ecosystems, 16*, 894-908.

Mittermeier, R. A., N. Myers, C. G. Mittermeier, & P. Robles Gil. (1999). *Hotspots: Earth's biologically richest and most endangered terrestrial ecoregions*: CEMEX, SA, Agrupación Sierra Madre, SC.

Mittermeier, R. A., N. Myers, J. B. Thomsen, G. da Fonseca, & S. Olivieri. (1998). Biodiversity Hotspots and Major Tropical Wilderness Areas: Approaches to Setting Conservation Priorities. *Conservation Biology, 12*, 516-520.

Montagnin, F., E. Gonzalez, C. Porras, & R. Rheningans. (1995). Mixed and pure forest plantations in the humid neotropics: a comparison of early growth, pest damage and establishment costs. *The Commonwealth Forestry Review, 74*, 306-314.

Moore, D. S. (1993). Contesting terrain in Zimbabwe's eastern highlands: political ecology, ethnography, and peasant resource struggles. *Economic Geography, 69*(4), 380-401.

Mosandl, R., S. Günter, B. Stimm, & M. Weber. (2008). Ecuador suffers the highest deforestation rate in South America. In E. Beck, J. Bendix, I. Kottke, F. Makeschin, R. Mosandl (*eds*), *Gradients in a Tropical Mountain Ecosystem of Ecuador*. Springer: Ecological Studies. *198*, 37-40.

Mulligan, M. (2010). Modeling the tropics-wide extent and distributionof cloud forest and cloud forest loss, with implications for conservation priority. In L.A. Bruijnzeel, F. N. Scatena, & L. S. Hamilton *(eds)*. *Tropical montane cloud forests: science for conservation and management*, International Hydrology Series: Cambridge University Press. 14.

Mulligan, M., & S. Burke. (2005*a*). FIESTA: Fog Interception for the Enhancement of Streamflow in Tropical Areas. Final Technical Report for AMBIOTEK, DFID FRP R7991.

Mulligan, M., & S. M. Burke. (2005*b*). Global cloud forests and environmental change in a hydrological context. Final Report, DFID FRP. Project ZF0216.

Muñiz-Castro, M. A., G. Williams-Linera, & J. M. R. Benayas. (2006). Distance effect from cloud forest fragments on plant community structure in abandoned pastures in Veracruz, Mexico. *Journal of Tropical Ecology, 22*, 431-440.

Myers, N. (1997). The world's forests and their ecosystem services. In G. Daily (ed) *Nature's Services: societal dependence on natural ecosystems*. Washington, DC: Island Press, 215-235.

Myers, N., R. A. Mittermeier, C. G. Mittermeier, G. A. Da Fonseca, & J. Kent. (2000). Biodiversity hotspots for conservation priorities. *Nature, 403*, 853-858.

Nagendra, H. (2009). Drivers of regrowth in South Asia’s human impacted forests. *Current Science, 97*, 1586-1592.

Nanni, A. S., & H. R. Grau. Agricultural adjustment, population dynamics and forests redistribution in a subtropical watershed of NW Argentina. *Regional Environmental Change*, *14*,1-9.

Nave, A. G., & R. R. Rodrigues. (2007). Combination of species into filling and diversity groups as forest restoration methodology. In R. R. Rodrigues, S. V. Martin & S. Gandolfi (*eds*). *High Diversity Forest Areas: Methods and Projects in Brazil*. New York: Nova Science Publishers, 103-126.

Nelson, A., & K. Chomitz. (2007). The Forest–Hydrology–Poverty Nexus In Central America: An Heuristic Analysis. *Environment, Development and Sustainability, 9*, 369-385.

Nepstad, D. C., C. Uhl, & E. A. Serrao. (1991). Recuperation of a degraded Amazonian landscape: forest recovery and agricultural restoration. *Ambio, 20,* 248-255.

Netting, R. M. (1972*a*). Of Ken and Meadows: Strategies of Alpine Land Use. *Anthropological Quarterly, 43*, 132-144.
Netting, R. M. (1972*b*). Of Ken and Meadows: Strategies of Alpine Land Use. *Anthropological Quarterly, 45*, 132-144.
Netting, R. M. (1976). What Alpine Peasants Have in Common: Observations on Communal Tenure in a Swiss Village. *Human Ecology, 4*, 135-146.
Netting, R. M. (1993). *Smallholders, householders: farm families and the ecology of intensive, sustainable agriculture*. Chigago: Stanford University Press.
Norden, N., R. L. Chazdon, A. Chao, Y. H. Jiang, & B. Vílchez-Alvarado. (2009). Resilience of tropical rain forests: tree community reassembly in secondary forests. *Ecology Letters, 12*, 385-394.
Noss, R. F. (1983). A regional landscape approach to maintain diversity. *BioScience, 33*, 700-706.
Oksanen, J. (2011). Multivariate analysis of ecological communities in R: vegan tutorial. *R package version, 1*(7).
Ortega-Pieck, A., F. López-Barrera, N. Ramírez-Marcial, & J. G. García-Franco. (2011). Early seedling establishment of two tropical montane cloud forest tree species: The role of native and exotic grasses. *Forest Ecology and Management, 261*, 1336-1343.
Osterhoudt, S. R. (2014). *The Forest in the Field: The Cultural Dimension of Agroforestry Landscapes in Madagascar.* (Ph.D. Dissertation), Yale.
Ostrom, E. (1985). *Institutional arrangements for resolving the commons dilemma: some contending approaches*. Paper presented at the the 46th National Conference of the American Society for Public Administration, Bloomington, Indiana.
Pagiola, S., A. Arcenas, & G. Platais. (2005). Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. *World development, 33*, 237-253.
Palmer, M. A., D. A. Falk, & J. B. Zedler. (2006). Ecological theory and restoration ecology. *Foundations of Restoration Ecology.* Island Press, Washington, DC, 1-10.
Parrotta, J. A. (1992). The role of plantation forests in rehabilitating degraded tropical ecosystems. *Agriculture, Ecosystems & Environment, 41*, 115-133.
Parrotta, J. A. (1995). Influence of overstory composition on understory colonization by native species in plantations on a degraded tropical site. *Journal of Vegetation Science, 6*, 627-636.
Parrotta, J. A. (2010). *The role of forest biodiversity in the sustainable use of ecosystem goods and services in agro-forestry, fisheries and forestry.* Paper presented at the International Symposium for the Convention on Biological Diversity, Tokyo, Japan.
Parrotta, J. A., & O. H. Knowles. (1999). Restoration of Tropical Moist Forests on Bauxite-Mined Lands in the Brazilian Amazon. *Restoration Ecology, 7*, 103-116.
Parrotta, J. A., J. W. Turnbull, & N. Jones. (1997). Catalyzing native forest regeneration on degraded tropical lands. *Forest Ecology and Management, 99*, 1-7.
Pascarella, J. B., T. M. Aide, M. I. Serrano, & J. K. Zimmerman. (2000). Land-use history and forest regeneration in the Cayey Mountains, Puerto Rico. *Ecosystems, 3*, 217-228.
Pattanayak, S. K., D. E. Mercer, E. Sills, & J.-C. Yang. (2003). Taking stock of agroforestry adoption studies. *Agroforestry Systems, 57*, 173-186.

Pattanayak, S. K., S. Wunder, & P. J. Ferraro. (2010). Show me the money: Do payments supply environmental services in developing countries? *Review of Environmental Economics and Policy*, *4*, 254-274.

Pedraza, R., & G. Williams-Linera. (2003). Evaluation of native tree species for the rehabilitation of deforested areas in a Mexican cloud forest. *New Forests, 26*, 83-99.

Pellissier, L., M. Anzini, L. Maiorano, A. Dubuis, J. Pottier, P. Vittoz, & A. Guisan. (2013). Spatial predictions of land-use transitions and associated threats to biodiversity: the case of forest regrowth in mountain grasslands. *Applied Vegetation Science, 16*, 227-236.

Peluso, N. L. (1992). *Rich forests, poor people: Resource control and resistance in Java*. Oakland: University of California Press.

Pena-Domene, M. D. L., C. Martinez-Garza, & H. F. Howe. (2013). Early recruitment dynamics in tropical restoration. *Ecological Applications, 23*,1124-1134.

Persha, L., A. Agrawal, & A. Chhatre. (2011). Social and ecological synergy: local rulemaking, forest livelihoods, and biodiversity conservation. *Science, 331*, 1606-1608.

Persha, L., & K. Andersson. (2014). Elite capture risk and mitigation in decentralized forest governance regimes. *Global Environmental Change, 24*, 265-276.

Perz, S. G. (2007). Grand theory and context-specificity in the study of forest dynamics: forest transition theory and other directions. *The Professional Geographer, 59*, 105-114.

Perz, S. G., & D. L. Skole. (2003). Secondary forest expansion in the Brazilian Amazon and the refinement of forest transition theory. *Society & Natural Resources, 16*, 277-294.

Peters, T., T. Drobnik, H. Meyer, M. Rankl, M. Richter, R. Rollenbeck, . . . J. Bendix. (2013). Environmental Changes Affecting the Andes of Ecuador. In J. Bendix, E. Beck, A. Bräuning, F. Makeschin, R. Mosandl, S. Scheu & W. Wilcke (*eds*). *Ecosystem Services, Biodiversity and Environmental Change in a Tropical Mountain Ecosystem of South Ecuador.* Springer, 19-29.

Phalan, B., M. Onial, A. Balmford, & R. E. Green. (2011). Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science, 333*, 1289-1291.

Piotto, D., F. Montagnini, M. Kanninen, L. Ugalde, & E. Viquez. (2004*a*). Forest plantations in Costa Rica and Nicaragua: performance of species and preferences of farmers. *Journal of Sustainable Forestry, 18*, 59-77.

Piotto, D., E. Vıquez, F. Montagnini, & M. Kanninen. (2004*b*). Pure and mixed forest plantations with native species of the dry tropics of Costa Rica: a comparison of growth and productivity. *Forest Ecology and Management, 190*, 359-372.

Pirard, R., G. de Buren, & R. Lapeyre. (2014). Do PES Improve the Governance of Forest Restoration? *Forests, 5*, 404-424.

Pompeu, G. d. S. S., L. S. Rosa, M. M. Santos, R. S. Modesto, & T. A. Vieira. (2012). Adoption of agroforestry systems by smallholders in Brazilian Amazon. *Tropical and Subtropical Agroecosystems, 15*, 165-172.

Porter-Bolland, L., E. A. Ellis, M. R. Guariguata, I. Ruiz-Mallén, S. Negrete-Yankelevich, & V. Reyes-García. (2012). Community managed forests and forest protected areas: An assessment of their conservation effectiveness across the tropics. *Forest Ecology and Management, 268*, 6-17.

Portillo-Quintero, C., A. Sanchez, C. Valbuena, Y. Gonzalez, & J. Larreal. (2012*a*). Forest cover and deforestation patterns in the Northern Andes (Lake Maracaibo Basin): a synoptic assessment using MODIS and Landsat imagery. *Applied Geography, 35*, 152-163.

Portillo-Quintero, C. A., A. M. Sanchez, C. A. Valbuena, Y. Y. Gonzalez, & J. T. Larreal. (2012*b*). Forest cover and deforestation patterns in the Northern Andes (Lake Maracaibo Basin): A synoptic assessment using MODIS and Landsat imagery. *Applied Geography, 35*, 152-163.

Posada, J. M., T. M. Aide, & J. Cavelier. (2000). Cattle and weedy shrubs as restoration tools of tropical montane rainforest. *Restoration Ecology, 8*, 370-379.

Power, A. G. (2010). Ecosystem services and agriculture: tradeoffs and synergies. *Philosophical Transactions of the Royal Society B: Biological Sciences, 365*, 2959-2971.

Prendergast, J. R., R. M. Quinn, & J. H. Lawton. (1999). The gaps between theory and practice in selecting nature reserves. *Conservation Biology, 13*, 484-492.

Pretty, J. (2003). Social capital and the collective management of resources. *Science, 302*, 1912-1914.

Putz, F. E., & C. Romero. (2014). Futures of tropical forests (sensu lato). *Biotropica, 46,* 495-505.

Raberg, L. M., & T. K. Rudel. (2007). Where are the sustainable forestry projects?: A geography of NGO interventions in Ecuador. *Applied Geography, 27*, 131-149.

Raffles, H. (1999). "Local Theory": Nature and the Making of an Amazonian Place. *Cultural Anthropology, 14*, 323-360.

Rainforest Concern. (2009). Management Plan Neblina Reserve Rainforest Concern. Supporting documentation for the areas declaration as protected forest and vegetation.: Rainforest Concern. Retrieved 10/15, 2010, http://www.rainforestconcern.org/projects/ecuador/

Ramírez, B. L., P. Lavelle, J. A. Orjuela, & O. Villanueva. (2012). Caracterización de fincas ganaderas y adopción de sistemas agroforestales como propuesta de manejo de suelos en Caquetá, Colombia. *Revista Colombiana de Ciencias Pecuarias, 25*, 391-401.

Ramirez-Villegas, J., F. Cuesta, C. Devenish, M. Peralvo, A. Jarvis, & C. A. Arnillas. (2014). Using species distributions models for designing conservation strategies of Tropical Andean biodiversity under climate change. *Journal for Nature Conservation, 22*, 391-404.

Reardon, T., & S. A. Vosti. (1995). Links between rural poverty and the environment in developing countries: asset categories and investment poverty. *World Development, 23*, 1495-1506.

Redo, D. J., H. R. Grau, T. M. Aide, & M. L. Clark. (2012). Asymmetric forest transition driven by the interaction of socioeconomic development and environmental heterogeneity in Central America. *Proceedings of the National Academy of Sciences, 109*, 8839-8844.

Ren, H., Z. Li, W. Shen, Z. Yu, S. Peng, C. Liao, . . . J. Wu. (2007). Changes in biodiversity and ecosystem function during the restoration of a tropical forest in south China. *Science in China Series C: Life Sciences, 50*, 277-284.

Rhemtulla, J. M., D. J. Mladenoff, & M. K. Clayton. (2007). Regional land-cover conversion in the US upper Midwest: magnitude of change and limited recovery (1850–1935–1993). *Landscape Ecology, 22*, 57-75.

Rhoades, C. C., G. E. Eckert, & D. C. Coleman. (1998). Effect of pasture trees on soil nitrogen and organic matter: implications for tropical montane forest restoration. *Restoration Ecology, 6*, 262-270.

Richards, P. W. (1996). *The tropical rain forest: an ecological study* (2ed.) Cambridge: Cambridge University Press.

