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FOREST LANDSCAPE RESTORATION: RECONCILING BIODIVERSITY CONSERVATION WITH LOCAL LIVELIHOODS IN ECUADOR

DOCTORAL DISSERTATION PRESENTED BY

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'Ik heb het nog nooit gedaan, dus ik denk dat ik het wel kan'
- Pippi Långstrump

In June 2012, I stepped out of the train and onto the platform of Louvain-la-Neuve. Little did I know, I would spend the next years of my life repeating that act five times every week. Finalizing this degree seems like both a thrilling adventure and an endless struggle. The reason this dissertation lies before you now is first and foremost thanks to the brilliant supervision of Eric. Keeping track of the bigger picture, providing motivation and encouragement when needed, and allowing a lot of freedom to pursue my own research objectives. Visiting you at Stanford has been one of most gratifying experiences during the course of my PhD. Thank you Eric, for your availability, confidence, and pragmatism.

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CONTENTS

SUMMARY

Tropical forest conversion and agricultural intensification are important drivers of loss of biodiversity and ecosystem services on which local communities depend. Resilient agricultural landscapes are crucial to safeguard food security and adapt to environmental and climate changes. An increasing number of policies and programs target forest landscape restoration but lack the scientific basis to ensure sustainable outcomes.

This dissertation explores the potential of forest landscape restoration and its financial instruments to reconcile biodiversity conservation, ecosystem services supply, and human wellbeing in tropical agricultural landscapes. Focusing on natural regeneration, tree plantings, and agroforestry in Ecuador, three research questions are answered: (i) What is the potential of different restoration types? (ii) How do landscape-level characteristics affect restoration objectives? (iii) How can market-based mechanisms support restoration?

Combing remote sensing, ecological fieldwork, forest modeling, and socioeconomic analyses, this dissertation shows that the value of restoration landscapes is context dependent. Spatial targeting of forest restoration can stimulate biodiversity and carbon sequestration during regeneration by accounting for landscape characteristics, including the presence of natural remnant forests, different restoration types, and land-cover heterogeneity. Payments for ecosystem services schemes and alternative markets for high-quality commodities can contribute to the maintenance and restoration of tree cover in tropical agricultural landscapes. This dissertation contributes to a deeper understanding of the challenges facing forest landscape restoration. In conclusion, conservation, sustainable management, and landscape restoration are complementary and need to be aligned to guarantee long-lasting environmental, social, and economic benefits. This requires active coalitions of public and private actors to develop effective supply chain and land use interventions.

Chapter 1

Introduction

Tropical forests hold tremendous value. They regulate our global climate via water transpiration, cloud formation, and atmospheric circulation (Devaraju et al., 2015; Lawrence and Vandecar, 2015; Spracklen et al., 2012), shelter over half of Earth's 5 to 20 million species (Scheffers et al., 2012), and provide about 1.5 billion people with food, timber, medicines, and other ecosystem services (Vira et al., 2015). Old-growth forests recycle essential plant nutrients and organic matter over long periods, resulting in enriched and fertile soils. Traditional shifting agriculture has employed this immense regeneration capacity for centuries. Farmers cultivated burned forest patches, implementing long rotation cycles during which fallows were regularly abandoned to allow natural restoration to occur. Over the last few decades, pressure from growing human populations and globalization has gradually reduced fallow periods, with severely degraded lands and increased deforestation rates of primary tropical forests as a result. Today, over half of the world's tropical forests have been cut down (Lewis et al., 2015). Deforestation and inadequate land use practices have transformed biodiverse and productive tropical forest ecosystems into areas with reduced supply of *ecosystem services* (i.e., the benefits people derive from nature (Isbell et al., 2017)) and unsuitable habitat for most native species (Foley et al., 2005).

It is unsurprising that United Nations Rio+20 conference participants recognized restoration of degraded areas as one of the world's top priorities to avoid further loss of biodiversity and ecosystem services (UN, 2012). Ecosystem restoration receives growing recognition, through the Aichi Targets, the Bonn Challenge to restore 150 million hectares of degraded and deforested land by 2020, and the New York Declaration on Forests launched at the UN Climate Summit 2014. Merely protecting remnant forest through reserves and conservation areas is insufficient to conserve biodiversity and sustain levels of ecosystem services required by growing human populations (Harvey et al., 2008; Chazdon et al., 2009a). Devoting large areas in human-modified tropical landscapes solely to forest conservation by excluding human activities seems improbable, let alone unethical, from a socioeconomic standpoint. Many people depend on forests for their livelihoods, cultural traditions, and well-being. Successful forest restoration activities must therefore include environmental, economic, and social benefits.

Forest landscape restoration (FLR) provides a promising new approach to reconcile biodiversity conservation, ecosystem services provision, and human well-being in human-dominated landscapes (Mansourian et al., 2005). FLR can be defined as a planned process that aims to regain ecological integrity and enhance human well-being in deforested or degraded landscapes' (Dudley et al., 2005). Through natural regeneration, targeted tree plantings, and agroforestry, FLR connects the well-being of local caretakers directly to the restoration of functioning forest landscapes (Chazdon, 2014). These land-use systems play an important role in mitigating global carbon emissions, safeguarding water supplies, preventing soil erosion and flooding, and provide resources to many species (see section 1.2). Secondary and planted forests, as well as agroforestry systems, dominate land-use in the tropics, providing an enormous scope for forest restoration (Gilroy et al., 2014; Chazdon, 2014) and are therefore the focus of this dissertation.

So far, we know little about the potential of different restoration methods to achieve FLR goals to consolidate biodiversity conservation, ecosystem services supply, and human well-being in human-dominated landscapes. In addition, we lack knowledge of the impacts of different economic strategies for FLR, which include payments for ecosystem services (e.g., carbon sequestration initiatives) and alternative market strategies for tropical commodities. The main aim of this dissertation is to better understand the potential of restoration methods and its financial instruments to restore biodiversity and enhance livelihoods in human-dominated tropical landscapes. We focus mainly on floristic biodiversity, as vegetation composition and structure largely determine faunal diversity in human-dominated landscapes (Bremer and Farley, 2010; Mendenhall et al., 2014, 2016). This dissertation will answer three main research questions: (i) What is the potential of different restoration types to achieve FLR objectives? (ii) How does the landscape-level pattern of restoration affect FLR objectives? (iii) How can market-based mechanisms support FLR objectives? FLR objectives of interest in this dissertation are biodiversity conservation, ecosystem services supply, and human well-being. Results presented in this dissertation contribute to a deeper understanding of challenges facing FLR and provide a basis for designing restoration strategies with long-lasting social and environmental benefits in the tropics.

This introductory chapter first reviews the forest landscape restoration literature. It touches on the scientific evidence of tropical forest transitions, emphasizes the need for a landscape-scale approach, and introduces the conceptual model applied in this dissertation to frame on human impacts on tropical landscapes. I then discuss evidence of environmental and socioeconomic outcomes for three reforestation methods that differ in their origin, dynamic properties, and landscape context: natural regeneration, plantations, and agroforestry. I continue with an overview of the main framework for financing and promoting restoration activities, including payments for ecosystem services and alternative markets for tropical commodities. Next, I clarify the motivation, novelty, and objectives of this dissertation, and outline the remaining structure of this dissertation. This section includes a brief description of the applied modeling approach and provides background on forest restoration activities in Ecuador, where two case study landscapes are selected.

1.1 Forest landscape restoration

We urgently need large-scale and long-term restoration efforts to reverse global trends of deforestation and forest degradation in the tropics. *Forest landscape restoration* (FLR) approaches restoration as a means to regain ecological integrity, while enhancing human well-being in human-modified tropical landscapes. To achieve sustainable food production, ecosystem service provisioning, and biodiversity conservation, forest and non-forest ecosystems, extractive land uses, and different restoration methods are combined (Chazdon and Uriarte, 2016; Strassburg et al., 2016). Instead of returning to the 'original' state of the forest landscape, FLR promotes a larger set of options for the delivery of forest-related ecosystem services across entire landscapes (Table 1.1). This section details the recent evidence of forest transitions in the tropics as well as the importance of a landscape-scale approach to restoration. It then introduces a conceptual model to frame the research on the human impacts on landscapes in the context of forest landscape restoration.

Table 1.1: Key elements of forest landscape restoration. Source: Dudley et al. (2005)

It is implemented at the landscape scale, rather than a single site. Restoration planning is done in the context of many elements: social, economic, and biological, meaning that trees are planted strategically to achieve a set of functions (e.g., provision of building materials, habitat for selected species, water retention)
It takes into account environmental and socioeconomic dimensions.
It addresses the root causes of forest loss and degradation.
It opts for multiple solutions depending on local conditions (e.g., agroforestry, enrichment planting, natural regeneration, policy analysis, research)
It includes different actors in the planning and decision process to reach consensus or compromise.
It identifies trade-offs between different solutions.
It emphasizes forest quantity as well as quality.
It aims to restore different forest goods, services and processes, rather than forest cover alone.

1.1.1 Forest transition in the tropics

Although tropical deforestation continues at an unprecedented rate, scholars have observed a shift from net deforestation to net reforestation for several countries in Latin America and South-East Asia (Meyfroidt and Lambin, 2010; Sloan and Sayer, 2015). Defined as *forest transitions* (Mather, 1992), these shifts were first reported in temperate regions (Mather, 2004; Mather and Fairbairn, 2000; Mather et al., 1999; Ramankutty et al., 2010) and resulted from complex interaction between economic, political, institutional, and cultural factors in the agricultural, forestry, and energy sector (Meyfroidt and Lambin, 2011).

Based on a cross-national study for the 1990s, Rudel et al. (2005) recognized two potentially overlapping pathways that could explain forest transitions: the economic development pathway and the forest scarcity pathway. The first pathway focuses on forest regrowth and tree planting on abandoned agricultural lands, following the spatial reorganization of agricultural activities in marginal regions. Main drivers of this pathway are the rural-urban migration of local workforce, the intensification of agricultural activities in the most suitable regions, and the decline of agricultural activities in the least suitable areas. The second pathway identifies the scarcity and rising prices of forest products following deforestation as the main driver, which encourages landowners to plant trees and intensify forest management (Rudel et al., 2005). Revising these two pathways, Lambin and Meyfroidt (2010) pointed out three additional pathways that better explain more recent tropical forest transitions: (i) the state forest policy pathway that identifies national policies as the main driver for forest transitions; (ii) the globalization pathway that places the economic development pathway in a global context of a more integrated world in which commodities, people, and ideas flow unceasingly through national economies; and (iii) the smallholder, tree-based land use intensification pathway that ascribes reforestation to the development of fruit orchards, agroforestry systems, home gardens, and hedgerow.

The drivers and patterns of forest transition vary strongly over space and time. Forest cover increases are mainly attributed to the forest scarcity pathway in densely populated and poor countries of Asia, whereas the economic development pathway plays a larger role in less densely populated and richer countries in America (Rudel, 2010; Rudel et al., 2005; Meyfroidt and Lambin, 2010). Nevertheless, due to the displacement of deforestation and degradation elsewhere following national forest transitions, a global forest transition seems unlikely (Meyfroidt and Lambin, 2010, 2011). This emphasizes the need for sound restoration strategies that aim to manage remaining forests sustainably, support natural regeneration, increase the efficiency of land-use practices, and promote nature-friendly farming over larger spatial scales.

1.1.2 Why a landscape-scale approach?

Conventional ecosystem-based restoration initiatives focus on small spatial scales. In contrast, global restoration initiatives play out over large spatial extents where multiple ecosystem types, land-use types, governance, and ownerships exist alongside a mosaic of production and conservation areas distributed over different forest types (Fig. 1.1). Landscape approaches are necessary to identify potential conflicts and synergies between environmental protection and human well-being. In this sense, restoration approaches need to take into account landscape patterns and context to optimize ecological outcomes. This dissertation defines *the landscape* from an ecological perspective as a mosaic of interacting ecosystems (at different scales) that are heterogeneous in at least one factor of interest (Turner et al., 2001) (e.g., vegetation composition). From a socioeconomic perspective, *the landscape* is an area delineated for a specific set of objectives in which people, organizations, and other entities interact (Sayer et al., 2013).

Processes of reforestation affect the spatial configuration of landscapes and therefore habitat distribution. Many scholars have suggested that landscape structure determines to a large extent the effectiveness of local biodiversity conservation management in human-modified landscapes (Tscharntke et al., 2012b; Turner et al., 2013). Nevertheless, the management effectiveness of restoration in relation to landscape characteristics is still largely debated (Tscharntke et al., 2005; Leite et al., 2013; Bullock et al., 2011; Melo et al., 2013a). Contrasting restoration results likely emerge from differences in biogeographic conditions (Quesada et al., 2009), landscape spatial configuration (Smith et al.,

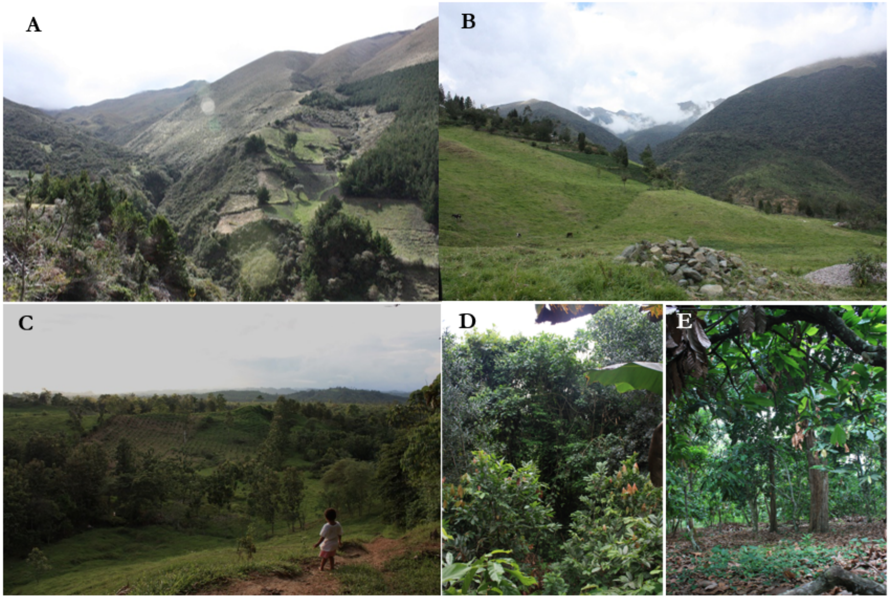

Figure 1.1: (a, b) Landscape mosaics in Ecuadorian Andes showing extractive land uses mixed with patches of native vegetation and remnant forest fragments. (c) Intensified agroforestry landscape interspersed with natural remnant forest, and pastures. (d, e) Traditional cocoa agroforestry systems in northern Ecuador shaded by a mix of fruit and timber species. Photos: Romaike Middendorp

2011), climatic aspects (Hirota et al., 2011), and intensity of human disturbance (Proulx and Fahrig, 2010).

In 84 per cent of recent empirical restoration studies reviewed (N = 54), the landscape context played a key role in restoration processes (Leite et al., 2013). Especially, the close proximity of neighboring forest patches and the amount of habitat cover positively affected restoration effectiveness. Lindenmayer (2010) have suggested that for successful biodiversity conservation at large scales, the maintenance of forest connectivity, landscape heterogeneity, and the stand structural complexity is imperative. Landscape characteristics, such as landscape heterogeneity and structural complexity, have been found to positively influence biodiversity conservation (Tscharntke et al., 2005; Fahrig et al., 2011), species recovery (Metzger et al., 2009; Osland et al., 2011), and ecological resilience (i.e., the capacity of landscapes to recover local species losses through immigration of species at larger spatial scales) (Pardini et al., 2010; Tambosi and Metzger, 2013; Tambosi et al., 2014). The size of restoration patches matters too. Forest edges increase with decreasing patch size and alter climatic conditions that favor ecologically plastic and generalist plant species over old-growth flora (Laurance et al., 2006). Small forest remnants transition towards early-successional states, limiting the long-term persistence of forest-obligate and forest-dependent species; a process called 'retrogressive succession' (Tabarelli et al., 2012a,b).

The larger landscape-scale context is crucial for the design and management of tropical agricultural systems (Harvey et al., 2014). Current certification schemes focus at the farm or plantation levels, whereas ecosystem benefits from well-designed agroforests are delivered at the landscape-level, emphasizing a scale mismatch (Tscharntke et al., 2015). Scholars have suggested two approaches to address this incongruence: certification mechanisms could be linked with broader landscape-scale approaches (Milder et al., 2014) and current certification models could consider the landscape as a certified unit through landscape labeling (Ghazoul et al., 2009). An important question remains whether and to what extent new forms of landscape certification will stimulate the adoption of more sustainable land use practices.

1.1.3 Framework for human impact on landscapes and restoration

Various disciplines uphold a range of concepts and terms that are relevant for landscape planning and environmental management in restoration policies and programs. The Drivers-Pressures-State-Impacts-Response (DPSIR) conceptual model has been selected to frame the research on the human impacts on landscapes in the context of forest landscape restoration. This framework has been developed originally to structure the cause-effect relationships in natural recourse management problems from a social, economic, and environmental perspective (Ness et al., 2010). It has been widely used for ecosystem management, assessment, and communication in both policy reports and scientific literature (Levin and Wells, 2008; Bowen and Riley, 2003). For the purpose of this dissertation, we have implemented the more general DPSIR framework in the context of restoration, following in part the propositions made by Vranken (2015) (Fig. 1.2).

Humans are the underlying *drivers* or causes of anthropogenic impact on landscapes and ecological processes. Several factors determine the strength of drivers and their subsequent impacts, including population density, revenues, and lifestyle (Lambin et al., 2003). The activities undertaken in a landscape that support human needs generate mostly negative environmental *pressures*, including different types of disturbances like infrastructure development, fire management, and agriculture intensification (Ness et al., 2010). In turn, these pressures affect the *state* of the landscape. The state represents the level of anthropogenic disturbance and determined using land use and cover data at

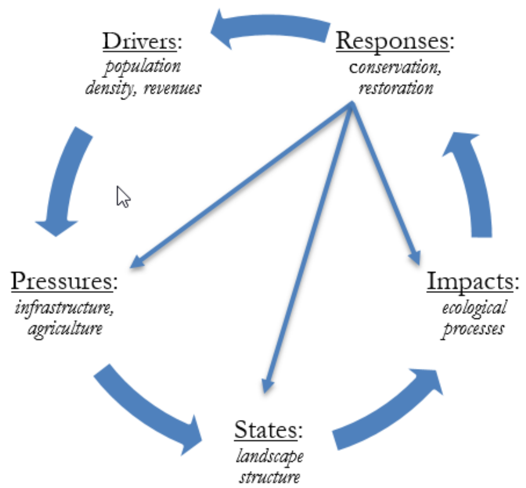

Figure 1.2: The DPSIR framework implemented for the human impact on landscape in the context of forest landscape restoration. Source: Ness et al. (2010)

the landscape (Foley et al., 2005; Bogaert et al., 2011). Concepts from landscape ecology, such as landscape composition and heterogeneity, are important variables used to determine the state of ecological processes. Changes in the landscape structure has *impacts* on various ecological processes and disturb local ecological communities. It should be noted that diversity may actually be increased as a result of moderate anthropogenic disturbances (Fahrig, 2016; Chakraborty and Li, 2011).

The *response* to environmental disturbances depend in part on the severity of changes and include several strategies. Conservation management is implemented to avoid further degradation within a delineated area that is protected from anthropogenic disturbance. Other forms of environmental management are aimed at rectifying or compensating degradation. Such activities include active or passive restoration and agroforestry, and are the focus of this dissertation. The type of response chosen depends largely on the societal willingness to invest in conservation and restoration activities (Ghazoul and Chazdon, 2016). The societal willingness can be stimulated gradually by increasing environmental concern of diminishing returns. Policy incentives may aim to increase societal willingness, but the net benefits of landscape restoration efforts depends strongly on the existing land cover and use patterns, as has been shown in four different regions in Latin America (Vergara et al., 2016).

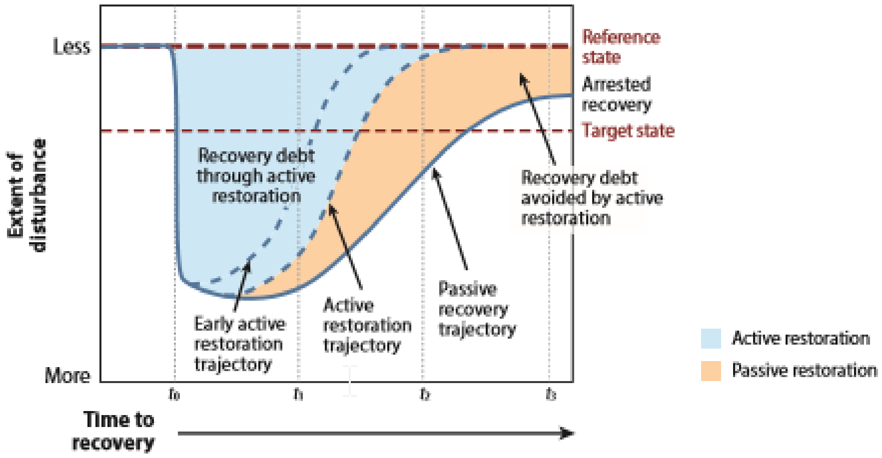

Figure 1.3: Degradation and subsequent recovery represented as recovery debt defined as the accumulative reduction of ecosystem functions during degradation and following recovery in relation to a target or reference state (Moreno-Mateos et al., 2017). Active restoration can accelerate recovery to the reference state and thus reduce the recovery debt compared to passive recovery. Source: Ghazoul and Chazdon (2016)

Most environmental and social research views degradation as a state with reduced ecosystem functions relative to a preferred reference state. One way of connecting degradation (a state) and recovery (a process) is by means of *recovery debt*: the cumulative loss of selected attributes relative to a *reference state* experienced during the recovery phase (Moreno-Mateos et al., 2017). This framework implies the process of returning to the preferred conditions by natural regeneration or active restoration (Ghazoul and Chazdon, 2016) (Fig. 1.3). The reference state refers to a prior state against which a recovering ecosystem is compared and serves as a departure point to measure impact of restoration activities. Defining the reference state depends on both the discipline, restoration objectives, and region under consideration. For example, botanist and ecologist refer to *naturalness* as the state of a system without any human activity or influence. The term ‘original naturalness’ is often used in restoration ecology to describe the ecosystem state before anthropogenic interferences (Bakker and Berendse, 1999; Winter et al., 2010).

Achieving original naturalness in tropical agricultural landscapes is practically impossible and precise forest restoration targets and reference states are rarely realistic (Hiers et al., 2016). For example, secondary forests are novel ecosystems with new species’ combinations and interactions never seen before, but nevertheless provide many ecosystem services. More realistic outcomes of restoration include alternative trajectories driven by process-based dynamics in a response to environmental changes (Hughes et al., 2012). The emphasis of scientific research has therefore shifted toward resilience-based targets of restoration, with *resilience* defined as the capacity to resist disturbance or recover from disturbance while maintaining ecosystem function over time (Chapin et al., 2010).

In view of the increasingly complex distinction between natural and anthropogenic disturbances, and between natural and novel ecosystems, impacts of recovery are more efficiently referred to as responses to change (i.e., resilience) rather than achieving a certain subjective reference state, abandoning the notion of returning to prior states (Ghazoul and Chazdon, 2016). Throughout this dissertation, the reference state is referred to as a *baseline* that represents the current landscape mix and is used as a point of departure to measure the impact of alternative restoration trajectories.

1.2 Environmental and socioeconomic aspects of restoration types

Even though many forest-obligate species depend on old-growth forests, forest restoration provides opportunities to restore biodiversity and ecosystem functions (Chazdon et al., 2009a,b). This dissertation identifies two important environmental aspects related to forest restoration, namely biodiversity conservation and ecosystem services. Ecological theory considers biodiversity and ecosystem services to relate positively, but not linearly, with the relationship varying with spatial and biological scales (Tilman et al., 2014; Perrings et al., 2010). Hence, restoration activities may provide a win-win situation for socioeconomic development objectives, as they likely increase biodiversity and ecosystem services simultaneously (Rey Benayas et al., 2009). Ecosystem services that are relevant for forest landscape restoration are grouped into provisioning, regulating and supporting, and cultural services (Table 1.2). Tradeoffs exists among biodiversity and multiple ecosystem services (Maskell et al., 2013) and still need to be assessed in depth for ecological restoration (Meli et al., 2017). Restoration initiatives should consider how biodiversity and ecosystem services respond to different management action and identify potential overlap and conflict.

Improving ecosystem service provision can provide considerable value for local people when supported financially. Many scholars aim to provide estimates of the monetary value of key ecosystem services. Costs and benefits of restoration should be distributed equitably in order to succeed (Bullock et al., 2011; Meli et al., 2017). Unfortunately, this is not always the case as

Table 1.2: Type of ecosystem services resulting from forest landscape restoration. Source: Chazdon (2014)

Type of ecosystem service	Description
Provisioning services [a]	
Water	Provision of safe and clean drinking water, stable water supplies, and irrigation services
Food and medicine	Gathering of fish, fruits, and medicinal plants
Raw materials	Source of materials for construction, fuel, and fiber
Regulating and supporting services [b]	
Carbon storage	Accumulation of carbon in above- and belowground biomass
Mitigation of extreme weather events	Protection from heavy rainfall or wind
Erosion prevention	Prevention of damage from landslides, erosion, and siltation
Pest-regulation	Regulation of populations of natural enemies of insect pests
Pollination and seed dispersal	Maintenance of populations of insects, birds, and mammals that pollinate flowers and disperse seeds
Nutrient cycling	Maintain and recycle nutrients in soils for plant uptake
Cultural services [c]	
Aesthetic beauty and spiritual experience	Provision of environments to appreciate nature, enjoy surroundings, and inspire spiritual or religious experiences
Cultural heritage	Share local ecological knowledge and cultural heritage
Educational value	Opportunities to study botany, ecology, and forestry during outings
Recreation	Enjoy sports, recreational fishing, hiking, and social event in natural settings

[a] (Martin-Ortega et al., 2013, Brauman et al., 2007, Adams et al., 2016, Ferraz et al., 2014)
[b] (Strassburg et al., 2016, Bullock et al., 2011, Mukul et al., 2016)
[c] (Brancalion et al., 2014, del Castillo et al., 2011, Aronson et al., 2010, Clewell and Aronson, 2013)

the poorest farmers and women have been excluded from project design and implementation (Corbera et al., 2007). Long-term benefits from restoration activities will only be achieved if socioeconomic benefits are present (Aronson et al., 2010) and stakeholders are involved (Lazos-Chavero et al., 2016). Nevertheless, few studies have explored the economic costs and benefits of FLR. The following sections reflects on the environmental and socioeconomic aspects and evidence of the different restoration types as highlighted in recent scientific literature.

1.2.1 Natural regeneration

Natural regeneration of tropical forests can occur spontaneously under suitable conditions. The structure, function, and composition of the pre-disturbance ecosystem can recover during natural regeneration, as tree species recolonize abandoned agricultural lands and pastures, and communities reassemble (Guariguata and Ostertag, 2001; Chazdon, 2008, 2014). *Natural regeneration* refers to the unassisted regrowth of forest after disturbance of the original forest vegetation (Chokkalingam and de Jong, 2001). According to the FAO's Forest Resources Assessments, *naturally regenerating forests* can follow selective logging of trees or forest regrowth on abandoned agricultural lands (FAO, 2015). Recent natural regeneration have been reported in Puerto Rico (Lugo and Helmer, 2004), Argentina (Turner II, 2010), Costa Rica (Arroyo-Mora et al., 2005), and Brazil (Baptista and Rudel, 2006), and many other examples exist (Sloan and Sayer, 2015). Areas under natural regeneration can be extensive: over 35 million hectare of woody vegetation has recovered in Latin America and the Caribbean between 2001 and 2010 (Aide et al., 2013). Restoration approaches that include natural regeneration can offer substantial environmental and socioeconomic benefits, but success depends on multiple factors, including local site characteristics, landscape dynamics, and socioeconomic conditions.

Forest recovery following natural regeneration can be especially rapid on sites where forest clearance occurred recently, propagule sources are still intact (e.g., soil seeds, seedlings, saplings, re-sprouting banks), and native forest are still present in the landscape (Holl, 2007). In the western Andes of Colombia, natural regenerating forest accumulated about half of the non-soil carbon stocks of primary forests within 15-30 years and recovered habitat for 33 out of 40 threatened bird species (Gilroy et al., 2014). Faunal species richness showed levels similar to mature forest within 20-40 years, but the recovery of faunal species composition took considerably longer (Dunn, 2004). Several studies have found an enhanced ecosystem services supply following natural regeneration, such as a more dependable stream flow during the dry season, lower stream erosion, and carbon sequestration (Gilroy et al., 2014; Molina et al., 2012; Gilman et al., 2016). Natural regeneration can also improve livelihoods of local communities. Reij and Garrity (2016) illustrated how assisted natural regeneration is improving livelihoods, lives, and landscapes in dryland ecosystems of sub-Saharan Africa.

Although numerous studies have emphasized the critical importance of natural regenerating forest in human-dominated landscapes (Aide et al., 2000; Chazdon et al., 2009b; DeClerck et al., 2010), forest gain is not the exact mirror image opposite of forest loss. Regenerating forests often contain only a subset of native plant and wildlife (Zimmerman et al., 2000; Gibson et al., 2011; Tabarelli et al., 2010). Old-growth forests serve as the main seed source in tropical landscapes and their absence has been found to be one of the most important barriers for forest regrowth (Norden et al., 2009; Guariguata and Ostertag, 2001; Mesquita et al., 2001; Holl et al., 2000). Dispersal limitation is exacerbated for large seeded plant species due to the lack of appropriate dispersal agents (Lewis et al., 2017). These plant species must be included in restoration projects; otherwise they will likely remain absent (Pärtel et al., 2011). Unsuitable microsites for seed establishment may limit natural regeneration (Holl and Aide, 2011), resulting in few pioneer species to become dominant (Mesquita et al., 2015).

Changes in vegetation structure and composition during natural regeneration determine the habitat quality of restoration areas for animal populations. Plant species provide resources, such as food, shelter, roosting, nesting, and mating sites. In turn, animals act as plant pollinators, dispersers, consumers, and protectors from herbivores. A number of studies indicate that animal diversity increases during the first stages of succession, but that faunal species composition recovers slowly, due to the slow influx of old-growth specialists (Dunn, 2004; Bowen et al., 2007; Chazdon et al., 2009b). Important plant indicators of animal biodiversity are the number of plant species, the number of plant stems, basal area, vegetation height, presence of invading plant species, and area covered by herbaceous species (Melo et al., 2013b).

The aboveground biomass in tropical forests contains about 50 per cent carbon, which releases into the atmosphere upon burning. Deforestation and forest degradation cause approximately 10 and 15 per cent of global carbon emissions (Van der Werf et al., 2009). However, secondary forests can play a major role in climate change mitigation by recapturing carbon emission (Chazdon et al., 2016). A growing number of studies have found large amounts of carbon in natural regenerating forests (Gilroy et al., 2014; Mukul et al., 2016; Strassburg et al., 2016). For example, secondary regrowth in the Peruvian Amazon provided an 18 per cent offset against total gross emissions between 1999 to 2009 (Asner et al., 2010). In addition, post-agricultural forest regeneration showed co-benefits for biodiversity and ecosystem services in western Andes of Colombia (Gilroy et al., 2014), the upland Philippines (Mukul et al., 2016), and the Atlantic forest in Brazil (Strassburg et al., 2016).

Landscape structure plays an important role in determining the effectiveness of local biodiversity conservation management in human-modified landscapes (Tscharntke et al., 2012b). Unfortunately, our understanding of the effect of landscape context on the recovery potential of naturally regenerating forest is still limited (Hall et al., 2012; Leite et al., 2013; Melo et al., 2013a). In a recent review, only two of the 68 reviewed studies explicitly considered the role of landscape context for recovery of faunal assemblages in natural regenerating forests (Bowen et al., 2007). Depending solely on natural regeneration for the restoration of forest ecosystems is likely the cheapest restoration methods, but can be a risky strategy. Thresholds for unassisted restoration may have been crossed and prevention for further disturbances is difficult (Lamb et al., 2005). Natural regeneration is often ignored as a viable restoration strategy. Whether natural regeneration initiates and persists in tropical landscapes depends largely on historical land-use decision by landowners or farmers (Chazdon and Uriarte, 2016). Passive restoration can provide the most cost-efficient restoration method and be potentially applied at large spatial scales (Chazdon and Uriarte, 2016). Especially in landscapes with intermediate disturbances, passive and large-scale restoration may be effective (Pardini et al., 2010). A modeling approach in dryland forest landscapes showed that passive restoration was cost effective, while the relatively high costs from active restoration outweighed the benefits involved (Birch et al., 2010). However, not all landscapes are suitable for passive restoration, emphasizing the need to set restoration priorities (Tambosi and Metzger, 2013) and explore other restoration options.

Monetizing the benefits of natural regeneration has received a lot of attention of recent years, especially in the context of payments for ecosystem services projects, including the Reducing Emissions from Deforestation and Forest Degradation (REDD+) program. For example, economic benefits from climate change mitigation and sediment retention could completely compensate the opportunity costs of agricultural production over 20 years in the Atlantic Forest of Brazil (Strassburg et al., 2016). Natural regeneration of abandoned pastures yielded up to US$7,440 per hectare over hectares for carbon, non-timber forest products, timber, and tourism in drylands of Latin America (Birch et al., 2010). Beneficiaries from an ecological restoration project in the Atlantic forest of Brazil, appreciated cultural ecosystem services, such as esthetic landscape improvement, tourism, recreation, as well as religious, spiritual, and educational services (Brancalion et al., 2014). The valuation of these types of services depend on spatial and temporal heterogeneity, making them difficult to extrapolate from case studies (Martín-López et al., 2009). Furthermore, the socioeconomic impact on local livelihoods of restoration programs depends on many other variables, such as availability of off-farm jobs, household characteristics, land productivity, land tenure, and markets for forest products and ecosystem services (Adams et al., 2016).

1.2.2 Plantations

In areas where natural regeneration potential is limited, assisted restoration through plantings of native tree species can catalyze recovery. In this dissertation *reforestation* refers to the re-establishment of forest through planting trees or deliberate seeding on land already classified as forest, while *afforestation* refers to the planting or seeding of trees on land previously unforested, as defined by the FAO's Forest Resource Assessment (FAO, 2015). With an estimated annual increase of approximately 2.95 per cent between 1990 and 2015, tree plantation contributed to reforestation worldwide (Sloan and Sayer, 2015). In 2015, planted forest covered about 59 million hectare in the tropics alone, encompassing semi-natural forests and intensively managed forest plantations of primarily introduced tree species (FAO, 2015). Recent increases in forest cover in Asia (e.g., China, India, Vietnam), but also Latin America (southern Brazil, the Rio de la Plata grasslands, Panama), were attributed to plantation establishment in response to the growing local demand for forest products, as well as for exports (Rudel, 2012; Song and Zhang, 2009; Baptista and Rudel, 2006; Baldi and Paruelo, 2008; Sloan, 2008). Studies have provided mixed evidence of the value of plantations for biodiversity and other ecosystems.

Active restoration in the form of tree planting can boost forest recovery in areas where natural regeneration potential is restricted (Omeja et al., 2011; Bertacchi et al., 2016; de Rezende et al., 2015). Planting bypasses some of the ecological restrictions that prevent successful seedling establishment during natural regeneration and can kick-start reestablishment of forest biodiversity and ecosystem services on degraded lands (Holl and Aide, 2011; Zahawi et al., 2013). A meta-analysis showed polycultural plantations of 39 years accumulated more aboveground biomass than naturally regenerating forests of 80 years (Bonner et al., 2013). Comparably, Gilman et al. (2016) found higher basal area of woody recruits in polyculture tree plantings compared to natural regenerating forests during a replicated reforestation experiment in Costa Rica. Forests that are planted under the Brazil's Mato Grosso State Plan for the Sustainable Development of Planted Forests (*Plano Estadual para o Desenvolvimento Sustentável de Florestas Plantadas de Mato Grosso do Sul*) are rehabilitating land degraded by cattle-ranching (Hirakuri and Saraiva, 2014). Furthermore, these planted forests produce about 10 times more wood and generate an average internal rate of return of approximately 15 per cent, compared to 5 per cent in sustainably managed natural forests. (Hirakuri and Saraiva, 2014). However, not all plantation initiatives enhance both biodiversity and ecosystem functions. For example, planting non-native species resulted in negative impacts on native vegetation cover and water availability in China (Cao et al., 2009) and the Ecuadorian Andes (Molina et al., 2012).

