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Forest ecosystem services and multifunctionality across the land use transition phases in Ecuadorian forest landscapes

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Abstract

Tropical forests host great biodiversity and provide a wide range of ecosystem services (provisioning, regulating, supporting, and cultural) essential for human well-being. On the local scale, tropical forests provide timber and non-timber forest products, facilitate biological connectivity in fragmented areas, and sustain different levels of biodiversity. On a global scale, tropical forests play a crucial role in climate regulation due to their capacity for carbon sequestration. Despite tropical forests' contribution to human well-being, they have been negatively impacted in the last decades due to anthropogenic activities (inter alia, forest conversion to pastures or crops, overexploitation of timber species), resulting in forest cover loss and forest degradation with negative effects on the provision of multiple ecosystem services.

Because of deforestation and forest degradation, different land use transition phases emerge at the landscape level, depicting a mosaic of land uses that evidence the transformation from forested to human-modified landscapes. Thus, landscapes can be composed of natural forests (old-growth forests, logged forests, and successional forests), planted forests, and agroforestry systems as a manifestation of the productive activities that drive the landscape dynamics. The existence of several land uses in the landscape could reflect ecological decline and recovery phases called land-use transition phases which are associated with changes in the provision of individual ecosystem services but also with changes in the multifunctionality of the landscape. Within the land use transition phases, we observe, on the one hand, the transition from undisturbed old-growth forests to logged forests which is frequently accompanied by a decline in ecosystem services. On the other hand, successional forests, plantations, and agroforestry systems can emerge as an effort to recover degraded services.

To stop deforestation and forest degradation, and to achieve human welfare goals, different international initiatives have been proposed in the last decades. Some of the well-known initiatives with the greatest global impact are the United Nations Framework Convention on Climate Change (UNFCC); the Convention on Biological Diversity (CBD); the mechanism to Reduce Emissions from Deforestation and Forest Degradation (REDD+); and the 2030 Agenda of the United Nations that comprises 17 sustainable development goals (SDGs). In all these initiatives, forests have an important role in securing biodiversity and ecosystem services, supporting climate change mitigation, and sustaining livelihood strategies. However, these

initiatives have not yet succeeded in fulfilling their goals as they pursue forest conservation and human welfare goals at the same time.

To harmonize conservation and development goals, it is required to adopt new approaches. The multifunctional landscape approach is positioned as one that contributes to sustainable landscape management. The multifunctional landscape approach aims to safeguard biodiversity, ecosystem services, and ecological functions while providing livelihood opportunities for people inhabiting the landscape. Under this approach, one must recognize the various land uses occurring in the landscape and needs to understand how these land-uses impact ecosystem services provision from which people depend to develop their livelihood strategies. It also requires acknowledging that landscape dynamics are influenced by institutional elements that can foster or discourage unsustainable land-uses.

In the tropics, studies addressing the effects of land-use transition phases on the provision of single and multiple ecosystem services are scarce. The multifunctional landscape approach is rarely considered in both research and practice. Moreover, the potential that institutions can have on landscape dynamics is usually overlooked because many studies only adopt the ecological perspective. In consequence, this dissertation provides more insights into human-land interactions and the associated environmental effects. The study was conducted in the tropical lowland rainforest landscapes of Ecuador and addresses three main issues: (i) the land use transition phases and their implication in the supply of ecosystem services and ecosystem service multifunctionality; (ii) the influence of the incentive-based forest conservation on ecosystem services provision across the land use transition phases; (iii) the contribution of incentive-based forest conservation to halt deforestation beyond the limits of the conservation program.

This study draws on in situ information from twelve landscapes (10 × 10 km) selected in the Ecuadorian Central Amazon and the Chocó. The selected landscapes are a mosaic of old-growth forests, logged forests resulting from timber extraction activities, successional forests resulting from natural regeneration after land abandonment, forest plantations, and agroforestry systems. In total, I installed 156 inventory plots (1600 m^2) where I collected information for indicators of provisioning, regulating, and supporting services, and biodiversity, to quantify the amount of ecosystem services provision in each land use transition phase. The ecosystem service multifunctionality index using the multifunctional average approach was also estimated. Ecological data was complemented with household data from surveys conducted in the selected landscapes; this data allowed us to analyze the effect of incentive-based conservation on household deforestation decisions.