Richter, M., K.-H. Diertl, P. Emck, T. Peters, & E. Beck. (2009). Reasons for an outstanding plant diversity in the tropical Andes of Southern Ecuador. *Landscape Online, 12*, 1-35.
Robbins, P. (2001). Tracking invasive land covers in India, or why our landscapes have never been modern. *Annals of the association of American Geographers, 91*, 637-659.
Rodrigues., R. R., S.V. Martin & S. Gandolfi (2007). *High Diversity Forest Areas: Methods and Projects in Brazil*. New York: Nova Science Publishers.
Rohter, L. (2004). Mapuche Indians in Chile struggle to take back forests. *New York Times, 11*, 3.
Rondon, X. J., D. L. Gorchov, & F. Cornejo. (2009). Tree species richness and composition 15 years after strip clear-cutting in the Peruvian Amazon. *Plant Ecology, 201*, 23-37.
Rudel, T. K. (2000). Organizing for sustainable development: conservation organizations and the struggle to protect tropical rain forests in Esmeraldas, Ecuador. *AMBIO: A Journal of the Human Environment, 29*, 78-82.
Rudel, T. K. (2009*a*). Three Paths to Forest Expansion: A Comparative Historical Analysis. In J. S. Harini Nagendra (*ed*), *Reforesting Landscapes Linking Pattern and Process*. Dordrecht: Springer.
Rudel, T. K. (2009*b*). Tree farms: driving forces and regional patterns in the global expansion of forest plantations. *Land Use Policy, 26*, 545-550.
Rudel, T. K., D. Bates, & R. Machinguiashi. (2002). A tropical forest transition? Agricultural change, out-migration, and secondary forests in the Ecuadorian Amazon. *Annals of the Association of American Geographers, 92*, 87-102.
Rudel, T. K., O. T. Coomes, E. Moran, F. Achard, A. Angelsen, J. Xu, & E. Lambin. (2005). Forest transitions: towards a global understanding of land use change. *Global Environmental Change, 15*, 23-31.
Ruiz-Jaén, M. C., & T. M. Aide. (2005). Vegetation structure, species diversity, and ecosystem processes as measures of restoration success. *Forest Ecology and Management, 218*, 159-173.
Ruiz-Jaen, M. C., & T. M. Mitchell Aide. (2005). Restoration success: how is it being measured? *Restoration Ecology, 13*, 569-577.
Salazar, L., J. Homeier, M. Kessler, S. Abrahamczyk, M. Lehnert, T. Krömer, & J. Kluge. (2013). Diversity patterns of ferns along elevational gradients in Andean tropical forests. *Plant Ecology & Diversity*, *7*, 1-12.
Sampaio, A. B., K. D. Holl, & A. Scariot. (2007). Regeneration of Seasonal Deciduous Forest Tree Species in Long-Used Pastures in Central Brazil. *Biotropica, 39*, 655-659.
Sánchez-Cuervo, A. M., T. M. Aide, M. L. Clark, & A. Etter. (2012). Land cover change in Colombia: surprising forest recovery trends between 2001 and 2010. *PloS one, 7*, e43943.
Santa Lucía Cloud Forest Reserve. Santa Lucia Cloud Forest Retrieved 20/12, 2013, from http://www.santaluciaecuador.com/
Sarmiento, F. O. (1995*a*). Human impacts on the cloud forests of the upper Guayllabamba river basin, Ecuador, and suggested management responses. In L. Hamilton, J. Juvik, F. Scatena (*eds*). *Tropical montane cloud forests.* Ecological Studies, Springer*, 110,* 284-295.
Sarmiento, F. O. (1995*b*). *Naming and knowing an Ecuadorian landscape: a case-study of the Maquipucuna Reserve.* Paper presented at the The George Wright Forum.
Sarmiento, F. O. (2002). Anthropogenic change in the landscapes of highland Ecuador. *Geographical Review, 92*, 213-234.

Sarmiento, F. O., S. P. Churchill, H. Balslev, E. Forero, & J. L. Luteyn. (1995). *Restoration of equatorial Andes: The challenge for conservation of trop-Andean landscapes in Ecuador.* In P. Churchill, H. Balslev, E. Forero, J. L. Luteyn *(eds)*. *Biodiversity and conservation of Neotropical montane forests. Proceedings of a symposium,* New York Botanical Garden, 21-26 June 1993. 541-549.
Sarmiento, F. O., & L. M. Frolich. (2002). Andean cloud forest tree lines: naturalness, agriculture and the human dimension. *Mountain Research and Development, 22*, 278-287.
Sarmiento, M. (2013, 07/24). Colombia Takes Lead In Latin American Biodiversity Offsetting, http://www.ecosystemmarketplace.com/pages/dynamic/article.page.php?page_id=9856
Satake, A., & T. K. Rudel. (2007). Modeling the forest transition: forest scarcity and ecosystem service hypotheses. *Ecological Applications, 17*, 2024-2036.
Scatena, F., L. Bruijnzeel, P. Bubb, & S. Das. (2010). Setting the stage. In L.A. Bruijnzeel, F. N. Scatena, & L. S. Hamilton *(eds)*. *Tropical montane cloud forests: science for conservation and management*, International Hydrology Series, Cambridge University Press, 1.
Schelhas, J. & R. S. Greenberg (1996). *Forest patches in tropical landscapes*. Washington, DC: Island Press.
Schelhas, J., & G. A. Sánchez-Azofeifa. (2006). Post-frontier forest change adjacent to Braulio Carrillo National Park, Costa Rica. *Human Ecology, 34*, 407-431.
Scherr, S. J. (2000). A downward spiral? Research evidence on the relationship between poverty and natural resource degradation. *Food Policy, 25*, 479-498.
Schmook, B., R. Palmer Dickson, F. Sangermano, J. M. Vadjunec, J. R. Eastman, & J. Rogan. (2011). A step-wise land-cover classification of the tropical forests of the Southern Yucatán, Mexico. *International Journal of Remote Sensing, 32*, 1139-1164.
Schneider, L. C. (2004). Bracken fern invasion in southern Yucatan: A case for Land-Change Science. *Geographical Review, 94*, 229-241.
Schroth, G., & C. A. Harvey. (2007). Biodiversity conservation in cocoa production landscapes: an overview. *Biodiversity and Conservation, 16*, 2237-2244.
Schroth, G., C. A. Harvey, & G. Vincent. (2004). Complex agroforests: their structure, diversity, and potential role in landscape conservation. *Agroforestry and biodiversity conservation in tropical landscapes.* Washington, DC: Island Press. 227-260.
Scoones, I. (1998). Sustainable rural livelihoods: a framework for analysis. IDS Working Paper 72.
Scoones, I. (2009). Livelihoods perspectives and rural development. *The Journal of Peasant Studies, 36*, 171-196.
Scotti-Saintagne1, C., C. W. Dick, H. Caron, G. G. Vendramin, V. Troispoux, P. Sire, . . . I. Scotti. 2013. Amazon diversification and cross-Andean dispersal of the widespread Neotropical tree species Jacaranda copaia (Bignoniaceae). *Journal of Biogeography, 40*, 707–719.
Sears, R. R., C. Padoch, & M. Pinedo-Vasquez. (2007). Amazon forestry tranformed: integrating knowledge for smallholder timber managemet in Eastern Brazil. *Human Ecology, 35*, 697-707.
Selden, T. M., & D. Song. (1994). Environmental quality and development: is there a Kuznets curve for air pollution emissions? *Journal of Environmental Economics and Management, 27*, 147-162.

Seuss, Dr. (1971). *The Lorax.* New York: Random House.
SER. (2004). The SER International Primer on Ecological Restoration, from http://www.ser.org
Shiferaw, B. A., J. Okello, & R. V. Reddy. (2009). Adoption and adaptation of natural resource management innovations in smallholder agriculture: reflections on key lessons and best practices. *Environment, Development and Sustainability, 11*, 601-619.
Shvidenko, A., C. V. Barber, & R. Persson. (2005). Forest and Woodland Systems. *Ecosystems and Human Well-being: Current State and Trends, Millennium Ecosystem Assessment.*
Sierra, R., & J. Stallings. (1998). The dynamics and social organization of tropical deforestation in Northwest Ecuador, 1983-1995. *Human Ecology, 26*, 135-161.
Silver, W. L., L. M. Kueppers, A. E. Lugo, R. Ostertag, & V. Matzek. (2004). Carbon sequestration and plant community dynamics following reforestation of tropical pasture. *Ecological Applications, 14*, 1115-1127.
Sloan, S. (2008). Reforestation amidst deforestation: simultaneity and succession. *Global Environmental Change, 18*, 425-441.
Smith, N. J. H., I. C. Falesi, P. d. T. Alvim, & E. A. S. Serrão. (1996). Agroforestry trajectories among smallholders in the Brazilian Amazon: innovation and resiliency in pioneer and older settled areas. *Ecological Economics, 18*, 15-27. doi: http://dx.doi.org/10.1016/0921-8009(95)00057-7
Sodhi, N. S., G. Acciaioli, M. Erb, & A. K.-J. Tan. (2008). *Biodiversity and human livelihoods in protected areas: case studies from the Malay Archipelago*. Cambridge: Cambridge University Press.
Solimano, A. (2003). Workers remittances to the Andean Region: mechanisms, costs and development impact. *Santiago: UN-ECLAC. Mimeo*, 5-8.
Sommer, B. W., & M. K. Quinlan. (2009). *The oral history manual*. Lanham: Rowman Altamira.
Sood, K. K., & C. P. Mitchell. (2009). Identifying important biophysical and social determinants of on-farm tree growing in subsistence-based traditional agroforestry systems. *Agroforestry Systems, 75*, 175-187.
Sood, K. K., & C. P. Mitchell. (2011). Household level domestic fuel consumption and forest resource in relation to agroforestry adoption: Evidence against need-based approach. *Biomass and Bioenergy, 35*, 337-345.
Stadtmüller, T. (1987). *Cloud forests in the humid tropics: a bibliographic review*: Bib. Orton IICA/CATIE.
Stanturf, J. A., B. J. Palik, M. I. Williams, R. K. Dumroese, & P. Madsen. (2014). Forest Restoration Paradigms. *Journal of Sustainable Forestry, 33*, S161-S194.
Stattersfield, A. J., M. J. Crosby, A. J. Long, & D. C. Wege. (1998). Endemic bird areas of the world: priorities for biodiversity conservation. Cambridge: Birdlife International.
Stern, M. J. (1998). Field comparisons of two rapid vegetation assessment techniques with permanent plot inventory data in Amazonian Peru. In Dallmeier, F., & Comiskey, J. A. *Forest Biodiversity Research, monitoring and modeling: conceptual background and old world case studies.* UNESCO: Parthenon Publishing Group.
Stevens, J. P. (2012). *Applied multivariate statistics for the social sciences*. Routledge, 5th ed.
Styger, E., H. M. Rakotondramasy, M. J. Pfeffer, E. Fernandes, & D. M. Bates. (2007). Influence of slash-and-burn farming practices on fallow succession and land degradation in the rainforest region of Madagascar. *Agriculture, Ecosystems & Environment, 119*, 257-269.

Suárez, A., G. Williams-Linera, C. Trejo, J. I. Valdez-Hernández, V. M. Cetina-Alcalá, & H. Vibrans. (2012). Local knowledge helps select species for forest restoration in a tropical dry forest of central Veracruz, Mexico. *Agroforestry Systems, 85*, 35-55.
Sunderlin, W. D., A. Angelsen, & S. Wunder. (2003). Forests and poverty alleviation. *State of the World's Forests*, 61-73.
Sveiby, K. E. (2001). A knowledge-based theory of the firm to guide in strategy formulation. *Journal of Intellectual Capital, 2*, 344-358.
Tanner, E., P. Vitousek, & E. Cuevas. (1998). Experimental investigation of nutrient limitation of forest growth on wet tropical mountains. *Ecology, 79*, 10-22.
Temperton, V. M., R. J. Hobbs, T. Nuttle, & S. Halle. (2004). *Assembly rules and restoration ecology: bridging the gap between theory and practice*. Washington: Island Press.
Terborgh, J., & R. B. Foster. (1996). Tropical Tree Communities: A Test of the Nonequilibrium Hypothesis. *Ecology, 77*, 561-567.
Thoreau, H. D. (1962). Walking. Boston: *Atlantic Monthly*, June, 1962.
Tobin, J. (1958). Estimation of relationships for limited dependent variables. *Econometrica: Journal of the Econometric Society*, 24-36.
Torres-Lezama, A., E. Vilanova, H. Ramirez-Angulo, G. Alciaturi. (2012). Socioeconomic and Environmental Basis for the Development of Small Scale Forestry in a Highly Degraded Watershed in the Venezuelan Andes. *Small-Scale Forestry, 11*, 321-337.
Trawick, P. (2001). The moral economy of water: Equity and antiquity in the Andean commons. *American Anthropologist, 103*, 361-379.
Tropek, R., O. Sedláček, J. Beck, P. Keil, Z. Musilová, I. Šímová, & D. Storch. (2014). Comment on "High-resolution global maps of 21st-century forest cover change". *Science, 344*, 981.
Tscharntke, T., Y. Clough, T. C. Wanger, L. Jackson, I. Motzke, I. Perfecto, . . . A. Whitbread. (2012). Global food security, biodiversity conservation and the future of agricultural intensification. *Biological Conservation, 151*, 53-59.
Turner, B., & P. Robbins. (2008). Land-change science and political ecology: Similarities, differences, and implications for sustainability science. *Annual Review of Environment and Resources, 33*, 295-316.
Turner, B. L., E. F. Lambin, & A. Reenberg. (2007). The emergence of land change science for global environmental change and sustainability. *Proceedings of the National Academy of Sciences, 104*, 20666-20671.
Turner, I., Y. Wong, P. Chew, & A. bin Ibrahim. (1997). Tree species richness in primary and old secondary tropical forest in Singapore. *Biodiversity & Conservation, 6*, 537-543.
Turner, I. M., & R. T. Corlett. (1996). The conservationv alue of small, isolated fragments of lowland tropical rain forest. *Trends in Ecology and Evolution, 11*, 330-333.
Uhl, C., R. Buschbacher, & E. Serrao. (1988). Abandoned pastures in eastern Amazonia. I. Patterns of plant succession. *The Journal of Ecology*, *76*, 663-681.
Uhl, C., K. Clark, H. Clark, & P. Murphy. (1981). Early plant succession after cutting and burning in the upper Rio Negro region of the Amazon basin. *The Journal of Ecology*, *69*, 631-649.
Uhl, C., & C. F. Jordan. (1984). Succession and nutrient dynamics following forest cutting and burning in Amazonia. *Ecology*, *65,* 1476-1490.
UNEP. (2013). The Billion Tree Campaign - Growing Green: United Nations Environment Program.

Uprety, Y., H. Asselin, Y. Bergeron, F. Doyon, & J.-F. Boucher. (2012). Contribution of traditional knowledge to ecological restoration: practices and applications. *Ecoscience, 19*, 225-237.

Vaillant, J. (2006). *The Golden Spruce: A True Story of Myth, Madness and Greed*. New York: W.W. Nordon and Company.

van der Hammen, T. (1974). The Pleistocene changes of vegetation and climate in tropical South America. *Journal of Biogeography*, *1*, 3-26.

Vargas-Rodriguez, Y. L., W. J. Platt, J. A. Vázquez-García, & G. Boquin. (2010). Selecting relict montane cloud forests for conservation priorities: the case of western Mexico. *Natural Areas Journal, 30*, 156-173.