Although polycultural plantations and enrichment plantings can boost biodiversity and ecosystem services, these projects have high establishment and maintenance costs. Hence, the majority of forest plantations established in the tropics are large-scale industrial monocultures. Some rare studies have found that monoculture plantations can accumulate a significant diversity of plant species (Keenan et al., 1997) and improve soil protection on heavily degraded lands (Temperton et al., 2014; Xu, 2011), but only if natural forest remnants were near and plantation thinning and logging were restricted. Nevertheless, numerous studies have argued large-scale monocultures are unlikely to deliver benefits needed to restore ecosystem functions, provide sustainable livelihoods, and support biodiversity (Barlow et al., 2007; Bremer and Farley, 2010). Furthermore, planting commercial, non-native tree species in dry regions imposes risks for local and downstream hydrology, due to increased evapotranspiration (Farley et al., 2004; Molina et al., 2009). Industrial monoculture plantations offer limited potential to contribute to socioeconomic FLR goals, as a few large corporations redeem most of the profits (Barlow et al., 2007; Brancalion and Chazdon, 2017).

The benefits of plantations for FLR thus depend mainly on the type of plantations (mono- versus polycultures), the type of species planted (natives versus exotics), the land cover they replace, and the management practices. Plantations can be a valuable restoration method if established on degraded lands and indigenous tree species are planted (Bremer and Farley, 2010). Large-scale tree planting should be avoided in natural ecosystems, such as forests, grasslands, and shrublands.

1.2.3 Agroforestry

Large parts of human-modified tropical landscapes consist of orchards, woodlots, agroforestry systems, home gardens, and hedgerows managed by smallholders. We define these *agroforestry systems* as the intentional integration of shade trees with woody perennials, crops and livestock (Bhagwat et al., 2008; Nair, 1993). Many agroforestry systems are adopted to human needs, while retaining some degree of the structural characteristics and ecological processes of natural forests (Wiersum, 2004). Many scholar have argued that these forms of nature-friendly farming consolidate production and biodiversity conservation in human-dominated landscapes, as they implement environmentally unharmful practices, improve ecosystem service provision, and maintain high landscape heterogeneity (Schroth and Harvey, 2007; Perfecto et al., 2007; Bhagwat et al., 2008; Steffan-Dewenter et al., 2007; Tscharntke et al., 2011, 2015; Vaast and Somarriba, 2014; Fischer et al., 2008). Agroforestry systems provide a promising approach for FLR through the production of tree crops for local use or export cultivated under forest canopies. Nevertheless, landscape impacts of agroforests are difficult to assess, as these systems resemble natural forests and therefore are neglected in many land-use statistics (Rudel, 2009).

Cocoa and coffee are tropical perennial crops that are most commonly grown under shade (Rice and Greenberg, 2000), covering an estimated 20 million hectare of land across the tropics (FAO, 2014). Compared to other tropical agricultural systems, shaded cocoa agroforests support high levels of biodiversity. A meta-analysis showed that converting natural forest to cocoa and coffee plantations reduced total species richness across different taxonomic groups by 11 per cent and ecosystem services by 37 per cent (De Beenhouwer et al., 2013). But compared to other agricultural types, agroforestry systems were more likely to include indigenous tree species and maintain high tree species richness (Michon et al., 2007; Perfecto et al., 2010). By storing large quantities of carbon stock in above- and belowground biomass, agroforests can play an important role in the carbon cycle (Kessler et al., 2012; Obeng and Aguilar, 2015; Somarriba et al., 2013; Albrecht and Kandji, 2003). Reestablishing cocoa agroforestry systems on degraded crop- and pastureland in the Brazilian Amazon led to significant environmental benefits, including reduced emissions of up to 135 Mg of carbon per hectare and improved soil conditions (Schroth et al., 2016). In southern Bahia, Brazil, over half of the landscape carbon stocks (59 per cent) were contained in traditional cocoa agroforests, compared to 30 per cent in natural forest remnants (Schroth et al., 2015). Community managed agroforests in Southeast Brazil contributed to overall FLR goals by improving carbon sequestration, soil protection, water infiltration, and habitat provision for wildlife, as well as local livelihoods by supplying valuable and culturally important plants (de Souza et al., 2016). Increases in tree cover at the landscape scale have been observed following development of rubber agroforestry in Borneo (De Jong, 2001) and coffee plantations in Colombia (Rueda and Lambin, 2013a).

Tree diversity of agroforests depends on the type of management, location, and socio-economic factors (Schroth and Harvey, 2007). The associated animal diversity and ecosystem services provision varies in turn with tree diversity and shade cover (Clough et al., 2010; Asase and Tetteh, 2016; Faria et al., 2007; De Beenhouwer et al., 2013, 2016). Worldwide, smallholders struggle with persistent crop diseases, low production, and labor-intensive management practices. In addition, the growing concentration of value chains and downward pressure on prices increase farmers' vulnerability to volatile international markets (Hernández et al., 2014). In an attempt to secure short-term income, many smallholders worldwide convert shaded agroforests to more intensively managed plantations by reducing the number of shade trees (Steffan-Dewenter et al., 2007; Clough et al., 2009b; Vaast and Somarriba, 2014; Philpott et al., 2008). Decreasing tree cover in tropical agricultural landscapes might affect landscape functioning far beyond the farm level. Intensification reduces the landscape-level potential to provide habitat for biodiversity and store carbon (Abou Rajab et al., 2016; Wade et al., 2010; Clough et al., 2009a). Moreover, this strategy does not secure farmers with a more stable income, as intensified farms are more susceptible to diseases, reduce soil fertility, and reduce landscape resilience (Tscharntke et al., 2011; Jacobi et al., 2014).

Agroforestry production systems may motivate community involvement in FLR efforts, providing economic benefits and food security to local communities and fostering restoration of degraded landscapes (Chappell and LaValle, 2011). Shaded agroforests can increase the resilience of tropical landscapes (Bray and Klepeis, 2005) and serve as a transition stage between conventional agriculture and forest restoration (Vieira et al., 2009). For example, the adoption of more diverse agroforestry practices can reconcile sustainable agriculture, livelihoods, and biodiversity conservation though the use of native tree species and restoration of soil fertility in Minas Gerais State in Brazil (de Souza et al., 2012). Nevertheless, revenues from many agroforests are low. Adequate agricultural diversification, such as multi-cropping and multiple crop rotations, can potentially reduce the yield gap with conventional agriculture (Ponisio et al., 2015). Furthermore, implementing multifunctional tree species, such as nitrogen-fixing shrub or fruit tree species can provide additional food security, while enhancing soil fertility, preventing erosion, and providing nutritious fodder for livestock (Erdmann, 2005). A better understanding of local livelihoods, forest use, and farming systems will be required for successful restoration initiatives Although many studies have quantified biodiversity and biomass along the shade intensification gradient at the local scale (Steffan-Dewenter et al., 2007; De Beenhouwer et al., 2013; Obeng and Aguilar, 2015; Vaast and Somarriba, 2014), the contribution of agroforestry systems to achieve FLR objectives has rarely been assessed.

1.3 Financing and promoting restoration

Well-designed forest landscape restoration initiatives have a large potential to improve biodiversity, provide ecosystem services, and enhance local livelihoods. One of the main challenges in FLR is to cover the costs. Scaling up tropical forest restoration is constrained by opportunity cost of land, restoration cost, and the lack of a well-defined, cost-effective approach (Birch et al., 2010). Financial incentives that cover incurring costs are crucial to implement ecological restoration successfully at large scales. Potential incentive-based approaches that stimulate FLR include payments for ecosystem services schemes (PES), such as for watershed management, biodiversity conservation, carbon sequestration, and improved commodity markets for tropical commodities, especially from agroforests. Menz et al. (2013) estimated a cost of US$18 billion per year for the restoration of 150 million hectares of disturbed and degraded land globally, as targeted by the 2012 United Nations Rio+20 Conference on Sustainable Development. However, these efforts would provide approximately US$84 billion per year to the global economy.

1.3.1 Payments for ecosystem services

In *payments for ecosystem services* (PES) schemes, beneficiaries of ecosystem services pay local landowners in exchange for adapting land uses that secure ecosystem conservation and restoration (Engel et al., 2008). PES programs are gaining global recognition. International and national climate policies, such as Reducing Emissions from Deforestation and Forest Degradation (REDD+) and Clean Development Mechanism (CDM) programs, have been implemented in various tropical countries (Farley, 2010). REDD+ schemes focus on payments for carbon sequestration thought reforestation, but the number of CDM projects that include objectives reforestation remains small (Thomas et al., 2010). The outcomes for biodiversity and ecosystem services depend on the objectives and implementation of initiatives. In Costa Rica, PES schemes have been used to stimulate commercial timber trees with limited ambitions in terms of ecological restoration (Pagiola, 2008). In contrast, the Madagascan 'Mantadia' PES initiative successfully facilitated carbon sequestration and biodiversity conservation, while reducing soil erosion and nutrient depletion through forest restoration (Wendland et al., 2010). Despite weak carbon markets, forest restoration was a viable option for Colombian smallholders, as pre-scheme cattle farming provided minimal economic returns (Gilroy et al., 2014). A growing number of PES initiatives specifically target agroforestry smallholders, such as in Costa Rica where smallholders were compensated on a basis of US$0.60 per tree (Rosa et al., 2004). Carbon market participation for smallholders offers a promising option to simultaneously maintain shaded agroforests, while increasing economic returns (Seeberg-Elverfeldt et al., 2009; Hamilton et al., 2010; Hamrick and Goldstein, 2017; Jayachandran et al., 2017).

An important threat to these market-based incentives that stimulate forest restoration is their long-term durability. The Chinese Under the Grain to Green (GTGP) project aimed to restore ecosystem services and biodiversity, but over 20 per cent of the land was reconverted to original use when payments ceased (Chen et al., 2009). PES schemes that focus on one particular service risk neglecting other services. It is unlikely that multiple ecosystem services and livelihoods are simultaneously improved, while costs are reduced (Jack et al., 2008). This threat has been confirmed in several REDD+ projects. When carbon market are weak, landowners tend to maximize carbon gains at the lowest costs, at the expense of biodiversity and social benefits (Stickler et al., 2009). Therefore, PES schemes need to carefully target objectives to prevent trade-offs between carbon sequestration and other co-benefits (Venter et al., 2009; Putz and Redford, 2009). Finally, PES schemes tend to focus on intermediate and large farms, neglecting smallholders to reduce transaction costs, as was shown in several projects in Americas (Rosa et al., 2004). Over 380 million smallholder farming households owning less than five hectares make up about 30 per cent of all agricultural land and are responsible for more than half of the food calories produced globally (Samberg et al., 2016). These farmers tend to maintain complex and diverse agro-ecosystems, but are more vulnerable compared to larger farmers. Inclusion of small farmers in financial initiatives is critical to achieve FLR objectives.

1.3.2 Differentiated market channels

To incentivize and maintain smallholders agroforestry practices, scholars have argued that value chains of tropical commodities need to be developed or strengthened (Reij and Garrity, 2016; Reardon et al., 2009). Two major market-based approaches to maintain agroforests are improving market demand for sustainability standards for tropical commodities and the development of specialty markets that explicitly market the environmental benefits of commodities. These market-based incentives stem from a combination of growing consumer awareness of sustainability issues, changing global trade patterns following globalization and trade liberalization, and recognition that intergovernmental collaboration fails to address global supply chain sustainability issues (Potts et al., 2014).

Voluntary sustainability standards are market-based interventions that promote 'green' or ecoconsumerism to consumers by means of certification labels, such as Fairtrade, Organic, Rainforest Alliance, or UTZ certified. These types of voluntary, corporate management, termed 'new corporate environmentalism', promote self-regulation of environmental impacts beyond the compliance of environmental laws (Meyfroidt and Lambin, 2011). Consumers pay a higher price for certified products; in turn, certified producers are incentivized to implement nature-friendly farming practices through price premiums offered for sustainably farmed commodities. Market shares of all standard-compliant commodities have increased steadily over the past half-decade: certified coffee and cocoa comprised 40 and 22 per cent of global production by 2012 (Potts et al., 2014).

A second approach to stimulate agroforestry practices has been the development of *specialty markets*. These markets develop around commodities where traditional production practices already incorporate 'environmentally-friendly' attributes and the focus lies on marketing those attributes explicitly to consumers. Examples of alternative markets are shade-grown coffee and bean-to-bar cocoa. Farmers' incorporate environmental management practices and services, such as post-harvest processes to improve quality, in seeking better entry terms and prices on the market. Specialty markets often stimulate more direct trade relationships between buyers and producers, supported by private actors and NGOs (Ton et al., 2008). Numerous studies have found positive impacts on smallholder wellbeing and environmental conservation from participations in specialty markets, but outcomes are context dependent (Borrella et al., 2015; Neilson, 2014; Tobin et al., 2015; le Polain de Waroux and Lambin, 2013).

Although sustainability standards and specialty markets gained popularity over the last decades, they also drew criticism. Standards can create barriers for market entry for poor marginal farmers with limited access to capital, leading to even more inequality (Matissek et al., 2012). On the other hand, corporate involvement has diminished certain standard requirements, undergirding their potential impacts (Jaffee and Howard, 2009). Farm level studies often lack compelling evidence of positive socioeconomic or environmental impacts of certification (Blackman and Rivera, 2010; Lyngbæk et al., 2001; DeFries et al., 2017) and specialty markets (Neilson, 2014; Petchers, 2003). Certification schemes focus at the farm levels, whereas ecosystem benefits from well-designed agroforests are delivered at the landscape-level (Tscharntke et al., 2015). This incongruence can be addressed by two approaches: certification mechanisms could be linked with broader landscape-scale approaches (Milder et al., 2014) and current certification models could consider the landscape as a certified unit through landscape labeling (Ghazoul et al., 2009).

Market-based initiatives can foster FLR in tropical developing countries, by promoting restoration as an effective approach in national and international programs and policies, valuing positive environmental outcomes to local landowners, and developing sustainable supply chains. Effective market-based initiatives need to provide landowners with adequate compensation for any restoration costs and initiate restoration actions using effective incentives (Bullock et al., 2011). Strong local and regional institutional frameworks are needed that can deal with complexity and distribute benefits equally among stakeholders (Meli et al., 2017; Le et al., 2012).

1.4 General framework

1.4.1 Motivation and objectives

Forest restoration, in the form of natural regeneration, reforestation, afforestation, and agroforestry, offer potential to reconcile biodiversity conservation, ecosystem services supply, and human well-being in human-dominated landscapes. To achieve these ambitious goals, FLR initiatives need to be designed based on scientific knowledge and account for local complexities. Scientific understanding lacks on the potential of different restoration methods, the impact of landscape characteristics on restoration outcomes, and the potential of different market-based mechanisms that can support FLR financially. The main aim of this dissertation is to better understand the potential of restoration methods and its economic instruments to restore biodiversity and enhance livelihoods in human-dominated tropical landscapes. This dissertation limits FLR objectives to biodiversity conservation, ecosystem services supply, and human well-being. Three research questions summarize this objective:

1. What is the potential of different restoration types (i.e., natural regeneration, reforestation, afforestation, and agroforestry) to achieve forest landscape restoration objectives?
2. How does the landscape-level pattern of restoration affect forest landscape restoration objectives?
3. How can market-based mechanisms (i.e., payments for ecosystem services schemes and development of niche markets for agroforestry commodities) support forest landscape restoration objectives?

1.4.2 Modeling approach: LANDIS-II

Most studies on the effects of restoration have applied costly and time consuming fieldwork or empirical work, often carried out over short time periods of less than five years. Tropical restoration data reflect a sampling bias towards young regenerating forests and long-term datasets are scarce (Chazdon et al., 2009a,b; Gardner et al., 2009). This dissertation benefitted from an approach that combines ecological modeling, ecological fieldwork, as well as socioeconomic surveys. Model simulations allow considerably more experimentation with various restoration types and landscape patterns and over much longer periods than possible in the field. This dissertation implemented the spatially, dynamic forest simulation model LANDIS-II. Forest simulation model simulates and projects "the spatiotemporal trends of forest changes at the landscape-level based on mechanisms of forest dynamics and the interaction of disturbances" (He, 2008). Forest simulation models are increasingly used in forest planning, forest management, habitat conservation, and ecological restoration to answer research questions, including how ecological processes interact and how this affects the spatial pattern of forest landscapes (Xi et al., 2009). Many applications of LANDIS-II exists. In Latin America for example, LANDIS-II has been used to support conservation planning in a dry-land environment in Chile (Newton et al., 2011), to assess the potential for forest restoration in two Mexican landscapes under different disturbance regimes (Cantarello et al., 2011), and to predict the spatial extent of forest restoration (Birch et al., 2010). This section briefly introduces LANDIS-II

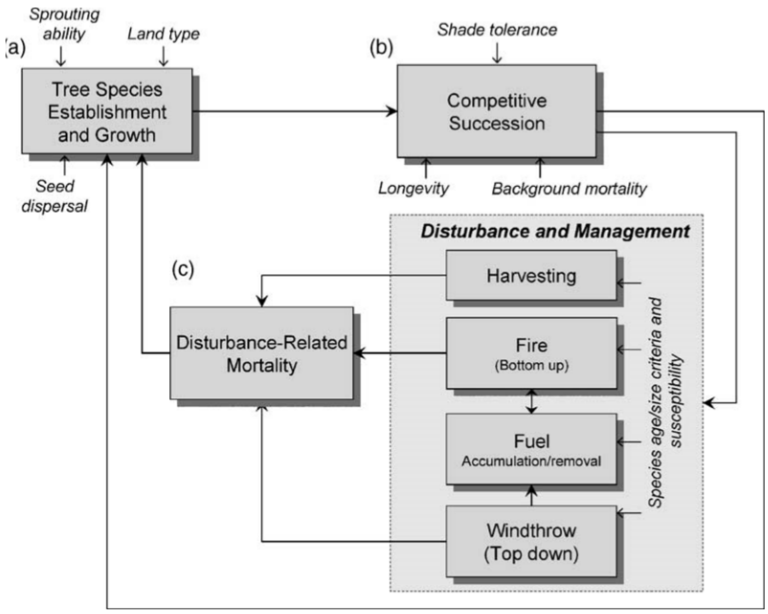

Figure 1.4: Major model dynamics of the LANDIS modelling framework, such as (a) tree species establishment and growth at site level, (b) competitive succession influenced by tree species characteristics, (c) disturbance extensions, such as harvesting, fire, and wind throw resulting in additional tree species mortality. Source: Mladenoff (2004)

and a more detailed description can be found elsewhere (He and Mladenoff, 1999; Mladenoff, 2004; Scheller et al., 2007; Foundation, 2015).

The well-established LANDIS-II model simulates the dynamics of forested landscapes on large spatial scales (10^4 to 10^7 hectares) by incorporating ecological processes, such as succession, disturbance, and seed dispersal (Fig. 1.4) (Scheller et al., 2007). The simulated landscape is represented as a raster with grid cells that are divided into sites and ecoregions. Sites represent individual raster cells and are assumed homogeneous in light levels that influence tree species succession. Ecoregions are represented by groups of sites that are homogeneous in climate and soils that determine the ability of tree species to establish.

LANDIS-II simulates the distribution of user-defined number of tree and shrub species and the species' characteristics (i.e., life history attributes) determine the successional strategy of each species, including where it establishes, how long it grows, and how it responds to disturbances. To improve computation efficiency, individual trees and shrubs are grouped into cohorts present at a particular site and encompass additional data, for example on aboveground live biomass. Furthermore, LANDIS-II includes a set of spatially dynamic processes, such as species dispersal and harvest, fire, and wind throw disturbances. The user can define the simulated disturbances by means of extensions (Fig. 1.4).

Model parameterization consists of three basic data types: tree species data, ecoregion data, and initialization data. Species data include the species' life history attributes and species establishment probabilities within each ecoregion. Life history attributes, such as species longevity, age of maturity, seed dispersal distance, and shade tolerance, are normally extracted from the literature and expert knowledge. In regions where species data are rare, species can be lumped into functional groups of species with similar characteristics. Species establishment probabilities for each ecoregion can be determined from expert knowledge or additional modeling, such as species distribution modeling (Newton et al., 2011; Cantarello et al., 2011). Initialization data include the community composition of species and ages across the simulation landscape. These data are often based on time-consuming and scarce inventory data.

Simulation of biomass are optional in LANDIS-II through the Biomass Succession extension (Scheller and Mladenoff, 2004) and is useful as a measure of tree species density and addressing carbon storage and ecosystem functioning. Biomass Succession simulates changes in the aboveground biomass (g m^2) of each cohort in relation to age, competition, and disturbance, neglecting changes in below ground carbon dynamics. This extension determines shade as a function of actual site biomass relative to the maximum site biomass. Shade levels in turn impact the ability of species to establish and grow on sites. For each species-ecoregion combination, the maximum growth rate (MaxANPP). The actual growth during simulations is limited by the presence of other cohorts and the biomass of the current cohort. It is assumed that growth ability is limited for low biomass cohorts, but increases with biomass. Final cohort biomass is a function of the maximum ANPP of each species (maxANPP), the maximum achievable live aboveground biomass of each species (maxBiomass), the age of the cohort, and inter- and intraspecific competition (Scheller and Mladenoff, 2004). MaxANPP and maxBiomass are input parameters that we have based on field observations and the literature. LANDIS-II goes through a 'spin-up' phase to calculate the starting biomass for each cell at the onset of each simulation. During this phase, each species-age cohort is grown from establishment to its age at the beginning of the simulation (year 0).

To simulate landscape-level restoration impacts, this dissertation has implemented the Harvest extension, quite similar to (Thompson et al., 2011). The

Harvest extension simulates biomass removal or planting of a user-specified selection of species cohorts during simulation. The simulation landscape is divided into a number of management areas that define the collection of stands (i.e., collecting of raster cells) to which specific harvest prescriptions will be applied. The prescriptions define the amount of forest biomass that is removes across all species on the site, the prevention of future establishment, and the reduction of maximum achievable biomass (maxBiomass) on the site. It also determines which cohorts will be removed and in which order sites will be harvested.

Model calibration intends to change model parameters to fit model output to empirical data. Data limitation is often a challenge, especially in tropical systems. For this dissertation, model output was compared with the dependent input data due to a lack of independent datasets. In addition, a sensitivity analyses on key parameters was performed to check the influence of model parametrization. Model validation intends to determine how well the model reflects empirical data. As forest simulation models, like LANDIS-II, intend to project future conditions, validation data is by definition non-existent. In this sense, it is important to underline that the modeling results presented in this dissertation should thus not be interpreted as predictions.

1.4.3 Restoration in Ecuador: two case studies

Major Latin American restoration projects in the 1970s and 1980s have emphasized tree plantations at industrial scales, disregarding biophysical suitability and sociocultural dimension (Chokkalingam and de Jong, 2001; Nagendra and Southworth, 2010; Hirsch et al., 2011). Today, community-based forest management is promoted by international environmental organization as a more sustainable approach to restoration (Ribot et al., 2006).

In Ecuador, land reforms and governmental programs (e.g., the payment for carbon sequestration program PROFAFOR) have stimulated forest plantations from the 1960s onwards, leading to the conversion of large areas of natural grasslands to exotic pine plantations in the highlands (Farley, 2007, 2010). Economic and demographic forces, including land abandonment with economic development resulted in fallow abandonment, mainly in the Ecuadorian Andes (Balthazar et al., 2015). Ecuadorian conservation policies (e.g., the payment for ecosystem services program Socio Bosque) have started to recognize the role of forest regrowth at the landscape-level and national reforestation programs incorporate restoration activities (de Koning et al., 2011; Cuesta et al., 2017). For example, the Socio Bosque program offers payments to individual landowners and communities between US$0.50 and US$60 per hectare per year to incentivize conservation stewardship (Rosa da Conceiçao et al., 2015). For 2014 onwards, the program shifted its focus specifically to restoration of natural ecosystems through reforestation or natural regeneration (Raes et al., 2017). Additionally, Ecuador has included the right of nature restoration explicitly in the Constitution under the National Plan for Good Living and the National Plan for Forest Restoration in 2014, creating financial incentives to encourage restoration of 500,000 hectares by 2017 (Meli et al., 2017; MAE, 2014b).

The government has launched a national economic diversification program 'Ecuador Productivo 2025' to develop higher value agricultural products (Ahmed and Hamrick, 2015). The cocoa segment is one of the high priority agriculture chains for Ecuador to develop. Over US$80 million will be invested to develop the sector in the next 10 years (Cepeda et al., 2013). Multiple laws have been put in place that aim to maintain and stimulate the production of high quality Nacional cocoa beans, including financial and technical support to improve cropping, pruning, and market access for smallholders. Shares of certified cocoa have also been growing in Ecuador. Major voluntary standards in the Ecuadorian cocoa sector have increased include organic, Fairtrade, UTZ Certified, and Rainforest Alliance, accounting for approximately 8, 4, 1, and < 1 per cent of total national production in 2014, respectively (Potts et al., 2014).

1.4.4 Outline

This section details the outline implemented in this dissertation to answer the three main research questions (Fig. 1.5). Chapter two explores the relationship between restoration and the potential to support biodiversity, taking into account the impact of different landscape characteristics. The Pangor watershed serves as a case study; a dynamic Andean landscape where patches of agricultural land, natural forest and grasslands, natural regenerating forest, and plantations coexist. Specific objectives were:

- To explore the impact of restoration, following either natural forest regeneration or plantation establishment, on the potential to restore native floristic biodiversity at the landscape-level;

- To analyze the relative importance of landscape characteristics of restoration (i.e., proximity to natural forest remnants, percentage of surrounding natural forest, compositional heterogeneity of surround natural land covers, restoration patch size, time since restoration onset, altitude) to restore native floristic biodiversity at the landscape scale.

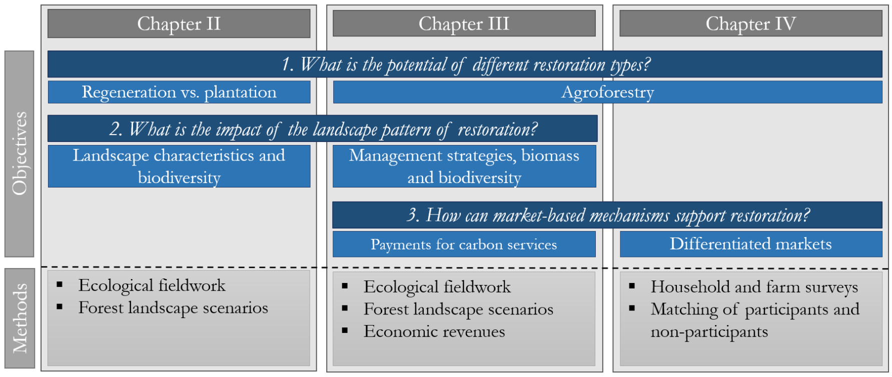

Figure 1.5: Outline showing how the individual chapters of this dissertation connect to answer the main research questions and implement various methods

This research serves as a test case to develop the LANDIS-II model in the context of Ecuador. To achieve the objectives, ecological fieldwork is conducted in the study area and to tree species distribution models are constructed to calibrate and validate model simulations. Then, restoration scenarios are developed based on observed historical land use change with different landscape characteristics of restoration. Model outcomes are compared to a baseline scenario to assess the impacts of different landscape restoration strategies.

Chapter three examines the potential of smallholder agroforestry systems to achieve FLR objectives. This research is conducted in the province of Esmeraldas; a main production region of the traditional, fine flavor cocoa variety Nacional produced mainly by poor smallholders owning less than five hectares of land. Specific objectives were:

- To understand the landscape-level impact of different agroforestry management strategies on aboveground biomass and biodiversity;
- To analyze the potential of payment for carbon services initiatives to support shade agroforestry practices financially.

To achieve the objectives, ecological fieldwork is carried out in agroforestry plots in the study area and forest inventory data obtained from the Ecuadorian Ministry of Agriculture is analyzed to calibrate and validate model simulations. Then, scenarios are constructed that test different agroforestry management practices and landscape characteristics. Finally, economic revenues from carbon markets are estimated based on model predictions.

Chapter four considers the socioeconomic and environmental outcomes of alternative markets for niche commodities, taking cocoa agroforestry in the province of Esmeraldas as a case study. Specific objectives were:

- To explore the impact of participation in specialty markets on smallholder livelihoods;
- To understand the impact of participation in specialty markets on the implementation of nature-friendly management practices and environmental conditions of farms.

For this research, socioeconomic household and environmental surveys are conducted, focusing on small farmers owning less than 10 hectares of land. Subsequently, a credible counterfactual is constructed that compares participants and non-participants. Additional data is collected from secondary sources to support quantitative results.

CHAPTER 2

RESTORATION THROUGH NATURAL REGENERATION AND PLANTATIONS: A MODELING APPROACH IN THE ECUADORIAN ANDES

This chapter is based on the following paper: R.S. Middendorp, A.J. Pérez, A. Molina, E.F. Lambin. 2016. The potential to restore native woody plant richness and composition in a reforesting landscape: a modeling approach in the Ecuadorian Andes. *Landscape Ecology*, 31(7): 1581-1599. doi:10.1007/s10980-016-0340-7.

Abstract

Natural regenerating forests are rapidly expanding in the tropics. Forest transitions have the potential to restore biodiversity. Spatial targeting of land use policies could improve the biodiversity benefits of reforesting landscapes. We explored the relative importance of landscape attributes in influencing the potential of tree cover increase to restore native woody plant biodiversity at the landscape scale. We developed land use scenarios that differed in spatial patterns of reforestation, using the Pangor watershed in the Ecuadorian Andes as a case study. We distinguished between reforestation through natural regeneration of woody vegetation in abandoned fallows and planted forests through managed plantations of exotic species on previously cultivated land. We simulated the restoration of woody plant biodiversity for each scenario using LANDIS-II, a process-based model of forest dynamics. A pair-case comparison of simulated woody plant biodiversity for each scenario was conducted against a random scenario. Species richness in natural regenerating fallows was considerably higher when occurring in: (i) close proximity to remnant forests; (ii) areas with a high percentage of surrounding forest cover; and (iii) compositional heterogeneous landscapes. Reforestation at intermediate altitudes also positively affected restoration of woody plant species. Planted exotic pine forests negatively affected species restoration. Our research contributes to a better understanding of the recolonization processes of regenerating forests. We provide guidelines for reforestation policies that aim to conserve and restore woody plant biodiversity by accounting for landscape attributes.

2.1 Introduction

Although tropical deforestation continues at a rapid rate, there is also evidence of a regrowth of tropical and subtropical forests in some regions. Regrowth occurs in various ways. For example, through regenerating forests which naturally establish after human or natural disturbance of the original forest vegetation (Chokkalingam and de Jong, 2001; Chazdon, 2014) or planted forests composed of trees established through planting of native or exotic species (FAO, 2015). To date, we still lack an accurate estimate of the extent of regenerating or planted tropical forests (Asner et al., 2009; Meyfroidt and Lambin, 2011). Numerous studies have emphasized the critical importance of regenerating forests in tropical landscapes (Dunn, 2004; Chazdon, 2008; DeClerck et al., 2010), but have also emphasized the differences between forest transition pathways to restore biodiversity and ecosystem functions (Meyfroidt and Lambin, 2011; Chazdon, 2014).

In Ecuador, land reforms and governmental programs (e.g., the payment for carbon sequestration program PROFAFOR) have stimulated forest plantations from the 1960s onwards, leading to the conversion of large areas of natural grasslands to exotic pine plantations in the highlands (Farley, 2007, 2010). Economic and demographic forces, such as land abandonment with economic development, have also favored natural regeneration of tropical forests (Lambin and Meyfroidt, 2010; Rudel et al., 2005) and have been observed in the Ecuadorian Andes (Balthazar et al., 2015). Ecuadorian conservation policies (e.g., the payment for ecosystem services program Socio Bosque) have started recognizing the role of forest regrowth at the landscape level and will incorporate reforestation programs in the future (de Koning et al., 2011). [1]

Reforestation and planted forests affect the spatial configuration and composition of landscapes and therefore habitat distribution. Landscape structure has been suggested as an important factor that determines the effectiveness of local biodiversity conservation management in human-modified landscapes (Tscharntke et al., 2012b). Even though landscape fragmentation has been recognized as a key factor in biodiversity loss (Fischer and Lindenmayer, 2007), the effect of landscape patterns on restoration of biodiversity and ecosystem functions in reforesting landscapes is still not well understood (Hall et al., 2012; Leite et al., 2013; Melo et al., 2013a). Well-designed approaches of re-

[1] After publication of this manuscript (from 2014 onwards), the Socio Bosque program focusses specifically on restoration of natural ecosystems through reforestation or natural regeneration (Raes et al., 2017). The program offers payments to individual landowners and communities between US$0.50 and US$60 per hectare per year to incentivize conservation stewardship (Rosa da Conceiçao et al., 2015).

forestation policies could improve natural regeneration of native species at a landscape scale.

The objective of this study was to better understand the relative importance of landscape attributes in influencing the potential of tree cover increase to restore native woody plant richness and composition at the landscape scale. We focused on human-modified landscapes in the Ecuadorian Andes, where extensive farming and grazing takes place. We used a process-based vegetation model to simulate forest cover and species dynamics. We distinguished between reforestation through woody vegetation gain by natural, unassisted regeneration of abandoned fallows, and planted forests through managed plantations of exotic species on previously cultivated lands. We focused on the richness and composition of woody plants, as the diversity and structural characteristics of vegetation in regrowth forests were important determinants of faunal assemblages (Evelyn and Stiles, 2003; Kanowski et al., 2006; Bowen et al., 2007). This research provides guidelines for reforestation policies that aim to enhance biodiversity restoration in human-modified landscapes.

2.2 Background

Many studies that have explored the effects of landscape patterns on species richness have been based on costly and time consuming fieldwork, carried out over short time periods (<5 years). In the tropics, available data reflect a sampling bias towards young regenerating forests (Chazdon et al., 2009b) and long-term datasets are scarce (Chazdon et al., 2009a; Gardner et al., 2009). Model simulations allow considerably more experimentation with various landscape patterns and over much longer time periods than possible in the field.

In this study, we used the spatially explicit landscape simulation model LANDIS-II (Scheller et al., 2007) to test various hypotheses on landscape factors influencing the recovery of woody species richness and composition resulting from tree cover increase in a tropical landscape. LANDIS-II is a well-established model that simulates the dynamics of forested landscapes over large spatial scales by incorporating ecological processes, such as succession, disturbance, and seed dispersal. In Latin America, LANDIS-II has been used to support conservation planning in a dry-land environment in Chile (Newton et al., 2011), to assess the potential for forest restoration in two Mexican landscapes under different disturbance regimes (Cantarello et al., 2011), and to predict the spatial extent of forest restoration (Birch et al., 2010). Thompson et al. (2011) showed that LANDIS-II could simulate forest to non-forest transitions.

Table 2.1: Overview of the hypotheses tested in this study

Hypothesis	Landscape attribute
	The recovery of woody species richness and composition in regenerating fallows are positively correlated with …
1	proximity to natural forest remnants.
2	cover percentage of surrounding natural forest remnants.
3	compositional heterogeneity of surrounding natural land covers.
4	the size of regenerating patches.
5	time since abandonment of regenerating patches.
	The recovery of woody species richness and composition in regenerating fallows is negatively correlated with …
6	the altitude of regenerating patches.
7	planted forest using exotic pine species.

The ecological value of forest regeneration is influenced by many factors. Beyond the local site characteristics (i.e., climate, soil fertility), landscape dynamics affect the initial establishment, species composition, and persistence of regenerating forests (Chazdon et al., 2009b). Landscape attributes (e.g., distance to forest remnants, local forest cover, heterogeneity of surrounding land covers) and temporal dynamics (e.g., time since abandonment) have a strong influence on the reforestation trajectory (Chazdon, 2003). We tested seven hypotheses based on the literature (Table 2.1), which are justified here.

Local forest regrowth is linked with features of the surrounding landscapes through source populations and processes of seed dispersal. The extent and distribution of forest cover in the surrounding landscape serves as the external ecological memory and influences the recruitment of forest species during recolonization (Chazdon, 2014). In previous studies, the presence of old-growth forest remnants promoted the reassembly of tropical communities (Norden et al., 2009). Proximity to mature forests was particularly important during forest regrowth, as remnant vegetation provided a seed source (Guariguata and Ostertag, 2001; Mesquita et al., 2001; Günter et al., 2007). The lack of dispersal of forest seeds was found to be one of the most important barriers to the restoration of tropical montane forest on abandoned pastures (Holl et al., 2000). We thus expect that the recovery of species richness and composition in regenerating fallows positively relates to a greater proximity to natural forest remnants (*hypothesis 1*) and a higher percentage of surrounding natural forest cover (*hypothesis 2*).