I conducted principal component analysis (PCA) to identify synergies and trade-offs among ecosystem services and the multifunctionality index. Additionally, I estimated the analysis of variance to evaluate differences in ecosystem services provision across the land-use transition phases, as well as, to assess the effect of the incentive-based conservation program on ecosystem services provision and deforestation reduction. I also estimated regression analysis to evaluate the influence of the incentive-based conservation program on household deforestation decisions. The main findings are as follows:

The analysis showed synergetic relations for ecosystem service multifunctionality, regulating services, provisioning services, and plant diversity indicators. Above-ground carbon stocks had synergies with various services and with the ecosystem service multifunctionality but contrasting patterns with soil-related services. Above-ground carbon stocks can be considered as an umbrella service since it is a sensitive indicator of forest integrity and had strong to moderate synergies with ecosystem service multifunctionality and several ecosystem services. Any anthropogenic activity that modifies above-ground carbon stocks can trigger the increase or decrease of additional ecosystem services. Above-ground carbon stocks can be a useful indicator to monitor several ecosystem services and biodiversity.

This research further demonstrates that timber extraction caused a decline of 16% to 18% of the ecosystem service multifunctionality in the study areas. Timber extraction has a high to moderate impact on provisioning and regulating services, and plant diversity. Logging activities strongly impact the timber volume potential, provoking a reduction between 40% to 49% of this provisioning service; those activities also decreased above-ground carbon stocks between 30% to 32% in the selected landscapes. The study demonstrates that ecosystem services in logged forests are highly impacted, even though logging is executed under the technical normative of the environmental authority.

To assess the recovery of ecosystem services and ecosystem service multifunctionality the successional forests, agroforestry systems, and plantations were considered. The results showed that successional forests are the most effective option to restore single and bundles of ecosystem services. Successional forests offer high values of provisioning services (timber volume potential and non-timber forest products), regulating services (above-ground carbon stocks and soil carbon stocks), and plant diversity but with a different structure and plant composition than old-growth forests. The agroforestry systems and the plantations offer lower ecosystem service multifunctionality, nevertheless, they have a great potential to recover soil-related services like soil carbon stocks, nitrogen, phosphorus, and potassium.

Results also showed that the incentive-based forest conservation program contributes to maintaining the evaluated ecosystem services, as no reduction of ecosystem services was observed in old-growth forests under the conservation program. In logged forests with no incentive-based forest conservation, there was a higher decline in ecosystem services with impacts on above-ground carbon stocks and species richness. Interestingly, the above-ground carbon stocks were reduced by 21% in the logged forests close to the areas under the incentives, meanwhile, logged forests in landscapes with no conservation program showed 41% less above-ground carbon stocks; this evidenced that incentive-based forest conservation helped to lessen the adverse effects of logging. To prevent a higher decline in ecosystem services provision the Ecuadorian environmental authority must establish strict monitoring and post-harvesting control measures.

In the selected landscapes, incentive-based forest conservation contributed to reducing deforestation. This study indicated a decrease in the annual net deforestation rate after the implementation of incentive-based forest conservation at the parish level. When evaluating the role of the incentive-based forest conservation program on household deforestation decisions, the results showed that households living close to areas under the program have lower odds (56% less) to deforest compared to households settled in landscapes with no influence of the incentives program. These results suggest that the incentive-based conservation program is having impacts beyond the areas of conservation. Incentive-based conservation programs have the potential to enhance multifunctional landscapes by combating deforestation beyond the limits of the conservation area. Incentive-based conservation can contribute to balancing and integrating forest conservation, timber production, and forest landscape restoration as observed in landscapes with the presence of the program.

This study links the ecosystem services framework to the landscape multifunctionality approach, giving helpful information on how land-uses affect single and multiple ecosystem services. Through the analysis of synergies and trade-offs, it is possible to identify ecosystem services that drive the landscape's multifunctionality. For the first time, the umbrella species concept is brought and adapted to the ecosystem services framework. Based on the umbrella species concept, I attempted to identify umbrella ecosystem services, which can be practical to prioritize ecosystem services conservation. The umbrella services concept could be refined and used in forest conservation, landscape restoration, and ecological monitoring projects.