Verma, S., S. Tsephal, & T. Jose. (2004). Pepsee systems: grassroots innovation under groundwater stress. *Water policy, 6*, 303-318.

Vieira, D. L. M., K. D. Holl, & F. M. Peneireiro. (2009). Agro-Successional Restoration as a Strategy to Facilitate Tropical Forest Recovery. *Restoration Ecology, 17*, 451-459.

Vieira, I. C. G., T. Gardner, J. Ferreira, A. C. Lees, & J. Barlow. (2014). Challenges of Governing Second-Growth Forests: A Case Study from the Brazilian Amazonian State of Para. *Forests, 5*, 1737-1752.

Vogelmann, H. (1973). Fog precipitation in the cloud forests of eastern Mexico. *BioScience*, *23,* 96-100.

Walters, B. B., A. Cadelina, A. Cardano, & E. Visitacion. (1999). Community history and rural development: why some farmers participate more readily than others. *Agricultural Systems, 59*, 193-214.

Walters, B. B., C. Sabogal, L. K. Snook, & E. d. Almeida. (2005). Constraints and opportunities for better silvicultural practice in tropical forestry: an interdisciplinary approach. *Forest Ecology and Management, 209*, 3-18.

Wassenaar, T., P. Gerber, P. Verburg, M. Rosales, M. Ibrahim, & H. Steinfeld. (2007). Projecting land use changes in the Neotropics: The geography of pasture expansion into forest. *Global Environmental Change, 17*, 86-104.

Watkins, J. E., C. Cardelús, & R. Colwell. (2006). Vascular epiphyte distribution patterns: explaining the mid-elevation richness peak. *Journal of Ecology, 94*, 144-156.

Waylen, K. A., P. J. McGowan, & E. Milner-Gulland. (2009). Ecotourism positively affects awareness and attitudes but not conservation behaviours: a case study at Grande Riviere, Trinidad. *Oryx, 43*, 343-351.

Weber, M., S. Günter, N. Aguirre, B. Stimm, & R. Mosandl. (2008). Reforestation of abandoned pastures: silvicultural means to accelerate forest recovery and biodiversity. In E. Beck, J. Bendix, I. Kottke, F. Makeschin, R. Mosandl (*eds*.), *Gradients in a Tropical Mountain Ecosystem of Ecuador*. Springer: Ecological Studies. *198*, 431-441.

Webster, G. L., S. Churchill, H. Balslev, E. Forero, & J. L. Luteyn. (1995). *The panorama of Neotropical cloud forests.* In P. Churchill, H. Balslev, E. Forero, J. L. Luteyn *(eds.)*. *Biodiversity and conservation of Neotropical montane forests. Proceedings of a symposium,* New York Botanical Garden, 21-26 June 1993. 541-549.

West, M. (2002). Sparkling Fountains or Stagnant Ponds: An Integrative Model of Creativity and InnovationImplementation in Work Groups. *Applied Psychology An International Review, 51*, 355-387.

West, M. A., C. A. Sacramento, & D. Fay. (2006*a*). Creativity and innovation implementation in work groups: The paradoxical role of demands. *Creativity and innovation in organizational teams*, 137-159.
West, P., J. Igoe, & D. Brockington. (2006*b*). Parks and peoples: the social impact of protected areas. *Annual Review of Anthropology, 35*, 251-277.
Whitmore, T. C. (1990). *An introduction to tropical rain forests*. New York: Oxford University Press.
Wilcke, W., S. Yasin, U. Abramowski, C. Valarezo, & W. Zech. (2002). Nutrient storage and turnover in organic layers under tropical montane rain forest in Ecuador. *European Journal of Soil Science, 53*, 15-27.
Williams, M. (1992). *Americans and their forests: a historical geography*. Cambridge: Cambridge University Press.
Williams, M. (2011). Deforesting the Earth: From Prehistory to Global Crisis, an Abridgment. *Journal of Agricultural and Environmental Ethics, 24*, 673-686.
Williams-Linera, G. (2002). Tree species richness complementarity, disturbance and fragmentation in a Mexican tropical montane cloud forest. *Biodiversity & Conservation, 11*, 1825-1843.
Williams-Linera, G., M. Toledo-Garibaldi, & C. G. Hernandez. (2013). How heterogeneous are the cloud forest communities in the mountains of central Veracruz, Mexico? *Plant Ecology, 214*, 685–701.
Wilson, E. O. (1992). *The diversity of life*. Cambridge: Harvard University Press.
Wilson, E. O. (2002). *The future of life*. New York: Random House.
Wilson, S. (2013). Making tree projects work for the rural poor in Latin America. Research to Practice – Strengthening Contributions to Evidence-Based Policymaking: ISID-CIDA. PB-2013-15.
Winterhalder, B., & R. B. Thomas. (1978). *Geoecology of Southern Highland Peru: A Human Adaptation Perspective.* A Joint Contribution to the Peruvian and United States Unesco Man and the Biosphere (MAB) Program, Project 6, Studies of the Impact of Human Activities on Mountain and Tundra Ecosystems. University of Colorado, Institute of Arctic and Alpine Research.
Wolf, J. H. (1993). Diversity patterns and biomass of epiphytic bryophytes and lichens along an altitudinal gradient in the northern Andes. *Annals of the Missouri Botanical Garden*, *80*, 928-960.
Wunder, S. (1996). Deforestation and the uses of wood in the Ecuadorian Andes. *Mountain Research and Development*, *16,* 367-381.
Wunder, S. (2001). Poverty alleviation and tropical forests—what scope for synergies? *World Development, 29*, 1817-1833.
Wunder, S. (2005). Payments for environmental services: some nuts and bolts. Jakarta: CIFOR, *42*.
Wunder, S. (2011). PES for improved ecosystem water services in Pimampiro town, Ecuador.
Wunder, S. (2013). When payments for environmental services will work for conservation. *Conservation Letters, 6*(4), 230-237.
Wunder, S., & M. Albán. (2008). Decentralized payments for environmental services: the cases of Pimampiro and PROFAFOR in Ecuador. *Ecological Economics, 65*(4), 685-698.
Wunderle Jr, J. M. (1997). The role of animal seed dispersal in accelerating native forest regeneration on degraded tropical lands. *Forest Ecology and Management, 99*, 223-235.

WWF. Retrieved 11/13, 2010, from http://www.worldwildlife.org/home-full.html
Yee, T. W. (2010). The VGAM package for categorical data analysis. *Journal of Statistical Software, 32*, 1-34.
Young, K. R. (1998). Deforestation in landscapes with humid forests in the central Andes: Patterns and processes. In K. S. Zimmerer & K. R. Young (*eds*). *Nature's geography: new lessons for conservation in developing countries*, Madison: University of Wisconsin Press, 75-99.
Young, K. R. (2009). Andean land use and biodiversity: humanized landscapes in a time of change. *Annals of the Missouri Botanical Garden, 96*, 492-507.
Young, K. R. (2011). Introduction to Andean Geographies. In S. K. Herzog, R. Martínez, P. M. Jørgensen, H. Tiessen (*eds*). *Climate change and biodiversity in the Tropical Andes*, Inter-American Institute for Global Change Research, Climate Change and Biodiversity in the Tropical Andes, 128-140.
Young, K. R., & B. León. (1995). Distribution and Conservation of Peru's Montane Forests: Interactions Between the Biota and Human Society. In L. Hamilton, J. Juvik, F. Scatena (*eds*). *Tropical montane cloud forests.* Ecological Studies: Springer*, 110,* 363-376.
Young, K. R. & B. León. (1995). Connectivity, social actors, and conservation policies in the central Andes. The case of Peru's montane forest. In P. Churchill, H. Balslev, E. Forero, J. L. Luteyn (*eds*). *Biodiversity and conservation of Neotropical montane forests. Proceedings of a symposium,* New York Botanical Garden, 21-26 June 1993.
Young, K.R. & B. León. (1999). Peru's humid eastern montane forests: an overview of their physical settings, biological diversity, human use and settlement, and conservation needs. Diversity of Andean Rainforests, DIVA. Technical Report:1–97.
Young, K. R., & P.L. Keating (2001). Remnant forests of Volcán Cotacachi, northern Ecuador. *Arctic, Antarctic, and Alpine Research, 33,* 165-172.
Young, K. R., & J. K. Lipton. (2006). Adaptive governance and climate change in the tropical highlands of western South America. *Climatic Change, 78*, 63-102.
Young, K. R., C. U. Ulloa, J. L. Luteyn, & S. Knapp. (2002). Plant evolution and endemism in Andean South America: an introduction. *The Botanical Review, 68*, 4-21.
Young, T. P. (2000). Restoration ecology and conservation biology. *Biological Conservation, 92*, 73-83.
Young, T. P., D. A. Petersen, & J. J. Clary. (2005). The ecology of restoration: historical links, emerging issues and unexplored realms. *Ecology Letters, 8*, 662-673.
Zahawi, R. A., & C. Augspurger. (1999). Early Plant Succession in Abandoned Pastures in Ecuador1. *Biotropica, 31*, 540-552.
Zahawi, R. A., K. D. Holl, R. J. Cole, & J. L. Reid. (2013). Testing applied nucleation as a strategy to facilitate tropical forest recovery. *Journal of Applied Ecology, 50*, 88-96.
Zambrano, A. M. A., E. N. Broadbent, & W. H. Durham. (2010). Social and environmental effects of ecotourism in the Osa Peninsula of Costa Rica: the Lapa Rios case. *Journal of Ecotourism, 9*, 62-83.
Zanella, M. A., C. Schleyer, & S. Speelman. (2014). Why do farmers join Payments for Ecosystem Services (PES) schemes? An Assessment of PES water scheme participation in Brazil. *Ecological Economics, 105*, 166-176.
Zbinden, S., & D. R. Lee. (2005). Paying for Environmental Services: An Analysis of Participation in Costa Rica's PSA Program. *World Development, 33*, 255-272.

Zhai, D.-L., J.-C. Xu, Z.-C. Dai, C. H. Cannon, & R. Grumbine. (2014). Increasing tree cover while losing diverse natural forests in tropical Hainan, China. *Regional Environmental Change, 14*, 611-621.
Zimmerer, K. S. (1998). The ecogeography of Andean potatoes. *BioScience*, *48*(6), 445-454.
Zorrilla, C. (2010). Guia de reforestacion para las comunidades de Intag, http://www.decoin.org. Retrieved May 20, 2010.

APPENDIX A: HOUSEHOLD SURVEY QUESTIONNAIRE

Consent Script

Mi nombre es Sarah Wilson y soy (o mi nombre es ------y trabajo para Sarah Wilson quien es) una estudiante investigadora del Departamento de Geografía de la Universidad de McGill en Montreal, Canadá. Estamos muy interesada en conocer más acerca de sus proyectos de plantar arboles. Específicamente, nos gustaría entender como ha afectado los proyectos de plantar arboles las vidas de la gente que vive aquí.

Si usted acepta en participar en esta investigación, me gustaría platicar con usted acerca de su granja, su ocupación y su participación en los proyectos de restauración, incluyendo por qué decidió restaurar los bosques, y cómo estos proyectos han impactado (beneficiado o afectado) a usted, a su familia y a su comunidad. Si usted esta de acuerdo, me gustaría visitar las áreas restauradas en el que usted participó como parte del proyecto de restauración. La hora y el lugar de la entrevista de acuerdo a lo que usted le convenga más.

Me gustaría mencionar que su participación es voluntaria, usted tiene el derecho a no contestar las preguntas que no quiera, adicionar información que usted crea conveniente, y puede parar su participación en la entrevista en cualquier momento que usted desee. Es probable que usted no se vea beneficiado directamente con su participación en éste estudio, pero la información que usted proporcione podrá servir para ayudar a otras personas y a otros proyectos, para entender los retos y las necesidades de la población rural en esta región. Además, la información que usted proporcione puede ser utilizada en el tesis de Sarah, en publicaciones académicas, presentada en forma de resumen para DECOIN, o presentados en una conferencia académica. Todas las transcripciones de las entrevistas se mantendrán en una computadora protegida con contraseña para que sólo yo tenga acceso a la información, o guardadas en un cajón bajo llave, la cual sólo yo tendré acceso.

¿Tiene alguna duda o pregunta para mi?.
¿Estaría interesado en participar en el proyecto? En caso de que aceptara me gustaría grabar las conversaciones. ¿Me daría su permiso para grabar las conversaciones? Las conversaciones solo van ser utilizadas para transcribir las respuestas de su entrevista.

(Voy a proporcionarles mis datos de contacto al final de las entrevistas).
Sarah Wilson: sarahjwilson00@yahoo.ca, 001-514-276-0698.

Código de la Familia (Hogar) ___________________________
Nombre del jefe de Familia (Hogar) y (esposa ó esposo) ____
____
Nombre del entrevistado (otro que los jefes de familia)____
Coordenadas Geográficas _____________________________ **Día**____
Entrevistadores____

A. Demografía

1. Por cada miembro de familia (presente o ausente): Cuales son los nombres de la personas en la familia?

Nombre y apellido	**Edad**	**Genero**	**Relación**	**Donde nació**	**Donde vive**	**Cuando salió/ llegó**	**Ocupación**	**Educación (Nivel de estudios, números de años en la escuela)**

2. En que año se formo está familia? ______________(año) (En que ano el jefe de familia y su esposa/o moviéndose a su residencia aparte?)

3. En que año llegó usted a esta comunidad? ______________(año) (Si vivía aquí siempre, pase a la pregunta 4)

→Dónde vivía antes?____ (Lugar y ciudad)

→Si es de otro lugar, porqué vino a vivir aquí?) (Cual fue la principal razón, tuvo que ver con la organización DECOIN, la creación de la reserva, o por otras actividades u oportunidades ofrecidas)_________________________
____
____
____

4. En qué trabaja usted? Y los miembros de su familia? (pregunta algunas preguntas de seguimiento considerando las preguntas anteriores para dirigir la discusión – Miriam, Sarah o Jake) ____

B. Tierras (en el año 2010)

1. Por favor, describa las tierras de su propiedad y los tierras en que trabaja si son aquilladas o son a medias. (Pregunte cada de los siguiente categorías).

→Por cada categoría, voy a preguntar que son los principales producidos desde la semana santa en el año 2010 y semana santa en 2011 – entonces, un ano en total.

→(Si es posible, hay que ir al campo con el entrevistado en este punto de la entrevista para conocer la tenencia de la tierra)

Categoría	**Área privada (Ha)**	**Área aquillado o a medias* (Ha)**	**Principales cultivos producidos y cosechados en el ultimo ano**		
Total de tierra propia/cultivada			-------------------	------------------	------------------
Bosque:					
Primerio o Monte					
Chaparro					
Plantados					
Tierra agrícola:					
Cultivos					
Potreros (con pasto cebolla o no, señalar)					
Potreros con arboles para la sombra (Silviopastoreo)					
Huerto /parcela familiar					
Huertos de café					
Huertos de plántanos					
Frutales					
Otros tipos de vegetación/ usos de la tierra (residencial, arbustos, pantanos, humedal)					
Las tierras alquiladas no incluídas anteriormente?					
Las tierras rentadas no incluídas anteriormente?					