The diversity of different natural land covers (i.e., compositional heterogeneity) is also important during species regeneration (Gardner et al., 2009). Fahrig et al. (2011) suggested that biodiversity is expected to increase through the accumulation of species as more natural cover types are added to the landscape. Compositional heterogeneity was a key component for explaining landscape species richness (Wagner et al., 2000). We expect that a higher compositional heterogeneity of surrounding natural land covers positively affects the recovery of species richness and composition in regenerating fallows (*hypothesis 3*). Island biography and metapopulation theories predict that the largest patches hold the most species as a result of lower extinction rates (MacArthur and Wilson, 1967; Hanski, 1999). In highly fragmented tropical landscapes, the size of remnant forest patches was positively related to plant species density (Arroyo-Rodríguez et al., 2009; Hernández-Stefanoni et al., 2011). We expect that a larger size of regenerating forest patches positively affects the recovery of species richness and composition (*hypothesis 4*).

It has been shown that plant species richness increased over time during forest regrowth (Guariguata and Ostertag, 2001; Barlow et al., 2007; Dent and Wright, 2009; Norden et al., 2009), making the age structure of regenerating forests an important determinant of landscape-level biodiversity. At initial stages of reforestation, typical old-growth species and endemic forest specialists may be missing, but they might be able to establish later (Chazdon, 2003). We expect that time since abandonment of regenerating fallows positively affects the recovery of species richness and composition (*hypothesis 5*).

Altitude is a well-known driver of compositional variation in Andean vegetation (Jorgensen and Ulloa Ulloa, 1994). Individuals of the same species have been found to differ in growth rate across their altitudinal ranges due to changes in temperature (Rapp et al., 2012). Recovery rates in cloud forests might lag behind due to lower temperatures at higher altitudes that cause trees to grow slowly. We expect that a higher elevation of regenerating abandoned fallows negatively affects the recovery of species richness and composition (*hypothesis 6*).

Finally, studies showed that the ecological benefits of plantations depend largely on the land cover they replace and whether planted trees are native or introduced species (Bremer and Farley, 2010). Most exotic forest plantations established in high altitude grasslands in Ecuador negatively affected the water retention and carbon storage capacity of páramo soils (Molina et al., 2012; Farley et al., 2013) and offered little value for biodiversity conservation (Hofstede et al., 2002). We expect that planted forests using exotic pine species negatively affects the recovery of species richness and composition in previously cultivated areas (*hypothesis 7*).

We simulated restoration of woody plant richness and composition using LANDIS-II over a series of land-use scenarios that explored a range of variation in the seven key landscape attributes (Table 2.1). This approach facilitated the comparison of different landscape architectures of reforestation to evaluate the relative importance of landscape attributes on biodiversity restoration.

2.3 Methods

2.3.1 Study area

The Pangor watershed (78°50′-79°01′W, 1°43′ - 1°58′S) located in the Chimborazo province is a headwater basin of about 282 km^2 and ranges from 1,431 to 4,333 masl (Fig. 2.1a). The mean annual precipitation is about 1,400 mm and the topography is characterized by steep slopes, with over half of the catchment above 25 °C. Soils developed on volcanic ash deposits (mainly Andosols) and are characterized by a high water retention capacity and soil organic matter content when undisturbed (Buytaert et al., 2005). However, these soils become susceptible to erosion when cleared or cultivated.

Temperature, precipitation, and solar radiation are strongly correlated with elevation, leading to an altitudinal zonation of vegetation observed throughout the Ecuadorian Andes. The treeless area above 3,400 masl is covered by grass páramo (i.e., *pajonal*), mainly consisting of bunch grasses with scattered herbs and shrubs. Forests of Polylepis trees occur sporadically in this zone, supporting additional woody species. Grass páramos can support a high number of species and are considered natural vegetation cover, although scientists have claimed it is man-made through burning during the Holocene, consequently lowering the historical treeline (Jorgensen and Ulloa Ulloa, 1994). Below the treeline, upper montane forest (>3,000 masl) and montane forest (<3,000 masl) cover the catchment. Upper montane forest is characterized by contorted tree growth with many multi-stemmed trees or giant shrubs. Montane forest (i.e., Andean cloud forest) is composed of a rich diversity of taller trees cover by diverse epiphyte flora and lianas.

Balthazar et al. (2015) highlighted a shift from net deforestation to net reforestation in the Pangor watershed in the early 1990s based on a 50-year time series consisting of aerial photographs and satellite images. Most of the lowland in the catchment has been cleared of its natural vegetation for small-scale agriculture of mainly beans, potatoes, and maize, while extensive grazing occurs on the high altitude grasslands. Since the late 1980s, pine plantations were established in the higher parts of the catchment (>3,200 masl), after being promoted by governmental programs that linked the Ministry of Agriculture with rural landowners (Farley, 2007, 2010). Approximately five percent of the watershed was covered with pine plantations by 2009 (Vanacker et al., 2013). Additionally, some abandoned agricultural lands were left for natural regeneration (Balthazar et al., 2015).

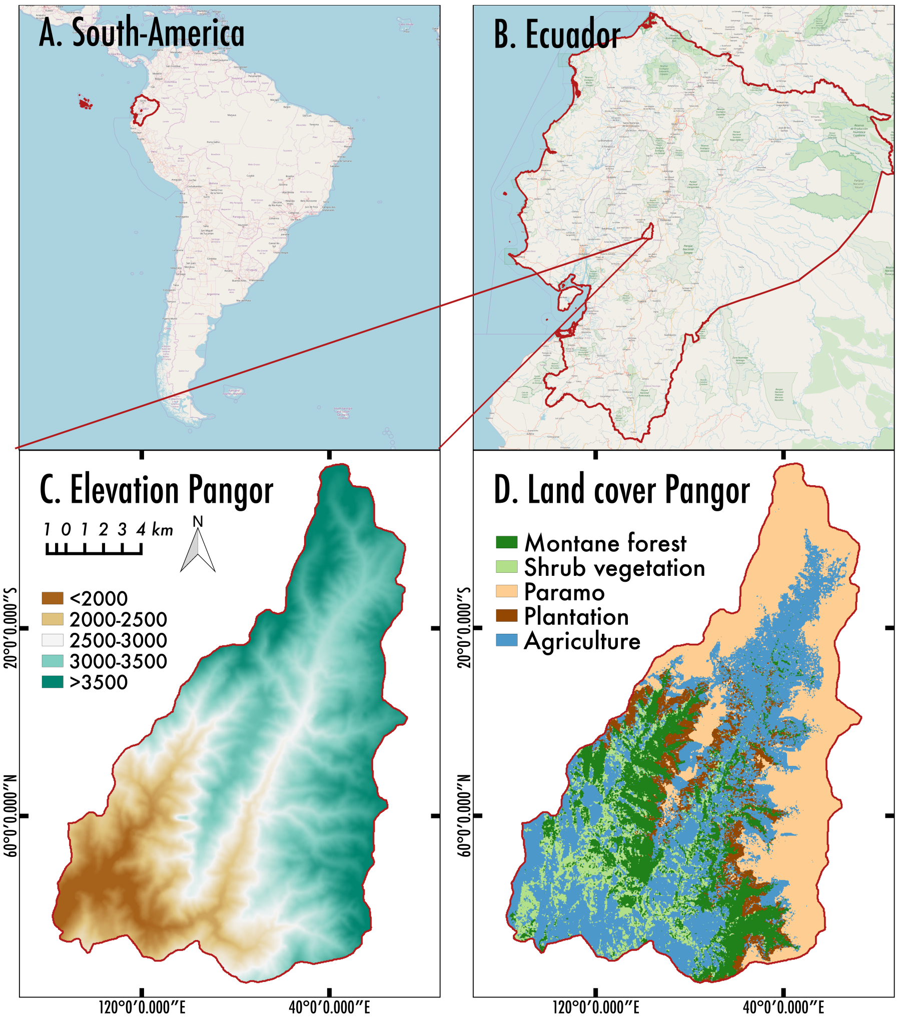

Figure 2.1: (a) Location of the Pangor watershed in Ecuador; (b) land-cover classification; (c) digital elevation model

2.3.2 Forest landscape model

LANDIS-II is a stochastic, spatially-explicit, raster- and process-based computer model that simulates the dynamics of forested landscapes on large spatial and temporal scales by incorporating ecological processes, such as succession, disturbance, and seed dispersal (Scheller et al., 2007). Species composition and succession is driven by the user-defined life history characteristics of each modeled species (i.e., longevity, minimum age at reproduction, shade tolerance, minimum/maximum seed dispersal distances, and resprout probability) and their establishment coefficient. This coefficient governs the probability of each species to establish in different subdivisions of the landscapes that are homogeneous in environmental conditions (i.e., ecoregions). For each cell, the presence and absence of 10-year age cohorts is monitored. A more detailed description of LANDIS-II can be found elsewhere (He and Mladenoff, 1999; Mladenoff, 2004; Scheller et al., 2007).

2.3.3 Field survey data

To determine the floristic composition across the Pangor watershed, we collected species data in September and October 2013 from twelve 0.02 hectare plots (40 x 5 m) (Table A.1). These plots were parallel to the contour lines of the slope in montane forest (n = 3), upper montane forest (n = 2), natural regenerating forest (n = 2), pine plantations (n = 4), and páramo (n = 1). Jorge Caranqui collected additional species data in 2007 from five 0.02 hectare plots (50 x 4 m) in montane forest (*unpublished data*), resulting in a total of eight montane forest plots. Plots were selected using a subjective sampling approach, to identify sample units that are assumed to be representative of the entire population.

In each plot, plant height and diameter at breast height (DBH) for all woody plants >1 cm DBH were recorded. Based on collected specimens, all individuals were subsequently identified at the Herbario QCA at the Pontificia Universidad Católica del Ecuador. Plant species names and authorities were according to Jorgensen and León-Yánez (1999). We excluded six individuals that could not be identified to genus level.

2.3.4 Model parameterization

LANDIS-II is designed to accept raster imagery as spatially explicit input. The required input maps are an ecoregion map that describes the ecological conditions affecting species establishment and an initial communities map that describes the distribution and age of cohorts for each species at the onset of simulations. All maps were produced and manipulated using ArcGIS 10.2.2 (©2014 ESRI Inc., Redlands, California, USA).

2.3.4.1 Ecoregions

Vegetation zonation in the Ecuadorian Andes is strongly related to altitude. We selected the elevation zonation as proposed by Jorgensen and Ulloa Ulloa (1994). We generated a total of 11 active ecoregions by overlaying a Digital Elevation Model and a beam solar irradiation map (Table A.2). All regions above 3,700 masl and all agricultural raster cells were inactive during simulations as they mainly consisted of herbaceous and small shrub vegetation in most of our study area. The species establishment probabilities were derived from MaxEnt models (Phillips et al., 2006) as the average probability of occurrence for each species in each ecoregion. Input data for these models included species occurrence data from the MOBOT Tropicos®database (©2014 Missouri Botanical garden, USA) and bioclimatic variables extracted from the Worldclim database (Hijmans et al., 2005).

2.3.4.2 Initial communities

A land cover map with a spatial resolution of 30 m was made for 2010 based on a supervised classification method using Worldview-2 imagery, with an average accuracy of 80.4% and a Kappa coefficient of 77.5% (Marie-Pierre de Wouters and Vincent Balthazar - *unpublished data*). Five land cover types were identified: (1) natural forest remnants (i.e., cloud and upper montane forest); (2) shrub vegetation mainly characterized by natural regeneration; (3) sub and grass páramo; (4) agriculture; and (5) pine plantations (Fig. 2.1b).

The initial distribution, composition, and age structure of forest communities for each cell in the landscape was determined by combining the land cover classification with the field survey data. We estimated ages of individuals by dividing the measured height with the maximum documented height, multiplied with the maximum documented longevity for each species.

2.3.4.3 Species attributes

We selected a subset of the 19 most abundant species based on their importance value index, which is the sum of relative density, relative frequency, and relative dominance found during the field survey (Mishra, 1968). For each modeled species, we extracted life history characteristics from the literature and from a consultation of local experts (Table A.3).

2.3.5 Scenarios

To explore the relative importance of landscape attributes on the re-colonization of biodiversity during reforestation, we developed alternative land-use scenarios that differed in the pattern of reforestation (Table 2.2). First, we estimated the rate of past natural regeneration based on post-classification change detection between land-cover classifications of two satellite imagery for 1991 and 2009 (Balthazar et al., 2015), using Markov Chain Analysis (MCA) within the Land Change Modeler (LCM) in IDRISI Taiga v.17.01 (©2015 Clark Labs, Clark University, Worcester, USA). We projected the area of natural regenerating forests from 2010 to the year 2055 to about 943 hectare (i.e., 10,476 raster cells). All natural regeneration during simulations took place in former agricultural raster cells and at equal time intervals of 15 years (i.e., year 2010, 2025, and 2040) to disintegrate the influence of time since onset of regeneration on species richness.

The spatial allocation of raster cells under reforestation varied for each scenario, altering the: (i) proximity to remnant forest; (ii) percentage of surrounding forest cover; (iii) compositional heterogeneity of surrounding natural land-cover; (iv) patch size; (v) elevation; and (vi) type of reforestation (i.e., planted pine forests). We created one random reforestation scenario with a similar area of reforestation that served as a reference to which all other scenarios were compared. The proximity to remnant forest was calculated as the Euclidean distance to the nearest forested cell. The percentage of forest cover in the surrounding landscape was calculated in a 200 m radius surrounding the focal cell. Compositional heterogeneity was determined as the Simpson's diversity index, which represents the probability that any two pixels selected at random were different patch types, calculated for a 200 m radius moving window using FRAGSTATS software V.4.2 (McGarigal et al., 2012). We varied patch size by increasing from 150 m to 250 m the buffer size in which reforestation occurred around randomly selected raster cells. For the plantation scenario, an age-cohort of pines was planted at the onset of regeneration at random locations across the landscape. We created a total of one random, one planted forest, and 15 reforestation scenarios (Table 2.2; Fig. A.1).

We implemented the Base Harvest (v2.2) extension of LANDIS-II (Gustafson et al., 2000) to simulate reforestation during simulations. At each harvesting time step, raster cells identified for reforestation were completely harvested until the onset of reforestation (i.e., 2010, 2025, and 2040). Simulations were conducted for 55 years. Each scenario was replicated three times with a different random seed number to account for within-scenario stochasticity. The time steps were five years for tree succession and one year for the harvesting extension.

Table 2.2: Names and description of 17 scenarios simulated in LANDIS-II

Scenario name	Reforestation occurs ...
random	randomly on previously cultivated lands
distance < 200m	at an Euclidean distance to the nearest forested cell smaller than 200 m
distance 200-400m	at an Euclidean distance to the nearest forested cell between 200-400 m
distance > 400m	at an Euclidean distance to the nearest forested cell larger than 400 m
forest cover < 50%	where the percentage forest cover in a 200 m radius lies between 0-50%
forest cover > 50%	where the percentage forest cover in a 200 m radius lies between 50-100%
forest cover < 30%	where the percentage forest cover in a 200 m radius lies between 0-30%
forest cover > 30%	where the percentage forest cover in a 200 m radius lies between 30-100%
diversity index < 50%	where the Simpson's diversity index in a 200 m radius is smaller than 0.5 (i.e., low compositional heterogeneity)
diversity index > 50%	where the Simpson's diversity index in a 200 m radius is larger than 0.5 (i.e., high compositional heterogeneity)
patch size <	in patch sizes with a diameter of maximum 150 m
patch size >	in patch sizes with a diameter of 150-250 m
altitude < 2,400m	randomly at altitudes below 2,400 masl
altitude 2,400-3,000m	randomly at altitudes between 2,400-3,000 masl
altitude 3,000-3,400m	randomly at altitudes between 3,000-3,400 masl
altitude > 3,400m	randomly at altitudes above 3,400 masl
plantation	after planting an age-cohort of exotic pine trees randomly on previously cultivated lands

2.3.6 Data analyses

The Age Cohort Statistics v1.0 extension of LANDIS-II generated raster maps for each time step containing for each pixel: (i) the minimum and maximum age across all species; (ii) the presence or absence of all species; and (iii) the total number of tree species. All output maps were analyzed using the raster package (Hijmans, 2014) in R (R Development Core Team, 2008). We recalculated species richness maps to contain only individuals older than one year.

At the landscape scale, we calculated the mean and standard deviations for the area in hectares containing a number of species and for the cover percentage of selected species over time for three repeated simulations of each scenario. We selected the following species for analysis, based on their life characteristics (e.g., shade-tolerance, seed dispersal limitations): *Aegiphila bogotensis*, *Baccharis latifolia*, *Boehmeria aspera*, *Clusia multiflora*, *Gynoxys* spp., *Hedyosmum luteynii*, and *Podocarpus glomeratus*.

We compared simulated species richness for reforesting fallows between each scenario and the random scenario using a pair matched case-comparison method (Table A.5). This approach accounts for the imbalance in co-variables (i.e., altitude, solar irradiance, time since abandonment) that influences species establishment during simulations. We conducted a genetic matching with replacement in R (Sekhon, 2011). This algorithm found the optimal pair of matching that achieves covariance balance. We assessed the causal effect of a given landscape attribute on the simulated species richness based on matched data by comparing means of each scenario against the random scenario, using Wilcoxon-Mann-Withney tests. In the subsequent results, we only included the results for scenarios that showed a significantly different average species richness compared to the random scenario for the matching analyses. These scenarios are indicated in the text as 'significant scenarios'. All other results are included in the appendices and are indicated in the text as 'non-significant scenarios'.

Descriptive statistics were used to calculate the mean cover percentage of reforesting fallows containing a number of species. In addition we calculated α-, and β-diversity values to assess the changes in landscape biodiversity, using the vegan package in R (Oksanen et al., 2007). We calculated α-diversity as the mean species richness in reforesting raster cells (Whittaker, 1972) and β-diversity as the dissimilarity between two reforesting raster cells based on counts at each cell, using the Bray-Curtis index (Bray and Curtis, 1957). We also calculated the cover percentage of selected species over time for reforesting fallows in each scenario.

2.4 Results

2.4.1 Field survey

A total of 80 woody species were encountered in the field sites, although taxonomic uncertainty over some genera led to congeneric species being grouped under the same species life history characteristics in the model. The most important tree species were *Hedyosmum luteynii* and *Aegiphila bogotensis* in montane forest, *Berberis* spp. and *Brachyotum ledifolium* in upper montane forest, and *Miconia* spp. and *Baccharis latifolia* in regrowth forest (for all species encountered, see Table A.4). A total of 49 species were found in montane forest with an average richness of 12 species per plot. Natural regenerated forests were species rich with about 15 species per plot. However, species composition did not resemble montane forests completely, containing more shade-intolerant pioneer species. In the páramo plot, we found a total of 12 woody species. Pine plantations contained little or no undergrowth. We encountered more than two shrub species only in one plantation that was less densely planted with pines and located in close proximity to remnant forest.

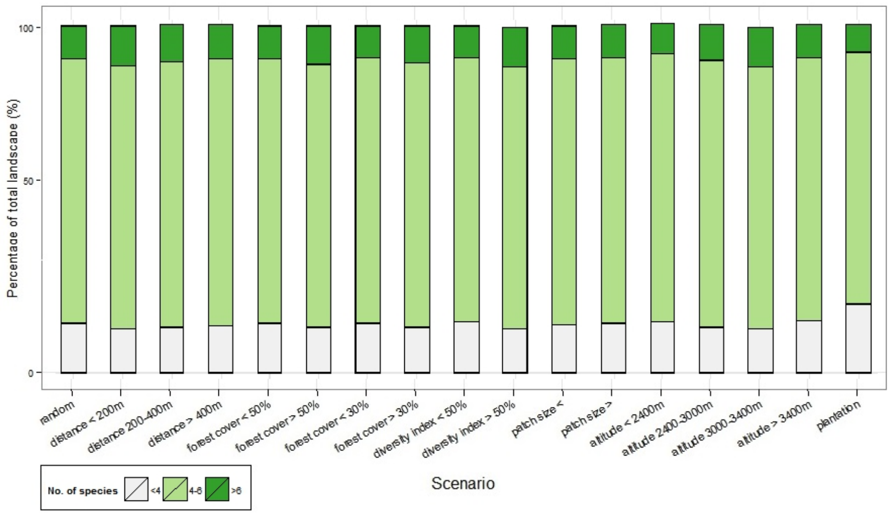

Figure 2.2: Percentage of the total landscape containing a number of woody species for all scenarios after 55 years of simulation. Location and type of reforestation affected the distribution of species richness at landscape scale

2.4.2 Species restoration, proximity and cover percentage of remnant forest patches

Consistent with hypotheses 1 and 2, we found significantly higher species richness for regenerating fallows located within 400 m from forest remnants and surrounded by more than 30 per cent cover of forest remnants compared to random reforestation ($P < 0.05$, two-sided Wilcoxon-test; Fig. 2.3). For these scenarios, α-diversity values were high but β-diversity values were low and decreasing, indicating high mean species richness and a small differentiation between species occurring in fallows (Fig. 2.5). When forest cover dropped below 30 per cent, species richness recovery in natural regenerating forests was significantly lower ($P < 0.05$, two-sided Wilcoxon-test; Fig. 2.3) and more natural regenerating forests remained uncolonized (Fig. 2.4).

Simulated species composition of regenerating fallows resembled natural forests more when occurring in the proximity to natural forest remnants and in areas with a high percentage of remnant forest cover in the surrounding. In these scenarios, montane forest species, such as *A. bogotensis*, *C. multiflora*, *H. luteynii*, and *P. glomeratus*, were present in a larger number of regenerating fallows compared to other scenarios (Fig. 2.6a, d, f, and h).

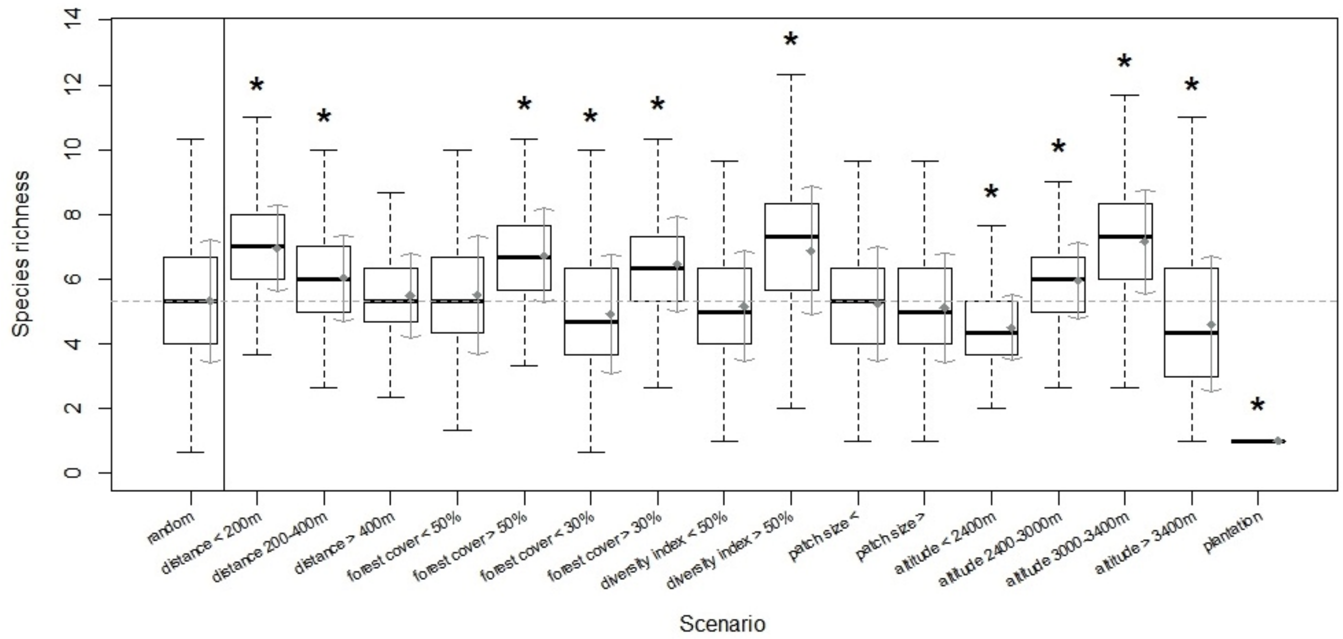

Figure 2.3: Boxplot showing the median, 25-75% quartiles, and non-outlier ranges (in black), and the average and standard deviations (in grey) of simulated species richness after 55 yrs in reforesting fallows over three runs for each scenario. The horizontal dashed line indicates average species richness for the random scenario. Compared to the random scenario, average species richness was significantly higher for the distance <200m, distance 200-400m, forest cover >50%, forest cover >30%, diversity index >50%, altitude 2,400-3,000m, and altitude 3,000-3,400m scenarios and significantly lower for the forest cover <30%, altitude <2,400m, altitude >3,400m, and plantation scenarios. Significant differences are indicated by asterisks ($P < 0.05$, two-sided Wilcoxon-test)

2.4.3 Species restoration and compositional heterogeneity

Consistent with hypothesis 3, we found significantly higher species richness in regenerating fallows with high compositional heterogeneity of surrounding natural cover types compared to random reforestation ($P < 0.05$, two-sided Wilcoxon-test; Fig. 2.3). Regenerating fallows located in compositional heterogeneous surroundings were increasingly dissimilar in species over time (Fig. 2.5b), indicating enhanced biodiversity at the landscape level (Fig. 2.2). These regenerating fallows were more often occupied by species typical of montane (Fig. 2.6a, d and h), shrub (Fig. 2.6c), and páramo (Fig. 2.6e) vegetation.

2.4.4 Species restoration and patch size

Contrary to hypothesis 4, our simulations showed no significant difference between the average species richness of regenerating forest of either small or large patch sizes compared to random reforestation (Fig. 2.3). Patch size scenarios also did not differ in terms of species richness at the landscape scale (Fig. 2.2) or recolonization dynamics over time (Fig. A.3).

2.4.5 Species restoration over time

Species establishment in abandoned fallows was fast during all natural regeneration scenarios (Fig. 2.4). Consistent with hypothesis 5, most species rapidly increased their cover percentage in fallows after abandonment in all scenarios, except for the plantation scenario (Fig. 2.6). The presence of short-lived, shade-intolerant shrub species declined several decades after colonization (Fig. 2.6c, and e), due to increasing levels of simulated shade during recolonization. This decline resulted in a decrease in species richness over time, shown by diminishing α-diversity values for all scenarios (Fig. 2.5a).

2.4.6 Species restoration and altitude

Consistent with hypothesis 6, species richness in regenerating fallows was significantly lower at high altitudes (>3,400 masl) and significantly higher at intermediate altitudes (between 2,400-3,400 masl) compared to random reforestation ($P < 0.05$, two-sided Wilcoxon-test; Fig. 2.3). Contrary to hypothesis 6, however, we found significant lower species richness for regenerating fallows located below 2,400 masl compared to random reforestation ($P < 0.05$, two-sided Wilcoxon-test; Fig. 2.3), because the lower basin of the catchment has been almost completely cleared of natural forest remnants. Regenerating fallows located between 3,000 and 3,400 masl showed the highest α-diversity val-

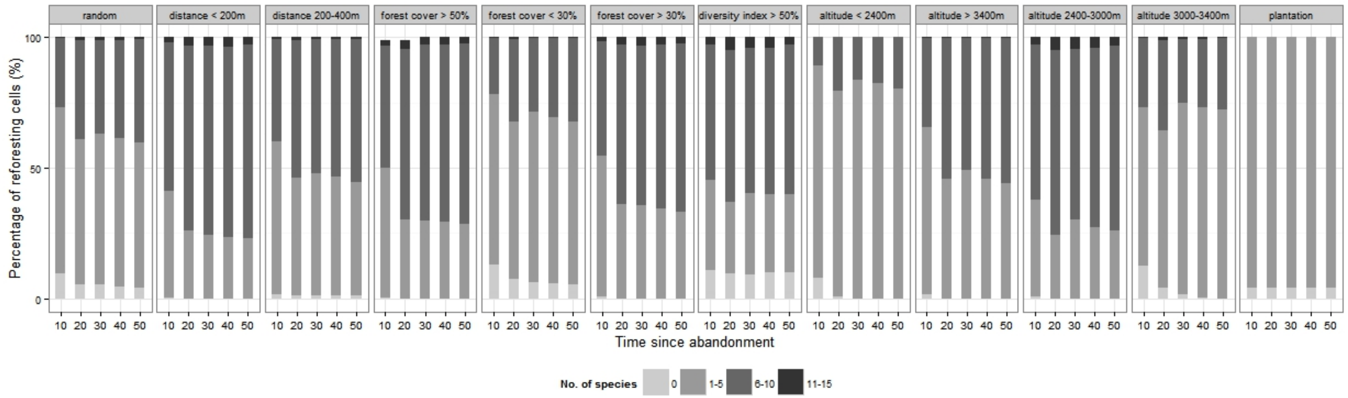

Figure 2.4: The percentage of reforesting fallows containing a number of woody species over time since abandonment for significant scenarios. Re-colonization of abandoned agricultural lands was dynamic during simulations. In particular, the distance <200m, forest cover >50%, forest cover >30%, diversity index >50%, and altitude 3,000-3,400m scenarios indicated a relatively large area that contained a large number of species

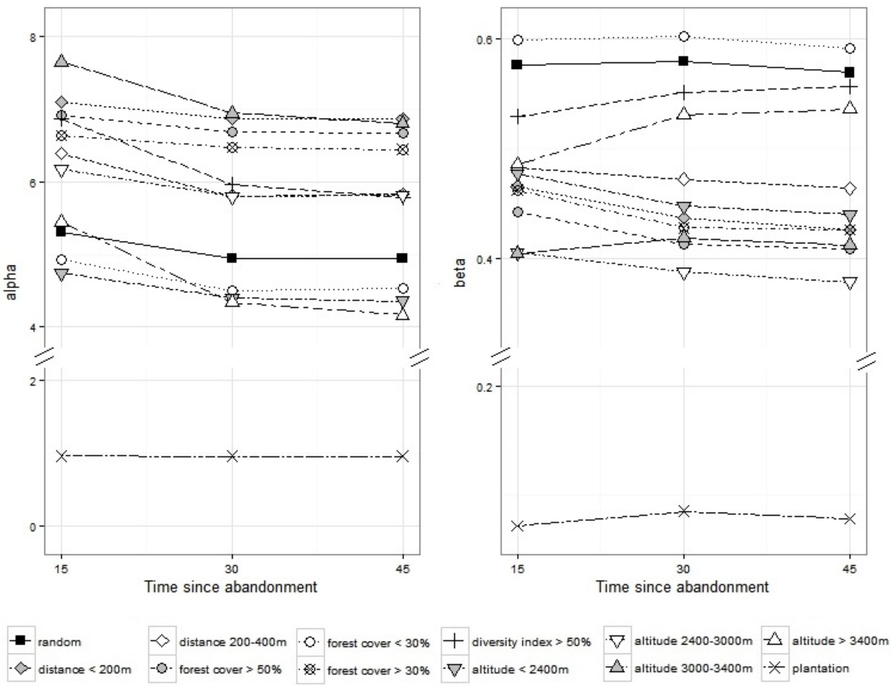

Figure 2.5: The development of the simulated (a) mean species diversity per cell (i.e., α-diversity) and (b) differentiation among these sites (i.e., β-diversity) over time for significant scenarios. α-Diversity decreased over time for all scenarios, whereas β-diversity increased for diversity index >50%, altitude 3,000-3,400m, and altitude >3,400m scenarios, whereas it decreased for the remaining scenarios. Note the discontinued y-axis

ues (Fig. 2.5a). Species associated with lower elevations, were more abundant in naturally regenerated forests below 2,400 m Fig. 2.6c, and g), while species that were able to establish and grow at high altitudes increased their cover percentage at intermediate altitudes (between 3,000-3,400 masl) (Fig. 2.6e, f, and h).

2.4.7 Species restoration in planted forests

Consistent with hypothesis 7, we found significantly lower species richness for planted forest on previously cultivated land compared to random reforestation (P <0.05, two-sided Wilcoxon-test; Fig. 2.3). Planted forests using exotic pine species prevented the establishment of all other species Fig. 2.4,

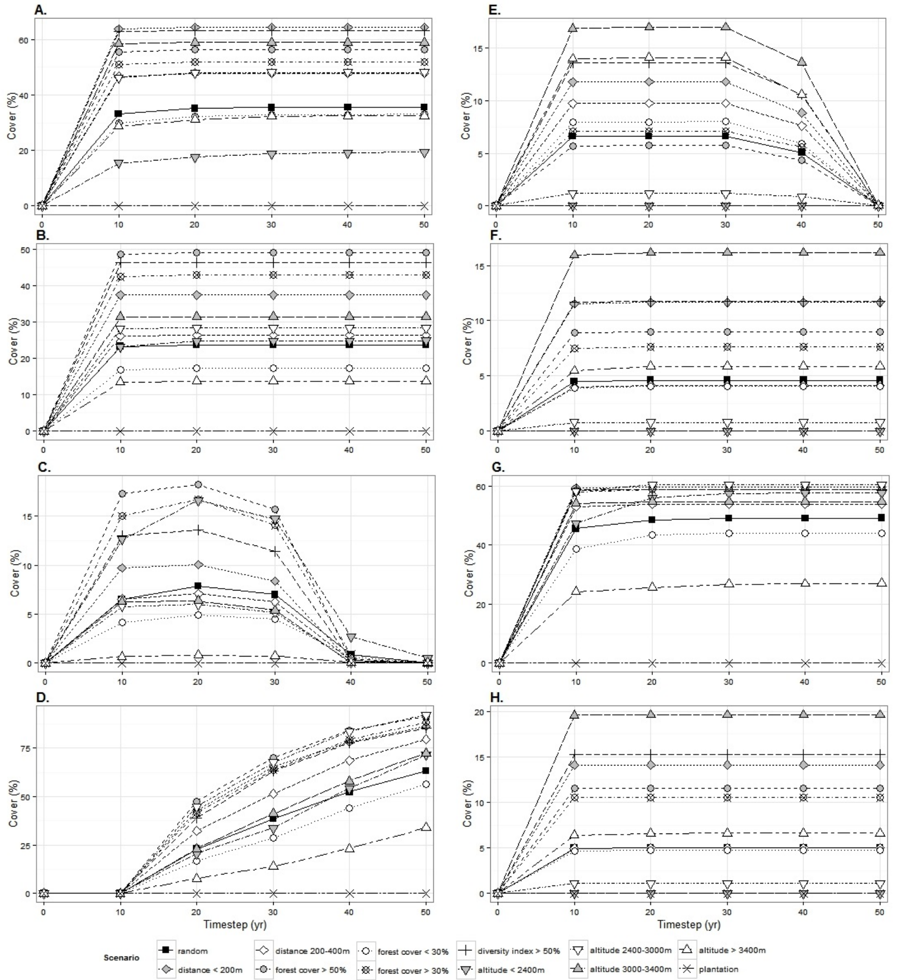

Figure 2.6: The cover percentage of natural regenerated fallows occupied by each selected species over time under significant scenarios. (a) *Aegiphila bogotensis*; (b) *Baccharis latifolia*; (c) *Boehmeria aspera*; (d) *Clusia multiflora*; (e) *Gynoxys* spp.; (f) *Hedyosmum luteynii*; (g) *Miconia* spp.; and (h) *Podocarpus glomeratus*. Shapes correspond to scenarios. Values presented are means for three repeated simulations for reforestation onset in 2010. Note that the y-axis values are not fixed

Fig. 2.5a, and Fig. 2.6), leading to a diminished overall species richness at the landscape scale (Fig. 2.2).

2.5 Discussion

2.5.1 Species restoration, proximity and cover percentage of remnant forest patches

Our results demonstrate that remnant vegetation was an important seed source that determined the recovery of both species richness and composition during natural regeneration of abandoned fallows. Consistent with our results, other studies indicate a smaller numbers of individuals and species in abandoned fields under reforestation located further away from forests (Mesquita et al., 2001; Robiglio and Sinclair, 2011). In the tropical mountain rainforest of Southern Ecuador, the total number of species found in natural regenerating forest decreased from 47 to 31 with increasing distance from the forest edge (Günter et al., 2007). In a review of more than 30 years of ecological restoration in Brazil, Rodrigues et al. (2009) have stated that the distance to the forest edge reduces species richness and diversity long after the onset of forest regeneration. Our results support the idea that below a 20-30 per cent threshold of natural surrounding habitat, isolation becomes an important factor determining species richness (Andrén, 1994; Fahrig, 1998, 2003).