This study also assessed how institutional aspects influence ecosystem services provision. For this purpose, I considered the incentive-based conservation program because this is the second most important conservation policy in Ecuador. The study revealed that ecosystem services can be benefited from such programs even beyond the areas under conservation. Likewise, the incentive-based conservation program showed important effects on households' land-use decisions. The effect of incentive-based conservation program on ecosystem services and households' behavior has not been evaluated previously for the country and is rarely assessed for the tropics.

Zusammenfassung

Tropenwälder beherbergen eine große Biodiversität und bieten eine breite Palette von Ökosystemleistungen (bereitstellend, regulierend, unterstützend und kulturell), die für das menschliche Wohlergehen unerlässlich sind. Auf lokaler Ebene liefern Tropenwälder Holz- und Nicht-Holz-Waldprodukte, erleichtern die biologische Konnektivität in fragmentierten Gebieten und erhalten unterschiedliche Artenvielfalt. Auf globaler Ebene spielen Tropenwälder aufgrund ihrer Fähigkeit zur Kohlenstoffbindung eine entscheidende Rolle bei der Klimaregulierung. Trotz des Beitrags der Tropenwälder zum menschlichen Wohlergehen wurden sie in den letzten Jahrzehnten durch anthropogene Aktivitäten (Umwandlung von Wäldern in Weiden oder Nutzpflanzen, Übernutzung von Holzarten) negativ beeinflusst, was zu einem Verlust der Waldfläche und Waldschädigung mit negativen Auswirkungen auf führte die Bereitstellung mehrerer Ökosystemleistungen.

Aufgrund von Entwaldung und Walddegradation entstehen auf Landschaftsebene verschiedene Phasen des Landnutzungsübergangs, die ein Mosaik von Landnutzungen darstellen, die die Transformation von bewaldeten zu vom Menschen veränderten Landschaften belegen. Somit können Landschaften aus natürlichen Wäldern (Urwälder, abgeholzte Wälder und Sukzessionswälder), gepflanzten Wäldern und Agroforstsystemen als Manifestation der produktiven Aktivitäten bestehen, die die Landschaftsdynamik antreiben. Das Vorhandensein mehrerer Landnutzungen in der Landschaft könnte ökologische Niedergangs- und Erholungsphasen widerspiegeln, sogenannte Landnutzungsübergangsphasen, die mit Änderungen in der Bereitstellung einzelner Ökosystemleistungen, aber auch mit Änderungen in der Multifunktionalität der Landschaft verbunden sind. Innerhalb der Übergangsphasen der Landnutzung beobachten wir einerseits, dass der Übergang von ungestörten Altwäldern zu abgeholzten Wäldern häufig mit einem Rückgang der Ökosystemleistungen einhergeht. Andererseits können sukzessive Wälder, Plantagen und Agroforstsysteme entstehen, um verschlechterte Leistungen wiederherzustellen.

Um Entwaldung und Waldschädigung zu stoppen und Ziele für das Wohlergehen der Menschen zu erreichen, wurden in den letzten Jahrzehnten verschiedene internationale Initiativen vorgeschlagen. Einige der bekannten Initiativen mit der größten globalen Wirkung sind das Rahmenübereinkommen der Vereinten Nationen über Klimaänderungen (UNFCC); das Übereinkommen über die biologische Vielfalt (CBD); der Mechanismus zur Reduzierung von Emissionen aus Entwaldung und Waldschädigung (REDD+); und die Agenda 2030 der

Vereinten Nationen, die 17 Ziele für nachhaltige Entwicklung (Nachhaltige Entwicklungsziele, SDGs) umfasst. Bei all diesen Initiativen spielen Wälder eine wichtige Rolle, um die Biodiversität und Ökosystemleistungen zu sichern, den Klimaschutz zu unterstützen und Strategien zur Sicherung der Lebensgrundlagen aufrechtzuerhalten. Diese Initiativen weisen jedoch etwas widersprüchliche Ziele auf, da sie gleichzeitig Walderhaltungs- und menschliche Wohlergehensziele verfolgen.