2. Vendió parte de sus tierras para la creación de la reserva? (Si no, pase a la pregunta 3) ________________

→ **Si la respuesta es afirmativa**: → a) Cuántas hectáreas (Qué extensión?) ?__________ha

→ b) Para qué era utilizada la tierra cuándo la vendió?____

→ c) Me podrías mostrar (indicar) que parte de la reserva usted solía ser dueño? (En caso afirmativo y si Sarah ya esta, utiliza la hoja para la historia del uso de la tierra. Si no tiene tiempo ahorita, o Sarah no esta, si es posible agenda una cita con Sarah para ver las tierras)?

→ d) Qué hizo usted con el dinero que recibió con la venta de la tierra?

Uso	Si?	Cantidad de moneda	Características
Gastos de la casa?			
- Comida			
- Ropa			
- Muebles / platos / sabinas / cobijas…			
Compra de equipo y herramientas? Cuáles?			
Paga los estudios de un hijo?			
Guarda en la casa para si hay un enfermedad?			
Compra semillas?			
Compra otra tierra? Donde?			
Herridos?			
Pone en el banco?			
Una boda o fiesta?			
Otro?			

3. ¿Usted alquilaba la tierra que vendió para crear la reserva en el momento de la venta? (en caso afirmativo, pase a la pregunta 2.3. (arriba)

C. Agricultura

1. ¿Cómo se mantiene el suelo fértil en sus tierras? (Roza tumba (tala) y quema? Fertilizantes químicos? Abono orgánico de animales? Abono orgánico de las plantas/ composta? Ceniza?)____________________________________

____

____

2. Ha cambiado la forma en que usted mantiene el suelo fértil en los últimos 8 años? __
→En caso afirmativo, Cómo ha cambiado? (disminución / aumento de la roza tumba y quema? usa mas/menos de las fertilizantes químicos? Más/menos abono orgánico? u otros métodos?)____________________________________

____

____

→En caso afirmativo, por qué ha cambiado la forma de mantiene el suelo fértil? (ej. La disminución de los bosques, disminución de la fertilidad del suelo, recibieron capacitación / los subsidios / asistencia del gobierno / organizaciones no gubernamentales (especificar), otro cambio en el ambiente (especificar)...____________________

____

3. Si usted **utiliza fertilizantes químicos**: a) En que cultivos? ____

____

→ b) Cuándo empezó a utilizar fertilizantes químicos? ____
→ c) Cómo se enteró o supo hacerca de los fertilizantes químicos? ____

____

4. Si utiliza **composta / abono orgánico de las hojas o animales**: a) En que los cultivos?__________________________

____

→b) Cuándo empezó a utilizar productos orgánicos (composta, abono, fertilizantes orgánicos? __________________

____

→c) Recibió cursos o capacitación para aprender ha hacer composta?________________ De quien? ______________

____

5. Utiliza algún otro químico? (pesticidas, herbicidas, fungicidas? ____________________________________
→ a) en caso afirmativo en que cultivos? ____
→ b) Cuándo empezó a utilizar estos químicos? ____

→ c) Porqué empezó a utilizar estos químicos? ____

6. Cultiva los frijoles de rama? ____________
→ Si la respuesta es si, tiene un sistema de alambre para colgar los matas? ____Hasta que ano tiene este sistema? _
→ Si no tiene un sistema de alambre: → usa maíz o chaparos?____
→quemar las tierras después de cosechar el frijoles?______________________

D. Ingresos a la casa

1. Que son la producción del hogar por en el año pasado (un ano en total)?
→ Si un cultivo es solo para la casa, pregunta cuantas veces por semana cosecha, y cuantas unidades por vez.
→ **Ponga la total cosechada por el hogar, de los tierras propias, aquilladas y a medias (la porción de hogar).**

Cultiva: Producto	**Tiene?**	**Se vende (poco)?**	**Unidad**	**cantidad cosechada en total**	**cantidad vendido**	**Por los tres cultivos más importantes, cosecho más o menos en los últimos 8 anos?**
Maíz y choclo			Quintal o libras			
Frijol de rama			Quintal o libras			
Frijol de mata			Quintal o libras			
Yuca			canastas			
Tomate de árbol			sacos llenos			
Naranjilla			Gavetas			
Plátanos			recemos			
Coles			Uni. por semana o unidades			
Zanahorias			Uni./ semana o libras			
Cebolla			Uni./semana o tallos (vender)			
Otras verduras			Uni./semana o libras (vender)			
Camote			canastas			
Limones			sacos			
Aguacates			canastas			
Otras frutas			Quintales			
Caña de azúcar			Libra de panela o plantas			
Café			Libra			**Cosecho más o menos de este cultivo en los últimos 8 años que antes?**
Vacunos			unidades			
Chanchos			unidades			
Cuyes			unidades			
Gallinas/ pollos			unidades			
Huevos			docena			
Queso			unidades			
Leche			litros			
Cabuya			quintales			
Madera*			Tablas			
Lena*			canastas			
Otras productos del bosque* (frutas, hongos, medicina,						

hojas…)						
Otras						

* Si la persona cosecha y vendió estos productos, en el fin de entrevista (antes de sección M) preguntarles sección de ingresos forestales. Si no cosecha, no usa esta sección.

2. En general, como ha cambiado su producción aquí desde 8 anos (**si llegaron después de 8 anos**, desde Ud. llegó aquí o desde que formó su familia aquí)?____

____

____

____

3. Ingresos Salariales
¿Algún miembro del hogar recibió salario del trabajo en el ano pasado?

Nombre de la persona	**Tipo de Trabajo**	**Días Trabajados**	**Salario por día**	**En la granja/ fuera de la granja?**	**Dónde? (localmente, en la ciudad (especifique)…**

4. Tienda/ ingresos del negocio
Usted es dueño de una tienda o de algún negocio? ____________________________________
→Si la respuesta es sí, que tipo de tienda-negocio? ____
→ Cuántos ingresos tuvo en el ano pasado?________($ E.U. dólares)
→ Desde hace cuánto usted adquirió la tienda? __________________________(años, o en que ano)

5. Ha vendido arboles en el ano pasado? __________
→ Si la respuesta es sí, de su propio vivero o un vivero comunitario? _________________________
→ Cuantos ingresos tuvo de vender arboles en el ano pasado?______________($)
→ Ha vendido a quien? ____

6. Otros ingresos: En el año pasado, recibió dinero de:
- Gobierno? ____________________($US). Por qué razón? ______________________________
- Algún miembro de la familia? __________________($US). Dónde viven? ___________________

____

- Alguna otra fuente no mencionada anteriormente? _____________________($US). Qué fuente? __________

______ ____

E. Activos de los hogares

1. Posesiones: Tiene un/una:

Tipo	**Tiene/ no tiene**	**Año de adquisición**
Carro/ camión		
Motocicleta		
Tractor		
Bicicleta		
Radio		
TV		

Teléfono		
Computadora		
Tipo	**Tiene/ no tiene**	**Año de adquisición**
Reproductor de Casete/CD/ VHS/DVD		
Concina de gas		
Cocina de leña		
Refrigerador/ congelador		
Motosierra		
Arado		
Rifle/ Carabina		
Carretilla		
Una bomba de fumiga		
Sistemas de riego		
Tienda		
Vivero		
Casa en otra parte		
Bodega/ Chingana		
Otros (precio de compra mas de cien cincuenta dólares)		

Vendió o perdió algunas posesiones en los ultimo 8 anos? ___________ Cuales posesiones y porque?_____________

____

____

3. Es dueño de ganado/ animales? ________→ En caso afirmativo:

Tipo de Ganado	**Número de cabezas de ganado**	**Han incrementado ó disminuido en el ano pasado? Porqué?**	**Han incrementado o disminuido en los últimos 8 anos? Porqué?**

→ Usa vitaminas, inyecciones o hormonas con el ganado? __________Si la respuesta es sí, desde que ano?_______ _______________

Si ahora no tiene ganado (vacas o chanchos): Ha tenido ganado en el pasado? ________________En caso afirmativo, Que tipo y cuántos?____ ______________________________

→ Cuándo dejo de tener ganado? ____

→ Porque no tiene ganado ahorita? ____

____

4. Usted es dueño de su casa o la renta?__________________________

De que material están hechas la mayoría de las paredes de su casa? ______________________________

De que tipo de material es el techo? ____

Cuántos pisos hay en su casa? ____

(El entrevistor puede evidenir pregunta 4 si es posible)

F. Tala del Bosque (deforestación)

1. ¿Alguien en su hogar corto (taló) árboles del **bosque monte** en el último ano? _______ **o Chaparos** en el último ano?__________ (No → pasa a pregunta 10).

→ Si la respuesta es 'si':

2. ¿Cuánta tierra fue deforestada?		ha	
3. Cuál fue la razón principal por la que talaron el bosque? (ejemplos, para cortar madera o leña, para hacer pastos, para plantar frijoles de ramas, aclarar el bosque para otros usos de la tierra?			
4. ¿Para qué se uso la tierra deforestada? *(Señalar orden de prioridad. Max. 3)*		si/ no	**Ha bosque/ chaparos**
	Cultivos		
	Potreros		
	usos no agrícolas (especificar)		
5. Si se usó para cultivos, ¿cuál fue el principal cultivos establecido? *Señale orden de prioridad. Max. 3*			
6. ¿Qué tipo de bosque tumbó? (Bosque primerio/ monte; chaparos/ bosques bajas; bosques plantados) (si hay mas de uno, especificar por cual razón cada tipo de bosque fue talado)			
7. Si era bosque secundario/ chaparos o plantado ¿cuál era la edad del bosque?		anos	
8. Quién es o era el dueño del bosque talado (deforestado)? Código: 1 = Privado (propiedad del hogar), 2 = privado (propiedad de un miembro de la familia), 3 = privado rentado/ a media, 4 = comunal, 5 = gobierno/ municipal, 6 = otro			
9. ¿A qué distancia de la casa estaba la tierra deforestada?			

10. ¿El hogar ha talado bosques montes o chaparos durante los últimos 8 años? _____(Si 'no', pasa a 12).

11. **Si la respuesta es 'sí':** ¿cuántas hectáreas, aproximada, han sido deforestada durante los últimos 8 años? ._____________________ha. Que tipo del bosque? (monte, chaparro o plantado)__________________________

Por cual razón? ____

Nota: Está pregunta debe incluir el área reportada en la pregunta 2.

12. El cantidad de bosque que usted ha talado por año ha cambiado en los últimos 8 años)? 1= menos tala, 2= igual, 3=más tala (No → pasa a 14).____________________________

→ Si ha habido un cambio, como ha cambiado y porqué? ____

____

13. ¿Su familia recolecta leña? ____

→ **Si la respuesta es 'sí'**, para usarla en la casa o para vender? ____

→ ¿Comparando con 8 años atrás, colectar más o menos leña que antes?_______________________________________

14. ¿Cuánta tierra usada por la familia ha sido dejada para la regeneración natural de su vegetación o chaparos cada ano (durante los últimos 8 años?) __________ Ha/ano. (Si necesario, elabore)________________________

G. Base de Recursos Forestales

1. ¿Cuál es la distancia entre la casa y el límite del bosque **primero/ monte** más cercano al que se tiene acceso y

puede ser utilizado? _____(minutos a pie)

2. Su familia ha plantado árboles en sus tierras? (si no, pase a la pregunta 6) ___________________

3. **Si la respuesta es "sí":** → a. Durante el año pasado? ______________________b. En los ultimo 8 anos? __________
c. Más tiempo antes que eso? _____________________________.

d. ¿cuáles son los principales propósitos de la plantación de estos árboles? Por favor, señale el orden de prioridad de los propósitos más importantes. Max. 3. Notas:	**Propósito**	**Si/ no**	**Rango 1-3**
	1. Leña para uso doméstico		
	2. Leña para la venta		
	3. Madera/postes para uso doméstico		
	4. Madera/postes para la venta		
	5. Otros usos domésticos		
	6. Otro productos para la venta		
	7. Calidad/ cantidad de agua		
	8. Otros servicios ambientales -especifique(e.g., control de erosión del suelo, protección contra el viento)		
	11. Delimitación de la tierra		
	12. Incrementar el valor de mi tierra		
	13. Permitir a mis hijos y nietos conocer estos árboles		
	14. Sombre para los animales		
	15. Para tener un bosque lindo		
	16. Otro		

4. Qué especies de árboles plantó en sus granjas?

Especies de árboles plantadas	**Cuántas?**	**En qué año las plantó?**	**Usted escogió estas especies?**

5. De dónde obtuvo las semillas o plantas? (ej. Usted las colectó, usted las produjo, o vienen del vivero de la comunidad?) ____
____

H. Adopciones

1. Tiene un sistema de biogás? ____________________________(si la respuesta es no, pasa a 2).
→ En que ano fue estableada?____________________Recibió apoya o capitación? De quien?_____________________
____
→ Porque decidió a usar esta sistema? ____
____

2. Que hace con la basura inorgánica de la casa? (y Donde bota?)____

3. ¿Tiene usted un sistema de compostaje? _________________(si la respuesta es no pasa a sección I)
→ En que ano empezó con una sistema de compostaje (y sin o con lombriz de tierra)?___________________________
→ Porque empezó con una sistema de compostaje? Recibió apoya o capitación de una organización? ____________

I. Membrecías en grupos comunitarios, conexiones con el gobierno local

1. Era usted o algún miembro de su hogar un miembro de algún grupo comunitario o ha participado en el gobierno local?

Nombre de la persona	Nombre del grupo	Año / mes de ingreso	¿Porqué se unió al grupo?

2. Usted o algún miembro de su hogar utiliza internet? ____________________ (si no pase a la pregunta 3).

→ Si la respuesta es sí: Cuándo empezó o empezaron usar internet?________________________________

→ Qué hace cuándo está en línea? (email, Actualización de páginas de internet / mantenimiento, búsqueda de mercado, otro (especifique?))____

→ . Usted o un miembro de su hogar recibió capacitación para usar el internet? De quien?_____________________
____

3. Alguna vez se ha comunicado con el gobierno local? (si no, pase a la pregunta 4).

Si la respuesta es sí→ Porqué motivo(s)? (subvenciones para equipamiento, información, otro)?

→Cuándo empezó a contactar al gobierno local? Cuándo se terminó la comunicación?

¿Cómo se le ocurrió la idea de contactar al gobierno y escuchar acerca de las subvenciones? (un amigo, trabajador de una ONG, funcionario del gobierno, grupo de la comunidad...)