2.5.2 Species restoration and compositional heterogeneity

Compositional heterogeneity increased the recovery of species richness and composition during our simulations, due to the accumulation of species associated with different natural cover types. Similarly, Kumar et al. (2006) found that compositional heterogeneity (i.e., Simpson's diversity index) was positively correlated with native and non-native plant species richness for 180 plots in the Rocky Mountains. This effect was strongest at an intermediate spatial extent (240 m) (Kumar et al., 2006), which is comparable to the spatial scale of 200 m at which we found a positive effect of compositional heterogeneity on species richness. An increase in patch density (i.e., increased compositional heterogeneity), which resulted from low levels of land use, enhanced plant diversity in a tropical forest landscape in Mexico (Hernández-Stefanoni et al., 2011). In a modeling approach, it was found that plant diversity was highest in heterogeneous landscapes with more small patches and low aggregation (Steiner and Kohler, 2003).

Besides compositional heterogeneity, it has also been suggested that species diversity is positively correlated with an increasing complexity of patch shapes of different cover types (i.e., configurational heterogeneity) (Fahrig et al., 2011). In practice, these two dimensions are difficult to control independently (Fahrig et al., 2011). This study only focused on compositional heterogeneity. The nature of landscape heterogeneity-biodiversity relationships still needs further exploration. The effect of compositional heterogeneity on biodiversity likely depends on contextual factors, such as historical landscape patterns, time frame, and the nature of reforestation.

2.5.3 Species restoration and patch size

Our finding that patch size of regenerating fallows did not affect species richness or composition contradicts other studies. Several researchers have suggested that smaller patches support smaller populations, which increases the probability of extinction (Ellstrand and Elam, 1993; Fischer and Lindenmayer, 2007). Harrison (1999) found a higher local endemic plant diversity (α-diversity) and among-site differentiation (β-diversity) on small patches when comparing communities from patchy and continuous habitats. In Yucatan Peninsula in Mexico, larger regenerating forest patches with fewer edges had higher woody species diversity compared to small regenerating forest patches of irregular shapes (Hernández-Stefanoni et al., 2011). Arroyo-Rodríguez et al. (2009) found that patch size explained density-area relationships of plants in highly fragmented landscapes. However, richness of native and non-native plants did not increase with size of planted revegetation established on agricultural land in Gippsland southeastern Australia (Munro et al., 2009), which might be the result of insufficient maturity to display significant species density-area relationships.

Fragmentation in human-modified landscapes leads to the decrease in mean patch size and thus the creation of more forest edges (Fischer and Lindenmayer, 2007). Several studies have suggested that light-demanding generalist, exotic, and invasive species may benefit from edge environments in tropical forest remnants (Harper et al., 2005; Tabarelli et al., 2010). Natural regenerating forests that persist as isolated small patches were recognized as edge-affected habitats (Tabarelli et al., 2012b) and represented a suitable habitat for a subset of species only (Barlow et al., 2007). The absence of species-area relationships in the LANDIS-II model could explain the lack of patch size effect during model simulations. Nonetheless, empirical work demonstrates the importance of patch size for species conservation (Arroyo-Rodríguez et al., 2009; Hernández-Stefanoni et al., 2011).

2.5.4 Species restoration over time

As found by other studies, age structure of regenerating fallows was an important determinant for biodiversity recovery (Guariguata and Ostertag, 2001; Barlow et al., 2007; Dent and Wright, 2009; Norden et al., 2009). In a quantitative review of the value of natural regenerating forests across the tropic, it was found that similarity to old-growth forests increased rapidly with age (Dent and Wright, 2009). Interestingly, we found a decrease in species richness over time for all scenarios, due to the decline of short-lived, shade-intolerant pioneer species several years after colonization. This process indicates a shift in successional stages of reforestation, with shade-intolerant, short-lived pioneers being replaced by long-lived pioneers and generalist species (Wirth et al., 2009). Long-term species time-series data are rare for tropical forest sites (Gardner et al., 2009). As a consequence, recolonization trajectories over longer time periods are not well understood. In temperate forests, however, the decline in species richness as forest succession shifts to later stages is well recorded (Horn, 1974; Vetaas, 1997; Bonet and Pausas, 2004; Amici et al., 2013).

2.5.5 Species restoration and altitude

In the Andes, altitude is an important factor that affects plant community composition (Young, 1993; Lauer and Rafiqpoor, 2000), which supports our findings. Variations in species composition related to altitude were most obvious in disturbed forest fragments and played a role in recolonization of regenerating pastures (MacLaren et al., 2014). It has been suggested that the creation of more open areas at high altitudes in the Andes, due to agriculture and pasture expansion, benefits the establishment of woody species associated with tree line, alpine shrub land, or páramo (MacLaren et al., 2014; Sarmiento, 2002). Consistent with our findings, regenerating pastures in highland areas may prove beneficial for woody species associated with high altitudes in lower regions previously covered with mature forest. This phenomenon is thought to be widespread in the Andes (Sarmiento, 2002). Nevertheless, the establishment of small pioneer shrub or tree species, such as *B. latifolia*, promoted the regrowth of other sensitive species (Posada et al., 2000). At very high altitudes (>3,400 masl), growth of woody species lagged behind due to lower temperatures causing trees to grow slowly (Rapp et al., 2012). Natural regeneration of abandoned fallows is therefore limited at high altitudes.

2.5.6 Species restoration in planted forests

Our simulations showed that planted pine forests provided no benefits for biodiversity recovery. Nonetheless, almost all forest plantation activities in

Ecuador have involved the planting of exotic species, such as *Pinus* and *Eucalyptus* spp. In regions with a high endemic biodiversity, such as the highlands of Ecuador, exotic plantations posed a high risk to native vegetation (Bremer and Farley, 2010) and altered hydrological properties of soils (Farley et al., 2004; Molina et al., 2012). Plantations may provide habitats for certain species and increase forest connectivity if managed with long rotations and if located around natural forest fragments (Meyfroidt and Lambin, 2011).

Lindenmayer et al. (2009) found that compositional heterogeneity at the landscape scale partly determined the level of biodiversity supported by plantation landscapes. Bird species richness was significantly reduced by 4 to 9 species in patches of native eucalypt forest where surrounding stands of exotic *Pinus radiata* were clear-felled compared to patches where surrounding pine stands remained unlogged (Lindenmayer et al., 2009). However, spatial heterogeneity was not accounted for in our plantation scenario and may affect species recovery at the landscape scale.

2.5.7 Limitations

Forest landscape models explicitly represent spatial processes such as seed dispersal and disturbance spread. The value of forest models does not lie in the exact prediction of future conditions, but in their potential to explore the consequences of explicitly stated assumptions. They allow for considerable experimentation with different landscape patterns and human processes that would not be possible in the field given time and money constraints. Nevertheless, the dependence on relationships estimated in the past to predict future dynamics is a limitation to all forest models (Gustafson, 2013), which makes it difficult to perform rigorous model validation. In the tropics, there is a lack of long-term data on the ecological behavior of forests, which hinders model parameterization. The greatest uncertainties for model parameterization in this study were: (i) the dispersal ability of modeled species; (ii) establishment probabilities of tree species across the landscape; and (iii) initial forest composition and age structure. We addressed these uncertainties by including high resolution remote sensing imagery, MaxEnt models, scientific literature, and consultations of local experts.

Some other factors that potentially affect future species distribution were not taken into account, such as climate change, differences in microhabitat, and seed dispersal. Andean plant species are predicted to shift their distributions to higher altitudes in response to changes in temperature and humidity (Feeley and Silman, 2010). Microhabitat differences in soil fertility can also strongly affect competition and growth during re-colonization (Guariguata and Ostertag, 2001). Additionally, animal movement is influenced by landscape patterns and affects seed rain (Jesus et al., 2012). We neglected the presence of seed banks from which seeds of mostly shrubs and short-lived pioneer trees may establish at the time of abandonment during natural regeneration (Holl, 2007). Finally, in the Ecuadorian Andes, many farmers leave remnant trees and shrubs on agricultural lands and, in some cases, allow vegetation to regenerate (Middendorp, *personal observation*). Our assumption that cultivated lands were bare may have underestimated the natural regeneration potential during model simulations.

2.6 Conclusion and policy recommendations

Forest regeneration might prove fundamental to future conservation initiatives, but the value of regenerating forests is context dependent. Conservation policies have started to recognize the role of forest regrowth at the landscape level (Harvey et al., 2008). Reducing deforestation and increasing forest cover became a national priority under the Ecuadorian National Development Plan, known as the Plan for Good Living 2009-2013. For example, reforestation plans will be incorporated in the Socio Bosque program, in which the Ministry of Environment (MAE) offers yearly monetary payments to private and communal landholders in return for maintaining forest cover (de Koning et al., 2011). A better integration of restoration ecology and landscape ecology is needed for mainstreaming successful programs (Metzger and Brancalio, 2013). Our research provides specific guidelines for the design of reforestation projects that enhance biodiversity restoration in human-modified landscapes. Spatial targeting of reforestation should take into account the presence of natural remnant forest patches, the spatial pattern of the landscape, altitudinal gradients, and the type of regenerating forest, as summarized below.

We emphasize that the conservation and regeneration of natural remnant forest patches is crucial. A considerable fraction of human-modified landscapes no longer serves as a source of seed dispersal due to habitat loss. This may impose a limitation for natural forest regeneration (Zimmerman et al., 2000). The proximity and cover percentage of surrounding remnant forest patches is an important factor that determines the success of natural regeneration after abandonment. Subsequent management and conservation of regenerating forests is essential, because regenerating forests need to reach a certain age to become a source of propagules (van Breugel et al., 2013).

Additionally, we suggest that diversity and spatial arrangement of habitat patches determines the successful recolonization of the species that inhabit different environmental conditions. Based on our research, we recommend maintaining compositional heterogeneity as an important guideline in conservation management. Even though our results did not show a significant effect of patch size on species restoration, we recommend stimulating the regeneration of large and aggregated patches of natural regenerating forests. Habitat-specialists that need large areas of contiguous habitat for their survival may disappear from landscapes with high fragmentation and compositional heterogeneity (Fahrig et al., 2011). The limitations of conservation efforts to small forest fragments only risks biotic homogenization of forest-requiring species, as shown in the northern parts of the Atlantic Forest region in Brazil (Lôbo et al., 2011).

Natural regeneration following fallow abandonment often occurs on marginal lands (Crk et al., 2009), such as upland areas (Asner et al., 2003). Our results suggest, however, that reforestation in upland areas might not be enough for species recovery. We recommend that natural regeneration be encouraged at various altitudes, allowing a variety of species associated with different elevation ranges to recover at the landscape scale. Finally, we found that planted forests using exotic pine species provided no benefits for species recovery. Active planting of native species might stimulate restoration of native biodiversity, especially in regions were natural land covers are strongly diminished.

CHAPTER 3

RESTORATION THROUGH COCOA AGROFORESTRY MANAGEMENT: ENHANCING CARBON SEQUESTRATION AND FUNCTIONAL DIVERSITY

This chapter is based on the following paper submitted to *Landscape Ecology*: R.S. Middendorp, V. Vanacker, E.F. Lambin. Shaded agroforestry management can enhance carbon sequestration and functional diversity in cocoa production landscapes.

Abstract

Conversion of biodiversity-friendly shaded agroforests to unshaded monocultures endangers the resilience of tropical landscapes. The landscape scale impact of alternative shade management strategies has rarely been assessed. This study explored plantation- and landscape-level impacts of different shade management strategies on aboveground biomass, functional species composition, and economic potential of cocoa production in northern Ecuador. We simulated several cocoa shade management scenarios, using the dynamic forest model LANDIS-II: (i) 'baseline' projections representing the current mosaic of traditional agroforests, planted agroforests, and unshaded monoculture plantations; (ii) 'traditional' agroforestry shaded by native fruit and timber trees; (iii) 'planted' agroforests shaded by planted fruit trees; and (iv) 'monoculture' unshaded plantations. The impacts of setting aside 20, 30, and 40% of cocoa plantations for natural regeneration was tested for the monoculture scenario (respectively iv-a, iv-b, and iv-c). Shaded agroforests stored up to seven per cent more aboveground biomass and had higher abundances of rare functional groups compared to monocultures after 50 years of simulation. Shaded plantations and land set aside for natural regeneration reduced forest fragmentation at the landscape-level. The estimated yield gap for monoculture and shaded plantations could not be compensated by additional revenues for carbon storage at a current carbon market price of US$5 per hectare per ton CO_2. Improving payment-for-ecosystem services and certification schemes are needed to incentivize smallholders to maintain substantial non-cocoa tree cover that may provide an environmental-friendly way to improve economic potential and food security for smallholders, while supporting biomass and functional species composition at the landscape level.

3.1 Introduction

Forest conversion and agricultural intensification are important causes of loss of biodiversity and associated ecosystem services (Foley et al., 2005). To adapt to environmental and climate changes, resilient agricultural landscapes are needed to safeguard ecosystem services and food security. It is therefore a paradox that agricultural intensification reduces the response capacity of land-use systems to environmental stresses (Tscharntke et al., 2011). Shaded agroforestry systems are widespread across the tropics. Many scholars have identified shaded agroforests as a biodiversity-friendly way to produce food and guarantee economic returns, while sustaining ecosystem services (Schroth and Harvey, 2007; Perfecto et al., 2007; Bhagwat et al., 2008; Steffan-Dewenter et al., 2007; Tscharntke et al., 2011, 2015; Vaast and Somarriba, 2014).

Cocoa is commonly grown under shade from non-cocoa trees (Rice and Greenberg, 2000) and covers about 10 million ha of land globally (FAO, 2014). Compared to other tropical agricultural systems, shaded cocoa agroforests can support high levels of biodiversity (Asase and Tetteh, 2016; De Beenhouwer et al., 2013) and play a role in the carbon cycle by storing carbon in above- and belowground biomass (Kessler et al., 2012; Obeng and Aguilar, 2015; Somarriba et al., 2013; Albrecht and Kandji, 2003). Trees in tropical agricultural landscapes determine key landscape characteristics and ecosystem services' delivery (Clough et al., 2009a; Tscharntke et al., 2011; Mendenhall et al., 2014).

Many farmers worldwide convert shaded agroforests to more intensively managed plantations by reducing the number of non-cocoa trees, in an attempt to increase short-term economic returns (Steffan-Dewenter et al., 2007; Clough et al., 2009b; Vaast and Somarriba, 2014). Cocoa cultivation has been estimated to cause 14-15 million hectares of tropical deforestation globally (Clough et al., 2011). Decreasing tree cover in tropical agricultural landscapes might affect landscape functioning far beyond the farm level. Some authors argue that agricultural intensification may free up other areas for nature conservation through land sparing, which may compensate for the loss of ecosystem services from intensification (Green et al., 2005; Phalan et al., 2011a,b). By contrast, wildlife-friendly farming integrates low-intensity agricultural production with natural landscape elements through land sharing, resulting in a patchy landscape (Milder et al., 2014; Tscharntke et al., 2012a). Some scholars have suggested that agroforestry farming methods may thus enhance the matrix quality of human-dominated agricultural landscapes (Perfecto et al., 2010, 2009; Fischer et al., 2014; Chappell and LaValle, 2011). While several studies have quantified biodiversity and biomass changes along intensification gradients in shaded agroforestry systems (Steffan-Dewenter et al., 2007; De Beenhouwer et al., 2013; Obeng and Aguilar, 2015; Vaast and Somarriba, 2014), the role of shade management and potential impact of land sparing at the landscape-level has rarely been assessed.

The objective of this study was to understand the landscape-level impacts of different cocoa shade management strategies on aboveground biomass (AGB), biodiversity, and the economic potential of cocoa plantations. We modeled seven alternative management scenarios representative of cocoa farmers in northern Ecuador, using the well-established ecological model LANDIS-II. Four scenarios explored the impact of alternative shade management ranging in type and percentage of non-coca trees on cocoa plantations and three scenarios assessed how much cocoa agroforestry land should be set aside for natural regeneration in a scenario of conversion to monoculture to reach levels of aboveground biomass at the landscape scale comparable to our baseline projections. Finally, to explore the impact of shade management and land sparing on economic potential, we estimated landscape-level revenues from cocoa production and payments for additional carbon stored in carbon markets. This study contributes to the literature on shaded agroforestry systems by quantifying landscape scale impacts of cocoa agroforestry systems. The results may inform the design of payment-for-ecosystem services and certification schemes to promote productive and biodiversity-friendly shade management regimes.

3.2 Methods

3.2.1 Study region

We conducted this study in the northern province of Esmeraldas, where we focused on the five main cocoa producing western cantons (8,470 km^2; 54 per cent of the surface area of Esmeraldas; Fig. 3.1). Esmeraldas is amongst the poorest provinces of Ecuador, with a high percentage of smallholders owning on average 5 hectare of land. The main commodities produced in this region are cocoa beans, fresh fruits (e.g., bananas, oranges, maracuyas) and, increasingly, palm oil. An agricultural survey conducted by the Ecuadorian Ministry of Agriculture in 2013 estimated that about 52,884 hectare in Esmeraldas was planted with cocoa, producing a total of 13,343 metric tons of dry cocoa beans (i.e., average yield of 252 kg per hectare) (ESPAC, 2014).

Chocolatiers have shown particular interest in this region for the high-quality fine flavor cocoa beans, known as 'Nacional', traditionally grown in shaded agroforestry systems. Most of the traditional cocoa plantations in Ecuador have a low productivity due to low-yielding planting material, aged cocoa trees and high vulnerability to diseases, such as witches' broom and moniliasis (Amores et al., 2011). Many smallholders have intensified production systems by reducing shade levels and replacing traditional Nacional varieties with clonal varieties, mainly CCN-51 (Hernández et al., 2014). CCN-51 trees are more disease-resistant and productive, less susceptible to sun damage, and are often planted densely without associated non-cocoa trees (Bentley et al., 2004). Production potential of CCN-51 trees was found to be between 53 and 275 per cent higher than Nacional-type trees in controlled production experiments (Amores et al., 2011; Boza et al., 2014). Monoculture plantations may provide increased yield, but require more agrochemical inputs and are more susceptible to droughts, soil erosion and degradation (Jacobi et al., 2014). On specialized markets, high-quality Nacional beans may receive up to 60 per cent above standard cocoa market price for their appreciated flavor. Nevertheless, a differentiated farm-gate price was absent for CCN-51 and Nacional beans in Esmeraldas for 2014, respectively US$1.93 and US$2.02 per kilogram (SINAGAP, 2015; PRAGMATICA, 2016).

We identified three cocoa plantation types in the study region based on shade level and type, following the classification proposed by Rice and Greenberg (2000): (i) cocoa with traditional shade, planted under thinned natural forest (e.g., *Aegihila alba, Guazuma ulmifolia, Schizolobium parahyba, Cecropia* spp.) with less than 20 per cent planted fruit, legume, or timber trees; (ii) cocoa with more than 20 per cent planted shade, such as leguminous trees (e.g., *Inga* spp., *Erythrina* spp.), fruit trees (e.g., *Citrus* spp., *Carica papaya, Persea americana, Mangifera indica*) and timber trees (e.g., *Cordia alliodora, Cedrela odorata, Handroanthus* spp.); and (iii) monoculture cocoa plantations without shade.

3.2.2 LANDIS-II parameterization

To simulate woody species establishment, growth and mortality, we used the spatially explicit forest landscape model LANDIS-II (Scheller et al., 2007). This well-established model simulates forest dynamics over large spatial scales by incorporating ecological processes, such as succession, disturbance, and seed dispersal. LANDIS-II has been applied in Latin America to support conservation planning in Chile (Newton et al., 2011), to assess the potential for forest restoration in Mexico (Cantarello et al., 2011) and Ecuador (Chapter 2), and to predict the spatial extent of forest restoration (Birch et al., 2010). The model can deal with multiple disturbances, such as timber harvest, forest conversion (Thompson et al., 2011), and land use (Thompson et al., 2016). LANDIS-II tracks age cohorts for each simulated species, ignoring individual trees, to achieve computational tractability. We implemented the Biomass Succession extension of the LANDIS-II model (v. 3.2) tracking live aboveground biomass (AGB) estimates for species cohorts influenced by growth, senescence, and management disturbance on all active landscape cells (i.e., cells for which for-

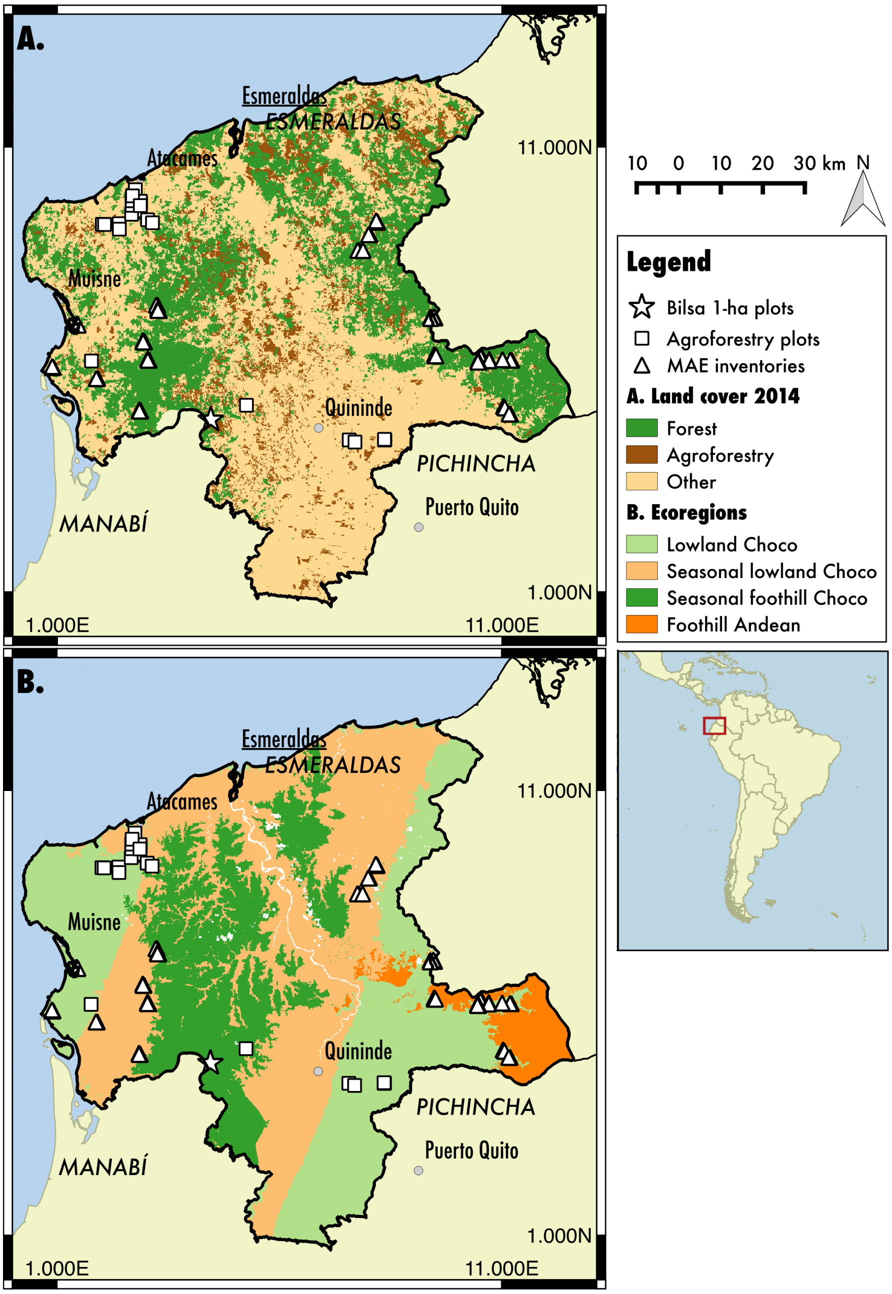

Figure 3.1: Study region in northern Ecuador comprised of the five western cantons of the Esmeraldas province (Muisne, Atacames, Esmeraldas, Rio Verde and Quininde). (a) 2014 Land cover classification created by the MAE (2014a); cover types relevant for simulations were forest (green), cacao agroforests (brown) and others (yellow). (b) Ecoregions were considered to be homogeneous in climatic and soil conditions (see Table 3.2 for details). Cacao agroforestry plots (squares), forest inventory plots collected by the Ecuador Ministry of Environment (MAE) (triangles) and 1-ha multi-census forest inventories in the Bilsa Biological station (stars) are indicated

est ecological processes took place, classified as forest and agroforests) for each 10-year time step.

3.2.2.1 Land cover map

The Ecuadorian Ministry of Environment (MAE) in collaboration with the Food and Agriculture Organization of the United Nations (FAO) created a land cover classification at 30 m spatial resolution, based on unsupervised classification with field validation for a LANDSAT and Aster mosaic consisting of satellite images from 2014 (Fig. 3.1a) (MAE, 2014a). The 2014 classification was validated using field observations for 500-1000 random sample points in each LANDSAT image, resulting in an average accuracy of 79 per cent and a Kappa coefficient of 76 per cent. LANDIS-II simulations were restricted to the five main cocoa producing western cantons and areas classified as natural forest remnants (about 31.4 per cent or 2,646 km^2) and cocoa agroforestry (about 10.5 per cent or 887 km^2), whereas remaining areas classified as agriculture, pastures, and build-up (about 58.1 per cent or 4,937 km^2) were ignored for simplification and lack of field data, resulting in an underestimation of total landscape AGB and functional group diversity.

3.2.2.2 Field data

We assessed the woody plant richness and composition of cocoa plantations in 43 plots of 20 x 50 m (0.1 ha) (Fig. 3.1a). Each individual taller than 2 m was identified to species level and the diameter at breast height (dbh) and height was recorded. To determine the woody plant richness and composition of natural remnant forests, we obtained forest inventory data collected by the Ecuadorian Ministry of Environment (MAE) in 172 patches of natural forests of 60 x 60 m (0.12 ha) located throughout our study region. The dbh, height, and genera of each live woody individual taller than 20 cm dbh was available for each plot. For both datasets, AGB estimates for all collected individuals were calculated with the Chave et al. (2005) allometric equation for moist forests, using the dbh and height measurements, as well as species-specific estimates of wood density for South-America from the Global Wood Density Database (Chave et al., 2009; Zanne et al., 2009).

To initialize dynamic aboveground net primary productivity (ANPP) rates of functional groups, we used growth data from three 1-ha multi-census forest inventory plots at the Bilsa Biological Station located in the southernmost part of Esmeraldas (00°21'N 79°44'W) (Clark et al., 2006). We derived ANPP estimates for each individual based on diameter increments for surviving trees only. Total ANPP estimates for these plots ranged from 772 to 1391 g biomass m^{-2} yr^{-1}, which fall in the range of other reported values (280-3010 g m^2 yr^1

(Clark et al., 2001); 2120 g m^{-2} yr^{-1} (Chave et al., 2010); 2470 g m^{-2} yr^{-1} (Keeling and Phillips, 2007)). ANPP estimates for cocoa trees were extracted from a study of 229 permanent sample plots in cocoa agroforestry systems in five Central American countries (Somarriba et al., 2013) (i.e., 360 g AGB m^{-2} yr^{-1}). Currently, data on ANPP are limited for tropical forests, thus the presented ANPP values must be viewed as rough estimates.

3.2.2.3 Initial composition and distribution of functional groups

We selected the 20 most dominant genera from the MAE forest inventories and clustered these into six functional groups based on wood density estimates from the Global Wood Density Database (Chave et al., 2009; Zanne et al., 2009) (Table 3.1). In tropical trees, wood density is a key functional trait positively correlated with competition ability, survival rate and shade tolerance, and negatively associated with growth rate (Kunstler et al., 2016; Poorter et al., 2010a,b). Abundant palm genera were omitted (i.e., *Wettinia* and *Iriatea*), as growing characteristics of monocots are not well represented in LANDIS-II. The initial distribution of functional groups across the landscape was mapped by combining the land cover map with the field data. MAE forest inventories were randomly distributed over cells classified as forests, whereas cocoa plantation plots were categorized following the three plantation types described in section 2.1 and then randomly distributed over cells classified as agroforestry.

Age estimates for each individual were made by dividing the maximum measured dbh from the MAE forest inventories with the maximum ANPP estimated from the Bilsa forest inventory plots for each functional group (Lieberman et al., 1985). Any error in the estimation of initial tree ages affects the initial distribution only, because LANDIS-II is independent of age-growth relationships. Longevity for each genus was determined based on the maximum observed dbh found in the MAE inventories and the maximum ANPP for that genera found in the Bilsa plots. Estimates were checked for consistency against maximum reported values from the literature (Table 3.1).

3.2.2.4 Species establishment probabilities

Four ecoregions were delineated by overlaying a digital elevation model with an annual average rainfall map (Table 3.2, Fig. 3.1b). Climatic and establishment conditions were assumed to be homogeneous within each ecoregion during the period of simulations. The maximum observed AGB in the MAE forest inventories was assigned for each functional group and for each ecoregion. Maximum AGB estimates for cocoa trees were extracted from Somarriba et al. (2013) (i.e., 1797 g AGB m^{-2}). Functional group establishment probabilities were derived from MaxEnt models (Phillips et al., 2006) as the average

Table 3.1: Functional group (Fg) characteristics. Long. = longevity (years); Mat = sexual maturity (years); ShTol = shade tolerance; Disp = effective and maximum seed dispersal distance (m); WD = mean wood density (g m^{-3})

Name	Long[a]	Mat	ShTol	Disp effective	Disp max	WD[b]	Forest genera	Agroforest genera	Description
Fg 1	120	22	2	50	400	0.28	*Apeiba , Cecropia*	*Cecropia, Erythrina*	Fast-growing native pioneers
Fg 2	340	66	3	50	400	0.64	*Brosimum, Miconia*	*Dussia, Psidium*	Understory to canopy trees
Fg 3	480	95	4	400	1500	0.79	*Castilla, Pouteria*	*Cupania, Pouteria*	High wood density timber trees
Fg 4	580	115	3	400	1500	0.54	*Inga, Matisia*	*Carica, Citrus, Inga*	Fruit trees (mainly planted)
Fg 5	340	67	2	50	400	0.39	*Otoba, Pourouma*	*Annona, Pourouma*	Fruit trees (native)
Fg 6	290	56	3	50	400	0.48	*Trattinnickia, Virola*	*Cedrela, Cordia*	Medium wood density timber trees
Cocoa	80	4	5	1	1	-	-	*Theobroma*	Shade-tolerant understory shrubs

[a] (Korning and Balslev 1994; Laurance et al. 2004; Lieberman et al. 1985)
[b] Means from the Global Wood Density Database (DRYAD) for South-America (Chave et al. 2009; Zanne et al. 2009)

Table 3.2: Description of ecoregions used to define homogeneous areas. Mean altitude and precipitation were used to delineate ecoregions, whereas mean PET and NPP were used to scale ANPP values over different ecoregions

Ecoregion	Description	Mean altitude (masl)	Mean precipitation [a] (mm/yr)	Mean PET [b] (mm)	Mean NPP [c] (gC/m2/yr)
1	Evergreen lowland Choco forest	137	2784	1442	9549
2	Evergreen seasonal lowland Choco forest	114	1736	1432	9453
3	Evergreen seasonal foothill Choco forest	293	1468	1398	5548
4	Evergreen foothill and mountain Andean forest	545	3845	1470	8372

[a] Worldclim database: BIO12 (Hijmans et al. 2005)
[b] Global Potential Evapo-Transpiration (Trabucco and Zomer 2009)
[c] Mean Net Primary Productivity for Terra Modis 2000-2015 maps (Running et al. 2004; Zhao et al. 2005)

probability of occurrence for each functional group in each ecoregion. Input data for these models included species occurrence data from the MOBOT Tropicos®database (©2014 Missouri Botanical garden, USA) and bioclimatic variables extracted from the Worldclim database (Hijmans et al., 2005). Occurrence data were checked for coordinate accuracy and only field observations were used, omitting controlled experiment observations.

3.2.2.5 Model validation and sensitivity

LANDIS-II is a stochastic model and does not predict actual events. We performed a sensitivity analyses on six key parameters: maximum ANPP, maximum AGB, ANPP shape, mortality shape, establishment probability, shade tolerance, and species longevity. Continuous parameters were altered by increments of 10 per cent and categorical parameters were altered by increments of 1 unit, after which the impact on the total AGB estimate was assessed at the onset and end of baseline simulations (Scheller and Mladenoff, 2004; Thompson et al., 2011). To evaluate the parameterization of the model, we compared AGB estimates at simulation onset to the values observed in the MAE forest inventories, using Pearson's correlation coefficient and RMSE. All model results were averaged over five simulation runs. Increasing the number of replicates did not decrease total between-run variability (less than 2 to 3 per cent), thus five replicates were deemed sufficient, as is common practice in LANDIS-II applications (Mairota et al., 2014; Duveneck et al., 2014; Cantarello et al., 2014, 2011; Thompson et al., 2011).

3.2.3 Simulation experiment

3.2.3.1 Agroforestry management scenarios

Agroforestry management was modeled using the Biomass Harvest extension (v. 3.0) of the LANDIS-II model. We designed seven landscape management scenarios, which ranged in the level and type of shade maintained in cocoa plantations (Fig. 3.3; Table 3.3). Four scenarios explored the impact of alternative shade management ranging in type and percentage of non-coca trees on cocoa plantations: (i) 'baseline' resembling a persistence of the current patchy landscape with a mix of traditional, planted, and monoculture plantations; (ii) 'traditional' agroforestry shaded by native fruit and timber trees; (iii) 'planted' agroforests shaded by planted fruit trees; and (iv) 'monoculture' unshaded plantations. Three additional scenarios assessed how much cocoa agroforestry land should be set aside for natural regeneration in a scenario of conversion to monoculture systems to reach levels of aboveground biomass at the landscape scale comparable to the baseline scenario. We tested percentages of land set aside of 20, 30, and 40% in the least accessible plots (i.e., greatest distance to roads) located in areas with high potential for biodiversity conservation, as identified by Cuesta et al. (2017) (respectively scenario iv-a, iv-b, and iv-c) (Fig. 3.3; Table 3.3; Fig. B.1). Extensive remnant forest patches are present in these high potential areas, likely favoring natural regeneration in abandoned plantations by providing a seed source for plant dispersal. For each scenario, we analyzed changes in AGB and in functional species composition at the plantation- and landscape-level.