Um Erhaltungs- und Entwicklungsziele in Einklang zu bringen, sind neue Ansätze erforderlich. Der multifunktionale Landschaftsansatz ist als Ansatz positioniert, der zu einer nachhaltigen Landschaftspflege beiträgt. Der multifunktionale Landschaftsansatz zielt darauf ab, die Biodiversität, Ökosystemleistungen und ökologischen Funktionen zu schützen und gleichzeitig Lebensgrundlagen für die Menschen zu schaffen, die die Landschaft bewohnen. Bei diesem Ansatz muss man zunächst die verschiedenen Landnutzungen in der Landschaft erkennen und verstehen, wie sich diese Landnutzungen auf die Bereitstellung von Ökosystemleistungen auswirken, von denen Menschen abhängig sind, um ihre Lebensunterhaltsstrategien zu entwickeln. Es muss auch anerkannt werden, dass die Landschaftsdynamik durch institutionelle Elemente beeinflusst werden kann, die nicht nachhaltige Landnutzungen fördern oder verhindern können.

In den Tropen gibt es kaum Studien, die sich mit den Auswirkungen von Übergangsphasen der Landnutzung auf die Bereitstellung einzelner und mehrerer Ökosystemleistungen befassen. Der multifunktionale Landschaftsansatz wird sowohl in der Forschung als auch in der Praxis kaum berücksichtigt. Darüber hinaus wird das Potenzial, das Institutionen für die Landschaftsdynamik haben können, normalerweise übersehen, da viele Studien nur die ökologische Perspektive einnehmen. Aus diesem Grund bietet diese Dissertation weitere Einblicke in Mensch-Land-Interaktionen und die damit verbundenen Umweltauswirkungen. Die Studie wurde in den tropischen Tiefland-Regenwaldlandschaften Ecuadors durchgeführt und befasst sich mit drei Hauptthemen: (i) die Landnutzungs-Übergangsphasen und ihre Auswirkungen auf die Bereitstellung von Ökosystemleistungen und die Multifunktionalität von Ökosystemleistungen; (ii) den Einfluss des anreizbasierten Waldschutzes auf die Bereitstellung von Ökosystemleistungen in den Übergangsphasen der Landnutzung; (iii) der Beitrag des anreizbasierten Waldschutzes, um die Entwaldung über die Grenzen des Schutzprogramms hinaus zu stoppen.

Diese Studie basiert auf In-situ-Informationen aus zwölf ausgewählten Landschaften (10 × 10 km) im ecuadorianischen Zentralamazonas und in Chocó. Die untersuchten Landschaften sind ein Mosaik aus alten Wäldern, abgeholzten Wäldern, die aus Holzgewinnungsaktivitäten resultieren, Sukzessionswäldern, die aus natürlicher Regeneration nach Landaufgabe resultieren, Waldplantagen und Agroforstsystemen. Insgesamt habe ich 156 Inventarplots (1600 m2) installiert, auf denen ich Informationen für Indikatoren der Bereitstellung, Regulierung und Unterstützung von Dienstleistungen und Biodiversität gesammelt habe, um die Menge der Bereitstellung von Ökosystemdienstleistungen in jeder Übergangsphase der Landnutzung zu quantifizieren. Der Ökosystemleistungs-Multifunktionalitätsindex wurde ebenfalls unter Verwendung des multifunktionalen Durchschnittsansatzes geschätzt. Ökologische Daten wurden durch Haushaltsdaten aus Erhebungen in den ausgewählten Landschaften ergänzt; Diese Daten ermöglichten es uns, die Wirkung des anreizbasierten Naturschutzes auf die Entwaldungsentscheidungen der Haushalte zu analysieren.

Ich habe eine Hauptkomponentenanalyse (PCA) durchgeführt, um Synergien und Kompromisse zwischen Ökosystemleistungen und dem Multifunktionalitätsindex zu identifizieren. Darüber hinaus habe ich die Varianzanalyse geschätzt, um Unterschiede in der Bereitstellung von Ökosystemleistungen in den Phasen des Landnutzungsübergangs zu bewerten und um die Wirkung des anreizbasierten Naturschutzprogramms auf die Bereitstellung von Ökosystemleistungen und die Reduzierung der Entwaldung zu bewerten. Ich habe auch eine Regressionsanalyse geschätzt, um den Einfluss des auf Anreizen basierenden Naturschutzprogramms auf die Entwaldungsentscheidungen der Haushalte zu bewerten. Die wichtigsten Erkenntnisse sind wie folgt:

Die Analyse zeigte synergetische Beziehungen für die Multifunktionalität von Ökosystemleistungen, Regulierungsleistungen, Bereitstellungsleistungen und Biodiversitätsindikatoren. Oberirdische Kohlenstoffvorräte hatten Synergien mit verschiedenen Dienstleistungen und mit der Ökosystemdienstleistung Multifunktionalität, aber kontrastierende Muster mit bodenbezogenen Dienstleistungen. Die synergetische Beziehung der oberirdischen Kohlenstoffvorräte legt nahe, dass sie als übergreifende Dienstleistung betrachtet werden kann, da sie ein empfindlicher Indikator für die Waldintegrität ist und starke bis mäßige Synergien mit der Multifunktionalität von Ökosystemleistungen und mehreren Ökosystemleistungen hatte. Jede anthropogene Aktivität, die oberirdische Kohlenstoffvorräte verändert, kann die Zunahme oder Abnahme zusätzlicher Ökosystemleistungen auslösen.

Oberirdische Kohlenstoffvorräte können ein nützlicher Indikator sein, um verschiedene Ökosystemleistungen und Biodiversität zu überwachen.

Diese Forschung zeigt weiter, dass die Holzentnahme in den Untersuchungsgebieten einen Rückgang der Ökosystemleistung Multifunktionalität um 16 % bis 18 % verursachte. Die Holzgewinnung hat einen hohen bis mittleren Einfluss auf Versorgungs- und Regulierungsleistungen sowie die Pflanzenvielfalt. Holzeinschlagsaktivitäten wirken sich stark auf das Holzvolumenpotenzial aus und führen zu einer Reduzierung dieser Bereitstellungsleistung zwischen 40 % und 49 %; Diese Aktivitäten verringerten auch die oberirdischen Kohlenstoffvorräte in den ausgewählten Landschaften um 30 % bis 32 %. Die Studie zeigt, dass Ökosystemleistungen in abgeholzten Wäldern stark beeinträchtigt werden, obwohl der Holzeinschlag unter der technischen Norm der Umweltbehörde erfolgt.

Um die Wiederherstellung von Ökosystemleistungen und die Multifunktionalität von Ökosystemleistungen zu bewerten, wurden Sukzessionswälder, Agroforstsysteme und Plantagen betrachtet. Die Ergebnisse zeigten, dass Sukzessionswälder die effektivste Option sind, um einzelne und gebündelte Ökosystemleistungen wiederherzustellen. Der Sukzessionswald bietet hohe Werte an Versorgungsleistungen (Holzvolumenpotenzial und Nichtholzwaldprodukte), Regulierungsleistungen (oberirdische Kohlenstoffvorräte und Bodenkohlenstoffvorräte) und Pflanzenvielfalt, jedoch mit einer anderen Struktur und Pflanzenzusammensetzung als Altholz Wälder. Die Agroforstsysteme und die Plantagen bieten eine geringere Ökosystemleistungsmultifunktionalität, haben jedoch ein großes Potenzial, bodenbezogene Leistungen wie Bodenkohlenstoffvorräte, Stickstoff, Phosphor und Kalium zurückzugewinnen.

Die Ergebnisse zeigten auch, dass das anreizbasierte Waldschutzprogramm zum Erhalt der bewerteten Ökosystemleistungen beiträgt, da in Altwäldern unter dem Schutzprogramm keine Verringerung der Ökosystemleistungen beobachtet wurde. Darüber hinaus kam es in abgeholzten Wäldern ohne anreizbasierten Waldschutz zu einem stärkeren Rückgang der Ökosystemleistungen mit Auswirkungen auf die oberirdischen Kohlenstoffvorräte und den Artenreichtum. Ein interessantes Ergebnis dieser Studie war, dass die oberirdischen Kohlenstoffvorräte in den abgeholzten Wäldern in der Nähe der Fördergebiete um 21 % reduziert wurden, während abgeholzte Wälder in Landschaften ohne Schutzprogramm 41 % weniger oberirdische Kohlenstoffvorräte aufwiesen. Um größere Auswirkungen auf die Bereitstellung von Ökosystemleistungen zu verhindern, ist es unerlässlich, dass die ecuadorianische Umweltbehörde die Einrichtung strenger Überwachungs- und Kontrollmaßnahmen nach der Ernte diskutiert. In den ausgewählten Landschaften trägt der Anreizwaldschutz dazu bei, die Entwaldung zu reduzieren. Diese Studie zeigte eine Verringerung der jährlichen Netto-Entwaldungsrate nach der Umsetzung des anreizbasierten Waldschutzes auf Gemeindeebene. Bei der Bewertung der Rolle des auf Anreizen basierenden Waldschutzprogramms bei den Entscheidungen der Haushalte zur Abholzung zeigten die Ergebnisse, dass Haushalte, die in der Nähe der vom Programm erfassten Gebiete leben, im Gegensatz zu Haushalten, die sich in Landschaften befinden, eine geringere Wahrscheinlichkeit (56 % weniger) zur Abholzung haben kein Einfluss des Anreiz-Programms. Die Ergebnisse deuten also darauf hin, dass das anreizbasierte Naturschutzprogramm über die Schutzgebiete hinaus wirkt.