Propósito del contacto	Año en que empezó	Año en que terminó	contacto a través de

4. Usted recibió capacitación para solicitar apoyo financiero o materiales o subsidios? Si no, salte la sección N.

→ En caso afirmativo ¿Qué tipo de capacitación? ____

¿En qué año?____

¿Quién lo proporcionó?____

I. La participación pasada en los proyectos de cuencas hidrográficas de la reserva

1. Usted o alguno de los miembros de su hogar han trabajado para la reserva? (Si no, pase a la pregunta 9)

--> Si la respuesta es sí, quién en el hogar participó en las actividades, y cómo (llene el cuadro siguiente)?

Nombre	Tipo de Trabajo (1=colectar semillas, 2=construir el vivero3 = producir árboles, 4 = plantar, mantenimiento 5 = otro, especifique)	Año/ Estación Participación Fecha de inicio	Año/ Estación Participación Fecha de término	Los trabajos se realizan durante meses o estaciones específicas? Cuáles?	# Contratos o periodo de trabajo por año Cuantos días por contrato?	Cuál es el salario por día, o **el pago por contrato**?	Cuántos árboles por contrato?

2. Cuánto dinero su hogar gano en total por los proyectos? ________________ .
3. Que hizo con el dinero que ganó? (Las siguientes categorías serán definidas con los grupos de trabajo)

Uso	**Cantidad**
Gastos de la casa?	
- Comida	
- Ropa	
- Muebles / platos / sabinas / cobijas…	
Compra de equipo y herramientas? Cuáles?	
Paga los estudios de un hijo?	
Guarda en la casa para si hay un enfermedad?	
Compra semillas?	
Compra otra tierra? Donde?	
Herridos?	
Pone en el banco?	
Una boda o fiesta?	
Otro?	

4. Cuáles fueron sus razones para participar en los proyectos de plantar árboles en la reserva? (Preguntar la pregunta anterior, después de que la persona terminé de responder, revisa las opciones. Finalmente pregunta al entrevistado que ordene las cinco principales razones)	**Razones**	**sí/ no**	**Orden de importancia**
	Mejorar la calidad del agua		
	Mejor manejo forestal, más beneficios en el futuro		
	Acceso a otros beneficios (ej. Apoyo del gobierno, subsidios, programas)		
	Mi deber de proteger el bosque para el beneficio de la comunidad ó mi familia en el futuro		
	Para recibir salario por mi trabajo en las reservas		
	Para recibir dinero por la venta de mis tierras		
	Aspectos sociales (ej. Conocer personas, trabajar en equipo)		
	Obligado por los líderes de la comunidad, vecinos,		
	Conocer mejor los productos forestales (madera?)		
	Aprender nuevas habilidades e información		
	Reducir la presión sobre los recursos forestales		
	otro, especifique		

5. Recibió entrenamiento o capacitación durante el proyecto? ________________________________
→ En caso afirmativo, Qué tipo de capacitación recibió? ___________________________________
____

6. Conozco a funcionarios del gobierno, o solicito apoyo al gobierno local a través de los trabajos de reforestación?
____
____

7. Aprendió nuevas técnicas agrícolas de las personas involucradas en los proyectos de reforestación? ________
→ En caso afirmativo, Que técnicas aprendió? ____
____
→ Ha utilizado estas técnicas en tu granja? _________En caso que si, Cómo funcionaron?______________
____
____

8. Gano algo más de los proyectos de reforestación? Recibió:	**Árboles?**	**Si/ no**	**Notas/ detalles**
	Acceso al vivero?		
	Información/ capacitación en ecoturismo?		
	Acceso a créditos?		
	Infraestructura para riego/ mejora del acceso al agua?		
	otro (especifica)		

9. Si no participó en los proyectos de plantar árboles en la reserva, enumere las 3 principales razones por la cuales no participó	**Razones**	**Sí/no**
	No me invitaron a participar	
	No me permitieron participar	
	Soy nuevo en la comunidad	
	Los miembros de los grupos pertenecían a otro grupo social, político, religioso que yo.	
	No tengo tiempo para participar	
	Enfermedad en la familia	
	No creo que los proyectos iban a cumplir con la promesa de producir más agua	
	Al ser miembro o participar iba a restringir mi uso del bosque, y quiero seguir utilizando el bosque según como lo vaya necesitando	
	No creo que el grupo es muy efectivo en plantar o manejar el bosque	
	No estoy interesado en las actividades realizadas por los proyectos de reforestación	
	Corrupción en los proyectos	
	Me interesaba participar en el proyecto pero no sabía mucho sobre el proyecto	
	Mi familia no se enteró de los proyectos de reforestación	
	No me gustan los objetivos del DECOIN (ONG)	
	Otro, especifique:	

10. En general, cómo considera que la creación de la reserva y la implementación de los proyectos de reforestación afectaron: (Código: 1=un gran efecto negativo; 2=pequeño efecto negativo; 3=no efecto; 4=pequeño efecto positivo;5=gran efecto positivo.)	Calidad del agua?	
	Cantidad del agua?	
	La habilidad de los miembros de la comunidad de trabajar juntos?	
	Su situación financiero?	
	Su situación personal o los beneficios que la familia obtiene del bosque?	

11. Ha trabajado con DECOIN en otras actividades en el pasado? ____

→ En caso afirmativo, antes o después de la implementación de los proyectos de la reserva? _______________ Qué hizo?

____

____

12. Ha trabajado con otras ONG´s o con proyectos del gobierno / otras organizaciones de apoya en el pasado? _________

→En caso afirmativo ¿Cuáles? ____

→ En qué años? Después o antes la reserva fue estableada?____

J. Participación actual en los proyectos de la cuenca hidrológica.

1. Actualmente o en el ano pasado participaba en las actividades de plantación y mantenimiento para las reservas, **o en otros grupos de trabajo en los bosques o para vender los arboles en la comunidad**? (si no, pase a la siguiente sección).

1. En caso afirmativo, En qué grupo?				
2. Cuántas personas-días (días de trabajo completos) miembros de la familia participaron en total en actividades de grupo (reuniones, vigilancia, juntas, etc.) **en el ano pasado?**	Nombre del miembro de la Familia	Posición	# Días por año	
3. ¿Su hogar recibió algún salario/ dinero en el año pasado?				
4. **Si la respuesta es 'sí':** ¿Cuánto recibió en este periodo? (US$)				
5. ¿Cuáles fueron sus razones para integrarse el grupo? Por favor, señale en orden de prioridad, las razones más importantes. Max 3. **Si hizo esta pregunta por el mismo grupo arriba, pasa a sección K.**				**Razón**

	Aumento en el acceso a productos forestales	**si/ no**	**rango (1-5)**
	Mejor manejo forestal y más beneficios para el futuro		
	Acceso a otros beneficios. Ej. apoyo gubernamental o de la cooperación internacional		
	Mi obligación de proteger el bosque para la comunidad y el futuro		
	Ser respetado y reconocido como una persona responsable en la comunidad		
	Aspectos sociales (conocer gente, trabajar colectivamente, temor de ser excluido, etc.)		
	Obligado por el Gobierno/líderes locales/vecinos		

5. El grupo ha beneficiado: Su situación personal/ financiero?________ El medio ambiente? ____________________La comunidad en general? __________La cantidad o calidad del agua?__________________(Usa los códigos arriba, I 10)

K. Servicios ambientales del bosque

2. En el año pasado, ¿El hogar ha recibido pagos en efectivo o en especie relacionados a los siguientes servicios ambientales del bosque?

Propósito principal	**¿Ha recibido?**	**Si sí, cantidades (valores) recibidas por año.**	**Durante que año(s)?**	**Qué año empezó?**
1. Turismo				
3. Proyectos de protección hídrica				
4. Conservación de biodiversidad				
5. Plantación de árboles				
6. Concesiones madereras				
7. Otros, especificar				

L. Crisis y gastos inesperados

¿El hogar ha enfrentado alguna escasez significativa de ingresos o grandes gastos inesperados, en el ano pasado?

Acontecimiento	**¿Qué tan severo?** [1]	**¿Cómo enfrentó las pérdidas o los costos?** ***Señalar en orden de prioridad. Max.3)*** [2]		
Pérdida seria de cultivos				
Enfermedad seria en la familia (adultos, económicamente activos, sin poder trabajar durante más de un mes debido a la enfermedad o al cuidado de un enfermo)				
Muerte de un adulto económicamente activo				
Pérdida de tierra (expropiación, etc.)				
Pérdida grande de ganado (robo, sequía, etc.)				
Pérdida grande de otros activos (fuego, robo,				
inundación, etc.)				
Pérdida de empleo				
Boda				

1. 0=no; 1=sí, crisis moderada; 2=sí, crisis severa.
2. 1. Cosechar más productos forestales
3. Cosechar más productos agrícolas
4. Gastar más ahorros en efectivo
5. Venta de activos (tierra, ganado, etc.)
6. Trabajo extra ocasional
7. Ayuda de amigos y parientes
8. Ayuda de ONGs, organizaciones comunales, religiosas o similares
9. Obtención de préstamo de un prestamista, asociación de crédito, banco, etc.
10. Tratar de reducir el gasto en el hogar
11. No se hizo nada en particular
12. Reducción del numero de comidas consumidas
13. prestado contra ingresos futuros.
14. vendieron tierras que se utilizarían para el consumo familiar,
15. alquilar la tierra,
16. Nuevo negocio,
17. cosecha temprana de los cultivos,
18. cambiaron los tipos de cultivos sembrados,
19. especificar otros.

2. En general, cuando había gastos inesperados, que hace? (usa los ejemplos arriba)____________________

M. Percepciones sobre el bienestar

1. ¿Qué tan satisfecho está usted con su vida entre semana santa 2010 y 2011?____________________

2. ¿La producción alimentaria del hogar y los ingresos entre semana santa 2010 y 2011 han sido suficientes para cubrir lo que consideran las necesidades del hogar?____________________

3. Usted puede recibir apoyo de otros miembros de la comunidad si lo llegará a necesitar? Por ejemplo si necesitará pedir dinero prestado porque algún miembro de su familia está enfermo? ____________________

4. En comparación con otros hogares en el pueblo (o comunidad), cuál es la situación económica de su casa?
(1 = peor situados, 2 = pertenecen a la media, 3 = mejor situación)____________________

5. Cuál es la actual situación económica de tu familia en comparación de hace 8 años?
(1 = menos pudientes ahora, 2 = más o menos igual, 3 = mejor ahora)____________________

6. Cuáles son las cosas más importantes que te han pasado durante los últimos 8 años? ____________________

7. Usted considera que su comunidad es un buen lugar para vivir? Porqué si o por qué no?____________________

N. Enumerator/researcher assessment of the household – Miriam, Jake or Sarah (circle one awesome interviewer)

1. ¿Durante la entrevista el entrevistado sonrió o rio?

(1) ni rio ni sonrió (serio); (2) solo sonrió; (3) sonrió y rio; (4) rio abiertamente y frecuentemente.

2. ¿De acuerdo con sus impresiones y con lo que ha visto (casas, bienes, etc.), cual es la situación económica de este hogar comparada con otros hogares de la comunidad?

1=peor; 2=promedio; 3=mayor

3. ¿Qué tan confiable es la información dada en general por este hogar?

1=pobre; 2=razonablemente confiable; 3=muy confiable

4. ¿Qué tan confiable es la información dada por este hogar sobre recolección y uso forestal?

1=pobre; 2=razonablemente confiable; 3=muy confiable

5. Si la información forestal no es confiable (código 1 arriba), ¿cree que la información dad subestima o sobrestima el uso real de los bosques?

1=subestima; 2=sobrestima; 3= no hay sobre o subestimación sistemática; 4=no sabe

6. ¿La persona ha mencionado que los arboles son buenas para evitar el erosión del suelo?

APPENDIX B: SOIL ANALYSIS METHODOLOGY

CHEMICAL ANALYSIS OF MACRO AND MICROELEMENTS IN SOIL

Instituto Nacional Autónomo de Investigaciones Agropecuarias (INIAP) soil laboratory, Santa Catalina Station, Panamericana Sur Km. 1, Sector Cutuglagua, Cantón Mejía, Pichincha

(Translated from Spanish, http://www.iniap.gob.ec/web)

North Carolina Method/Dr. Hunter

REACTIVES

Sodium Bicarbonate, E.D.T.A. Superfloc 127 and sodium hydroxide (Olsen).

PREPARATION

a. Dissolve 420 g of NaHCO3 (baking soda) in distilled water

b. Dissolve 37.2 g of disodium E

c. Dissolve 1 g of Superfloc 127 to 200 to 400 ml with distilled water.

d. Mixing three solutions and lead to a volume of 10 l with distilled water

e. Bring the solution to pH 8.5 with 10N NaOH.

* Modified Olsen

DETERMINATION OF NITROGEN IN SOILS - AMONIACAL

APPLIANCES

Coleman Spectrophotometer 295

Analytical scale

Automatic axial agitator

Diluter dispenser

Plates with trays

Trolleys to transport plates

Polyethylene bottles (or jars)

Automatic washer

Measurements with 2.5-5-10 ml capacity.

REACTIVES

Basic phenol

Sodium Hydroxide

Sodium hypochlorite solution (Cloretol)

Distilled water

Ammonium Chloride

PREPARATION

Preparing the "Basic Phenol"

Dissolve 100 g NaOH in 500 ml of distilled water, cool, and add 138 g of phenol in cristals, or 130 ml of 92 % liquid phenol, and bring to a volume of 1 liter.

Sodium Hypochlorite Solution

Dissolve 1 volume of NaClO (Clorex) 5.25% with 9 volumes of distilled water.

Preparing pattern solution of N

Weigh 9.69 g of NH_4Cl and dissolve in distilled water to a volume of 1 liter, this preparation has a concentration 2500 ug / ml, then taken 10 ml from this solution and bring to a volume of 1 liter with the extractant solution to obtain a final concentration 25 ug / ml.

PROCEDURE

1. Take 2.5 ml of soil
   Add 25 ml of extractant solution (OLSEN modified)
   Stir 10 minutes and filter

2. Take 2 ml of filtrate
   Add 8 ml of phenol basic
   Add 10 ml of NaClO (Cloretol)
   Allow to stand for 3 hours without exposure to direct sunlight to maintain an stable color
3. Make the calibration curve by taking as the highest point the solution pattern of 25 ug / ml and as zero the extractant solution, do the same dilutions to the number 2.

Read the percent of transmission with wavelength of 630 nm.

DETERMINATION OF PHOSPHORUS IN SOIL *

APPLIANCES

The same as for nitrogen analysis, above.

REACTIVES

Tartrate of Potassium and Antimonio

Concentrated of sulfuric acid

Molybdate of Ammonium

Acacias gum Q.P.2 /

Ascorbic Acid

Fosfate monobasic

PREPARATION

SOLUTION "A" REAGENT CONCENTRATE:

1. Dissolve 1 g of Tartrate of Potassium and Antimonio in 400 ml of distilled water into one volumetric bottleof one liter

2. Add slowly while mixing, 165 ml of H_2SO_4 concentrated, cool.

3. Dissolve 7.5 g of Molybdate of Ammonium in about 300 ml of distilled water.

4. When the acid solution has cooled, add the Molybdate of Ammonium solution and bring to a volume of 1 liter with distilled water.