3.2.3.2 Economic potential estimations

To explore the impact of shade management and land sparing on economic potential, we estimated landscape-level revenues from cocoa production and payments for additional carbon stored compared to the baseline levels in carbon markets. First, we estimated the economic potential from cocoa bean production for each scenario at the plantation- and landscape-level based on the mean aboveground net primary productivity (i.e., ANPP) modeled for cocoa trees. As LANDIS-II does not simulate cocoa bean production directly, dry bean yield was estimated by assuming that 76 per cent of cocoa ANPP was partitioned to producing beans, as reported for traditional, planted, and

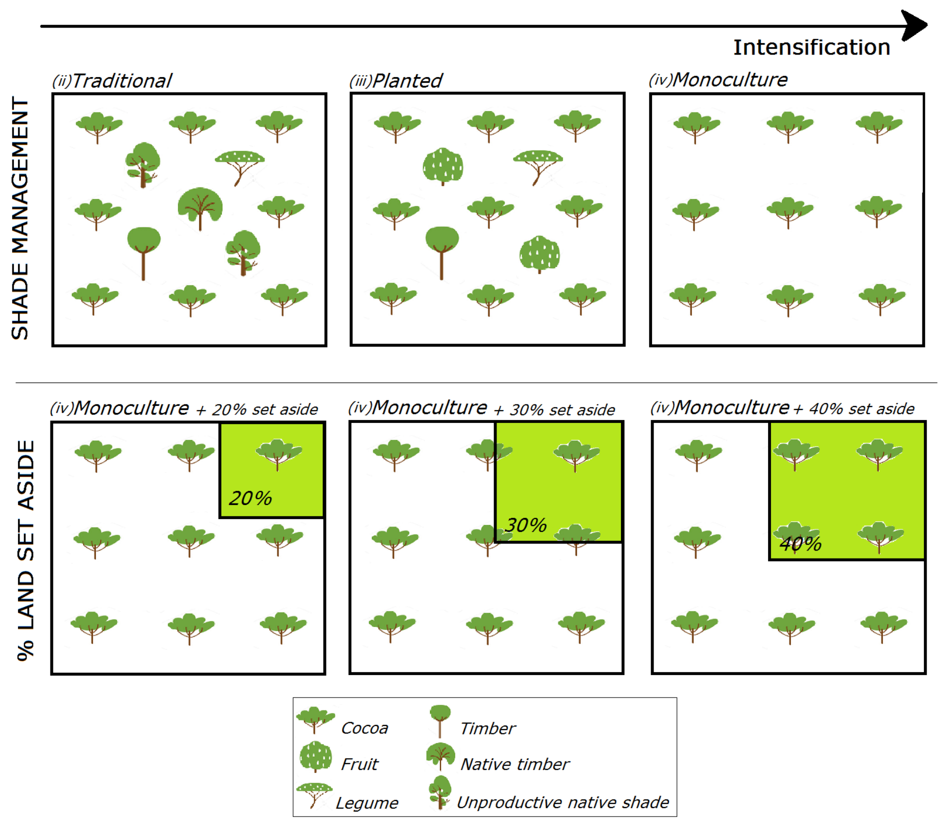

Figure 3.2: Schematic representation of the shade management scenarios tested in this study. Scenarios were grouped according to shade management and land set aside strategies along an intensification gradient. Land set aside scenarios (iv-a, iv-b, and iv-c) are indicated with resp. 20, 30, and 40% of agroforestry land cover left for natural regeneration after abandonment (lightgreen area). Different tree icons represent different types of functional groups represented in the LANDIS-II simulations

Table 3.3: Description of cocoa plantation management prescriptions for shade management scenarios tested. Productive shade included both fruit and timber tree species. Harvest management prescription represent percentage of aboveground biomass reduction

Pathway description	'Harvest' management prescription [a]
(i) *Baseline*: cocoa farm represent a spatial mix of three shade management types (traditional, planted, monoculture)	In traditional agroforests, remove 10% of all functional group age cohorts. In planted agroforests, remove 70% of functional group 1, 2, 5, and 6 age cohorts; remove 20% of functional group 3 and 4 age cohorts and plant new age cohorts for functional group 3 and 4. In monoculture plantations, remove all functional group age cohorts except cocoa and prevent future establishment of all functional groups.
(ii) *Traditional*: all cocoa farms convert to shade management with naturally regenerating native tree species	At scenario onset, remove 30% of functional group 3 and 4 on previously planted agroforests. Subsequently remove 10% of all functional group age cohorts on all cocoa plantations. Natural regeneration of native shade was allowed.
(iii) *Planted*: all cocoa farms convert to shade management with a planted mix of fruit, legume, and timber tree species	Remove 70% of functional group 1, 2, 5 and 6 age cohorts; remove 20% of functional group 3 and 4 age cohorts and plant new age cohorts for functional group 3 and 4.
(iv) *Monoculture*: all cocoa farms covert to full-sun management (i.e., without shade)	Remove all functional group age cohorts except cocoa and prevent future establishment of all functional groups.

[a] total agroforestry area affected = 88,700 ha / 10 years

monoculture cocoa plantations in Indonesia (Abou Rajab et al., 2016). The partitioning of available carbohydrates among different organs (e.g., fruits, branches, leaves) as a function of total tree biomass is typical for perennials (Niklas and Enquist, 2002) and implemented in several physiological growth and production models for cocoa (Zuidema et al., 2005; Beer et al., 1990). Dry bean yield was multiplied by the average farm-gate price for CCN-51 beans and Nacional beans, respectively US$1.93 and US$2.02 (PRAGMATICA, 2016; SINAGAP, 2015), to estimate cocoa revenues per hectare and at the landscape-level. We assumed that monoculture plantations exclusively produced CCN-51 beans whereas traditional and planted plantations exclusively produced Nacional beans. In all simulations, cocoa trees were represented by a single functional group (Table 3.1), neglecting differences in growth characteristics between Nacional and CCN-51 cocoa varieties.

Secondly, we estimated potential revenues from payments for additional carbon stored above baseline levels in carbon markets based on the mean CO_2 storage modeled for cocoa plantations (CO_2 = AGB x 1.84). We used a current price of US$5 per ton CO_2 in voluntary carbon markets based on the minimum credit price on the over-the-counter global market for agroforestry projects, which was also the average offset price for Latin America in 2015 (Seeberg-Elverfeldt et al., 2009; Hamilton et al., 2010; Somarriba et al., 2013). Other authors have used lower prices (US$1.2 per ton; Gockowski et al. (2011), but average carbon credits for agroforestry have been increasing to US$9.9 per ton of CO_2 in 2015 (Hamrick and Goldstein, 2017). Additionally, we made a second estimate for potential carbon revenues using a hypothetical price of

US$30 per ton CO_2, which has been suggested by scholars to cover the social and environmental costs of carbon (Nordhaus, 2017) and has been found high enough to incentivize smallholders to maintain shaded agroforestry systems (Seeberg-Elverfeldt et al., 2009).

3.2.3.3 Landscape metrics

To assess the landscape scale impacts of cocoa plantation management on forest landscape patterns, modeled AGB maps were reclassified for each scenario and time step into 'forested' and 'unforested' cells, using a threshold of 130 Mg per hectare AGB (i.e., 65 Mg per hectare carbon). Schroth et al. (2015) found that non-cocoa tree AGB stocks above this threshold repressed cocoa yields on cocoa plantations in southern Bahia, Brazil. Average total AGB estimates from the MAE inventories were around 140 Mg per hectare (Fig. A.2. The resulting binary maps were used to compute four landscape pattern indices calculated for cells classified as forested to capture the effect of shade management on forest fragmentation: (i) percentage of landscape, (ii) number of patches, (iii) effective patch size, and (iv) patch cohesion index, using the R package SDMTools (VanDerWal et al., 2014; R Development Core Team, 2008) based on FRAGSTATS statistics (McGarigal et al., 2012; McGarigal and Marks, 1995). These indices represented for each scenario respectively the (i) reduction in forested area; (ii) increase in number of forest patches; (iii) decrease in size of forest patches; and (iv) increase in isolation of forest patches.

3.2.4 Model assumptions and limitations

General limitations of the LANDIS-II model are the simplification of complex processes and data availability for model parameterization. Knowledge of primary productivity rates remain scarce in the tropics (Malhi et al., 2011; Clark et al., 2013; Zuidema et al., 2013), making the ANPP estimates used in this study a source of uncertainty. We could not control for the influence of climate and soil conditions that are known to influence ANPP (Aragao et al., 2009), due to lack of data. Furthermore, modeling functional species groups makes it difficult to determine if shifts in functional groups abundances would relate to shifts in abundance of rare, forest-dependent, or generalist species in real world landscapes. For example, Kessler et al. (2012) found that transforming natural forest to cacao agroforests led to a significant loss in rare forest-related species richness in 4 plant and 8 animal groups in Sulawesi, Indonesia. We neglected anthropogenic disturbances other than agroforestry management in all scenarios.

To estimate the economic potential from cocoa plantations from simulations, we assumed a linear relationship between increase in total cocoa tree AGB and cocoa fruit biomass. Even though this relation has been applied for physiological production models of cocoa growth, carbon partitioning over different organs depends on several factors, such as water availability and the slope of the allometric distribution function (Beer et al., 1990; Zuidema et al., 2005). Different hybrids and varieties may have different distribution equations, for example storing more biomass in reproductive organs. These differences were accounted for in our estimates by the bean yield to ANPP ratio from Indonesian cocoa plantations (Abou Rajab et al., 2016), but estimates for Ecuadorian plantations were unavailable. Furthermore, the effect of shade management on cocoa yield is still disputed among scholars. Even though many scholars have found that cocoa growth and production decreased non-linearly with increasing shade (Steffan-Dewenter et al., 2007; Gockowski et al., 2011; Wade et al., 2010; Blaser et al., 2017), others have shown that intermediate levels of canopy shade of 40 to 60 per cent protected cocoa trees from drought (Abou Rajab et al., 2016) and allowed to maintain high cocoa production (Waldron et al., 2012).

3.3 Results

3.3.1 Model calibration and sensitivity analysis

The LANDIS-II 'spin-up' phase (i.e., biomass initialization prior to simulation onset) estimated 37.26 Tg AGB at the landscape scale with an average AGB for all ecoregions of 107.41 Mg per hectare at simulation onset. About 22 per cent of total landscape AGB was allocated in cocoa agroforestry cells. AGB values for LANDIS-II simulations and observed MAE forest inventories were positively correlated (Pearson's correlation = 0.36, n = 172, $p < 0.001$) with a root mean square error of 78.9 Mg per hectare at the cell level (Fig. 3.3). A sensitivity analysis showed that model parameterization was not very sensitive to variations in most key parameter values, indicated by a less than 10 per cent change in total AGB (Table 3.4). The maximum biomass parameter affected the model outcome the most, with an increase in total AGB at year 50 of about 14 per cent following a 10 per cent initial increase. We kept the initial parameterization for this value given the large number of plots (N = 172) used to determine initial values.

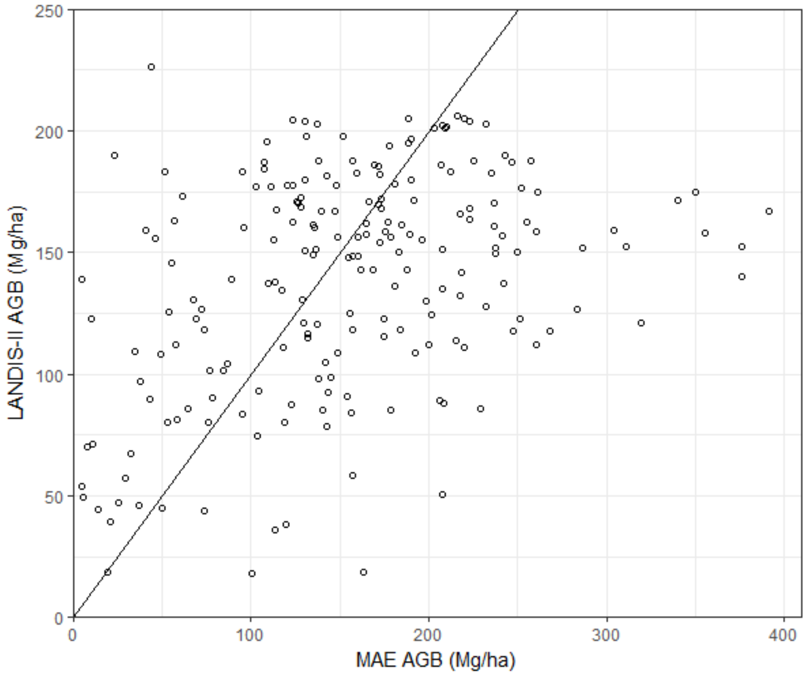

Figure 3.3: Estimates of total aboveground biomass for 172 inventory plots collected by the MAE (Ecuadorian Ministry of Environment) against the LANDIS-II spin-up representation of these plots (y-axis). Each plot corresponds to one dot, in contrast with the random communities map used to initialize simulations. The black line corresponds to a 1:1 perfect fit

3.3.2 Impact of cocoa management on aboveground biomass

The absence of natural or anthropogenic disturbances (other than cocoa management) throughout all scenarios allowed for continued forest growth and succession, incrementing AGB stocks (Fig. 3.4). For the baseline scenario (i), total AGB increased by 7.4 per cent (i.e., 40.04 Tg) 50 years after the spin-up initialization (Fig. 3.4; Table 3.5). The removal of all non-cocoa AGB on cocoa plantations in the monoculture scenario (iv) resulted in a 9.1 per cent decrease of total AGB to 36.38 Tg after 50 years (Fig. 3.4; Table 3.5). The transition to traditional and planted agroforestry systems resulted in a respective 6.9 and 3.4 per cent increase of total landscape AGB compared to baseline. Setting aside 40 per cent of agroforestry land for natural regeneration could compensate for the loss in landscape AGB following conversion to monoculture systems after 50 years (i.e., monoculture + 40% set aside; scenario iv-c; Fig. 3.4; Table 3.5).

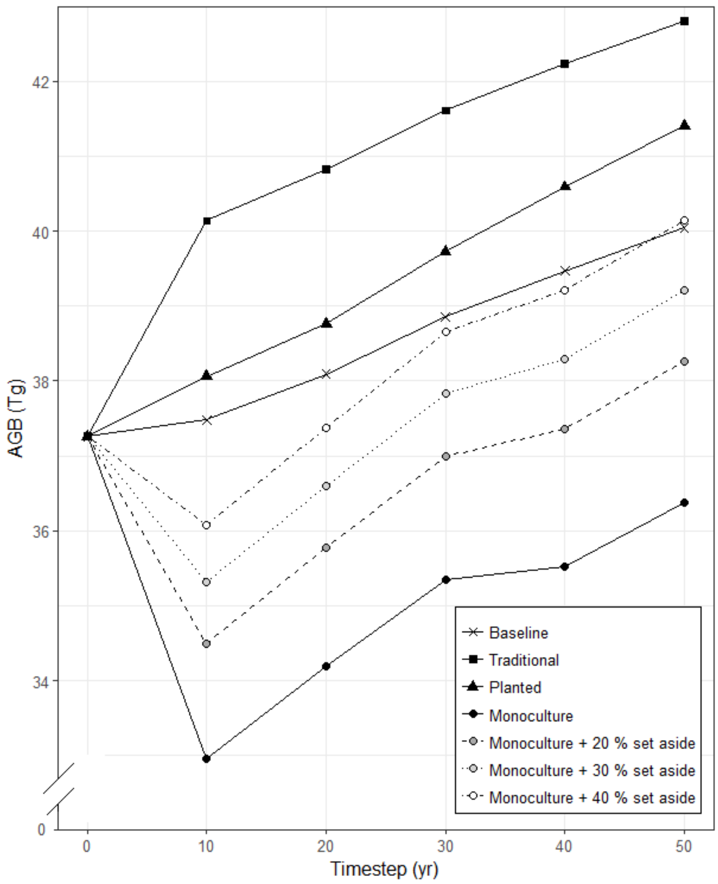

Figure 3.4: Total simulated landscape aboveground biomass (Teragram) from onset to year 50 for all scenarios. See Table 3.3 for scenario descriptions

Table 3.4: Sensitivity analysis results for six key parameters at year 0 and year 50 for the baseline scenario. Continuous parameters were adjusted 10 per cent and categorical parameters were adjusted 1 unit

Parameter	Change	Year 0		Year 50	
		AGB (Tg)	Change (%)	AGB (Tg)	Change (%)
Baseline scenario	0	37.26	0.00	40.04	0.00
	-10%	35.61	-4.43	38.38	-4.15
	10%	38.71	3.89	43.44	8.49
	-10%	33.24	-10.79	36.32	-9.29
	10%	41.22	10.63	45.72	14.19
	-10%	37.62	0.97	41.06	2.55
	10%	36.88	-1.02	39.67	-0.92
	-1	36.60	-1.77	39.38	-1.65
	1	38.04	2.09	42.41	5.92
	-10%	37.26	0.00	39.6	-1.10
	10%	37.26	0.00	41.75	4.27
	-1	37.26	0.00	38.35	-4.22
	1	37.26	0.00	43.70	9.14

3.3.3 Impact of cocoa management on functional composition

Shade management affected composition of functional groups strongly at the plantation level (Fig. 3.4a). In the monoculture scenario (iv), cocoa trees accumulate almost all of the total AGB in plantations. The AGB in the rare functional groups 1, 2, and 6 increased in the traditional scenario (v) relative to the planted and monoculture scenarios, but with a decrease of cocoa AGB (Fig. 3.5a). At the landscape-level, the effect of management scenarios on composition of functional groups was minor (Fig. 3.5b).

3.3.4 Impact of cocoa management on economic potential

Cocoa plantation management affected economic potential during simulations. Estimated revenues from cocoa bean production in the monoculture scenario were 3.5, 5.7, and 7.4 times larger compared to, respectively, baseline, planted,

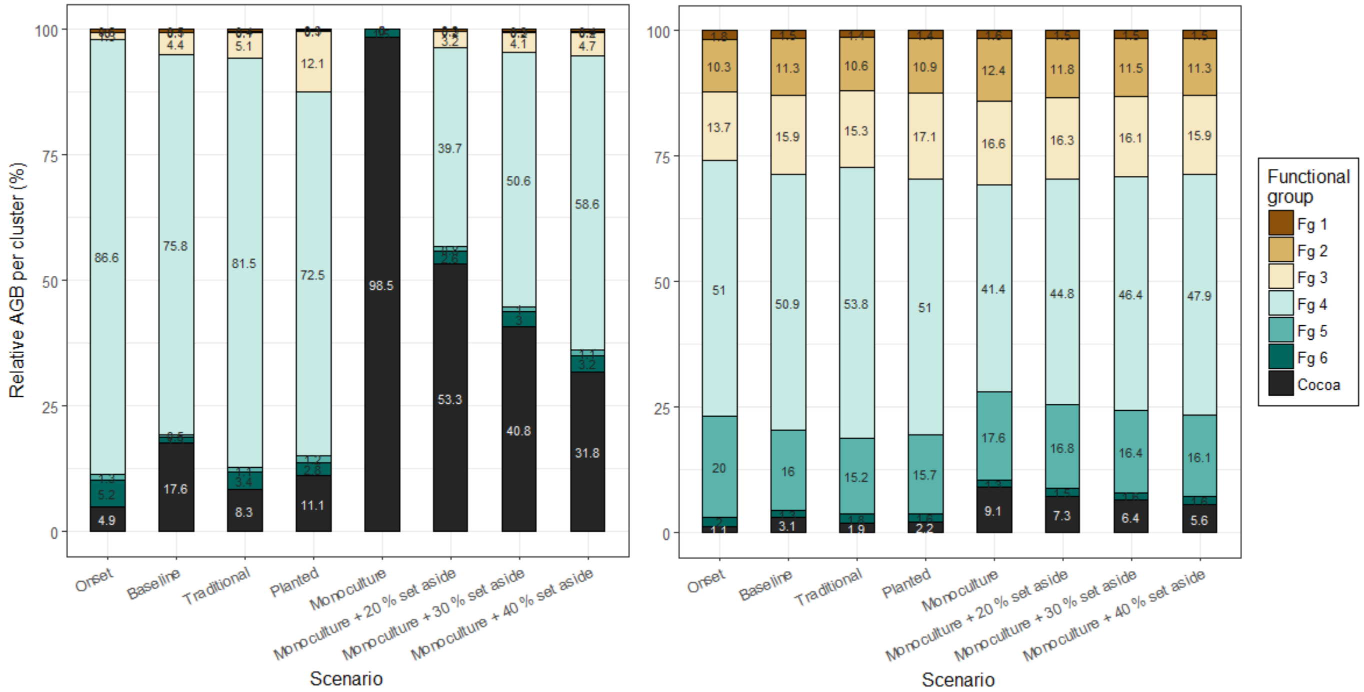

Figure 3.5: Functional group composition by aboveground biomass (AGB) at onset ('Current') and at year 50 for the seven LANDIS-II scenarios for (a) agroforestry cells and (b) at landscape scale. Numbers in bars indicate the percentage AGB accumulated in each functional group relative to total AGB; parenthesized numbers above bars indicate total AGB values in Tg

Table 3.5: Overview of the total AGB scenario outcomes and percentage variation compared to baseline and within scenario change for year 20 and year 50. Total AGB at 'spin-up' (i.e., year 0) equaled 37.26 Tg for all scenarios

Scenario	Year 20		Year 50		
	AGB	Change compared to baseline (%)	AGB	Change compared to baseline (%)	Within scenario change from 'spin-up' (%)
Baseline	38.09	0.00	40.04	0.00	7.37
Traditional	40.83	7.19	42.81	6.92	14.80
Planted	38.77	1.79	41.41	3.42	11.05
Monoculture	34.19	-10.24	36.38	-9.14	-2.44
Monoculture + 20% set aside	35.77	-6.09	38.26	-4.45	2.60
Monoculture + 30% set aside	36.59	-3.94	39.22	-2.05	5.18
Monoculture + 40% set aside	37.38	-1.86	40.15	0.27	7.67

and traditional scenarios (Table 3.6). Cocoa bean yield estimates from model output were in the range of average yield values for our study region (ESPAC, 2014). Compensating for the loss in landscape AGB following conversion to monoculture plantations by setting aside 40 per cent of agroforestry land for natural regeneration resulted in an estimated twofold increase in cocoa revenues at the landscape-level compared to baseline (i.e., monoculture + 40% set aside; scenario iv-c; Table 3.6).

We estimated additional revenues from carbon stored above baseline levels in carbon markets for the traditional and planted scenarios of respectively US\$25.7 and US\$12.6 million at assumed US\$5 per hectare per ton CO_2 (Table 3.6). Summing cocoa and carbon estimated revenues at the landscape scale showed that, at current carbon market price (i.e., US\$5 per hectare per ton CO_2), the income gap between monoculture and shaded agroforestry systems could not be filled. If carbon market price would rise to US\$30 per ton CO_2, a landscape with planted or traditional agroforestry systems would generate, respectively, equal or double revenues compared to a landscape with merely monoculture plantations (Table 3.6).

3.3.5 Impact of cocoa management on landscape metrics

Landscape pattern indices for the binary forest maps indicated a more fragmented forest landscape for the monoculture scenario (iv): forested area decreased in combination with low patch cohesion and effective mesh size (Fig. 3.6). By contrast, the traditional scenario (ii) resulted in a progressive increase in forested area, patch cohesion, and effective mesh size, indicating a decrease in landscape fragmentation. Land set aside for natural regeneration in the monoculture scenarios (iv-a, iv-b, and iv-c) resulted in increased patch cohesion and effective mesh size, indicating increased forest connectivity in abandoned areas (Fig. 3.6).

3.4 Discussion

3.4.1 Impacts of cocoa shade management

Our simulation study found that shade management strategies on cocoa plantations affected AGB stocks, functional species composition and economical potential at both the plantation- and landscape-level. Due to the limitations and assumptions discussed in section 2.4, simulations outcomes cannot be interpreted as exact predictions. Our modeling framework, like all models, is a simplification of real-world agroforestry landscape dynamics. Hence, care should be taken with the interpretation of exact numbers and estimates from all simulations.

3.4.1.1 Aboveground biomass

We found that plantations with traditional and planted shade management strategies had a large potential to store carbon (resp. 111.3 and 95.3 Mg per hectare after 50 years of simulation), storing almost three times more AGB compared to unshaded monoculture plantations (38.3 Mg per hectare) (Table 3.5; Table B.1). These results were in agreement with previous studies that showed that the level of shading in cocoa plantations affected overall carbon storage potential. For example, in Sulawesi, Indonesia, cocoa plantations with multiple-species shade contained five times more AGB (100 Mg per hectare) compared to monoculture plantations (17 Mg per hectare) (Abou Rajab et al., 2016). In Ghana, traditional cocoa plantations with more than 25 per cent shade cover stored over three times more AGB than intensified cocoa plantations with less than 25 per cent shade (resp. 262 and 78 Mg per hectare) (Wade et al., 2010).

Changes at the plantation level also affected biomass accumulation at the landscape scale during simulations, emphasizing the important role of agroforestry systems for landscape carbon mitigation efforts. Traditional shaded cocoa management strategies increased the total landscape AGB by almost seven per cent during our simulations compared to baseline. In southern

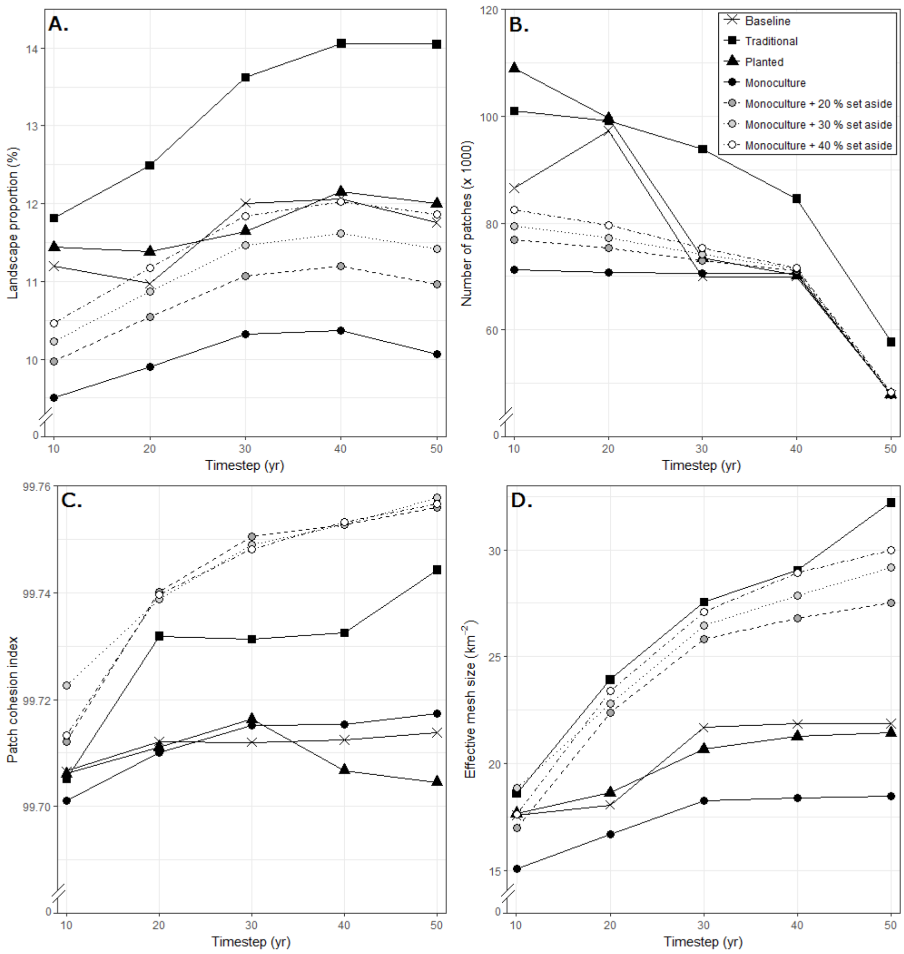

Figure 3.6: Landscape pattern indices captured four spatial effects of fragmentation on forest habitat pattern following shade management scenarios simulated in LANDIS-II: (a) reduction in forested area (percentage of landscape); (b) increase in the number of forest patches (number of patches); (c) decrease in size of forest patched (effective patch size); and (d) increase in isolation of forest patch (patch cohesion index)

Bahia, Brazil, over half of the landscape carbon stocks (about 59 per cent) were contained in traditional cocoa agroforests, compared to approximately 30 per cent in natural forest remnants (Schroth et al., 2015), illustrating the critical importance of shaded agroforestry systems for carbon storage.

3.4.1.2 Biodiversity

Besides impacts on biomass storage, we found a shift in functional species composition at the plantation level following shade management, resulting in small changes at the landscape scale (Fig. 3.4). Traditional management increased the abundance of rare functional groups, which are absent in monoculture and most planted cocoa plantations. Besides affecting vegetation structure and composition on cocoa plantations (Deheuvels et al., 2012), farmers' management strategies were found to influence beta-diversity of terrestrial plants, epiphytes, amphibians, and soil and litter invertebrates (Deheuvels et al., 2014). For example, the diversity and composition of the vegetation on cocoa plantations has been positively correlated with richness and diversity of bats, birds (Wilsey and Temple, 2011; Faria et al., 2006), soil fauna (da Silva Moço et al., 2009), and mammals (Vaughan et al., 2007; Pardini, 2004). The botanical composition of agroforestry systems offers a distinct set of morphological and functional traits, emphasizing the importance of a structurally complex and optimized canopy design. A diverse and structurally complex shade canopy of native species can conserve plant biodiversity on and off-farm by serving as seed source, as well as providing valuable habitats for many other organisms.

3.4.1.3 Economic potential

Our simulations estimated cocoa yield was over seven times larger for monoculture plantations compared to traditional agroforestry systems. The simulated yield gap in our study characterizes yields under optimized conditions, neglecting other management difficulties including lack of agricultural inputs or weak management skills. Hence, extensive household surveys on cash income and subsistence needs of cocoa smallholders in Northern Ecuador found smaller yield gaps between shaded and monoculture plantations (Blare and Useche, 2013). When discounting for differences in labor costs and market price, Blare and Useche (2013) found profits for monoculture CCN-51 plantations (US$1,223 per hectare) that were twice as large as for shaded Nacional plantations (US$608 per hectare). Even though production revenues are smaller, farmers often prefered traditional production systems due to non-market benefits, such as biodiversity, improved soil quality, and access to food and medicine (Useche and Blare, 2013; Steffan-Dewenter et al., 2007; Blare and

Useche, 2013). An increase in direct profits from shaded agroforestry systems may nevertheless be essential for the maintainance of traditional agroforestry systems. Without additional carbon markets, the simulated yield gap between the baseline and traditional agroforestry scenario indicated that farmgate price premiums should double to maintain the economic attractivity of these shaded systems (i.e., about US$400 per ton dry cocoa beans). In 2014, Fairtrade certified beans capture a stipulated premium of US$200 per metric ton, whereas premiums for organic, UTZ Certified, and Rainforest Alliance certified beans are based on market demand, ranging between US$140 and US$200 per ton (ICCO, 2016; Potts et al., 2014).

Our simulations indicated that a landscape planning strategy combining conversion to monoculture systems with about 40 per cent land set aside for natural regeneration might double total cocoa production while retaining similar levels of landscape aboveground biomass compared with baseline projections (Fig. 3.4; Table 3.6). On the other hand, increasing aboveground biomass accumulation on cocoa plantations in shaded management strategies offers smallholders the possibility to participate in carbon markets. We estimated a total additional revenue for traditional and planted agroforestry systems from payments for ecosystem services schemes of US$290 to US$143 per hectare at an assumed current carbon price of US$5 per ton CO_2. At the landscape-level, this current market price did not compensate for revenue loss following decreased production compared to monoculture systems. Nevertheless, average carbon credits for agroforestry increased up to US$9.9 per ton of CO_2 in 2015 (Hamrick and Goldstein, 2017). We estimated that, if carbon market prices would increase to US$30 per ton CO_2, total revenues from traditional plantations would become almost twice as high as for monocultures. Seeberg-Elverfeldt et al. (2009) estimated that increasing carbon prices up to about US$32 per ton CO_2 would incentivize Indonesian cocoa smallholders to sustain shade intensive agroforestry systems. Currently, agroforestry projects take up a small percentage (1 per cent) of total carbon volumes covered by global carbon markets (1.3 $MtCO_2e$ in 2009 and 7.5 $MtCO_2e$ in 2015), leaving potential for expansion globally (Hamrick and Goldstein, 2017; Hamilton et al., 2010). Increasing the extent of carbon markets and rising carbon prices may be essential to value social costs and reward smallholders for the environmental benefits their shaded agroforestry systems provide.

3.4.2 Implications and recommendations

The question remains what forms of agriculture can improve income and food security with the minimal environmental impact of land use. Visions range from ecomodernism with large-scale mechanized farming in combination with preservation of natural habitats (Asafu-Adjaye et al., 2015), to propositions of small-scale, labor-intensive agroecological systems (Altieri and Toledo, 2011). A growing consensus among ecologists and resource managers suggests the need for smarter landscape management through sustainable intensification (Fischer et al., 2017; Andres and Bhullar, 2016). A meta-data approach showed that agricultural diversification, such as multi-cropping and multiple crop rotations, potentially reduces the yield gap with conventional agriculture (Ponisio et al., 2015). Shade canopy optimization for carbon stocks can be achieved by selecting tree species with distinct morphological and functional traits, such as tree species with tall and thick stems, small and light foliage, or rapid growth and high density timber (Somarriba et al., 2013). Besides botanical composition, appropriate spatial arrangement of shade components could improve yield and reduce competition (Deheuvels et al., 2014; Schroth et al., 2016). The maintenance of large trees in agroforestry systems has also been suggested as a valuable conservation strategy (Schroth et al., 2015). These trees store by far the most biomass and compete less with cocoa trees for light in the understory. Large trees provide valuable habitat, and services for other flora and fauna species, such as nesting sites, cavities and food, and can be important source of seeds for surrounding areas. Nevertheless, sustainable management is labor and time intensive and its feasibility on larger scales in developed countries remains questionable (Andres and Bhullar, 2016).

In contrast, large-scale intensified farming may increase food production for urban consumers, but its contribution to local food security remains debatable (Meyfroidt, 2017; Fischer et al., 2014). In addition, agricultural intensification can result in land spared for preservation and restoration of natural habitat. Nevertheless, intensification of commodity crops, including, have been associated with a rebound effect associated locally through the expansion of agricultural land in intensifying regions or globally through the redistribution of agriculture (Hertel et al., 2014; Jadin et al., 2016). Different interventions, such as value chain upgrading, certification schemes, and payments for ecosystem services, should thus focus on increasing benefits for farmers and wage laborers and controlling for land use expansions to neutralize possible rebound effects (Meyfroidt, 2017).

Despite all efforts by conservationists, nature reserves only represent a small fraction of biodiversity-rich areas. Surrounded by a low quality landscape matrix, the isolation of biodiversity-rich areas results in greater local extinctions, which cannot be replenished. Mixed, small-scale farming systems, like shaded agroforestry systems, may provide an increase in landscape connectivity that can stimulate the matrix quality for other species (Asare et al., 2014; Perfecto et al., 2010). Additionally, agroforestry provides other services, such as watershed protection (Garrity, 2004), improved pollination (Forbes and Northfield, 2017), increased food security and accessibility (Tscharntke et al., 2012a; Altieri and Toledo, 2011; Kremen, 2015), pest, disease, and erosion control (Tscharntke et al., 2011; Smith Dumont et al., 2014), nutrient cycling (Asase and Tetteh, 2016), soil fertility improvement (Obeng and Aguilar, 2015), and capacity building (Lal et al., 2015). The biological diversity in cocoa agroforestry cannot equate that of primary forests and might only be economically viable if smallholders are able to gain access to improved carbon markets.

The larger landscape-scale context is crucial for the design and management of tropical agricultural systems (Harvey et al., 2014). Current eco-certification schemes focus at the farm or plantation levels, whereas ecosystem benefits from well-designed agroforests are delivered at the landscape-level, emphasizing a scale mismatch (Tscharntke et al., 2015). Scholars have suggested two approaches to address this incongruence: certification mechanisms could be linked with broader landscape-scale approaches (Milder et al., 2014) and current certification models could consider the landscape as a certified unit through landscape labeling (Ghazoul et al., 2009). The question remains whether and to what extent such activities will stimulate the adoption of agroforestry practices among farmers.

3.5 Conclusion

Our study found that shade management strategies on cocoa plantations affected AGB stocks, functional species composition, and economic potential at plantation- and landscape-level. Shaded cocoa management strategies (i.e., traditional and planted scenarios) increased the total aboveground biomass, preserved functional species diversity on plantations, and decreased fragmentation of forested areas, in contrast with unshaded monoculture management strategies. We found indications that, with the current low price for high-quality cocoa beans for specialty markets and carbon payments, smallholders might not be compensated for the yield gap between shaded and monoculture plantations. Our simulation experiments emphasize the important role agroforestry systems can play for biomass and biodiversity conservation in agricultural landscapes, which underlines the importance of increasing carbon and cocoa bean prices to maintain shaded production systems.