Die Ergebnisse zeigten, dass anreizbasierter Naturschutz ein vielversprechendes Potenzial zur Verbesserung multifunktionaler Landschaften hat, indem die Entwaldung über die Grenzen des Schutzgebiets hinaus bekämpft wird. Der anreizbasierte Naturschutz kann auch dazu beitragen, den Waldschutz, die Holzproduktion und die Wiederherstellung von Waldlandschaften auszugleichen und zu integrieren, wie dies bei Landschaften mit Anwesenheit des Programms beobachtet wurde. Diese Studie verbindet den Ökosystemleistungsrahmen mit dem landschaftsmultifunktionalen Ansatz und liefert nützliche Informationen darüber, wie Landnutzungen einzelne und mehrere Ökosystemleistungen beeinflussen. Durch die Analyse von Synergien und Zielkonflikten können herausragende Ökosystemleistungen identifiziert werden, um die Multifunktionalität der Landschaft zu erhalten. Zum ersten Mal wird die Schirmart in den Ökosystemdienstleistungsrahmen aufgenommen; Das Dachdienstleistungskonzept könnte verfeinert und in Waldschutz-, Landschaftswiederherstellungs- und ökologischen Überwachungsprojekten eingesetzt werden. Diese Studie bewertete auch, wie institutionelle Aspekte die Bereitstellung von Ökosystemleistungen beeinflussen können; In diesem Fall habe ich das anreizbasierte Naturschutzprogramm in Betracht gezogen, da dies die zweitwichtigste Naturschutzpolitik in Ecuador ist. Die Studie ergab, dass Ökosystemleistungen auch über die Schutzgebiete hinaus von solchen Programmen profitieren können. Ebenso zeigte das anreizbasierte Naturschutzprogramm wichtige Auswirkungen auf die Landnutzungsentscheidungen der Haushalte. Die Wirkung von anreizbasierten Naturschutzprogrammen auf Ökosystemleistungen und das Verhalten von Haushalten wurde bisher für das Land nicht bewertet und wird für die Tropen selten bewertet.

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1. Introduction

To tackle the environmental challenges of deforestation and forest degradation, several international initiatives have emerged to conserve and restore forest ecosystem services while accomplishing welfare objectives for society (FAO and UNEP, 2020). Among the initiatives with the greatest global impact is the United Nations Framework Convention on Climate Change (UNFCC); the Convention on Biological Diversity (CBD); the mechanism to Reduce Emissions from Deforestation and Forest Degradation (REDD+); and the 2030 Agenda of the United Nations that comprises seventeen sustainable development goals (SDGs). These global initiatives are interconnected and have in common the fact that they are aimed at conserving biodiversity and avoiding its rapid decline while fighting against climate change and securing a sustainable world for people. Within all these initiatives, forests play a vital role as their maintenance contributes to biodiversity conservation, climate change mitigation, and ecosystem services supply to humanity (Maes et al., 2013; MEA, 2005; Tsioumani, 2022).

The underlying synergies between the abovementioned global initiatives, point out that conventional forest conservation approaches might not be enough to successfully address current environmental and development challenges. In this sense, the need to adopt a multifunctional landscape approach is now recognized as a strategic tool to address a problem that goes beyond the environmental dimension and that also includes social, economic, and institutional aspects (Arts et al., 2017; Milder et al., 2012). The multifunctional landscape approach aims to safeguard biodiversity, ecosystem services, and ecological functions while providing livelihood opportunities for people living within the landscape. Ecosystem service multifunctionality refers to the capacity of a forest to co-supply simultaneously multiple ecosystem services to society (Manning et al., 2018). By recognizing forests as a key component of multifunctional landscapes, stakeholders can act in favor of biodiversity and ecosystem services maintenance while satisfying a mix of human demands from different actors (farmers, companies, conservation authorities, and the local and international community).