NOTE: This solution "A" should be stored in refrigerator to keep it without breaking down. (It is sensitive to heat and to light)

SOLUTION “B” REAGENT OF COLOR FOR PHOSPHORUS

a. The day you are going to use this solution, dilute 1g of acacia gum per liter, 1 g of ascorbic acid, mix these two suluciones, add 150 ml of solution "A" and lead to a volume of one liter with distilled water.

NOTE: The gum has to be dissolved in hot water and has to be prepared the same day of use.

PREPARATION OF PATTERN SOLUTION P

Weigh 4.39 g of KH_2PO_4 and dissolved in distilled water to a volume of one liter, this solution contains 1000 ug / ml, of P of this solution taking an aliquot of 12 ml and bring to a volume of one liter with the same extractant solution to get a final concentration of 12ug / ml P

PROCEDURE

1. Place 2.5 ml of soil in 25 ml of the extractant solution, stir for 10 minutes at a speed of 400 rpm

2. Take 2 ml aliquot of the filtrate

   Add 8 ml of distilled water

   Add 10 ml of the “B” reagent of color of Molivdenato of ammonium.

   Let stand 1 hour

3. Make the calibration curve taking as highest point the solution of 12 ug / ml of P and as zero point the extractant solution, do the same dilutions of the number 2.

4. Read the percent of transmission in a spectrophotometer at a wavelength of 680 nm.

DETERMINATION OF POTASSIUM, CALCIUM AND MAGNESIUM, ATOMIC ABSORPTION METHOD *

APPLIANCES

A spectrophotometer of atomic absorption, and the same materials that were used for the other analyzes.

REACTIVES

Lanthanum oxide 1%, HCl concentrated.

PREPARATION

Wet 58.64 g of lanthanum oxide La_2O_3 with approximately 50 ml of distilled water. Slowly and carefully add 100 ml of 37% concentrated HCl (400 ml to 20 L) and then bring up to 5 liters with distilled water.

Preparation of the pattern solution: Containing the following concentrations : 5000 ug / ml of K, 12 500 ug / ml of Ca, 5000 ug / ml of Mg.

Weigh: 9.53 g of KCl, 45,253 g of $CaCl_2$, $2H_2O$ and 41,793 g $MgCl_2.6H_2O$; and dissolve separately at a volume of one liter, then from the solution of K take 10 ml of 19 Ca take 20 ml of Mg take 10 ml and placed together in a liter flask (bottle or container?) and brimg to 1000 ml with extractent solution. The final concentrations are: 50 ug / ml of K, 250 ug / ml of Ca, and 50 ug / ml of Mg.

PROCEDURE

1. 2.5 ml of soil, 25 ml of the extractant solution, stir for 10 minutes and filter

2. Take 1 ml aliquot of the filtrate, add 9 ml of distilled water, and 15 ml of 1% lanthanum

3. Make the calibration curve using as the highest point the pattern solutions of 50-250- 50 ug / ml of K, Ca and Mg respectively and as zero the extractant solution.

4. Perform readings with the spectrophotometer of absorption atomic.

DETERMINATION OF TRACE MINERALS IN THE SOIL * (Cu-Fe-Mn-Zn)

APPLIANCES

The same equipment used for the analysis of K-Ca-Mg.

REACTIVES

Copper sulfate

Ferrous ammonium sulfate

metallic zinc

Manganese sulfate

Dilute nitric acid

PREPARATION

Preparation of pattern solution

Weigh 2.51 g of SO_4Cu, 7.02 $(SO_4)_2Fe(NH_4)_2 6H_2O$ and dissolve in aproximatly 300 ml of 1% HNO_3 and lead to a volume of 1000 ml with distilled water H_2O.

1 g of zinc metal dissolved in aproximatly 300 ml of 1% HCl and fill up to one liter with distilled H2O.
Weigh 3.076 g of SO4Mn, H2Oen about 300 ml of 1% HN03 and bring to 1000 ml with distilled water, this results in concentrations of 1000 ug / ml of each of the elements; then 4-10-5-3 ml aliquots is taken, from each of the respectivly solutions and volume takes the extractant solution and concentrations have 4-10-5-3 ug / ml of Cu-Fe-Mn-Zn

PROCEDURE

1. Add 2.5 ml of soil to 25 ml of extracting solution OLSEN. Stir 30 minutes and filter

2. Make the calibration curve using as the highest point 4-10-5-3 ug / ml of Cu-Fe-Mn-Zn, respectively, and as zero point the extractant solution.

3. Perform readings with the absorption spectrophotometer directly from the filter.

APPENDIX C: Species found in primary and secondary forests in Intag

Species	Site															
	APPA	APPL	BI	ECPA	ECPL	EPPA	EPPL	J	LC	LEPA	LEPL	NA	PVPA	PVPL	SLP	Total
Achariaceae																
Carpotroche platyptera	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Actinidiaceae																
Saurauia prainiana Buscal.	-	-	4	-	-	-	-	-	-	-	-	-	3	1	-	8
Saurauia pseudostrigillosa Buscal.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	2
Saurauia tomentosa (Kunth) Spreng.	-	-	-	-	1	-	-	-	-	-	-	4	-	-	-	5
Saurauia sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	2
Adoxaceae																
Viburnum mathewsii (Oerst.) Killip & Smith.	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	2
Anacardiaceae																
Mauria heterophylla Kunth.	-	-	-	-	-	-	-	-	-	-	-	5	-	-	-	5
Toxicodendron striatum (Ruiz & Pav.) Kuntze .	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Annonaceae																
Guatteria venosa Erkens & Maas.	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Aquifoliaceae																
Ilex guayusa Loes.	-	-	-	-	-	-	-	-	-	-	-	-	-	7	-	7
Ilex inundata Poepp. ex Reissek	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	3
Ilex sp.	-	-	4	-	-	-	-	-	-	-	-	-	-	-	-	4
Apocynaceae																
Tabernaemontana sp.	-	-	-	-	-	-	-	-	11	-	-	-	-	-	-	11
Araliaceae																
Dendropanax macrophyllus Cuatrec.	-	-	-	-	-	-	-	1	1	-	-	1	-	-	-	3
Dendropanax sp.	-	-	-	2	3	-	-	-	-	-	-	-	-	-	-	5
Oreopanax andreanus Marchal.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Oreopanax corazonensis Harms	-	3	-	-	-	3	7	-	-	-	-	-	-	143	-	156
Schefflera angulata Pav. Harms	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1
Schefflera dielsii Harms	-	6	-	-	-	-	-	-	-	-	-	-	-	-	-	6
Schefflera lasiogyne Harms	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1
Arecaceae																
Aiphanes chiribogensis Borchs. & H. Balslev	-	-	-	-	-	-	-	6	7	-	-	-	-	-	1	14
Ceroxylon alpinum Bonpl. ex DC.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Ceroxylon ventricosum Burret	-	-	1	-	1	-	-	-	-	-	-	-	-	1	-	3
Geonoma undata Klotzsch	-	-	-	-	-	-	-	17	21	-	-	-	-	-	-	38
Prestoea acuminata (Willd.) H.E. Moore	-	-	-	-	-	-	-	18	3	-	-	-	-	-	11	32
Asteraceae																
Asteraceae sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	2
Baccharis latifolia	1	-	-	-	43	44	76	-	-	-	-	-	-	-	-	164
Baccharis nitida	5	1	-	3	5	15	13	-	-	-	1	-	15	-	-	56
Barnadesia parviflora Spruce ex Benth. & Hook. f.	-	-	6	-	-	-	-	-	-	-	-	-	-	-	-	6
Baccharis trinervis	46	58	-	32	45	1	-	-	-	7	67	-	13	8	-	257
Baccharis sp.1	-	5	-	-	-	-	-	-	-	-	-	-	-	4	-	9
Baccharis sp.2	-	-	-	-	-	-	-	-	-	-	1	-	17	16	-	34

Species	Site															
	APPA	APPL	BI	ECPA	ECPL	EPPA	EPPL	J	LC	LEPA	LEPL	NA	PVPA	PVPL	SLP	Total
Clibadium eggersii Hieron.	-	-	-	-	2	-	6	-	-	1	-	-	1	-	-	10
Critoniopsis occidentalis (Cuatrec.) H. Rob.	-	-	-	-	-	-	-	11	3	-	-	-	-	-	3	17
Escallonia paniculatas (Ruiz & Pav.) Roem. & Schult.	-	-	-	-	-	-	-	-	-	-	-	-	4	-	-	4
Liabum igniarium (Bonpl.) Less.	-	-	-	-	2	4	3	-	-	-	-	-	5	-	-	14
Piptocoma discolor (Kunth) Pruski	-	-	5	-	-	-	-	-	-	-	-	-	-	-	-	5
Vernonanthura patens (Kunth) H. Rob.	-	-	-	-	-	-	-	-	-	-	-	1	-	1	-	2
Vernonia sp.	-	2	-	-	-	-	-	-	-	-	-	-	-	-	-	2
Verbesina sodiroi Hieron	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	1
Begoniaceae																
Begonia cf. Octopetala L'Hér.	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Begonia parviflora Poepp. & Endl.	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	2
Betulaceae																
Alnus acuminata Kunth	-	13	-	2	-	-	29	-	-	-	1	-	-	-	-	45
Bignoniaceae																
Arrabidaea sp.	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	1
Delostoma integrifolium D. Don	1	470	-	-	32	1	54	-	-	2	6	18	-	104	-	688
Tecoma stans (L.) Juss. ex Kunth	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	2
Boraginaceae																
Tournefortia fuliginosa Kunth	-	-	-	-	-	4	-	-	-	-	-	-	-	-	-	4
Tournefortia maculata Jacq.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1
Tournefortia sp.	-	-	-	-	-	-	22	-	-	-	8	-	-	1	-	31
Brunelliaceae																
Brunellia acostae Cuatrec.	-	-	1	-	-	-	-	-	-	-	6	-	-	-	-	7
Campanulaceae																
Centropogon solanifolius Benth.	-	-	-	-	-	-	-	-	-	-	1	-	-	4	-	5
Cannabaceae																
Celtis iguanaea (Jacq.) Sarg.	-	-	-	-	-	-	1	-	-	-	-	-	-	-	-	1
Cardiopteridaceae																
Citronella aff. Incarum (J.F. Macbr.) R.A. Howard	-	-	-	-	-	-	-	32	-	-	-	-	-	-	-	32
Caricaceae																
Carica papaya L.	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	1
Carica pubescens Lenné & C. Koch	-	-	-	-	-	-	-	-	-	-	1	-	-	8	-	9
Celastraceae																
Maytenus ebenifolia Reissek	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Chloranthaceae																
Hedyosmum cuatrecazanum Occhioni	-	-	5	-	-	-	-	-	-	-	-	24	-	-	-	29
Hedyosmum goudotianum var. goudotianum Todzia	-	-	-	-	-	-	-	-	7	-	-	1	-	-	-	8
Hedyosmum racemosum (Ruiz & Pav.) G. Don	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	2
Chrysobalanaceae																
Licania durifolia Cuatrec.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4	4
Licania grandibracteata Prance	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	3
Licania macrocarpa Cuatrec.	-	-	-	-	-	-	-	-	6	-	-	-	-	-	-	6
Licania megalophylla Prance	-	-	-	-	-	-	-	6	-	-	-	-	-	-	-	6
Cleomaceae																
Podandrogyne brevipedunculata Cochrane																

Species	Site															
	APPA	**APPL**	**BI**	**ECPA**	**ECPL**	**EPPA**	**EPPL**	**J**	**LC**	**LEPA**	**LEPL**	**NA**	**PVPA**	**PVPL**	**SLP**	**Total**
Clethraceae																
Clethra ovalifolia Turcz.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Clusiacaee																
Chrysochlamys colombiana (Cuatrec.) Cuatrec.	-	-	-	-	-	-	-	2	4	-	-	11	-	-	8	25
Chrysochlamys dependens Planch. & Triana	-	-	-	-	-	-	-	1	3	-	-	-	-	-	-	4
Clusia alata Planch. & Triana	-	-	6	-	-	-	-	-	2	-	-	-	-	-	-	8
Clusia crenata Cuatrec.	-	2	-	-	6	-	6	1	-	-	-	3	-	2	-	20
Clusia flaviflora Engl.	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	2
Clusia multiflora Kunth	-	-	-	-	-	-	-	-	2	-	-	-	1	12	1	16
Clusia pallida Engl.	-	-	-	-	-	-	-	-	4	-	-	-	-	-	-	4
Tovomita weddelliana Planch. & Triana	-	5	-	6	15	-	5	-	-	12	13	-	-	-	-	56
Cunnoniaceae																
Weinmannia balbisiana Kunth	-	-	-	-	-	-	-	7	2	-	-	1	-	-	14	22
Weinmannia macrophylla Kunth	-	-	-	-	-	-	-	-	1	-	-	1	-	-	-	2
*Weinmannia pinnat*a L.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Weinmannia pubescens Kunth	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	1
Cyatheaceae																
Alsophila erinacea (H. Karst.) D.S. Conant	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Alsophila sp.1	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Alsophila sp.2	-	-	-	-	-	-	-	6	-	-	-	-	-	-	-	6
Cyathea caracasana (Klotzsch) Domin	-	-	-	-	-	-	-	-	-	-	-	-	-	-	14	14
Cyathea pallescens (Sodiro) Domin	-	-	-	-	-	-	-	6	28	-	-	2	-	-	-	36
Cyathea sp.1	-	-	-	-	-	-	-	-	-	-	-	3	-	-	-	3
Cyathea sp.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4	4
Cyathea tortuosa R.C. Moran	-	-	18	-	-	-	-	1	-	-	1	1	-	-	-	22
Dichapetalaceae																
Stephanopodium angulatum (Little) Prance	-	-	-	-	-	-	-	12	2	-	-	-	-	-	-	14
Elaeocarpaceae																
Sloanea multiflora H. Karst	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	2
Ericaceae																
Cavendishia bracteata (Ruiz & Pav. ex J. St.-Hil.) Hoerold	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Psammisia sodiroi Hoerold	-	-	-	-	-	-	-	-	25	-	-	-	-	-	-	25
Euphorbiaceae																
Acalypha diversifolia Jacq.	-	-	-	-	-	-	-	-	-	-	-	27	-	-	-	27
Alchornea glandulosa Poepp.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Alchornea latifolia Sw.	-	-	9	-	-	-	-	-	-	-	-	-	-	-	-	9
Alchornea leptogyna Diels	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Alchornea triplinervia (Spreng.) Müll.	-	-	-	-	-	-	-	6	-	-	-	-	-	-	1	7
Croton floccosus B.A. Sm.	-	-	1	-	1	-	14	-	2	-	3	-	-	13	-	34
Croton mutisianus Kunth	-	-	-	-	-	-	-	-	-	-	-	-	-	6	-	6
Hyeronima scabrida (Tul.) Mull. Arg.	-	-	-	-	2	-	-	-	-	-	1	-	-	-	-	3
Hyeronima sp.	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	2
Sapium laurifolium (A. Rich.) Griseb.	-	-	1	-	-	-	-	-	-	-	-	-	-	1	-	2
Sapium stylare Müll. Arg.	-	-	-	-	-	-	-	1	-	-	-	-	-	-	2	3