Table 3.6: Estimated potential revenues from carbon production (left) and carbon markets at US$5 and US$30 per hectare per ton CO_2 above baseline projections (right) for seven shade management scenarios. Total landscape revenues represent the sum of cocoa and carbon landscape revenues (in million US$)

	Cocoa production						*Carbon markets*						
								US$5 ha^{-1} ton^{-1}			*US$30 ha^{-1} ton^{-1}*		
Scenario	Mean cocoa ANPP [a] ($Mg\ ha^{-1}\ yr^{-1}$)	Dry bean yield [b] ($kg\ ha^{-1}\ yr^{-1}$)	Dry bean price [c] (US$ kg^{-1})	Cocoa revenues (US$ ha^{-1})	Cocoa plantation area (ha)	**Cocoa landscape revenues (x10^6 US$)**	Mean CO_2 plantations [d] ($Mg\ ha^{-1}$)	Carbon revenues [e] (US$ ha^{-1})	**Carbon landscape revenues (x10^6 US$)**	**Total landscape revenues (x10^6 US$)**	Carbon revenues [f] (US$ ha^{-1})	**Carbon landscape revenues (x10^6 US$)**	**Total landscape revenues (x10^6 US$)**
Baseline	0.179	136.99	1.98	270.56	88,706	**24.00**	146.81	0	**0**	**24.00**	0	**0**	**24.00**
Traditional	0.085	64.69	2.02	130.68	88,706	**11.59**	204.74	289.62	**25.69**	**37.28**	1,737.70	**154.14**	**165.74**
Planted	0.110	84.20	2.02	170.09	88,706	**15.09**	175.32	142.51	**12.64**	**27.73**	855.05	**75.85**	**90.94**
Monoculture	0.653	498.26	1.93	961.64	88,706	**85.30**	70.40	0	**0**	**85.30**	0	**0**	**85.30**
Monoculture + 20%	0.653	498.26	1.93	961.64	70,965	**68.24**	70.40	0	**0**	**68.24**	0	**0**	**68.21**
Monoculture + 30%	0.653	498.26	1.93	961.64	62,093	**59.71**	70.40	0	**0**	**59.71**	0	**0**	**59.71**
Monoculture + 40%	0.653	498.26	1.93	961.64	53,223	**51.18**	70.40	0	**0**	**51.18**	0	**0**	**51.18**

[a] From LANDIS-II simulation output over 50 years (i.e., all time steps); [b] Dry bean yield was estimated by multiplying the mean aboveground net primary productivity (ANPP) in cocoa trees from LANDIS-II simulations with the average dry bean yield to ANPP ratio of 0.76 calculated in Indonesian traditional, planted, and monoculture cocoa plantations (Abou Rajab et al. 2016). Mean dry cocoa bean yield in the area was ~252 kg ha^{-1} (ESPAC 2014); [c] Assuming an average farm-gate price of US$1.93 and US$2.02 per kilogram for respectively CCN-51 (associated with monoculture plantations) and Nacional beans (associated with traditional and planted plantations) in Esmeraldas for 2014 (US$89 and US$93 per quintal;1 quintal = 46 kg) (PRAGMATICA 2016; SINAGAP 2015); [d] From LANDIS-II simulation output after 50 years of simulation, assuming Mg CO_2 = Mg AGB x 1.84 (Chave et al. 2005); [e] US$ = Mg CO_2 x US$5 [Mg CO_2]$^{-1}$ for additional carbon stored above the baseline projections; [f] US$ = Mg CO_2 x US$30 [Mg CO_2]$^{-1}$ for additional carbon stored above the baseline projections.

CHAPTER 4

SOCIOECONOMIC IMPACTS OF COCOA AGROFORESTRY: IMPROVING LIVELIHOODS AND ECOSYSTEMS THROUGH DIRECT TRADE

This chapter is based on the following paper submitted to *Agriculture, Ecosystems and Environment*: R.S. Middendorp, O.M. Boever, X. Rueda, E.F. Lambin. Improving smallholder livelihoods and ecosystems through direct trade relations: high-quality cocoa producers in Ecuador.

Abstract

Globalization has increased the influence of consumers' choices on land use change and livelihoods in developing rural areas worldwide. New niche commodity markets for fine cocoa – produced by old tree varieties frequently grown in shaded agroforestry systems – created more direct linkages between producers and buyers. We explored the socioeconomic and environmental outcomes for cocoa smallholders that participated in direct trade innovations compared with smallholders that sold through mainstream markets. Household surveys were conducted with 57 cocoa smallholders in northern Ecuador. Biodiversity conditions at farm-level were monitored for 75 per cent of surveyed households. Using a credible counterfactual based on genetic matching, we found indications that smallholders engaged in direct trade: (i) captured superior prices for cocoa sales; (ii) had greater access to agricultural training, technical assistance, and improved social networks; and (iii) applied more nature-friendly management practices, compared to smallholders selling through mainstream markets. Shade level and species richness and abundance in plantations were unrelated to market participation. Additional qualitative analyses supported these findings. This study provides insights on the potential of developing value chain innovations for high-quality commodity trade. The success of value chain innovations hinges on the competitiveness of farmers' cooperatives and involvement of governments, NGOs, and private actors.

4.1 Introduction

Smallholder agroforestry systems are widespread across the tropics. The intentional integration of trees in orchards, woodlots, and home gardens provides an opportunity to consolidate agricultural production and biodiversity conservation in human-dominated landscapes. Indeed, many scholars have argued that agroforests can maintain high levels of biodiversity and ecosystem services supply, while supporting livelihoods of local communities (Schroth and Harvey, 2007; Perfecto et al., 2007; Bhagwat et al., 2008; Steffan-Dewenter et al., 2007; Tscharntke et al., 2011, 2015; Vaast and Somarriba, 2014; Fischer et al., 2008). Yet smallholders struggle worldwide with low access to credit, technology, and market outlets. The growing concentration of value chains and downward pressure on prices meanwhile increase farmers' vulnerability to volatile international markets (Hernández et al., 2014). As a result, many smallholders convert shaded agroforests to more intensively managed plantations by reducing the number of shade trees, attempting to secure short-term income (Steffan-Dewenter et al., 2007; Clough et al., 2009b; Vaast and Somarriba, 2014; Philpott et al., 2008). This trend threatens the resilience of tropical agricultural landscapes, while failing to ensure more stable household incomes (Tscharntke et al., 2011; Jacobi et al., 2014).

Governments, corporations, consumers, and scholars alike have expressed concern about the impacts on sustainability of highly concentrated global commodity value chains. Two major market governance mechanisms have emerged that aim to increase the social and environmental benefits of tropical commodity markets: voluntary sustainability standards and direct trade of high-quality agricultural products in specialty markets. Private corporations promote sustainable practices to consumers by means of sustainability standards with specific certification labels, such as Fairtrade, Organic, Rainforest Alliance, or UTZ certified (Potts et al., 2014). Specialty markets meet rising consumer demands for high-quality agricultural products that are sourced sustainably and often that originate from specific terroirs (Lambin, 2015). Manufacturers establish direct trade relationships with producers, thus involving no or few intermediaries, and commit to improve the social, economic, and environmental conditions in producing regions. Farmers that participate in these alternative market channels may receive price premiums for certified or high-quality products and may benefit from additional agricultural training, reinforced social networks, and access to credit (Gereffi et al., 2005; Lee et al., 2012). A growing number of studies have found ambiguous evidence on environmental and socioeconomic impacts of market governance mechanisms (Blackman and Naranjo, 2012; Blackman and Rivera, 2010). A recent meta-analysis of 24 unique certification programs reported positive outcomes for 34 per cent of response variables with inconsistencies across cases (DeFries et al., 2017). Few studies have explored the outcomes of direct trade of high-quality agricultural products for specialty markets, reporting ambiguous outcomes and often without adequate statistical methods that compare observations for treatment and control groups (Borrella et al., 2015; Neilson, 2014; Tobin et al., 2015; le Polain de Waroux and Lambin, 2013). To our knowledge, no peer-reviewed study has yet compared economic, social, and environmental outcomes of direct trade of high-quality cocoa beans versus mainstream trade. This emphasizes the need for rigorous analyses and independent evaluation to assess effectiveness of market governance mechanisms, especially for direct trade in high-quality specialty markets.

This study took the case of cocoa smallholders in northern Ecuador. We aimed to evaluate the socioeconomic and environmental outcomes of market governance mechanisms, focusing on the impacts of specialty markets of fine cocoa beans versus mainstream trade of bulk cocoa. Constructing a credible counterfactual based on a statistical pair matching method, we compared households engaged in direct trade and households selling to the mainstream market in northern Ecuador. As direct trade relations and certification schemes were partially overlapping in our region, additional qualitative data were collected to refine and interpret our quantitative results. We conducted in-depth interviews and subsequently organized a workshop with international market actors, focusing on environmental, economic, and social aspects of the Ecuadorian cocoa market. We also relied on secondary sources, including scientific, governmental, and private publications. Section two first describes the organization of the Ecuadorian cocoa sector and details the hypotheses tested in this study. This study offers insight on the potential of developing value chain innovations for high-quality agricultural commodities.

4.2 Background

4.2.1 Ecuadorian cocoa sector

Cocoa remains an agricultural commodity produced mainly by smallholders: an estimated five to six million farmers who own on average three hectares of land produce over 90 per cent of world's production (WCF, 2013). Ecuador has a long tradition of cocoa production, providing up to 50 per cent of world production during the 19th century (Poelmans and Swinnen, 2016). Attacks of fungal pathogens, namely witch's broom (*Crinipellis perniciosa*) and frosty-pod (*Moniliophthora roreri*), decimated many plantations and led to the collapse of the industry by the 1930s (Griffith et al., 2003). Today, cocoa still serves as a cash crop for about 100,000 small and medium-size Ecuadorian farmers that cultivate an estimated 490,000 hectares (ESPAC, 2014). Although in 2015, Ecuador was responsible for a mere six per cent of global exports (261,000 of 4,251,000 tons), it remains the world's largest producer of fine cocoa beans originating from the genetically distinct Nacional Forastero tree (about 60 per cent of global supply) (ICCO, 2016; FAO, 2014; ANECACAO, 2016). Chocolatiers and chefs worldwide desire these beans for their unique taste and floral aromas. High-quality sustainable dark chocolates have gained popularity amongst urban consumers, resulting in a rising demand for fine cocoa beans of known origin (Lambin, 2015).

In contrast with rising demand, supply of high-quality Nacional beans from Ecuador have decreased. In an attempt to increase short-term income, many Ecuadorian farmers have replaced Nacional varieties with the presumably more productive and disease-resistant hybrid variety, named CCN-51 (Griffith et al., 2003; Melo and Hollander, 2013). This clone produces beans with a sour or bitter taste and lacks the flavorful aromas present in Nacional beans. Economic marginalization, genetic erosion of the original genotype, changes in management practices, and the continued mixing of varieties resulted in the progressive loss of Ecuadorian cocoa quality (Deheuvels et al., 2004). The International Cocoa Organization has downgraded the country's cocoa rating from 100 per cent fine flavor to 75 per cent in 2004 and has cautioned against further downgrading (ICCO, 2015). Today, CCN-51 beans comprise 30 per cent of total exports, compared to 47, 18, and 5 per cent for Nacional beans with ascending quality ratings of, respectively, ASE (*Arriba Superior Epoca*; default rating), ASS (*Arriba Superior Selecta*), and ASSS (*Arriba Superior Summer Selecta*) (ANECACAO, 2016). Nacional and CCN-51 varieties are complementary in the Ecuadorian market, as each satisfies the demand of a different market (Fig. 4.1).

Scholars have cautioned that the area expansion of CCN-51 may concur with a decrease in shaded agroforestry systems (Bentley et al., 2004; Useche and Blare, 2013). As CCN-51 is less susceptible to sun damage, it is often grown under reduced shade levels or in full-sun conditions. Shaded agroforestry systems, traditionally associated with Nacional varieties, can harbor high levels of biodiversity (Clough et al., 2011; De Beenhouwer et al., 2013), sequester large amounts of carbon (Somarriba et al., 2013; Wade et al., 2010), and improve soil management (Vanhove et al., 2016). Shade reduction in agroforestry systems has been found to reduce biodiversity levels (Maas et al., 2015; Deheuvels et al., 2014; Clough et al., 2009b) and ecosystem services (Franzen and Borgerhoff Mulder, 2007; Tondoh et al., 2015). In addition, full-sun cacao, especially CCN51, encourages ground weeds and farmers tend to use more herbicides and pesticides (Jano and Mainville, 2007; Bentley et al., 2004). Market governance mechanisms can incentivize the maintenance of traditional cocoa agroforests and associated nature-friendly management practices by compensating farmers for lower yields (Tscharntke et al., 2012a, 2015).

4.2.2 Differentiated cocoa markets

4.2.2.1 Mainstream cocoa

In the mainstream cocoa chain, individual farmers sell partially dried or dry beans to local intermediaries after which the product moves from exporters to foreign buyers (Fig. 4.1). A study from the German International Cooperation estimated that about 1,000 intermediaries are active in Ecuador. Monopsony conditions (i.e., only one potential buyer) occur frequently in remote areas (van der Kooij, 2013). Intermediaries have been found to take advantage of smallholders who have little access to information by paying reduced prices (Jano et al., 2013; Jano and Mainville, 2007). Exporters ship to countries where processing and manufacturing occur, through value chains that are dominated by a few large multinational companies located mainly in North America and Europe (Kaplinsky, 2004; WCF, 2014). Ecuadorian enterprises process a very small (less than five per cent) but growing fraction of the cocoa.

Concerns about the sustainability in the cocoa value chain has resulted in new arrangements between private chain partners, the state, and NGOs (Ton et al., 2008). Following devaluation of national cocoa quality, Ecuadorian policy makers have become involved in the cocoa industry, putting in place policies and projects intended to vitalize the niche market for fine Nacional beans. The state has facilitated the establishment of value chain partnerships and development efforts, while reducing traditional state-led regulatory functions in the cocoa value chain (Vellema et al., 2016). It has likewise implemented several national plans and laws to strengthen its position on the global national fine cocoa market (van der Kooij, 2013; Ahmed and Hamrick, 2015).

The Ecuadorian Association of Cocoa Exporters (ANECACAO) sets a daily 'referential' price based on data from Stock Exchanges in New York and London, which determines cocoa prices in the mainstream market. This gate price reflects the price for Arriba Superior Epoca (ASE) beans, the default rating of Nacional. CCN-51 beans capture roughly the same price (PRAGMATICA, 2016; ANECACAO, 2016). The farm-gate price is subsequently determined by subtracting intermediation costs from the referential price. In other words, the gate price negatively correlates with the number of actors in the value chain (Melo and Hollander, 2013). In Esmeraldas, the average farm-gate price was US$1.93 and US$2.02 per kilogram for CCN-51 and Nacional beans for 2014 (US$89 and US$93 per quintal; 1 quintal equals 46 kg) (SINAGAP, 2015; PRAGMATICA, 2016).

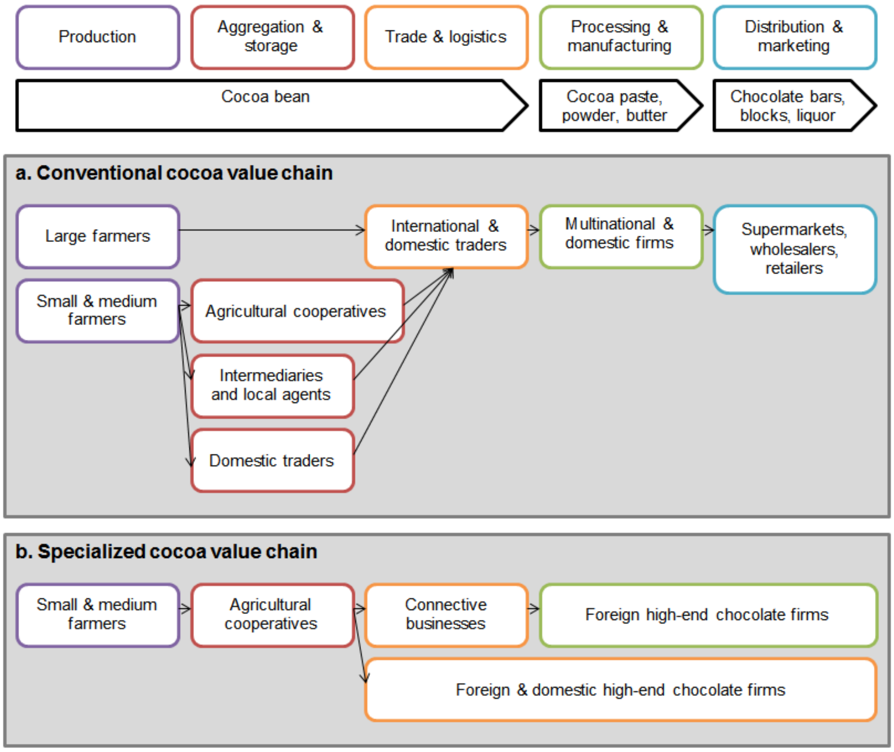

Figure 4.1: Outline of the (a) mainstream and (b) specialized cocoa marketing channels in Ecuador, based on Donovan (2006); Jano and Mainville (2007); Fernandez-Stark et al. (2012); Melo and Hollander (2013); Lehmann and Springer-Heinze (2014); Ahmed and Hamrick (2015); Poelmans and Swinnen (2016); Vellema et al. (2016)

4.2.2.2 Certified cocoa

Private companies have invested heavily in third-party certifications mainly as a response to growing consumer awareness around sustainability issues and recognition that intergovernmental collaboration fails to address global supply chain sustainability issues (Potts et al., 2014; Rueda and Lambin, 2013a). These private investments aim to improve living conditions of farmers, while promoting better environmental practices (Lee et al., 2012). In Ecuador, certified farmers often sell wet beans directly after harvesting to a farmers' cooperative that ferments and dries beans collectively. If possible, the cooperative acts as an exporter and sells the beans directly to foreign buyers at a higher price through a mandated or market-based premium (Tampe, 2016).

Farmer's price-setting differs across standards (Tampe, 2016). Fairtrade certified beans always capture a stipulated premium of at least US$200 and a minimum price of US$1,750 per metric ton, whereas premiums for organic, UTZ Certified, and Rainforest Alliance are based on market demand. Cooperatives hold the certificate and manage the internal control system, ensuring that member farmers comply with labor, environmental, and record keeping requirements. Voluntary sustainability standards started gaining ground in Ecuador during the early 2000s, facilitated by donor funding for rural developmental projects (Tampe, 2016). Today, major sustainability standards in the Ecuadorian cocoa sector include organic, Fairtrade, UTZ Certified, and Rainforest Alliance, accounting for respectively 8, 4, 1, and <1 per cent of total national production in 2011 (Potts et al., 2014).

Several studies have found positive economic and environmental impacts of coffee certification in Latin America, mainly for Fairtrade (Ruben and Fort, 2012; Ruben and Zuniga, 2011; Arnould et al., 2009; Ruben et al., 2009) and Rainforest Alliance (Rueda and Lambin, 2013b; Rueda et al., 2015). These studies explicitly accounted for the risk of selection bias in comparing certified and non-certified farmers, using rigorous matching procedures and constructing a counter factual control group. In contrast, to our knowledge, no peer-reviewed study has explored outcomes of cocoa certification, implementing these rigorous criteria (DeFries et al., 2017; Blackman and Rivera, 2010). In a non-peer-reviewed study, Cepeda et al. (2013) found positive effects of organic, Fairtrade, and Rainforest Alliance certification on quality improvements, income, and food security for 550 Ecuadorian cocoa smallholders, accounting for selection bias. Due to the evolving nature of sustainability standards, it is difficult to draw clear conclusions on impacts from individual certifications (Cepeda et al., 2013).

Sustainability standards have drawn criticisms despite their growing popularity over the recent years. As compliance is costly, standards can create barriers for market entry for poor marginal farmers with financial constraints or generate dependency on financial aid from developing organizations (Matissek et al., 2012). Less than half of certified cocoa beans are eventually sold as such to final consumers, reflecting similar issues found in coffee and other commodity certification markets (Potts et al., 2014). This oversupply of certified beans diminishes premiums paid, resulting in low producers prices as faced by the mainstream market (Raynolds et al., 2007; Jaffee and Howard, 2009). Certification outcomes are context-dependent and, in the case of Ecuadorian cocoa, only few certified cooperatives were documented as being able to create lasting benefits for producers (Melo and Hollander, 2013; Tampe, 2016).

4.2.2.3 Direct trade arrangements for high-quality cocoa

These shortcomings have prompted other forms of corporate engagement in commodity value chains beyond 'traditional' certification schemes. Direct trade is a form of engagement in which buyers that are looking for scarce high-quality products establish strategic alliances with farmers that meet quality and production requirements (Holland et al., 2016). Buyers can share a larger portion of the final price with the farmers by skipping traders and thus ensuring a better traceability and supply of the product (Porter and Kramer, 2011). These markets develop around commodities where traditional production practices already incorporate 'environmentally-friendly' attributes. The focus lies on marketing those attributes explicitly to consumers. Examples of direct trade markets are relationship coffee (Borrella et al., 2015; Hernandez-Aguilera et al., 2015) and bean-to-bar cocoa (Butcher and Wilson, 2014).

Farmers' cooperatives play an important role in establishing direct trade relationships by improving the bargaining position and knowledge appropriation of individual farmers (Gereffi, 1999; Gereffi et al., 2005; Lee et al., 2012). Buyers who are informationally close to farmers, particularly farmers' cooperatives, have been found to reward farmers for higher value characteristics of quality beans, compared to buyers who purchase at distant spot markets (Jano et al., 2013; Moe, 2008). Access to specialty markets for fine cocoa requires improved production methods and post-harvest processing systems, which are unavailable to smallholders and cooperatives with low asset endowment. Some Ecuadorian farmers' cooperatives have gained access to specialty markets, mainly through the establishment of sustainable buyer ties and recursive knowledge appropriation (Tampe, 2016; Melo and Hollander, 2013). A tendency for increased exports through farmers' cooperatives has been observed in Ecuador, but still accounted for only five to ten per cent of total exports in 2015 (PRAGMATICA, 2016; Ahmed and Hamrick, 2015). These cooperatives focus mainly on smallholders producing Nacional beans, with less than ten per cent of their sales comprised of CCN-51 beans (PRAGMATICA, 2016).

Several studies on other commodities have found positive impacts on smallholder wellbeing and environmental conservation from participations in direct trade markets for high-quality commodities, but outcomes are context dependent (Borrella et al., 2015; Neilson, 2014; Tobin et al., 2015; le Polain de Waroux and Lambin, 2013). For example, Indonesian coffee farmers received increased prices and revenues, as well as access to knowledge on coffee production, processing and new technologies through the direct relationships with private exporters and retailers (Neilson, 2014; Neilson and Shonk, 2014). However, the impacts of specialty markets on smallholder livelihoods and environment remain contentious.

4.2.3 Hypotheses

We tested four hypotheses (Table 4.1) to answer our research question, namely what are the socioeconomic and environmental outcomes for smallholders participating in direct cocoa trade compared to farmers in the mainstream cocoa market in northern Ecuador? We expected smallholders engaged in direct trade to capture a larger portion of the value added through superior prices paid for quality (*hypothesis 1*) and to receive more nonmonetary benefits such as technical assistance, improved market access, and improved social networks (*hypothesis 2*) than smallholders selling cocoa through mainstream markets. We expected smallholders engaged in direct trade to adopt more nature-friendly management practices such as using organic fertilizers, shading by non-cocoa trees, and restricting the use of chemicals (*hypothesis 3*) and to maintain more Nacional than CCN-51 cocoa varieties, higher non-cocoa species richness and abundance, and increased shade levels in plantations (*hypothesis 4*) than smallholders selling cocoa through mainstream markets.

Table 4.1: Hypotheses tested in this study

Hypothesis	Compared to smallholders who sell cocoa to mainstream markets, we expect that smallholders engaged in direct markets …
Smallholder livelihoods	
1	…capture a larger portion of the value added through price premiums for quality
2	…receive more nonmonetary benefits such as technical assistance, improved market access, or investments into their social networks
Nature-friendly management practices and biodiversity conditions	
3	…adopt more nature-friendlier management practices, such as the use of organic fertilizers, shading by non-cocoa trees, and the restricted use of chemicals
4	…maintain diversity of both cocoa and non-cocoa tree species, and shade-levels associated with traditional agroforestry systems as measured at farm-level

4.3 Methods

4.3.1 Study area

We conducted our study in the provinces of Esmeraldas and Santo Domingo de los Tsáchilas in northern Ecuador, which are responsible for 17 per cent of national cocoa production (SINAGAP, 2015). This region is recognized as a main origin of high-quality Nacional beans. It is also among the poorest regions in Ecuador with the highest percentage of smallholders owning less than 5 hectares of land (ICCO, 2016; ESPAC, 2014). A majority of cocoa producers in this region still cultivate cocoa in traditional agroforestry systems, but struggle with limited market access and low productivity in aged plantations. International buyers have shown particular interest in this region to establish direct trade initiatives, increasingly branding chocolate bars as 'Ecuador' or 'Nacional'.

4.3.2 Quantitative analyses

4.3.2.1 Household surveys

We conducted a survey with 57 households during August and September 2015 (Fig. 4.2), to collect data on socioeconomic conditions and management practices of cocoa smallholders participating in mainstream market and direct trade initiatives. We selected three farmers' cooperatives that established direct trade connections with national or international buyers. We surveyed 27 households that sold most of their cocoa through these cooperatives (labeled 'treatment group'). We also surveyed 30 households that sold cocoa to three other cooperatives selling cocoa on the mainstream market or to local independent buyers (labeled 'control group').

We applied a snowball sampling strategy to identify interviewees. With the help of farmers' cooperatives, local market actors, and governmental institutions, we randomly selected an initial household per fieldwork day and subsequently localized nearby members of the cocoa farmer community. During the course of data collection, we increased the number of interviewees in the control group with comparable farm size to smallholders selling through direct trade to diminish sampling bias. This increased the likelihood of a successful posterior matching of pairs. The survey included socioeconomic variables related to land use and management practices, farmer demographics, cocoa varieties cultivated, and land-use decision-making process.

4.3.2.2 Plantation surveys

We randomly established a north facing 20 x 50 m (0.1 ha) plot on 75 per cent of the surveyed cocoa farms to assess tree species richness and forest structure. We recorded woody species richness as the number of different woody species >2 m within the plot. Along the longitudinal 50 m plot midline, we classified: (i) ground cover as saplings, weeds, dead leaves, or bare soil at each 0.5 m, and (ii) shade levels as canopy cover above cocoa trees ranging from <25, 25-50, 50-75, and >75 per cent at each 10 m. We recorded the complexity of forest structure as the presence or absence of four height strata (<0.5 m, 0.5 – 1.5 m, 1.5 – 3 m, and >3 m). Within four 10 m quadrants located at the corners of each plot, we measured diameter at breast height (dbh) and height of each woody individual >2 m.

Figure 4.2: Locations of the surveyed households engaged in mainstream (triangles) and direct (squares) cocoa trade from Esmeraldas and Santo Domingo, Ecuador

4.3.2.3 Matching analyses

We recognize that the most reliable method to detect treatment effects is a randomized sample selection model, which would be difficult to implement in a setting with widely dispersed smallholders, limited geographical information, and high data collecting costs. Smallholders that already meet requirements for alternative markets have strong incentives to self-select into these markets, which results in a selection bias. A credible evaluation of the impacts of differentiated market participation must therefore construct a counterfactual outcome (Blackman and Rivera, 2010). The counterfactual estimated the socioeconomic and environmental outcomes for treatment group observations, had they sold their beans on the mainstream market. Many ex post statistical methods are available to estimate such a counterfactual, but a common method of selection and adjustment is matched sampling (Rosenbaum and Rubin, 1983, 1985). Matched sampling is able to estimate effects even with a relative small sample population, due to its refined control observation selection (Quigley, 2003). Matched sampling creates a control group of modest size from a larger pool of potential controls that resembles the treatment group with respect to a selection of observed covariates. Matching on a correct propensity score model thus means that the treatment and control groups have the same joint distribution of observed covariates (Rosenbaum and Rubin, 1983).

Here, we estimated the average treatment effect on the treated (ATT) by one-to-one pairwise matching with replacement, using the genetic matching algorithm (i.e., GenMatch) in the matching package in R (R Development Core Team, 2008; Sekhon, 2011) that optimizes post matching covariate balance (Diamond and Sekhon, 2013). Standard errors were Kolmogorov-Smirnov bootstrapped with 1,000 replications, following Dehejia and Wahba (2002). We conducted several nearest-neighborhood matching methods to test the robustness of genetic matching outcomes (Caliendo and Kopeinig, 2008; Holmes and Olsen, 2010), which resulted in similar qualitative conclusions as presented here (Table C.1- C.3).

4.3.2.4 Covariate and outcome variable selection for matching

We selected five farm and grower characteristics on which we matched mainstream and direct households (i.e., covariates): total farm size (in ha); area cultivated in cocoa (in ha); altitude of the farm (in masl); education of the household head (in number of years of school attendance); and family size (in number of family members in the household) (Table 4.3). Farm size and area in cocoa affect the grower's production potential and her/his access to additional resources. Elevation influences biophysical conditions for cocoa production and market access. Education and family size represent important aspects of the growers' ability to learn and implement specific management practices required for alternative market participation.

Outcome variables were grouped based on: (i) economic benefits; (ii) non-monetary benefits; (iii) management practices; and (iv) biodiversity conditions. Variables in the first three groups originated from household survey data and the fourth group from farm survey data. Farm-gate prices were determined as the average price received for wet cocoa beans over the last three months as indicated by the producer. Due to the mixing of beans from different varieties in the mainstream market, we were unable to segregate pricing based on cocoa varieties. Before matching, we compared outcome variables between households from both test groups using Wilcoxon-rank-sum tests for continuous variables and Fisher's exact tests for binary variables (Table 4.3). Matching was done independently for household and farm survey data, as the latter were only collected for 75 per cent of the participants. For all analyses, all covariates achieved balance without dropping treatment observations (Table 4.2).

4.3.2.5 Underlying matching assumptions and limitations

Although matching is the analog of randomization in ideal observation experiments, it can only balance the distribution of observed covariates (Rubin and Thomas, 2000). All matching methods assume underlying 'confoundedness' and require therefore that treatment selection occurs based on the selected covariates. Matching cannot directly account for unmeasured third factors associated with both potential outcomes and treatment selection (Imbens, 2015; Imbens and Wooldridge, 2009). Different matching estimators provide somewhat different interpretations of causal effects, but no 'best' practice has yet been identified (Morgan and Winship, 2014). We tested for the sensitivity of results from matching estimators to hidden bias with Rosenbaum bounds, although these tests cannot directly detect hidden bias. There is a risk of endogeneity between unobserved variables and selected covariates or outcome variables that matching cannot account for. Therefore, caution is required when inferring strong causality between treatment participation and outcomes.

4.3.3 Qualitative analyses

We conducted additional qualitative analyses to address the limitations in matching analyses and interpret our quantitative results. Firstly, unstructured interviews were conducted with private, public, and market actors. Public actors included governmental officers involved in state-led cocoa market initiatives (*Minga del Cacao and Proyecto de Reactivación del cacao Fino de Aroma* initiated by the Ecuadorian Ministry of Agriculture, Livestock, Aquaculture, and Fisheries), Ecuadorian NGO officers (*Conservación and Desarrollo* and VECO Andino), and two Ecuadorian scientists (*Escuela Superior Politécnica del Litoral*, Guayaquil and *Instituto Nacional de Investigaciones Agropecuarias*, Santo Domingo). Private actors included presidents of all farmers' cooperatives included in this study, several mainstream intermediaries working in the field, two managers of mainstream cocoa collection centers, two Ecuadorian chocolatiers, and one international exporter. Insights from these interviews supported interpretation of field observations.

Secondly, a workshop was organized in collaboration with the Ecuadorian NGO VECO Andino on 27 July 2016 in Quito. The aim of this workshop was to bring together international and national market actors to discuss economic, social, and environmental sustainability of the Ecuadorian cocoa chain. The main questions addressed during roundtable sessions were: (i) how to improve the relationship between smallholders and the private sector? (ii) how to enhance the inclusion of smallholders and young farmers? and (iii) how to incentivize environmentally-friendly management on cocoa plantations?

Finally, we relied on secondary sources, including scientific, governmental, and private reports, to gain insights on long-term trends in cooperative export prices.

4.4 Results

4.4.1 Quantitative results

4.4.1.1 Cocoa smallholders in northern Ecuador

Interviewees owned on average about 16 hectares of land with six hectares cultivated with cocoa trees (Table 4.3). Households engaged in direct trade tended to own less land with a significantly smaller area cultivated with cocoa compared with households engaged in mainstream trade. Cocoa provided the main source of income for approximately 80 per cent of the surveyed households. Production diversification was limited to trading fruits (bananas, maracuya, guavas) or livestock (chickens, pigs) on local markets. Few farmers had access to credit, additional labor, or agricultural inputs, such as insecticides, fungicides, or herbicides.

Direct trade arrangements and certification schemes were partially overlapping in our study region. Over 80 per cent of farmers engaged in direct trade were also organic certified, compared with no organic certified farmers selling through mainstream markets. As a result, it is impossible to attribute to direct trade rather than to organic certification the observed differences in outcome variables between treatment and control groups. In contrast, an equal share of farmers from direct and mainstream trade groups were Fairtrade certified. We conducted an additional matching analysis comparing Fairtrade and non-Fairtrade farmers to evaluate the impact of Fairtrade certification on selected outcome variables, independently from the marketing strategy (Table 4.4). This second analysis showed that the impacts of direct trade on outcome variables were unrelated to Fairtrade certification. We found no indication that Fairtrade improved farmers' well-being or stimulated the adoption of nature-friendly farming. For example, prices for Fairtrade and non-Fairtrade cocoa were similar. Cooperatives did capture a price premium for Fairtrade production that they invested in communal goods and services, such as farmers' pension funds or post-harvesting capacity at cooperative level.

4.4.1.2 Socioeconomic outcomes

We found that high-quality cocoa beans captured superior prices in the direct trade market compared with bulk cocoa sold through the mainstream chain (Table 4.4), thus supporting hypothesis 1. Farmers stated that they received

Table 4.2: Test for covariate balance before and after genetic matching on household and farm surveys. Balance is achieved for all covariates, as indicated by an insignificant Kolmogorov-Smirnov test value after matching, based on 1,000 bootstrap samples

Covariate	Household survey data		Farm survey data	
	Before matching	After matching	Before matching	After matching
Farm size (ha)	0.17	0.15	0.27	0.24
Area in cocoa (ha)	0.32*	0.15	0.41*	0.14
Altitude (masl)	0.19	0.22	0.22	0.24
Education hh (yrs)	0.23	0.11	0.18	0.14
Family size	0.15	0.11	0.25	0.19

* $p < 0.05$

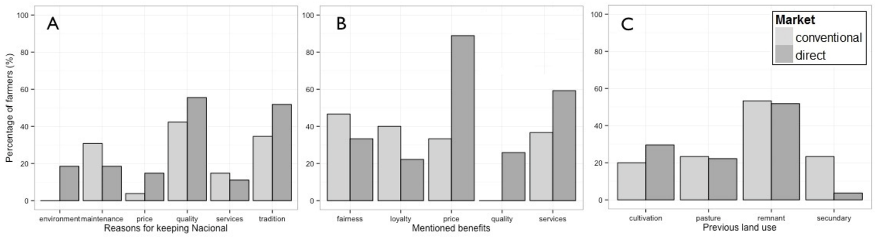

Figure 4.3: Qualitative survey responses of smallholders engaged in direct or mainstream trade on: (a) reasons for keeping Nacional varieties; (b) mentioned benefits of cooperative membership; and (c) previous land cover of plantations expansions, if applicable

price premiums for both quality and certification. Consistent with hypothesis 2, more households engaged in direct trade received technical assistance in the form of agricultural training and on-farm assistance by farmers' cooperatives or governmental institutions (Table 4.4). Farmers participated in the social networks of farmers' cooperatives to access direct markets. Besides improved prices, farmers engaged in direct trade regularly mentioned other benefits of cooperative membership, such as cocoa quality enhancement, access to logistic or technical assistance, better information, and provision of seeds (Fig. 4.3a, b).

4.4.1.3 Environmental outcomes

More smallholders engaged in direct trade applied nature-friendly management practices compared with smallholders engaged in mainstream market

(Table 4.4), supporting hypothesis 3. Significantly more farmers engaged in direct trade applied organic fertilizer and avoided chemical fertilizers or herbicides compared with the control group. Most farmers preferred to plant Nacional trees, but more farmers engaged in direct trade confirmed the presence of Nacional trees on their farms. The majority of the farms were shaded with non-cocoa trees, but farmers engaged in direct trade mentioned a significantly greater number of tree species during interviews than farmers selling through mainstream markets. Farmers from both groups who expanded their plantation in the last five years did so by replacing remnant forest, with more plantations of farmers engaged in mainstream trade that replaced secondary forests (Fig. 4.3c). This indicates that deforestation may be related to cocoa plantation expansion in the region.

Even though farmers engaged in direct trade adopted more nature-friendly management practices, we did not find a difference in biodiversity conditions of the tree cover at farm-level compared to farmers selling through mainstream markets (Table 4.4), thus not supporting hypothesis 4. We found comparable species richness, abundance, and basal area for non-cocoa trees on farms owned by households from both test groups. The density of Nacional trees was comparable on farms from both test groups, while only the density of CCN-51 plots was significantly higher on farms from households engaged in mainstream trade.