In the tropics despite well-intentioned global initiatives and the increased awareness of human dependency on ecosystem services provided by forests (Costanza et al., 2017), there is an ongoing land use pressure in forested lands due to the need to satisfy multiple human demands. The unsustainable use of forested lands results in highly degraded or deforested landscapes with serious implications for the supply of ecosystem services (FAO and UNEP, 2020; MEA, 2005). Land use practices transform previously forested landscapes into mosaics of different forest covers that reflect land-use transition phases and evidence decline and recovery ecological processes. Land-use transition phases are defined as changes in a land-use system and are associated with changes in the supply of ecosystem services (Bremer and Farley, 2010; Lambin and Meyfroidt, 2010; Wilson et al., 2017).

The Ecuadorian ecosystems have similar dynamics of deforestation and landscape degradation. Ecuador is a tropical country with 12.6 M hectares of forests (around 50% of the national territory), 80% of the Ecuadorian forest is concentrated in the Amazon and Chocó region, where the provinces of Pastaza, Napo, Orellana, and Esmeraldas represent the largest area of native forest. Despite the importance of the Ecuadorian Amazon and Choco regions for forest cover maintenance in the country, these regions have a high forest cover loss (51000 ha/year) (MAE, 2017), in addition, these regions are also affected by ongoing land demand, much of it taking place at the forest frontiers. The unsustainable development in consequence in the Ecuadorian Amazon and Choco provokes deforestation, forest degradation, and the fragmentation of natural ecosystems (Sierra et al., 2021). Human-modified landscapes in the Ecuadorian forest frontiers usually present a mosaic of intact old-growth forest remnants, logged forests, forest plantations, successional forests resulting from land abandonment, and agroforestry systems (Figure 1).

In Latin America and Ecuador, forests with no protection status frequently undergo unsustainable practices (timber extraction, charcoal production, or fires) (Hososuma et al., 2012; Kissinger et al., 2012), resulting in the decrease or loss of bundles of ecosystem services (Bremer and Farley, 2010; Lambin and Meyfroidt, 2010; Wilson et al., 2017). Despite that the Ecuadorian government states that logging shall be conducted under technical procedures and only with a legal permit issued by the environmental authority, logging usually precedes land use change (Sierra et al., 2021). In the Ecuadorian forest frontiers, it is common to observe that once the species of commercial interest disappear due to logging, landholders introduce pastures or crops that are later abandoned due to the loss of productivity. Land abandonment promotes natural regeneration and the appearance of secondary forests that help offset the loss of ecosystem services. Forest plantations and agroforestry systems are also part of the landscape mosaic and can be used to avoid a complete decline of ecosystem services while balancing conservation and development goals. Natural regeneration, plantations, and agroforestry systems are part of the forest landscape restoration (FLR) to halt ecosystem degradation, enhance ecosystem integrity and promote the recovery of multiple ecosystem services (Chazdon et al., 2020b; Sabogal et al., 2015).

Due to the threats that old-growth forests face, the Ecuadorian government implemented two major strategies to avoid deforestation and forest degradation: (i) the national system of protected areas and (ii) the Socio Bosque program. The national system of protected areas is a command and control policy that forbids the extraction of natural resources (Dudley and Stolton, 2008); currently, all protected areas are owned by the State. The Socio Bosque program, on the other hand, is an alternative mechanism created in 2008 to stop deforestation in private lands (individual or communal); it is an incentive-based program aimed to conserve forest areas or ecosystem services, while forest owners receive monetary compensation. Besides ecosystem protection, incentive-based forest conservation may help poverty alleviation and biodiversity conservation (Lewis et al., 2011; Sims and Alix-Garcia, 2017). Socio Bosque is among the ten largest conservation programs in the world due to its coverage (FAO and UNEP, 2020), however, its effect on ecosystem services conservation is not clear as many studies have focused on forest cover maintenance, likewise, its influence on deforestation decisions has not been studied.