Species	Site															
	APPA	APPL	BI	ECPA	ECPL	EPPA	EPPL	J	LC	LEPA	LEPL	NA	PVPA	PVPL	SLP	Total
Tetrorchidium euryphyllum Standl.	-	-	-	-	-	-	-	2	-	-	-	4	-	-	-	6
Tetrorchidium macrophyllum Müll. Arg.	-	-	4	-	-	-	-	-	-	-	-	2	-	-	5	11
Fabaceae																
Calliandra pittieri Standl.	-	73	-	-	11	-	5	-	-	-	7	-	-	7	-	103
Dalea mutissi	-	-	-	-	-	-	-	-	-	6	-	-	-	-	-	6
Dussia lehmannii Harms	-	-	-	-	-	-	-	1	2	-	-	-	-	-	1	4
Erythrina edulis Triana ex Micheli	-	-	2	-	2	-	5	-	-	-	5	-	-	-	-	14
Erythrina megistophylla Diels	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Inga densiflora Benth.	-	4	-	-	3	-	-	-	-	-	1	23	-	3	-	34
Inga edulis Mart.	-	-	3	-	-	-	-	-	-	-	-	-	-	-	-	3
Inga lallensis Spruce ex Benth.	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Inga aff. Striata Benth.	-	-	-	-	-	-	-	-	6	-	-	6	-	-	-	12
Inga sp.1	-	-	-	-	-	-	-	-	-	-	-	-	-	6	-	6
Inga sp.2	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Machaerium sp.	-	-	-	-	-	-	-	26	-	-	-	-	-	-	-	26
Tachigali sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Gesneriaceae																
Besleria solanoides Kunth	-	-	-	-	-	-	-	3	13	-	-	-	-	-	29	45
Icacinaceae																
Calatola costaricensis Standl.	-	-	-	-	-	-	-	16	-	-	-	-	-	-	3	19
Juglandaceae																
Juglans sp.	-	2	-	-	-	-	-	-	-	-	-	-	-	-	-	2
Lamiaceae																
Aegiphila alba Moldenke	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	2
Aegiphila novogranatensis Moldenke	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1
Amasonia sp.1	-	4	-	-	-	1	-	-	-	6	-	-	-	-	-	11
Lauraceae																
*Aniba coto (*Rusby) Kosterm.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Aniba sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Beilschmiedia alloiophylla (Rusby) Kosterm.	-	-	-	-	-	-	-	1	-	-	-	4	-	-	-	5
Beilschmiedia costaricensis (Mez & Pittier) C.K. Allen	-	-	-	-	-	-	-	4	2	-	-	25	-	-	1	32
Beilschmiedia tovarensis (Klotzsch & H. Karst. ex Meisn.) Sach. Nishida	-	-	-	-	-	-	-	17	4	-	-	-	-	-	2	23
Cinnamomum triplinerve (Ruiz & Pav.) Kosterm.	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Endlicheria formosa A.C. Sm.	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	2
Endlicheria sp.1	-	-	-	-	-	-	-	-	-	-	-	-	-	4	-	4
*Endlicheri*a sp.2	-	-	-	-	-	-	-	4	1	-	-	-	-	-	-	5
Lauraceae sp.1	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	2
Lauraceae sp.2	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Licaria sp.	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Nectandra acutifolia (Ruiz & Pav.) Mez	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Nectandra guadaripo Rohwer	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	3
Nectandra obtusata Rohwer	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-	2
Nectandra purpurea (Ruiz & Pav.) Mez	-	-	-	-	-	-	-	1	15	-	-	1	-	-	-	17
Nectandra sp.1	-	-	-	-	-	-	-	6	-	-	-	-	-	-	-	6

Species	Site															
	APPA	APPL	BI	ECPA	ECPL	EPPA	EPPL	J	LC	LEPA	LEPL	NA	PVPA	PVPL	SLP	Total
Nectandra sp.1	-	-	-	-	-	-	-	-	-	-	-	12	-	-	-	12
Nectandra sp.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4	4
Nectandra sp.4	-	-	-	-	-	-	-	-	-	-	-	5	-	-	-	5
Ocotea architectorum Mez	-	-	-	-	-	-	-	2	1	-	-	-	-	-	-	3
Ocotea cernua (Nees) Mez	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Ocotea ferruginea (Meisn.) Mez	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Ocotea floccifera Mez & Sodiro	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Ocotea glaucosericea Rohwer	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	1
Ocotea insularis (Meisn.) Mez	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Ocotea javitensis (Kunth) Pittier	-	-	-	-	-	-	-	1	3	-	-	-	-	-	-	4
Ocotea rugosa van der Werff	-	-	-	-	-	-	-	3	-	-	-	-	-	-	4	7
Ocotea sericea Kunth	-	-	7	-	-	-	-	-	-	-	-	-	-	-	-	7
Ocotea stuebelii Mez	-	-	-	-	-	-	-	5	33	-	-	-	-	-	-	38
Ocotea sp.1	1	-	-	-	-	-	-	15	-	-	-	-	-	-	-	16
Ocotea sp.2	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Ocotea sp.3	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Ocotea sp.4	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Persea aff. Americana Mill.	-	-	11	-	-	1	2	-	-	-	-	-	-	-	-	14
Persea areolatocostae (C.K. Allen) van der Werff	-	-	-	-	1	-	-	2	5	-	-	-	-	-	-	8
Persea nubigena L.O. Williams	-	-	-	-	-	-	-	-	15	-	-	-	-	-	-	15
Persea aff. Povedae W.C. Burger	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Persea rigens C.K. Allen	-	-	-	-	-	-	-	6	-	-	-	-	-	-	-	6
Pleurothyrium glabritepalum van der Werff	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Lecythidaceae																
Eschweilera caudiculata R. Knuth	-	-	-	-	-	-	-	-	-	-	-	-	-	-	6	6
Eschweilera cf. Integrifolia (Ruiz & Pav. ex Miers) R. Knuth	-	-	-	-	-	-	-	12	-	-	-	-	-	-	-	12
Eschweilera pittieri R. Knuth	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Malvaceae																
Matisia castano H. Karst. & Triana	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Sida poeppigiana Fryxell	-	1	-	-	-	1	-	-	-	-	-	-	-	-	-	2
Sida rhombifolia	-	-	-	-	-	-	-	-	-	-	-	-	-	17	-	17
Melastomataceae																
Axinaea macrophylla (Naudin) Triana	-	-	-	-	-	-	-	25	-	-	-	-	-	-	-	25
Axinaea sp.	-	-	-	-	-	-	-	-	9	-	-	-	-	-	-	9
Blakea rotundifolia D. Don	-	-	-	-	-	-	-	-	-	-	-	2	-	-	2	4
Clidemia sp.1	-	-	-	-	-	-	-	5	-	-	-	-	-	-	-	5
Clidemia sp.2	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	2
Conostegia apiculata Wurdack	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Graffenrieda cucullata (Triana) L.O. Williams	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	3
Leandra longicoma Cogn.	-	-	-	-	-	-	-	-	14	-	-	-	-	-	-	14
Leandra aff. Subseriata (Naudin) Cogn.	-	-	-	-	-	-	-	-	-	-	-	-	24	1	1	26
Meriania maxima Markgr.	-	-	-	-	-	-	-	-	-	-	-	3	-	-	-	3
Meriania tomentosa (Cogn.) Wurdack	-	-	7	-	-	-	-	-	15	-	-	-	-	-	8	30
Miconia affinis DC.	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Miconia bracteolata (Bonpl.) DC.	-	-	2	-	-	-	-	-	8	-	-	-	-	-	-	10

Species	Site															
	APPA	**APPL**	**BI**	**ECPA**	**ECPL**	**EPPA**	**EPPL**	**J**	**LC**	**LEPA**	**LEPL**	**NA**	**PVPA**	**PVPL**	**SLP**	**Total**
Miconia corymbosa (Rich.) Judd & Skean	-	-	-	-	-	-	-	-	4	-	-	-	-	-	-	4
Miconia af. corymbiformis Cogn.	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Miconia crocea (Desr.) Naudin	-	-	22	-	-	-	-	-	-	-	-	-	-	-	-	22
Miconia dapsiliflora Wurdack	-	-	-	-	-	-	-	1	-	-	-	1	-	-	-	2
Miconia rivetii Danguy & Cherm.	-	-	-	-	-	-	1	3	-	-	-	-	-	-	-	4
Miconia sp.1	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	2
Miconia sp.2	-	24	-	-	-	-	-	-	-	2	6	-	9	3	-	41
Miconia sp.3	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Miconia sp.4	-	1	-	-	-	-	-	-	-	-	-	-	-	15	-	16
Miconia sp.5	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Miconia sp.6	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	2
Miconia sp.7	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Miconia sp.8	-	-	-	-	1	-	19	-	-	-	1	-	-	-	-	21
Miconia theaezans Cogn.	-	-	16	-	-	-	-	1	1	-	-	-	-	-	1	19
Ossaea micrantha (Sw.) Macfad. ex Cogn.	-	-	-	-	-	-	-	-	1	-	-	67	-	-	16	84
Ossaea robusta (Triana) Cogn.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	6	6
Meliaceae																
Carapa guianensis Aubl.	-	-	-	-	-	-	-	107	-	-	-	-	-	-	-	107
Carapa megistocarpa A.H. Gentry & Dodson	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Cedrella odorata L.	-	1	-	-	-	-	-	-	-	-	-	-	-	-	-	1
Guarea kunthiana A. Juss.	-	-	-	-	-	-	-	4	2	-	-	11	-	-	3	20
Guarea sp.	-	-	-	-	5	-	-	-	-	-	-	-	-	-	-	5
Ruagea glabra Triana & Planch.	-	-	-	-	-	-	-	1	2	-	-	-	-	-	-	3
Ruagea tomentosa Cuatrec.	-	-	-	-	-	-	-	-	-	-	-	7	-	-	-	7
Trichilia pallida Sw.	-	-	-	-	-	-	-	-	-	-	-	-	-	5	-	5
Monimiaceae																
Mollinedia latifolia (Poepp. & Endl.) Tul.	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	3
Siparuna pilosolepidota Heilborn	-	-	2	-	-	-	-	-	1	-	-	-	-	-	-	3
Moraceae																
Castilla elastica subsp. Gummifera (Miq.) C.C. Berg	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	2
Ficus citrifolia Mill.	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Ficus cuatrecasasiana Dugand	-	-	-	-	-	-	-	4	3	-	-	1	-	-	1	9
Ficus lacunata Kvitvik	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Ficus maxima Mill.	-	-	3	-	-	-	-	-	1	-	-	-	-	-	-	4
Ficus mutisii Dugand	-	-	-	-	-	-	-	1	-	-	-	3	-	-	-	4
Helicostylis tovarensis (Klotzsch & H. Karst.) C.C. Berg	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Morus insignis Bureau	-	-	-	-	-	-	-	-	-	-	-	2	-	34	1	37
Naucleopsis capirensis C.C. Berg	-	-	-	-	-	-	-	16	-	-	-	-	-	-	-	16
Naucleopsis naga Pittier	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	3
Pseudolmedia rigida (Klotzsch & H. Karst.) Cuatrec.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Sorocea jaramilloi C.C. Berg	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Sorocea trophoides W.C. Burger	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Myristicaceae																
Otoba gordoniifolia (A. DC.) A.H. Gentry	-	-	1	-	-	-	-	3	-	-	-	-	-	-	-	4
Otoba novogranatensis	-	-	-	-	-	-	-	-	-	-	2	-	-	1	-	3

Species	Site															
	APPA	**APPL**	**BI**	**ECPA**	**ECPL**	**EPPA**	**EPPL**	**J**	**LC**	**LEPA**	**LEPL**	**NA**	**PVPA**	**PVPL**	**SLP**	**Total**
Myrtaceae																
Calyptranthes maxima McVaugh	-	-	-	-	-	-	-	-	32	-	-	27	-	-	-	59
Calyptranthes sp.1	-	-	4	-	-	-	-	-	-	-	-	-	-	-	-	4
Calyptranthes sp.2	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Calyptranthes sp.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	62	62
Eugenia crassimarginata M.L. Kawasaki & B. Holst	-	-	-	-	-	-	-	8	-	-	-	-	-	-	-	8
Eugenia florida DC.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1
Eugenia grossa B. Holst & M.L. Kawasaki	-	-	9	-	-	-	-	1	-	-	-	2	-	-	-	12
Eugenia myrobalana DC.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3	3
Myrcianthes alaternifolia Grifo	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Myrcia aff. Fallax (Rich.) DC.	-	-	-	-	-	-	-	1	6	-	-	-	-	-	9	16
Myrciaria floribunda (H. West ex Willd.) O. Berg	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	2
Myrcianthes hallii (O. Berg) McVaugh	-	-	6	-	-	-	-	-	-	-	-	-	-	-	-	6
Myrcia macrophylla DC.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8	8
Myrcianthes orthostemon (O. Berg) Grifo	-	15	-	-	-	-	-	-	-	-	-	-	1	1	-	17
Myrcianthes rhopaloides (Kunth) McVaugh	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Myrcia splendens (Sw.) DC.	-	-	-	-	-	-	-	2	-	-	-	16	-	-	-	18
Myrcia sp.	-	-	-	1	8	-	-	1	-	-	-	-	-	-	-	10
Myrtaceae sp.	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Psidium guajava L.	-	5	-	-	-	-	5	-	-	-	-	-	-	-	-	10
Onagraceae																
Fuchsia macrostigma Benth.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3	3
Papaveraceae																
Bocconia integrifolia Bonpl.	-	-	-	-	-	-	-	-	-	-	-	-	6	-	-	6
Pentaphylacaceae																
Ternstroemia lehmannii (Hieron.) Urb.	-	-	1	-	-	-	-	1	-	-	-	-	-	-	-	2
Phyllanthaceae																
Hieronima macrocarpa Müll. Arg.	-	-	8	-	-	-	-	-	-	-	-	-	-	-	-	8
Hyeronima macrocarpa Müll. Arg.	-	-	-	-	-	-	-	-	-	-	-	23	-	-	7	30
Hyeronima sp.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Piperaceae																
Peperomia armadana																
Piper aduncum L.	-	-	-	1	1	2	2	-	1	-	-	-	-	-	-	7
Piper andreanum C. DC.	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Piper auritum Kunth	-	-	13	-	-	-	-	-	-	-	-	-	-	16	-	29
Piper carpunya Ruiz & Pav.	1	-	2	-	-	-	-	-	-	-	-	-	-	-	-	3
Piper crassinervium Kunth	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Piper hispidum Sw.	-	-	-	-	-	-	-	-	-	-	-	12	-	-	-	12
Piper imperiale (Miq.) C. DC.	-	-	-	-	-	-	-	-	4	-	-	-	-	24	-	28
Piper nodulosum Link	-	-	4	-	-	-	-	-	-	-	-	-	-	-	-	4
Piper obliquum Ruiz & Pav	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Piper oblongifolium (Klotzsch) C. DC.	-	-	-	-	-	-	-	-	-	-	-	3	-	-	-	3
Piper obtusifolium L.	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Piper obtusilimbum C. DC.	-	-	-	-	-	-	-	12	-	-	-	15	-	-	-	27
Piper aff. Pedunculatum C. DC.	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Piper sp.	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-	2