4.4.2 Qualitative results

Interviews with presidents of the farmers' cooperatives that established direct relationships with multiple buyers confirmed that cooperatives were increasing farmer's price for cocoa due to higher prices from buyers. Part of these premiums was invested in improvements of cooperative's infrastructure, such as drying or fermentation installations, or to contract local labor for transport or extension and technical services, directly benefitting member farmers. Although several buyers stated that their main interest for direct relationships was access to high-quality cocoa beans, many favored beans with organic or Fairtrade certification to satisfy consumer demand for chocolates with an environmental or social label. Cooperatives invested some of the price premiums for high-quality beans to cover certification costs, resulting in a reduction of the price transfer to farmers. Data collected by PRAGMATICA (2016), comparing cocoa bean export volumes and sales from mainstream exporters and farmers' cooperatives at the national level, showed that farmers' cooperatives participating in this study captured higher prices for Arriba Superior Epoca (ASE) quality beans during some years (Fig. 4.4a). Our field observations confirmed that these prices were reflected in higher farm gate-prices compared to mainstream farmers. Total export volumes showed that farmers' cooperatives mainly focused on Nacional cocoa beans, whereas CCN-51 beans comprised over 50 per cent of exports through mainstream markets between January 2012 and September 2015 (Fig. 4.4b).

Participants from the workshop recognized the need to consolidate the sustainability of the Ecuadorian cocoa market. Ecuador has a unique position on the global cocoa market by being recognized as the main producer of Nacional fino de aroma beans. According to participants, a growing number of cases of direct associative commercialization exist, which successfully improve sustainability. Combined with a growing demand for high-quality chocolates in specialty markets, this offers opportunities for further differentiation in the Ecuadorian cocoa value chain.

Nonetheless, participants expressed concern about the decreasing quality of Nacional beans and the lack of smallholder and intergenerational inclusion in the value chain. Most Nacional producing smallholders struggle economically because of low productivity on aged plantations, poor access to genetic material, and lack of agricultural inputs and assistance. Participants recognized that an emphasis should be placed on improving cocoa quality to exploit the country's competitive benefit, requiring a long-term vision of both producers and buyers for direct trade relationships to succeed. It was recognized that the Ecuadorian Ministry of Agriculture, Livestock, Aquaculture, and Fisheries could play a key role in strengthening the competitive advantage of Nacional producers by setting a price difference between Nacional and CCN-51 beans, improving smallholder access to genetically improved Nacional clones with high yield potential, and promoting commercialization based on specific cocoa flavors and production methods. Although smallholders expressed a strong preference for agroforestry systems, they felt that environmentally-friendly management practices were not sufficiently valued and rewarded economically by buyers.

4.5 Discussion

4.5.1 Smallholder livelihoods

We found that smallholders engaged in direct trade captured about 70 per cent per pound dry cocoa beans above average farm-gate prices for mainstream cocoa. This finding supports other studies that have found greater earnings and reduced price volatility for high-quality products sold through differentiated market channels (Reardon et al., 2009; Rueda and Lambin, 2013a; Neilson, 2014; Borrella et al., 2015). Nevertheless, total household income needs to be considered to assess the overall economic impact on households. Superior

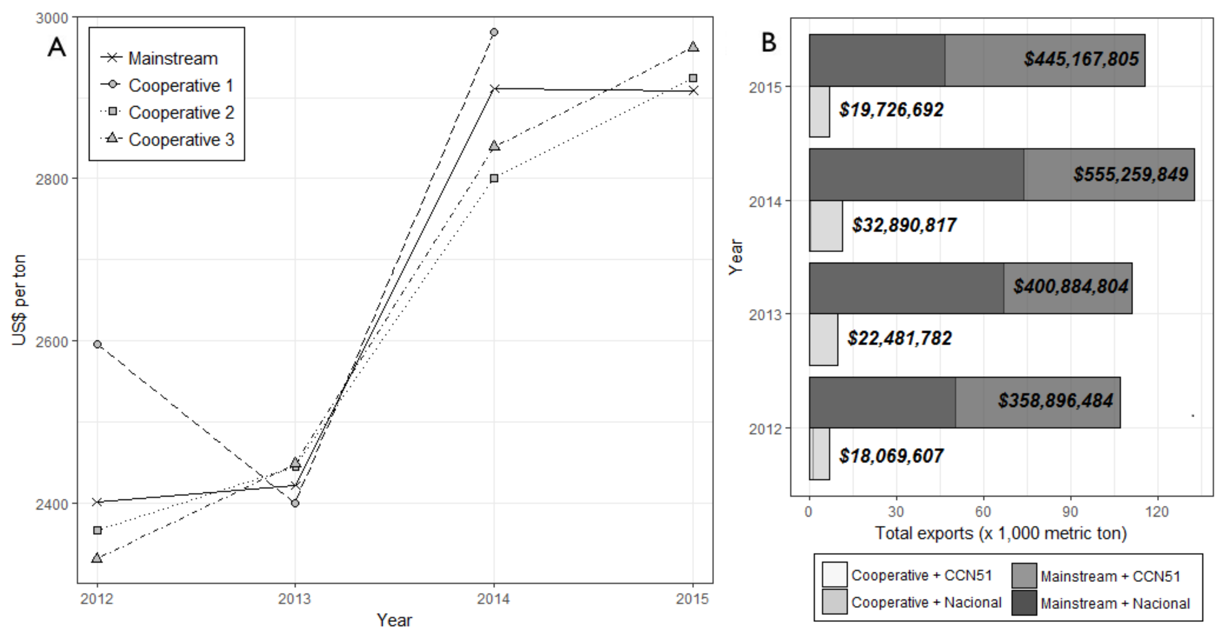

Figure 4.4: (a) Average market price (in US$) per ton of cocoa beans for mainstream cooperatives (ASE quality; solid line) and three farmers' cooperatives visited in northern Esmeraldas engaged in direct trade relationships with buyers between 2012 and September 2015. (b) Total volumes (in 1,000 metric ton) exported by mainstream exporters (dark grey) and small farmers' cooperatives (light grey) for CCN-51 and Nacional beans between 2012 and September 2015. Accumulated market prices (in US$) for all exports are shown. Data from PRAGMATICA (2016)

prices for differentiated products can fail to increase per hectare gross margins and profits due to low yields, production costs, large family size, and land or labor constraints (Beuchelt and Zeller, 2011; Ruben and Fort, 2012; le Polain de Waroux and Lambin, 2013; Schipmann and Qaim, 2010). Few studies have explored the impacts of market participation on total household income. In addition to raising rural incomes, quality-standards trade can also alleviate poverty by stimulating local labor markets (Maertens and Swinnen, 2009). Research suggests that improving food security and reducing poverty over the medium term can be achieved through increasing farmers' income as well as agricultural labor productivity (de Janvry and Sadoulet, 2010). This might be the case in northern Ecuador, as farmers' cooperatives hired local workforce to participate in transport and extension services.

Besides superior prices, we found that smallholders engaged in direct trade had greater access to agricultural training, technical assistance, and improved social networks through cooperative membership, compared to smallholders selling through mainstream markets. Our results are consistent with other studies that have shown that farmers engaged in direct trade had more access to credit, a better understanding of the market, a more optimistic view of the future, and increased homogeneity and quality of fermentation and drying processes through cooperative-managed processing units (Hernandez-Aguilera et al., 2015; Neilson, 2014). Several studies have raised concerns about the exclusion of small farmers from high-quality value chains, due to initial quality adoption constraints affecting disadvantaged farmers (Reardon et al., 2009). Non-land assets such as inputs, credits, membership of farmers' cooperatives, and extension services can be important entry barriers for differentiated markets. This underlines the importance of interventions to facilitate compliance, such as training and finance, to guarantee success of upgrading.

4.5.2 Nature-friendly management practices and on-farm biodiversity

We found that farmers engaged in direct trade adopted more nature-friendly management practices, such as the use of organic fertilizers and less CCN-51 monocultures. Despite the adoption of different management practices, we observed insignificant differences in shade levels and non-coca tree species richness and abundance in plantations of farmers engaged in direct and mainstream trade. Positive impacts on biodiversity and vegetation structure may lag behind direct trade participation. A more likely explanation is that most interviewees were small farmers who maintain traditional agroforestry systems due to insufficient financial assets to switch to more intensive production systems. In an attempt to increase yields by reducing shade, many farmers deliberately fell non-cocoa trees in traditionally shaded agroforestry systems. Even though many scholars have found that cocoa growth and production decreased non-linearly with increasing shade (Steffan-Dewenter et al., 2007; Gockowski et al., 2011; Wade et al., 2010; Blaser et al., 2017), others have shown that intermediate levels of canopy shade of 40 to 60 per cent protected cocoa trees from drought (Abou Rajab et al., 2016) and allowed to maintain high cocoa production (Waldron et al., 2012). Intermediate shade levels might thus be combined with high yields (Clough et al., 2011), given appropriate management knowledge and more labor-intensive farming strategy (Somarriba et al., 2013; Andres and Bhullar, 2016). Useche and Blare (2013) found that Ecuadorian cocoa producers preferred agroforestry production systems to more intensified systems when nonmarket ecological and social benefits are accounted for.

The absence of an observable biodiversity benefit following direct trade participation underlines the need to include specific ecosystem criteria in contracts between producers and buyers. Contract farming may also reduce market risks for smallholders (Henson et al., 2005). We found indications that continued clearance of remnant forest is associated with cocoa agroforestry. Even though shaded agroforestry systems provide many ecosystem services, they do not equate to primary forests. Eliminating deforestation for cocoa plantations needs to be an explicit standard to prevent misinforming consumers on the conservation value of cocoa agroforestry.

4.5.3 Improving Ecuadorian value chains: keys to success

Farmers' cooperatives played an important role in improving the bargaining position, establishing agreements with buyers, and appropriating management knowledge of smallholders in our study region. Research suggests that certification may play an important role for smallholders to gain access to niche markets, including direct trade, through improvements of economic, social, and environmental conditions, following the standards-as-catalyst view proposed by Jaffee and Henson (2005). For example, Rueda et al. (unpublished manuscript) have explored the causal link between certification and upgrading for Ecuadorian cocoa farmers, suggesting that certification may serve as a 'stepping-stone' for smallholders to enter high-value niche markets.

This idea is supported by our observations that direct trade engagements and certification are largely overlapping in our region, with certification schemes preceding direct trade relationships in most cases and direct buyers demanding certification in some instances. Farmers, cooperatives, and buyers stated during interviews that price premiums for quality were offered on top of price premiums for organic cocoa, suggesting that the economic impacts we observed were unlikely caused by organic certification alone. Fairtrade certification alone could not explain positive outcomes of direct trade relationships but we were unable to separate outcomes deriving from organic certification. Furthermore, farmers, cooperatives, and some market actors confirmed in interviews that the organizational improvements, upgraded post-harvesting processes, and improved management practices following quality recommendation by buyers were made possible by initial access to certification. Access to niche markets for fine cocoa requires improved production methods and post-harvest processing systems, which are unavailable to smallholders and cooperatives with low asset endowment. Certification schemes may support necessary quality improvements as participating farmers comply with standards for farm management practices, including the pre-selection of fungus-free beans, and an adequate agrochemical use. Quality is essential for consumers looking to buy Nacional chocolates, as they pay a price premium primarily for quality and then for certification (Cepeda et al., 2013). Entry to niche markets is likewise facilitated by a certain level of infrastructure, agricultural inputs, and technical skills, often associated with success in certification schemes (Donovan and Poole, 2013).

Melo and Hollander (2013) have suggested that differentiated cocoa trade may be successful if: (i) premiums for Nacional beans compensate for the lower productivity compared to CCN-51; (ii) farmers' collectivism is stimulated through investments in social networks; and (iii) farmers refrain from replanting CCN-51 following incentives to conserve traditional plantations. In a matched case comparison of two certified rural cooperatives in Ecuador, Tampe (2016) found that standards failed to improve conditions of farmers but suppliers could leverage standards to create value from vertical relationships with buyers. Only one of the certified cooperatives in that study was able to become competitive in an upgraded market with significant price premiums. The key to success was developing a close and learning-oriented relationship with the buyer, investing in multiple redundant buyer ties and recursively appropriating knowledge (Tampe, 2016). The mistrust in the value chain following bad commercialization experiences and contractual insecurity, as well as weak and insufficient farmers associations are important barriers to successful upgrading in the Ecuadorian value chain (Ahmed and Hamrick, 2015; Lehmann and Springer-Heinze, 2014).

4.6 Conclusion

Globalization continuously expands the assortment of exotic commodities produced worldwide for the tastes of wealthy consumers. These new niche commodity markets create a more direct linkage between consumers' choices, and livelihoods and ecosystems in places of production. Our study suggested that consumers who prefer chocolate made from directly sourced high-quality cocoa beans might help to enhance smallholder livelihoods and the adoption of some nature-friendly management practices. In addition, certification standards may provide smallholder with easier access to high-value niche markets. Additional research is needed to test this hypothesis, as this study was unable to separate outcomes from organic certification and direct trade markets. Outcomes for farmers' overall welfare and ecosystems thus remain contentious. Longitudinal studies are needed to reveal long-term socioeconomic and environmental outcomes. Producers, NGOs, private companies, and governments play complementary roles in governing market innovations in the cocoa value chain.

Table 4.3: Covariates and outcome variables, means, and p-value for unmatched observations for direct (treatment) and mainstream (control) households surveyed. Covariates served as input for matching

Variable	Mean all	Mean treatment (n = 27)	Mean control (n = 30)	Test p-value †
Covariates				
Farm characteristics				
Farm size (ha)	15.99	12.86	18.80	0.33
Area in cocoa (ha)	5.76	4.46	6.92	0.02*
Altitude (masl)	117.9	130.3	106.8	0.30
Producer's characteristics				
Education hh (yrs)	7.56	8.41	6.80	0.07
Family size	5.95	5.96	5.93	0.96
Outcome variables				
Economic benefits (hypothesis 1)				
Wet price (ct pound^{-1})	46.38	58.54	35.43	<0.001***
Premium for certification (1/0)	0.37	0.70	0.07	<0.001***
Premium for quality (1/0)	0.49	0.93	0.10	<0.001***
Nonmonetary benefits (hypothesis 2)				
Access to credit (1/0)	0.04	0.04	0.03	1.00
Technical assistance (1/0)	0.74	0.93	0.57	0.003*
Farmers' cooperative membership (1/0)	0.77	1.00	0.57	<0.001***
Workers hired on farm (1/0)	0.44	0.56	0.33	0.11
Cocoa as the main source of income (1/0)	0.77	0.78	0.77	1.00
Management practices (hypothesis 3)				
Organic fertilizer (1/0)	0.47	0.67	0.30	0.008*
Chemical fertilizer (1/0)	0.11	0.00	0.20	0.03*
Insecticides (1/0)	0.05	0.00	0.10	0.24
Fungicides (1/0)	0.02	0.00	0.03	1.00
Herbicides (1/0)	0.28	0.11	0.43	0.009**
Presence of Nacional (1/0)	0.93	1.00	0.87	0.11
Presence of CCN-51 (1/0)	0.42	0.22	0.60	0.007**
Preference for Nacional (1/0)	0.67	0.78	0.57	0.16
Shade of non-cocoa trees (1/0)	0.96	1.00	0.93	0.49
Non-cocoa tree species richness stated by farmer (1/0)	4.33	5.04	3.70	<0.001***
Irrigation (1/0)	0.05	0.00	0.10	0.24
Grafting of cocoa trees (1/0)	0.54	0.44	0.63	0.19
Pruning of cocoa trees (1/0)	0.91	0.96	0.87	0.36
Used fire before planting (1/0)	0.16	0.04	0.27	0.03*
Area in cocoa increased in past 5 years (1/0)	0.51	0.52	0.50	1.00
Biodiversity conditions (hypothesis 4)				
Non-cocoa species richness (species ha^{-1})	6.95	7.57	6.36	0.46
Abundance of non-cocoa trees (ha^{-1})	21.81	25.29	18.50	0.09
Basal area of non-cocoa trees (m^2 ha^{-1})	52.12	53.40	50.89	0.65
Average height of non-cocoa trees (m)	12.97	13.41	12.55	0.62
Average shade level index	2.1	2.3	1.90	0.32
Nacional density (trees ha^{-1})	619.8	627.4	612.5	0.85
CCN51 density (trees ha^{-1})	140.7	11.00	275.0	<0.001***

*** P < 0.001; ** P < 0.01; * P < 0.05;

† The Wilcoxon-rank-sum test p-value was calculated for continuous variables; the Fisher's exact p-value was calculated for binary variables.

Table 4.4: Estimates of the average treatment effect on the treated (ATT) for matched observations of direct vs. mainstream (*treatment 1*) and fairtrade vs. non-fairtrade (*treatment 2*) households surveyed, using genetic matching. The critical value of Rosenbaum's Γ is provided

Outcome variables	Direct vs. mainstream trade				Fairtrade vs. non-fairtrade			
	ATT	S.E.	t-test	Γ †	ATT	S.E.	t-test	Γ †
Economic benefits (hypothesis 1)								
Wet price (ct pound^{-1})	24.35	0.75	25.18***	7.7	4.80	5.00	0.96	-
Premium for certification (1/0)	0.67	0.13	5.77***	6.7	0.26	0.12	2.15*	2.6
Premium for quality (1/0)	0.78	0.10	6.43***	7.9	0.00	0.22	0.00	-
Nonmonetary benefits (hypothesis 2)								
Access to credit (1/0)	0.00	0.08	0.00	-	-0.22	0.11	-1.95	-
Technical assistance (1/0)	0.30	0.12	2.43*	3.0	0.22	0.16	1.43	-
Farmers' cooperative membership (1/0)	0.70	0.11	5.77***	7.1	0.30	0.13	2.36*	3.0
Workers hired on farm (1/0)	0.22	0.18	1.87	-	0.19	0.20	0.93	-
Cocoa as the main source of income (1/0)	-0.07	0.13	-0.51	-	-0.19	0.13	-1.42	-
Management practices (hypothesis 3)								
Organic fertilizer (1/0)	0.48	0.16	2.86**	2.0	0.52	0.17	2.99*	2.5
Chemical fertilizer (1/0)	-0.22	0.10	-3.10**	2.3	0.04	0.09	0.41	-
Insecticides (1/0)	-0.18	0.09	-1.06	-	-0.11	0.09	-1.29	-
Fungicides (1/0)	0.00	0.00	0.00	-	0.00	0.00	0.00	-
Herbicides (1/0)	-0.37	0.12	-2.94**	3.7	0.00	0.00	0.00	-
Presence of Nacional (1/0)	0.30	0.11	2.58**	3.0	0.00	0.07	0.00	-
Presence of CCN-51 (1/0)	-0.26	0.12	-2.18*	1.7	0.19	0.18	1.81	-
Preference for Nacional (1/0)	0.19	0.16	1.87	-	0.11	0.17	-1.03	-
Shade of non-cocoa trees (1/0)	0.11	0.08	0.73	-	-0.04	0.05	-0.71	-
Non-cocoa tree species richness stated by farmer (1/0)	2.37	0.65	3.67***	3.0	1.33	0.75	0.88	-
Irrigation (1/0)	0.00	0.00	0.00	-	0.07	0.07	1.03	-
Grafting of cocoa trees (1/0)	-0.04	0.17	0.20	-	0.37	0.16	1.47*	1.9
Pruning of cocoa trees (1/0)	0.14	0.11	1.24	-	0.30	0.14	1.93*	1.9
Used fire before planting (1/0)	-0.19	0.11	-0.73	-	-0.11	0.15	-0.53	-
Area in cocoa increased in the past 5 years (1/0)	0.07	0.16	0.20	-	0.00	0.18	0.00	-
Biodiversity conditions (hypothesis 4)								
Non-cocoa species richness (species ha^{-1})	2.10	1.65	1.27	-	2.27	1.79	0.74	-
Abundance of non-cocoa trees (ha^{-1})	4.33	5.41	0.75	-	8.53	4.85	0.78	-
Basal area of non-cocoa trees (m^2 ha^{-1})	5.96	30.24	0.44	-	2.11	27.9	0.11	-
Average height of non-cocoa trees (m)	4.85	2.80	1.90	-	2.37	2.57	0.43	-
Average shade level index	0.67	0.34	1.95	1.6	0.59	0.52	0.26	-
Nacional density (trees ha^{-1})	108.3	143.2	0.76	-	-186.7	127.4	-2.40	-
CCN51 density (trees ha^{-1})	-297.6	126.1	-2.36*	3.6	5.00	102.7	0.63	-

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$; † Critical value of odds of differential assignment to direct market due to unobserved factors (i.e., value above which ATT is no longer significant).

Chapter 5

Conclusion

A major challenge for sustainable development remains the reconciliation of biodiversity conservation, ecosystem services' supply, and human well-being in tropical agricultural landscapes. Forest landscape restoration (FLR) provides a unique opportunity to achieve these development goals. The lack of scientific understanding however, limits our potential to design effective FLR initiatives. This dissertation aimed to better understand the potential of FLR methods and economic instruments to restore biodiversity and enhance livelihoods in human-dominated tropical landscapes. First, Chapter 2 explored the relationship between forest restoration methods and the potential to support biodiversity, accounting for the impact of different landscape characteristics. Secondly, Chapter 3 examined the potential of cocoa agroforestry systems to achieve FLR objectives, focusing on landscape impacts of different management strategies on aboveground biomass, biodiversity, and economic revenues in northern Ecuador. Finally, Chapter 4 assessed the socioeconomic and environmental outcomes of smallholder participation in specialty markets for high-quality cocoa beans in northern Ecuador.

Here, I will first summarize the main findings presented in this dissertation by answering the main research questions: (i) What is the potential of different restoration types to achieve FLR objectives? (ii) How does the landscape-level pattern of restoration affect FLR objectives? (iii) How can market-based mechanisms support FLR objectives? This dissertation focuses on biodiversity conservation, ecosystem services supply, and human well-being as the main FLR objectives. Then, I will discuss the main contributions of this work with emphasis on both methodological and policy relevance. Finally, I will briefly point out some of the main limitations and the future prospective.

5.1 Restoration through natural regeneration, plantations, and agroforestry

Natural regeneration, targeted tree plantings, and agroforestry are dominant land-uses throughout the tropics and offer an enormous scope for forest restoration. These land-use systems play an important role in mitigating global carbon emissions, safeguarding water supplies, preventing soil erosion and flooding, and provide resources to many species, including human beings (see section 1.2 to 1.4). The main findings of Chapter 2 suggest that natural forest regeneration has a large potential to restore species richness and composition after abandonment of agricultural fields, especially in comparison to plantings of exotic pine trees. But the recovery of plant biodiversity was context dependent. Besides the important influence of landscape characteristics during regeneration, such as surrounding forest cover, land-cover diversity, and altitudinal range, local and landscape diversity positively related to time since abandonment. This suggests that second growth forest require sufficient time to recover species richness and composition after restoration onset. Although we found second growth forests contained a large species diversity and positively affected diversity at the landscape scale, their composition was not a mirror image of 'original' old-growth forest. Furthermore, planted exotic pine forests prevented the establishment of all other species, diminishing both local and landscape diversity.

The main findings of Chapter 3 suggest that maintaining shade trees on cocoa plantations might have substantial environmental benefits. Shaded cocoa agroforestry management increased the total biomass, preserved functional species diversity on plantations, and decreased fragmentation of forested areas at the landscape scale, compared to unshaded monocultures. Converting all cocoa plantations to monocultures resulted in a significant decrease in total landscape aboveground biomass compared with baseline projections. In contrast, converting cocoa plantations to traditional shaded agroforestry systems significantly increased total landscape biomass estimates as well as functional diversity.

The results from Chapter 4 indicated that specialty markets for high-quality cocoa beans incentivized the use of nature-friendly management practices (e.g., use of organic instead of chemical fertilizers, refrain from herbicide use), compared to mainstream markets. Setting aside land in combination with an monoculture intensification strategy resulted in similar aboveground biomass levels as our baseline projections and an increase in landscape connectivity, while increasing estimated revenues from cocoa production. Conservation, sustainable management, and landscape restoration are complementary and need to be aligned to guarantee long-lasting environmental, social, and economic benefits.

5.2 Forest restoration at the landscape-level

The results in this dissertation suggest that second growth forests and small-scale agroforestry systems may improve the quality of the landscape matrix and thus stimulate biodiversity and biomass accumulation at both local and landscape scales. The type of landscape matrix plays an important role in determining the success of forest restoration activities. In Chapter 2, evidence is presented that landscape attributes play an important role in the recovery of species richness and composition during forest restoration. Remnant vegetation proved an essential seed source that determined biodiversity outcomes of natural regeneration. Isolation effects reduced the biodiversity recovery in restoration area if natural forest remnants were located further than 400 m and natural forest cover dropped below a 30 per cent threshold. Heterogeneous landscapes (i.e., spatial mix of different land cover types) enhanced the accumulation of species during natural forest regeneration and improved the landscape level outcomes of biodiversity restoration. Restoration in mountainous areas was influenced by an altitudinal component, as species are restricted to elevation gradients, limiting their establishment and dispersal abilities.

The impact of potential land sparing from land abandonment following intensification of cocoa plantations was explored in Chapter 3. Abandonment of 40 per cent of cocoa agroforestry farms in combination with a conversion to monoculture plantations resulted in similar levels of aboveground biomass but a doubling of revenues at the landscape compared to baseline projections. Even when including additional revenues from carbon markets, we found that at current carbon market price (i.e., US$5 per hectare per ton CO_2), the income gap between monoculture and shaded agroforestry systems could not be filled at the landscape scale. Shaded plantations and land set aside for natural regeneration reduced forest fragmentation at the landscape-level. We conclude that improving payments for ecosystem services and certification schemes are needed to incentivize smallholders to maintain substantial non-cocoa tree cover that may provide an environmental-friendly way to improve economic potential and food security for smallholders, while supporting biomass and functional species composition at the landscape-level.

The larger landscape-scale context is crucial for the design and management of tropical agricultural systems (Harvey et al., 2014). Current certification schemes focus at the farm or plantation levels, whereas ecosystem benefits from well-designed agroforests are delivered at the landscape-level, emphasizing a scale mismatch (Tscharntke et al., 2015). Scholars have suggested two approaches to address this incongruence: certification mechanisms could be linked with broader landscape-scale approaches (Milder et al., 2014) and current certification models could consider the landscape as a certified unit through landscape labeling (Ghazoul et al., 2009). For example, landscapes that deliver ecosystem services are granted a 'label' that could publicize and with it conserve its cultural and symbolic attributes. Such schemes differs from payments for ecosystem services schemes as a wider range of people may benefit from funds invested in community-based projects (Ghazoul et al., 2009). Today, several issues with potential landscape labals remain unsolved, such as dealing with freeriders, managing conditionality, and avoiding displacement of degrading activities elsewhere. The question remains whether and to what extent such activities will stimulate the adoption of agroforestry practices among farmers.

5.3 Financial instruments of forest landscape restoration

From an economical point of view, smallholders' incentives to maintain shaded agroforestry systems may be limited. Estimated cocoa yields were about seven times higher for monocultures compared with traditional agroforests and current carbon markets are unsuffient to address this yield gap (Chapter 3). Furthermore, current premiums for certified cocoa beans should be about doubled to cover reduced productivity estimates in traditional agroforestry systems. Specialty markets for high-quality cocoa studied in Chapter 4 may provide an interesting incentive for smallholders to maintain traditional agroforestry systems. We found evidence that smallholders participating in specialty markets for high-quality cocoa captured superior prices for cocoa sales and had greater access to agricultural training, technical assistance, and improved social networks, compared to smallholders selling through mainstream markets. Farmers' cooperatives that participated in this study were found to capture prices above mainstream markets in the past five years, supporting empirical results. Nevertheless, sustainable economic benefits from these market channels are uncertain due to the lack of long-term data on overall household income impacts.

A growing consensus among ecologists and resource managers suggests the need for smarter landscape management through sustainable intensification (Fischer et al., 2017). Nevertheless, sustainable management is labor and time intensive and its feasibility on larger scales in developed countries remains questionable (Andres and Bhullar, 2016). Agricultural intensification can result in land spared for preservation and restoration of natural habitat. Nevertheless, intensification of commodity crops, including cocoa and coffee, have been associated with a rebound effect associated locally through the expansion of agricultural land in intensifying regions or globally through the redistribution of agriculture (Hertel et al., 2014; Jadin et al., 2016). It remains debatable how certification and payments for ecosystem services schemes can deliver appropriate and sufficient community-based investments

5.4 Main contributions of this dissertation

5.4.1 Methodological contributions

This dissertation benefitted from a multidisciplinary approach that combined ecological modeling, ecological fieldwork, as well as socioeconomic surveys. We used the well-established forest landscape model LANDIS-II for all ecological modeling efforts. Although LANDIS-II has been implemented in the tropics of South-America before (Cantarello et al., 2011; Newton et al., 2011; Birch et al., 2010), its applicability in the context of land-use change in Ecuador was demonstrated here. Our implementation of model simulations allowed for the testing of a large number of land-use scenarios that differed in restoration methods and landscape patterns over much longer periods than possible in the field. Especially in regions where data is scarce, like the tropics, modeling efforts provide an important framework to test a range of hypotheses. The value of these types of forest models thus lies in their potential to explore the consequences of explicitly stated assumptions, not in the exact prediction of future conditions. In all chapters, ecological modeling was supported by both empirical data collection and the use of open source databases. This supported the calibration and validation of models and strengthened the interpretation of research context and outcomes. The presented framework could be readily implemented in other research areas.

5.4.2 Policy relevance

Natural forest remnants are of utmost importance for biodiversity conservation. We emphasize that the conservation of remnant natural forest patches is crucial and cannot be substituted completely by anthropogenic land-uses.

A considerable fraction of human-modified landscapes no longer serves as a source of seed dispersal due to habitat loss. This may impose a considerable limitation for biodiversity conservation. Nevertheless, excluding human activities from tropical agricultural landscapes would be both unethical and unlikely. Despite all efforts by conservationists, nature reserves only represent a small fraction of biodiversity-rich areas. Natural forest regeneration might hence prove fundamental to future conservation initiatives. Improving the matrix quality in agricultural landscapes will increase landscape connectivity with resulting decreases in the isolation of biodiversity-rich areas (Asare et al., 2014; Perfecto et al., 2010), as our main findings confirmed.

Our findings emphasize that restoration approaches need to account for landscape patterns and context to optimize ecological outcomes. Specifically, spatial targeting of reforestation should take into account the (i) close proximity of remnant forests; (ii) surrounding forest cover percentage; (iii) compositional heterogeneity of the surrounding landscapes; and (iv) diversity in altitudinal gradients of restoration efforts. Not all restoration efforts provide similar benefits and exotic plantations should be restricted to highly isolated and degraded areas. Subsequent management and conservation of regenerating forests is essential, because regenerating forests need to reach a certain age to become a source of propagules (van Breugel et al., 2013).

Besides natural regeneration, active restoration efforts may stimulate restoration of native biodiversity, especially in regions were natural land covers are strongly diminished. One example is the establishment of polycultural plantations that can kick-start reestablishment of forest biodiversity and ecosystem services on degraded lands (Holl and Aide, 2011; Zahawi et al., 2013). Another example is the agroecological management of agroforestry systems. By actively planting a diverse shade canopy and maintaining adequate shade levels, smallholders may intensify yields. In line with our results, trade-offs between shade biomass, biodiversity, and yield may be balanced through appropriate agroecological design (Ponisio et al., 2015; Fischer et al., 2017). These efforts need to be support by improved access to extension services, technical knowledge, and credit systems.

The question remains what forms of agriculture can improve income and food security with the minimal environmental impact of land use. Visions range from ecomodernism with large-scale mechanized farming in combination with preservation of natural habitats (Asafu-Adjaye et al., 2015), to propositions of small-scale, labor-intensive agroecological systems (Altieri and Toledo, 2011). Different interventions, such as value chain upgrading, certification schemes, and payments for ecosystem services, should thus focus on increasing benefits for farmers and wage laborers and controlling for land use expansions to neutralize possible rebound effects and guarantee long-lasting environmental and social benefits.

Effective financial incentives for restoration, such as payments for ecosystems and certification schemes, should provide landowners with adequate compensation for any restoration costs and loss in potential revenues (Bullock et al., 2011). As our main findings indicate, current carbon markets are too weak and inaccessible to provide this kind of compensation for agroforestry smallholders. In addition, certification premiums may not compensate for production losses. Development of financial incentives is therefore of tremendous importance to maintain and incentivize environmental-friendly agricultural practices. A potential opportunity is the development of more direct specialty markets for high-quality commodities. This would require the generation of more trust, stronger actor ties, and increased transparency across global value chains.

We have shown that the larger landscape-scale context is essential for the design and management of tropical agricultural systems. Current financial incentives, such as payments for ecosystem services and commodity certification schemes, mainly focus at the farm or plantation levels, whereas ecosystem benefits from well-designed agroforests are delivered at the landscape level (Tscharntke et al., 2015). Two approaches can be suggested to solve this scale mismatch: certification mechanisms could be linked with broader landscape-scale approaches (Milder et al., 2014) and current certification models could consider the landscape as a certified unit through landscape labeling (Ghazoul et al., 2009).

5.5 Limitations and prospective

Although models provide an important contribution to scientific knowledge, all models are merely representations of real-world dynamics. Specific limitations are discussed in more depth in each chapter, but in general our modeling approach was restricted by the simplification of complex processes and data availability for model parameterization. The greatest uncertainties for model parameterization in this study were: (i) the dispersal ability of modeled species; (ii) establishment probabilities of tree species across the landscape; and (iii) initial forest composition and age structure. In addition, aboveground net primary productivity rates were a source of uncertainty in Chapter 3. We addressed these uncertainties by including high resolution remote sensing imagery, maximum entropy species distribution models based on independent databases, empirical data from other studies, scientific literature, and consultations of local experts. Rigorous model validation was difficult, as relationships estimated in the past are used to predict future dynamics, as is common in all forest models (Gustafson, 2013). The lack of long-term data and understanding of ecological behavior of tropical forests hindered model calibration and validation. Some other factors that potentially affect future species distribution were not taken into account, such as climate change, differences in microhabitat, seed dispersal, and additional anthropogenic or natural disturbances.

This dissertation opens several interesting and promising research avenues. Firstly, Chapter 2 and 3 highlighted the lack of long-term databases on dynamics of tropical remnant and regenerating forests. Active forest regeneration activities are scarce in Ecuador, but might provide substantial environmental benefits. The contribution of payments for ecosystem services in natural and active restoration activities needs to be quantified for successful design, including the involvement of local landowners. Chapter 4 pointed out the lack of empirical evidence concerning alternative market channels for niche commodities. An important question remains whether and to what extent payments for ecosystem services and alternative commodity markets will incentivize landowners and smallholders to adopt or maintain environmental practices.