Species	Site															
	APPA	**APPL**	**BI**	**ECPA**	**ECPL**	**EPPA**	**EPPL**	**J**	**LC**	**LEPA**	**LEPL**	**NA**	**PVPA**	**PVPL**	**SLP**	**Total**
Polygalaceae																
Monnina hirta (Bonpl.) B. Eriksen	1	7	-	4	11	-	-	-	-	21	7	-	5	-	-	56
Monnina pseudopilosa Ferreyra	1	4	-	-	21	-	12	-	-	-	-	-	-	-	-	38
Monnina sodiroana Chodat	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	2
Primulaceae																
Ardisia websteri Pipoly	-	-	2	-	-	-	-	4	5	-	-	1	-	-	-	12
*Cybianthus kayap*ii (Lundell) Pipoly	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
*Cybianthus spruce*i (Hook. f.) G. Agostini	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Geissanthus andinus Mez	-	-	5	-	-	-	-	-	-	-	-	-	-	-	-	5
*Geissanthus occidentali*s Cuatrec.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Geissanthus pichinchae Mez	-	-	-	-	-	2	-	-	-	-	-	-	-	-	-	2
Geissanthus quindiensis Mez	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	2
Geissanthus sp.1	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Geissanthus sp.2	-	-	-	-	-	-	-	5	-	-	-	-	-	-	-	5
Geissanthus sp.3	-	3	-	-	-	-	-	-	-	-	-	-	-	-	-	3
Myrsine andina (Mez) Pipoly	-	-	-	-	-	-	-	5	-	-	-	2	-	-	-	7
Myrsine coriacea (Sw.) R. Br.	-	2	-	1	21	-	26	-	-	-	22	-	3	4	-	79
Myrsine sodiroana (Mez) Pipoly	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Stylogyne ambigua (Mart.) Mez	-	-	-	-	-	-	-	12	2	-	-	-	-	-	-	14
Stylogyne venezuelana Mez	-	-	-	-	-	-	-	-	12	-	-	1	-	-	-	13
Proteaceae																
Roupala obovata Kunth	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	1
Rosaceae																
Prunus debilis Koehne	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8	8
Prunus huantensis Pilg.	-	-	-	-	-	-	-	1	3	-	-	-	-	-	-	4
Prunus sp. 1	-	-	-	-	-	-	1	-	-	2	-	-	3	1	-	7
Rubiaceae																
Agouticarpa grandistipula C. Persson	-	-	-	-	-	-	-	4	-	-	-	2	-	-	-	6
Agouticarpa aff. Velutina C. Persson	-	-	-	-	-	-	-	4	1	-	-	-	-	-	-	5
Bertiera guianensis Aubl.	-	-	-	-	-	-	-	-	14	-	-	-	-	-	-	14
Cinchona pubescens Vahl	-	-	3	-	-	-	-	-	-	-	-	4	1	4	-	12
Elaeagia karstenii Standl.	-	-	-	-	-	-	-	4	-	-	-	3	-	-	-	7
Elaeagia mariae Wedd.	-	-	-	-	-	-	-	-	-	-	-	3	-	-	-	3
Faramea calyptrata C.M. Taylor	-	-	-	-	-	-	-	27	1	-	-	-	-	-	24	52
*Faramea langlasse*i Standl.	-	-	-	-	-	-	-	29	2	-	-	-	-	-	-	32
Faramea multiflora Standl.	-	-	-	-	-	-	2	-	-	-	-	-	-	6	-	8
Faramea oblongifolia Standl.	-	-	21	-	-	-	-	-	-	-	1	-	-	24	6	52
Faramea sp.	-	-	-	-	-	-	-	-	-	-	2	-	-	95	-	97
Hippotis albiflora H. Karst.	-	1	-	-	1	-	-	-	-	1	-	-	-	1	-	4
Hoffmannia latifolia Bartl. Ex DC	-	-	-	-	-	-	-	-	-	-	6	-	1	25	-	31
Notopleura sp.	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Palicourea amethystina (Ruiz & Pav.) DC.	-	-	3	-	-	-	-	1	3	-	-	-	-	1	-	8
Palicourea angustifolia Kunth	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	2
Palicourea apicata Kunth	-	-	1	-	-	-	-	-	-	-	-	-	1	-	-	2
Palicourea chignul C.M. Taylor	-	-	34	-	-	-	-	-	-	-	-	-	-	-	-	34
Palicourea demissa Standl.	-	-	59	-	-	-	-	106	89	5	28	129	9	35	47	507

Species	Site															
	APPA	**APPL**	**BI**	**ECPA**	**ECPL**	**EPPA**	**EPPL**	**J**	**LC**	**LEPA**	**LEPL**	**NA**	**PVPA**	**PVPL**	**SLP**	**Total**
Palicourea lasiorrhachis Oerst.	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	2
Palicourea calothyrsa K. Schum. & K. Krause	-	41	-	-	-	-	-	-	-	-	-	-	-	-	-	41
Palicourea sodiroi Standl.	-	-	-	-	-	-	-	6	-	-	-	-	-	-	4	10
Palicourea thyrsiflora (Ruiz & Pav.) DC.	-	-	-	-	-	-	-	1	18	-	-	65	-	-	-	84
Palicourea sp.	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Posoqueria coriacea M. Martens & Galeotti	-	-	-	-	-	-	-	5	3	-	-	-	-	-	7	15
Posoqueria aff. Panamensis (Walp. & Duchass.) Walp.	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Psychotria acuminata Benth.	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	2
Psychotria allenii Standl.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	2
P*sychotria amethystina* Ruiz & Pav.	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-	1
Psychotria capitata Ruiz & Pav.	-	-	6	-	-	-	-	4	-	-	-	1	-	-	-	11
Psychotria aff. Hazenii Standl.	-	-	1	-	-	-	-	2	83	-	-	12	-	-	-	98
Psychotria marginata	-	-	-	-	-	-	-	-	-	-	-	-	-	88	-	88
Psychotria pilosa Ruiz & Pav.	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Psychotria setifera Benth.	-	-	39	-	-	-	-	-	-	-	-	-	-	-	-	39
Psychotria sp.1	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	2
Psychotria sp. 2	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	2
Rudgea sp.	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Simira sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Rutaceae																
Citrus limon (L.) Burm. f.	-	-	-	-	-	-	-	-	-	-	-	-	-	2	-	2
Zanthoxylum mauriifolium L.	-	7	-	-	-	-	-	1	-	-	-	-	-	2	-	10
Sabiaceae																
Meliosma novogranatensis Cuatrec. & Idrobo	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Meliosma occidentalis Cuatrec.	-	-	-	-	-	-	-	3	-	-	-	31	-	-	-	34
Salicaceae																
Banara guianensis Aubl.	-	-	2	-	-	-	3	-	-	-	-	-	-	-	-	5
Casearia cajambrensis Cuatrec.	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Casearia fasciculata (Ruiz & Pav.) Sleumer	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Casearia pitumba Sleumer	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Casearia silvestris Sw.	-	-	-	-	-	-	-	1	5	-	-	15	-	-	-	21
Casearia sp.	-	-	-	-	4	-	11	-	-	1	6	-	-	3	-	25
Sapindaceae																
Allophylus excelsus (Triana & Planch.) Radlk.	-	-	-	-	-	-	-	-	3	-	-	-	-	12	-	15
Allophylus peruvianus Radlk.	-	-	-	-	-	-	-	-	3	-	-	-	-	-	-	3
Billia columbiana Planch. & Linden ex Triana & Planch.	-	-	-	-	-	-	7	-	-	-	-	-	-	-	-	7
Billia rosea (Planch. & Linden) C. Ulloa Ulloa & P. Jørg.	-	-	-	-	-	-	-	27	6	-	-	-	-	-	13	46
Paullinia aff. Capreolata (Aubl.) Radlk.	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Talisia cerasina (Benth.) Radlk.	-	-	-	-	-	-	-	4	3	-	-	-	-	-	-	7
Talisia equatoriensis Acev. -Rodr.	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	3
Sapotaceae																
Pouteria baehniana Monach.	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1

Species	Site															
	APPA	**APPL**	**BI**	**ECPA**	**ECPL**	**EPPA**	**EPPL**	**J**	**LC**	**LEPA**	**LEPL**	**NA**	**PVPA**	**PVPL**	**SLP**	**Total**
Scrophulariaceae																
Buddleja bullata Kunth	1	1	-	-	1	-	-	-	-	-	-	-	-	-	-	3
Siparunaceae																
Siparuna aspera (Ruiz & Pav.) A. DC.	-	-	-	-	-	-	-	-	-	-	-	12	-	-	-	12
Siparuna aspera (Ruiz & Pav.) A. DC.	-	-	-	-	-	-	-	7	1	-	-	-	-	-	-	8
Siparuna aff. croatii S.S. Renner & Hausner	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	2
Siparuna decipiens (Tul.) A. DC.	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Siparuna lepidota (Kunth) A. DC.	-	-	-	-	-	-	-	1	-	-	-	2	-	-	-	3
Siparuna olivaceo -velutina Sleumer	-	-	-	-	-	-	-	3	-	-	-	-	-	-	-	3
Solanaceae																
Cestrum megalophyllum Dunal	-	-	-	-	-	-	-	5	1	-	-	1	-	-	19	27
Cestrum peruvianum Willd. ex Roem. & Schult.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4	4
Cestrum sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Cuatresia harlingiana Hunz.	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	2
Cuatresia sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	2
Solanum abitaguense S Knapp	-	-	-	-	-	-	-	-	-	-	-	-	-	4	-	4
Solanum appressum K.E. Roe	-	-	-	-	-	-	7	-	-	61	76	-	1	4	-	150
Solanum cf. arboretum Dodson & A. H. Gentry	-	-	-	-	-	-	-	-	-	-	-	-	-	4	-	4
Solanum asperolanatum Ruiz & Pav.	-	-	-	-	-	1	-	-	-	-	-	-	-	-	-	1
Solanum aturense Dunal	-	-	-	-	1	-	-	-	-	1	4	-	24	6	-	36
Solanum barbulatum Zahlbr.	-	-	5	-	-	-	-	-	-	-	-	-	-	-	-	5
Solanum confertiseriatum Bitter	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Solanum cucullatum S. D.Knapp	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1
Solanum dolosum C.V. Morton ex S. D.Knapp	-	-	-	-	-	-	-	-	-	-	-	6	-	-	-	6
Solanum hypaleurotrichum Bitter	-	-	-	-	-	-	-	-	-	3	-	3	1	1	-	8
Solanum aff. Laevigatum Dunal	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Solanum coconilla Huber	-	-	-	-	-	-	-	-	-	-	-	-	-	8	-	8
Solanum lepidotum Dunal	-	-	-	-	-	-	-	-	-	-	-	6	-	-	-	6
Solanum nigrescens M. Martins & Galeotti	-	-	-	-	-	-	-	-	-	-	-	-	3	7	-	10
Solanum ovalifolium Dunal	-	-	-	-	6	1	2	-	-	-	1	-	-	-	-	10
Solanum stenophyllum Dunal	-	-	5	-	-	-	-	-	-	-	-	-	-	-	-	5
Solanum sp.1	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	1
Solanum sp.2	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Solanum sp.3	-	24	-	-	-	1	1	-	-	-	1	-	-	19	-	46
Solanum sp.4	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	1
Solanum sp.5	-	-	-	-	-	-	-	-	-	-	-	-	-	5	-	5
Solanum sp.6	-	-	-	-	-	-	-	-	-	-	1	-	-	-	-	1
Solanum sp.7	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-	1
Solanaceae sp.1	-	-	-	-	-	-	-	-	-	-	-	-	-	3	-	3
Solanaceae sp.2	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Staphyleaceae																
Turpinia occidentalis (Sw.) G. Don	-	-	-	-	-	-	-	-	1	-	-	5	2	1	2	11
Styracaceae																
Styrax argenteus C. Presl	-	-	-	-	-	-	-	-	3	-	-	-	-	-	-	3
Styrax heterotrichus Perkins	-	-	-	-	-	-	-	-	1	-	-	3	-	-	-	4

Species	Site															
	APPA	**APPL**	**BI**	**ECPA**	**ECPL**	**EPPA**	**EPPL**	**J**	**LC**	**LEPA**	**LEPL**	**NA**	**PVPA**	**PVPL**	**SLP**	**Total**
Tapisciaceae																
Huertea glandulosa Ruiz & Pav.	-	-	-	-	-	-	-	-	-	-	-	2	-	-	-	2
Theaceae																
Freziera verrucosa (Hieron.) Kobuski	1	2	-	1	-	-	3	-	-	-	1	-	-	1	-	9
Gordonia fruticosa (Schrad.) H. Keng	-	-	-	-	-	-	-	-	-	-	-	27	-	-	-	27
Thelypteridaceae																
Thelypteris sp.1	-	-	-	-	-	-	-	5	2	-	-	-	-	-	-	7
Thelypteris sp. 2	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	2
Thymelaeaceae																
Daphnopsis grandis Nevling & Barringer	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-	1
Daphnopsis zamorensis Domke	-	-	-	-	-	-	-	7	-	-	-	-	-	-	-	7
Urticaceae																
Cecropia angustifolia Trécul	-	-	-	-	-	-	-	-	1	-	-	2	-	-	-	3
Cecropia maxima Snethl.	-	-	1	-	-	-	-	-	-	-	5	1	-	-	-	7
Cecropia reticulata Cuatrec.	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	2
Phenax uliginosus Wedd.	-	-	-	1	1	-	-	-	-	-	-	-	-	-	-	2
Pilea sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-	1
Pourouma sp.	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	2
Urera baccifera (L.) Gaudich. ex Wedd.	-	-	47	-	-	-	-	-	-	-	-	-	-	-	-	47
Urera caracasana (Jacq.) Gaudich. ex Griseb.	-	2	-	-	-	-	-	-	-	1	42	2	2	49	-	98
Verbenaceae																
Cytharexylum sp.	-	2	-	-	1	1	6	-	-	-	6	-	43	-	-	59
Lantana camara L.	-	9	-	1	1	-	3	-	-	-	-	-	-	-	-	14
Violaceae																
Gloeospermum grandifolium Hekking	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	2
Woodsiaceae																
Diplazium roemerianum (Kunze) C. Presl	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	1
Zygophyllaceae																
Guaiacum sp.	-	2	-	-	-	-	-	-	-	-	-	-	-	-	-	2