APPENDIX A

APPENDIX A: SUPPLEMENTARY MATERIAL FOR CHAPTER 2

Table A.1: Overview of twelve 0.02 ha transects (40 x 5 m) for which woody species data was collected in September and October 2013. Land covers are: MF, montane forest (n = 3); UMF, upper montane forest (n = 2); RF, regenerating forest (n = 2); PP, pine plantations (n = 4); and PA, páramo (n = 1)

PLOT NO.	LAND COVER	COORDINATES EASTING	NORTHING	ALTITUDE (masl)	SLOPE (deg)	AVG. HEIGHT	SPC. RICH.
1	MF	737806	9796039	3,594	20.8	5.4	5
2	MF	738012	9796058	3,641	19.2	4.0	9
3	MF	725796	9794246	2,715	28.5	5.5	14
4	MF	724074	9791138	2,334	27.3	8.1	16
5	PP	737423	9796260	3,636	23.9	13.5	3
6	PP	737405	9796242	3,636	23.9	20.6	1
7	MF	727851	9788300	2,508	34.3	6.5	14
8	PP	733150	9794329	3,251	35.5	18.7	1
9	PP	736165	9787553	3,590	32.9	11.0	6
10	PA	735548	9787693	3,578	32.9	2.3	12
11	RF	727527	9787953	2,430	27.2	3.6	15
12	RF	722531	9784947	2,000	40.7	3.5	14

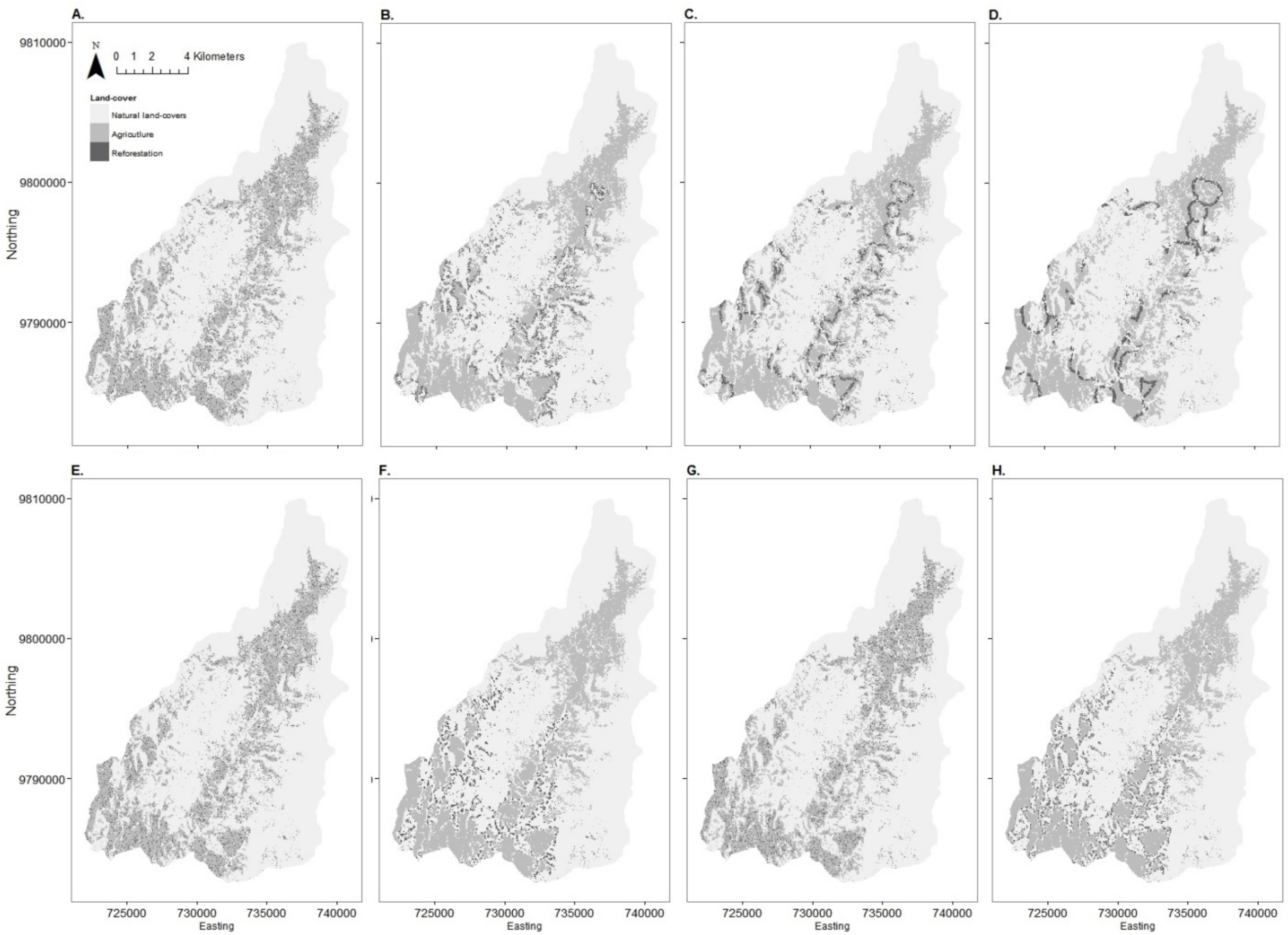

Figure A.1: Overview of all scenarios simulated using LANDIS-II. Scenarios were: (a) random; (b) distance <200m; (c) distance 200-400m; (d) distance >400m; (e) forest cover >50%; (f) forest cover <50%; (g) forest cover >30%; (h) forest cover <30%; (j) diversity index <50%; (k) diversity index >50%; (l) patch size <; (m) patch size >; (n) altitude <2,400m; (o) altitude 2,400-3,000m; (p) altitude 3,000-3,400m; and (q) altitude >3,400m

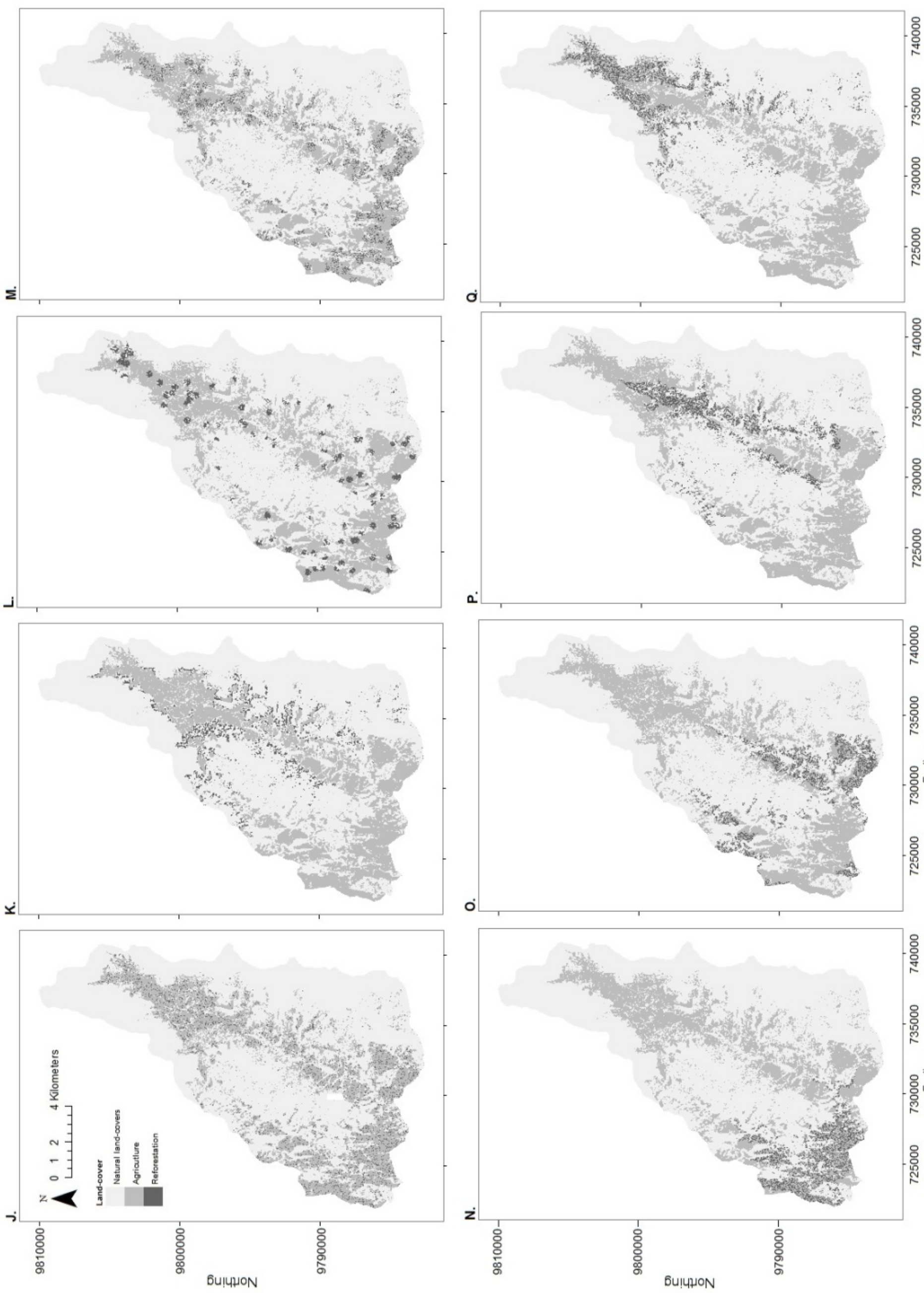

Figure A.1 (Continued): Continued Fig. A.1

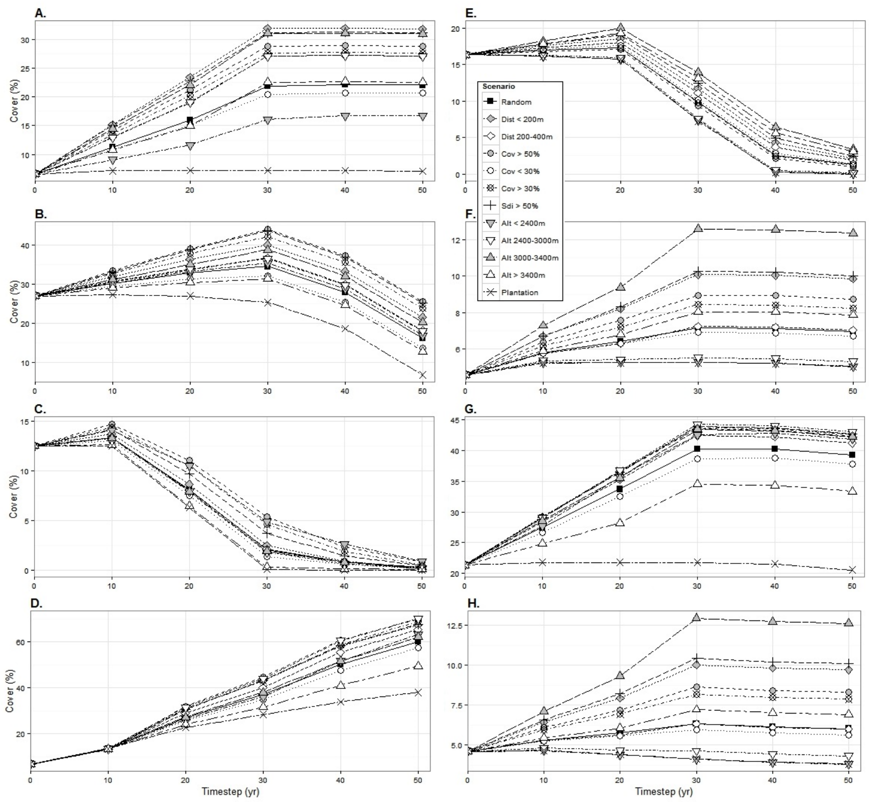

Figure A.2: The cover percentage of the study area occupied by each selected species over time under significant scenarios. (a) *Aegiphila bogotensis*; (b) *Baccharis latifolia*; (c) *Boehmeria aspera*; (d) *Clusia multiflora*; (e) *Gynoxys* spp.; (f) *Hedyosmum luteynii*; (g) *Miconia* spp.; and (h) *Podocarpus glomeratus*. Shapes correspond to scenarios. Values presented are means for three repeated simulations for reforestation onset in 2010. Note that the y-axis values are not fixed

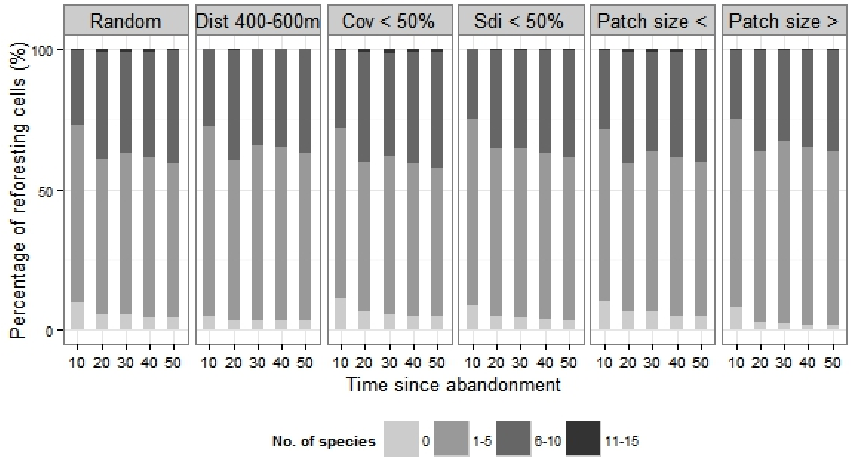

Figure A.3: The percentage of reforesting fallows containing a number of woody species over time since abandonment for non-significant scenarios. Re-colonization of abandoned agricultural lands was dynamic during simulations but comparable to the random scenario

Table A.2: Details of the ecoregion map in the Pangor study area. Vegetation zonation as proposed by Jorgensen and Ulloa Ulloa (1994). All regions above 3,700 masl were excluded from model simulations as these regions mainly consist of herbaceous and small shrub vegetation

ACTIVE	CODE	ELEVATION (masl)	SOLAR RADIATION ($*10^6$ WH/m^2)
Yes	1	1,431 – 2,400	0.85 – 1.44
Yes	2	1,431 – 2,400	1.44 – 2.03
Yes	3	1,431 – 2,400	2.03 – 2.62
Yes	4	2,400 – 3,000	0.85 – 1.44
Yes	5	2,400 – 3,000	1.44 – 2.03
Yes	6	2,400 – 3,000	2.03 – 2.62
Yes	7	3,000 – 3,400	0.85 – 1.44
Yes	8	3,000 – 3,400	1.44 – 2.03
Yes	9	3,000 – 3,400	2.03 – 2.62
Yes	10	3,400 – 3,700	1.44 – 2.03
Yes	11	3,400 – 3,700	2.03 – 2.62
No	12	> 3,700	0.85 – 2.62

Table A.3: Ecological characteristics for the tree and shrub species encountered in the Pangor study area that were included in model simulations. Long, longevity (yrs); Mat, age of maturity (yrs); ShT, shade tolerance class (1-5, with 1 for most shade intolerant and 5 for most shade tolerant); EffSD, effective seeding distance (m); MaxSD, maximum seeding distance (m); VRP, vegetative reproduction probability; MinVRP, minimum age of vegetative reproduction (yrs); MaxVRP, maximum age of vegetative reproduction (yrs); P-FiR, post-fire regeneration (form of reproduction that species adopts after fire events)

SPECIES	SPC. ABBR.	LONG	MAT	SHT	FIT	EFFSD	MAXSD	VRP	MIN VRP	MAX VRP	P-FIR
Aegiphila spp. (d, k, h, i)	Aegspp	150	15	2	1	400	1500	0	0	0	None
Baccharis latifolia (a, c, g, k, n, r)	Baclat	60	8	1	2	50	500	1	10	35	Resprout
Berberis spp (c, g, k, t)	Berlut	70	10	2	3	400	1500	0	0	0	None
Boehmeria aspera (h, m)	Boeasp	30	6	3	3	50	500	0	0	0	None
Brachyotum ledifolium (t, s, k, o)	Braled	80	10	4	4	400	1500	1	10	50	Resprout
Cervantesia tomentosa (i, l)	Certom	450	25	3	1	400	1500	0	0	0	None
Clusia multiflora (c, d, f, l, m, v)	Clumul	450	25	5	4	400	1500	1	25	150	Resprout
Erythrina edulis (b, d, l, n)	Eryedu	160	15	3	3	400	1500	1	15	60	Resprout
Escallonia spp. (a, l, m, t, v)	Escspp	250	15	3	2	50	500	0	0	0	None
Ficus dulciaria (d, e, i, l)	Ficdul	300	25	4	1	400	1500	1	15	100	Resprout
Gynoxys spp. (c, l, n, t)	Gynspp	40	6	2	4	50	500	0	0	0	None
Hedyosmum luteynii (e, l, p, v)	Hedlut	300	15	3	2	50	500	0	0	0	None
Miconia spp. (c, f, l, m, o, q)	Micspp	90	15	1	3	400	1500	0	0	0	None
Myrcianthes spp. (d, f, h, l, m, q)	Myrrho	460	35	5	3	50	500	0	0	0	None
Pinus radiata (j)	Pinrad	80	15	2	3	1	10	0	0	0	Serotiny
Podocarpus glomeratus (e, q)	Podglo	190	15	2	1	50	500	1	10	80	Resprout
Polylepis lanuginosa (d, q, t)	Pollan	250	30	3	4	50	500	1	15	100	Resprout
Solanum spp. (c, i, l, m, t)	Solspp	60	10	3	1	400	1500	0	0	0	None
Tournefortia fuliginosa (d, l, m)	Touful	150	15	4	1	400	1500	0	0	0	None

a. Aguirre et al (2011); b. Cantarello et al (2011); c. Castellanos-Castro and Bonilla (2011); d. Cuamacas and Tipaz (1995); e. Dislich et al (2009); f. Duivenvoorden and Cuello A. (2012); g. Easdale et al (2007a); h. Easdale et al (2007b); i. Gentry (1996); j. He and Mladenoff (1999); k. Jørgensen and León-Yánez (1999); l. Jørgensen and Ulloa Ulloa (1994); m. Ledo et al (2012); n. Lozano (2011); o. Macek et al (2009); p. Montenegro and Vargas (2008); q. Palacios (2011); r. Patzelt (1996); s. Pennington et al (2004); t. Pillajo and Pillajo (2011); u. Ramsay and Oxley (1996); v. Smith et al (2004); w. Ulloa Ulloa and Jorgensen (1995)

Table A.4: Overview of all 80 species found during the field survey, including authorities and common names

FAMILY	SPECIES	AUTHORITY	COMMON NAME[a]
Lamiaceae	*Aegiphila bogotensis*	(Spreng.) Moldenke	Luca del monte
Euphorbiaceae	*Alchornea* sp.		
Sapindaceae	*Allophylus excelsus*	(Triana & Planch.) Radlk.	Pandala blanco
Betulaceae	*Alnus acuminata*	Kunth	Aliso blanco
Asteraceae	*Andromachia igniaria*	Bonpl.	
Annonaceae	*Annona quinduensis*	Kunth	Chirimoyo
Asteraceae	*Baccharis latifolia*	(Ruiz & Pav.) Pers.	Chilca
Berberidaceae	*Berberis lutea*	Ruiz & Pav.	Espino amarillo
Berberidaceae	*Berberis multiflora*	Benth.	
Urticaceae	*Boehmeria aspera*	Wedd.	
Melastomataceae	*Brachyotum ledifolium*	(Desr.) Triana	Zarcilejo blanco
Calceolariaceae	*Calceolaria adenanthera*	Molau	Zapatitos
Arecaceae	*Ceroxylon echinulatum*	Galeano	Palma de cera
Santalaceae	*Cervantesia tomentosa*	Ruiz & Pav.	Olivo
Solanaceae	*Cestrum* sp.		Sauco
Poaceae	*Chusquea jamesonii*	Steud.	Suru
Clusiaceae	*Clusia multiflora*	Kunth	Incienso
Primulaceae	*Cybianthus* spp.		
Thymelaeaceae	*Daphnopsis macrophylla*	(Kunth) Gilg	
Fabaceae	*Erythrina edulis*	Triana ex Micheli	Poroto hortiga
Escalloniaceae	*Escallonia paniculata*	(Ruiz & Pav.) Roem. & Schultes	Chacharo
Escalloniaceae	*Escallonia pendula*	(Ruiz & Pav.) Pers.	
Moraceae	*Ficus dulciaria*	Dugand	Matapalo
Primulaceae	*Geissanthus* sp.		Maconsa
Asteraceae	*Gynoxys fuliginosa cf.*	(Kunth) Cass.	Macho
Asteraceae	*Gynoxys miniphylla*	Cuatrec.	
Rosaceae	*Hesperomeles obtusifolia*	(Pers.) Lindl.	Quique
Phyllanthaceae	*Hieronyma macrocarpa*	Müll. Arg.	Motilón
Aquifoliaceae	*Ilex gabinetensis*	Cuatrec.	Cacho de venado
Aquifoliaceae	*Ilex myricoides cf.*	Kunth	
Fabaceae	*Inga* sp.		Guaba
Ericaceae	*Macleania rupestris*	(Kunth) A.C. Sm.	Walicón
Sabiaceae	*Meliosma frondosa*	Cuatrec. & Idrobo	Pacche
Melastomataceae	*Miconia aggregata cf.*	Gleason	Alamoja
Melastomataceae	*Miconia crocea*	(Desr.) Naudin	Colca
Melastomataceae	*Miconia papillosa cf.*	(Desr.) Naudin	Tostado de pajarito
Melastomataceae	*Miconia* spp.		
Monimiaceae	*Mollinedia* sp.		

Table A.4 (Continued): Continued Table A.4

Polygalaceae	*Monnina phillyreoides*	(Bonpl.) B. Eriksen	Azulina
Myricaceae	*Morella parvifolia*	(Benth.) Parra-O.	Laurel
Myrtaceae	*Myrcianthes* sp.		Arrayán
Primulaceae	*Myrsine coriaceae*	(Swartz) Brown ex Roemer & Schultes	Tupial/charmuelán macho
Lauraceae	*Nectandra* spp.		Chachajillo
Araliaceae	*Oreopanax cf. ecuadorensis*		Pumamaki
Rubiaceae	*Palicourea amethystina*	(Ruiz & Pav.) DC.	Naranjo negro
Pinaceae	*Pinus radiata*	D. Don	Pino
Piperaceae	*Piper barbatum*	Kunth	Lunkug
Piperaceae	*Piper* sp.		
Rosaceae	*Polylepis lanuginosa*	Kunth	Árbol de papel/yagual
Rubiaceae	*Psychotria* sp.		
Dennstaedtiaceae	*Pteridium arachnoideum*	(Kaulf.) Maxon	
Dennstaedtiaceae	*Pteridium* sp.		
Bromeliaceae	*Puya* sp.		
Rosaceae	*Rubus niveus*	Thunb.	Morillo
Lamiaceae	*Salvia corrugata*	Vahl	
Lamiaceae	*Salvia* sp.		
Actinidiaceae	*Saurauia pseudostrigillosa*	Buscal.	Moquillo
Araliaceae	*Schefflera* sp.		Pata de gallo
Siparunaceae	*Siparuna aspera*	(Ruiz & Pav.) A. DC.	
Solanaceae	*Solanum* spp.		Hierba mora
Boraginaceae	*Tournefortia fuliginosa*	Kunth	Escorpión
Loranthaceae	*Tristerix longebracteatus*	(Desr.) Barlow & Wiens	Mato palo de cerro
Ericaceae	*Vaccinium floribundum*	Kunth	Mortiño
Adoxaceae	*Viburnum hallii*	(Oerst.) Killip & A.C. Sm.	
Asteraceae	*Viguiera quitensis*	(Benth.) S.F. Blake	
Boraginaceae	*Wigandia crispa*	(Tafalla ex Ruiz & Pav) Kunth	

[a] Not all common names were known by local communities

Table A.5: Based on pair matched case-comparison, we compared simulated species richness for reforesting fallows between each scenario and the random scenario. For all scenarios, we randomly selected a total of 1500 reforesting raster cells (i.e., 500 points for each onset of reforestation: year 2010, 2025, and 2040). The algorithm found the optimal pair of matching points for all comparisons, indicated by the non-significant KS test p-values after matching. To test for the significance of the causal effect of the landscape attribute on the simulated species richness, we compared the means of each scenario against the random scenario using the Wilcoxon-Mann-Withney test

SCENARIO	CALIPER	NO. OF PAIRS	MIN. KS TEST *P*-VALUE[a]		WILCOXON-TEST *P*-VALUE[c]
			BEFORE[b]	AFTER	
forest cover < 50%	0.8	1500	0.117	0.583	0.83
forest cover > 50%	0.18	1406	< 2e-16*	0.096	< 0.0001*
forest cover < 30%	0.8	1500	< 2e-16*	0.497	< 0.0001*
forest cover > 30%	0.14	1357	< 2e-16*	0.106	< 0.0001*
distance < 200m	0.23	1469	0.009*	0.279	< 0.0001*
distance 200-400m	0.8	1500	< 2e-16*	0.975	< 0.0001*
distance > 400m	0.5	1500	0.016*	0.709	0.066
diversity index < 50%	0.8	1500	< 2e-16*	0.279	0.76
diversity index > 50%	0.07	1029	< 2e-16*	0.111	< 0.0001*
patch size <	0.8	1500	0.039*	0.230	0.34
patch size <	0.8	1500	< 2e-16*	0.419	0.44
altitude < 2,400m	0.12	1295	< 2e-16*	0.212	< 0.0001*
altitude 2,400-3,000m	0.8	1498	< 2e-16*	0.509	0.99
altitude 3,000-3,400m	0.36	1493	< 2e-16*	0.296	0.008*
altitude > 3,400m	0.8	1500	< 2e-16*	0.209	0.35
plantation	0.9	1500	0.232	0.196	< 0.0001*

[a] smallest p-value obtained when testing difference in each covariate histogram distribution across the two matched groups; [b] asterisks indicate significant imbalance in co-variables (i.e., altitude, solar irradiance, time since abandonment) before matching of pairs of points; [c] asterisks indicate significantly different average species richness between the tested and the random scenario (Wilcoxon-Mann-Whitney test, $P < 0.05$)

APPENDIX B

APPENDIX B: SUPPLEMENTARY MATERIAL FOR CHAPTER 3

Table B.1: Overview of the average AGB (Mg ha^{-1}) across all sites and cocoa agroforestry sites for each scenario. Total AGB at 'spin-up' (i.e., year 0) equaled 107.41 and 60.31 Mg ha^{-1} for respectively all and agroforestry sites in all scenarios. Land abandonment scenarios included abandoned agroforests in cocoa agroforests sites

Scenario	All sites		Agroforestry only	
	Year 20	Year 50	Year 20	Year 50
Baseline	112.68	117.71	74.66	79.79
Traditional	118.59	124.00	105.73	111.27
Planted	114.21	120.81	82.31	95.28
Monoculture	103.80	109.37	30.55	38.26
Monoculture + 20% set aside	107.40	113.65	48.37	59.53
Monoculture + 30% set aside	109.27	115.83	57.68	70.44
Monoculture + 40% set aside	111.06	117.95	66.60	81.03

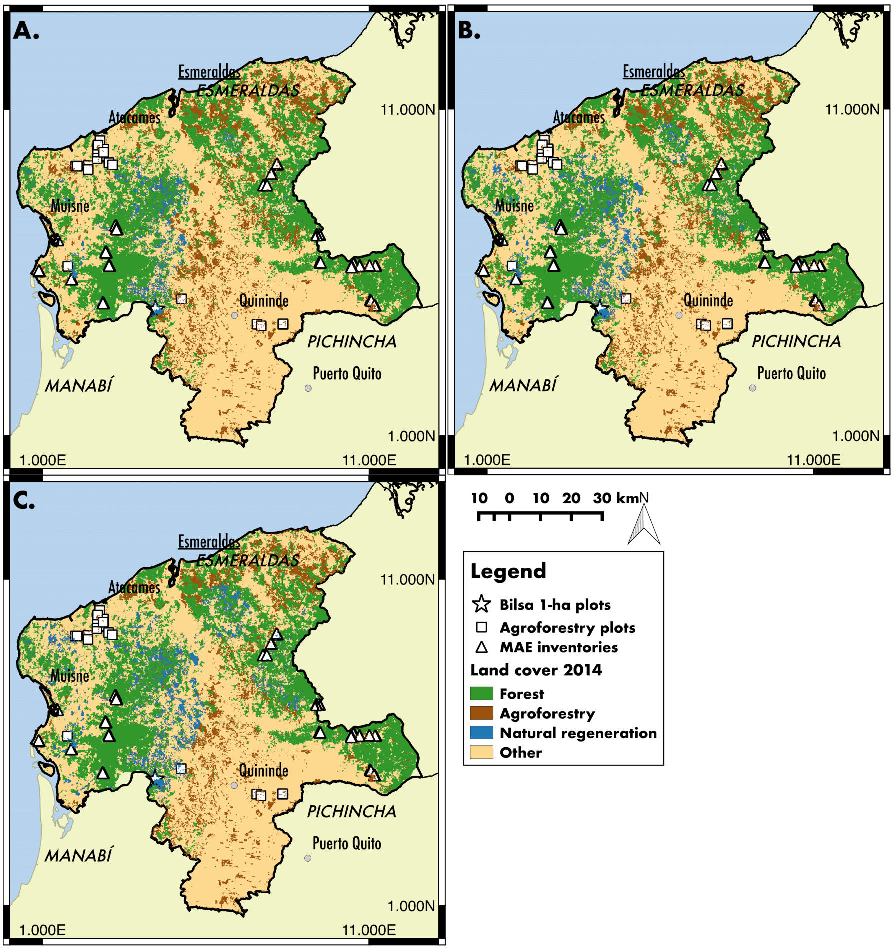

Figure B.1: Maps showing the location of set aside areas in the monoculture with (a) 20, (b) 30, and (c) 40 per cent scenarios

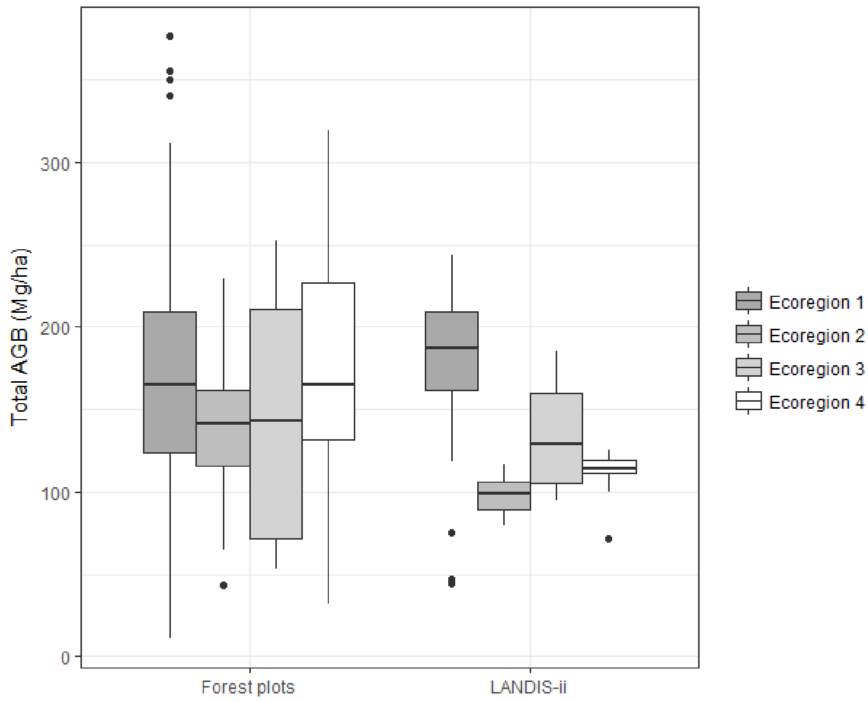

Figure B.2: Boxplot of total aboveground biomass (Mg ha^{-1}) per forest ecoregions as calculated for forest inventory plots (left) and simulated by LANDIS-ii (right) for the control scenario at the start of simulation (i.e., t = 0). Outliers were omitted from forest inventory data. LANDIS-II estimates total AGB within the range of values estimates from inventory observations

APPENDIX

C

APPENDIX C: SUPPLEMENTARY MATERIAL FOR CHAPTER 4

Table C.1: Test for covariate balance before and after matching for household and farm survey data, using genetic matching (GM) as presented in the manuscript and nearest neighborhood matching (NN) 1-1 and 1-2. Insignificant Kolmogorov-Smirnov test values based on 1,000 bootstrap samples after matching indicated that covariate balance was achieved. If after matching all covariate test values were insignificant, an optimal pair of matched points was found

Covariate	Household survey data				Farm survey data			
	Before matching	GM	NN 1-1	NN 1-2	Before matching	GM	NN 1-1	NN 1-2
Farm size (ha)	0.17	0.15	0.39	0.30**	0.27	0.24	0.33	0.30*
Area in cocoa (ha)	0.32*	0.15	0.36*	0.28**	0.41*	0.14	0.29	0.19
Altitude (masl)	0.19	0.22	0.25	0.20	0.22	0.24	0.24	0.35*
Education hh (yrs)	0.23	0.11	0.11	0.13	0.18	0.14	0.14	0.12
Family size	0.15	0.11	0.21	0.22 .	0.25	0.19	0.24	0.23 .

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

Table C.2: Propensity scores (i.e. the likelihood of participation in direct trade as predicted by a regression model) for input variables based on probit regression results with direct market as dependent variable for household and farm survey data. Scores were used for nearest neighbor matching methods

Variable	Household survey data			Farm survey data		
	Coefficient	S.E.	p-value	Coefficient	S.E.	p-value
Farm size (ha)	-0.79	0.01	0.43	-0.60	0.01	0.55
Area in cocoa (ha)	-1.59	0.05	0.11	-1.90	0.7	0.06 .
Altitude (masl)	0.52	0.00	0.60	0.78	0.00	0.44
Education hh (yrs)	1.91	0.05	0.06 .	1.49	0.06	0.14
Family size	1.19	0.08	0.24	0.37	0.08	0.71
Intercept	-1.08	0.82	0.28	-0.30	0.90	0.77

. $p < 0.1$

Table C.3: Estimates of the average treatment effect on the treated (ATT) for matched observations of direct vs. mainstream household surveyed, comparing nearest neighbor 1-1 and nearest neighbor 1-2 with genetic matching results. NN matching resulted in similar qualitative conclusions as presented in the main text

Outcome variables	Genetic matching			Nearest neighbor 1-1			Nearest neighbor 1-2		
	ATT	S.E.	t-test	ATT	S.E.	t-test	ATT	S.E.	t-test
Economic benefits (hypothesis 1)									
Wet price (ct pound^{-1})	24.35	0.75	25.18***	23.48	1.10	21.26***	23.61	0.96	24.72***
Premium for certification (1/0)	0.67	0.13	5.77***	0.56	0.19	2.93**	0.59	0.16	3.80***
Premium for quality (1/0)	0.78	0.10	6.43***	0.69	0.14	4.9***	0.78	0.11	7.27***
Nonmonetary benefits (hypothesis 2)									
Access to credit (1/0)	0.00	0.08	0.00	0.04	0.06	0.65	0.04	0.05	0.75
Technical assistance (1/0)	0.30	0.12	2.43*	0.43	0.19	2.26*	0.41	0.14	2.92**
Farmers' cooperative membership (1/0)	0.70	0.11	5.77***	0.44	0.15	2.96**	0.52	0.12	4.36***
Workers hired on farm (1/0)	0.22	0.18	1.87	0.02	0.20	0.09	0.24	0.16	1.53
Cocoa as the main source of income (1/0)	-0.07	0.13	-0.51	0.15	0.16	0.94	0.04	0.13	0.28
Management practices (hypothesis 3)									
Organic fertilizer (1/0)	0.48	0.16	2.86**	0.09	0.22	0.41	0.30	0.17	1.80
Chemical fertilizer (1/0)	-0.22	0.10	-3.10**	-0.56	0.07	-0.83	-0.22	0.09	-2.56*
Insecticides (1/0)	-0.18	0.09	-1.06	-0.30	0.14	-2.15*	-0.17	0.08	-2.10*
Fungicides (1/0)	0.00	0.00	0.00	0.00	0.00	0.00	-0.02	0.03	-0.62
Herbicides (1/0)	-0.37	0.12	-2.94**	-0.50	0.19	-2.64**	-0.26	0.14	-1.80*
Presence of Nacional (1/0)	0.30	0.11	2.58**	0.26	0.13	1.96*	0.15	0.08	1.95*
Presence of CCN-51 (1/0)	-0.26	0.12	-2.18*	-0.50	0.17	-2.93**	-0.31	0.15	-2.13*
Preference for Nacional (1/0)	0.19	0.16	1.87	0.24	0.19	1.26	0.11	0.14	0.78
Shade of non-cocoa trees (1/0)	0.11	0.08	0.73	0.30	0.14	2.15*	0.15	0.08	1.95*
Non-cocoa tree species richness stated by farmer (1/0)	2.37	0.65	3.67***	1.74	1.16	1.50	1.48	0.94	1.57
Irrigation (1/0)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Grafting of cocoa trees (1/0)	-0.04	0.17	0.20	-0.28	0.20	-1.42	-0.19	0.16	-1.16
Pruning of cocoa trees (1/0)	0.14	0.11	1.24	0.00	0.08	0.00	0.13	0.10	1.33
Used fire before planting (1/0)	-0.19	0.11	-0.73	-0.52	0.17	-3.02**	-0.31	0.10	-3.06**
Area in cocoa increased in the past 5 years (1/0)	0.07	0.16	0.20	0.28	0.20	1.36	0.07	0.19	0.40
Biodiversity conditions (hypothesis 4)									
Non-cocoa species richness (species ha^{-1})	2.10	1.65	1.27	2.43	1.22	1.98*	2.10	1.10	1.91
Abundance of non-cocoa trees (ha^{-1})	4.33	5.41	0.75	5.52	4.87	1.14	4.83	4.28	1.13
Basal area of non-cocoa trees (m^2 ha^{-1})	5.96	30.24	0.44	14.91	26.98	0.55	11.12	27.18	0.41
Average height of non-cocoa trees (m)	4.85	2.80	1.90	3.09	2.65	1.16	3.02	2.71	1.11
Average shade level index	0.67	0.34	1.95	0.70	0.42	1.65	0.38	0.37	1.04
Nacional density (trees ha^{-1})	108.3	143.2	0.76	3.29	5.62	0.58	-0.31	4.52	-0.07
CCN51 density (trees ha^{-1})	-297.6	126.1	-2.36*	-13.76	4.64	-2.96**	-10.61	3.69	-2.88**

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

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