Given the complexity of tropical and Ecuadorian forest landscapes, the attempts to harmonize the ecosystem services conservation and development goals are often unsuccessful, this is partially due to the lack of understanding of forest land use transition phases occurring at the landscape level and the relation with the ecosystem service multifunctionality. Deep knowledge about landscape mosaics and ecosystem service multifunctionality is the basis for building up multifunctional landscapes (Hölting et al., 2020). Therefore, quantitatively knowing the levels of ecosystem services supply throughout the land use transition phases, can help stakeholders to fulfill different purposes. Similarly, understanding the effect of extraction activities or restoration actions on single and multiple ecosystem services could shed light on the most effective ways to manage tropical forest landscapes. In addition, forest landscape management is not complete until the stakeholders are considered. In Ecuador, farmers are key actors to achieve conservation and sustainable development goals (Terlau et al., 2019); they are at the same time the precursors of forest landscapes' transformation into human-modified mosaics. Farmers face multiple socioeconomic constraints that force them to manage the land unsustainably, revealing again the need to incorporate the multifunctional landscape approach when addressing the tropical landscape challenges. In recent decades, monetary incentives have emerged as a strategy to motivate local actors to conserve the forest, reduce deforestation, and move towards more sustainable production systems (Blundo-Canto et al., 2018; Wunder, 2015). Thus, incentive-based forest conservation might contribute to achieving conservation aims and improving welfare with positive impacts on the construction of multifunctional landscapes. Unfortunately, there is no concluding evidence on how incentive-based forest conservation influences ecosystem services supply and deforestation beyond the conserved area; understanding this will help to orient programs and funding more efficiently.

To address multiple demands of society and to reduce forest loss and ecosystem services decline, research needs to integrate different perspectives to comprehend the impacts of land uses on ecosystem services as well as the repercussions of conservation policies on deforestation. This dissertation attempts to provide more evidence on the human-land interactions and the associated environmental effects, shedding light on decision-making processes related to the ecosystem's conservation, decline, and recovery (Balvanera et al., 2012; Daily, 1997; MEA, 2005). The study was conducted in the tropical lowland landscapes of Ecuador and encompasses three main issues: (i) the land use transition phases and their implication in the supply of ecosystem services and ecosystem service multifunctionality; (ii) the influence of the incentive-based forest conservation on ecosystem services provision across the land use transition phases; (iii) the contribution of incentive-based forest conservation to halt deforestation beyond the limits of the conservation program.

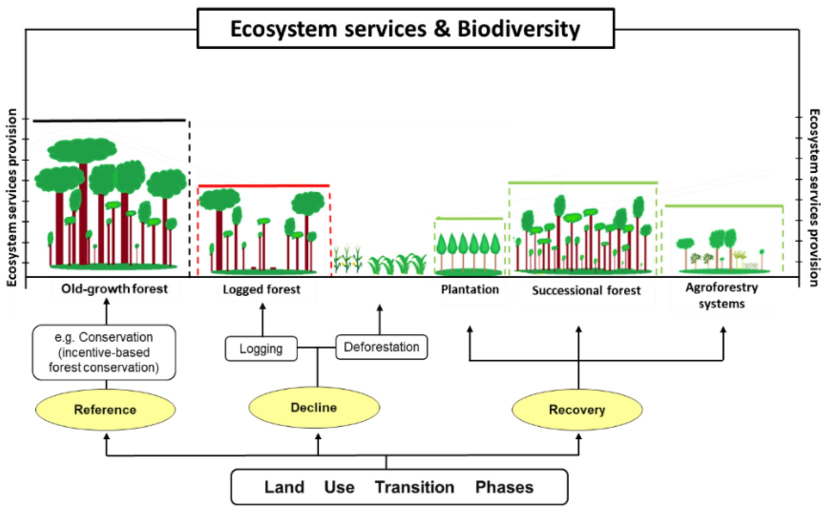

Figure 1. Schematic representation of the land use transition phases based on the ecosystem services maintenance, decline, and recovery. The order of the recovery phases does not imply a process, these are examples of the most common phases that promote ecosystem services recovery at the landscape level in Ecuador.

2. Research gap and thesis aim

Tropical forest landscapes are highly diverse and provide different levels of ecosystem services to society. The ecosystem services supply is closely linked to land use practices attached to human demands and policies implemented (De Groot et al., 2002; Queiroz et al., 2015). Tropical forested landscapes nowadays are being transformed into mosaics due to different land use practices; such transformation implies the supply of ecosystem services and highlights the need for studies that evaluate how land uses affect the quantity in which an ecosystem service is delivered.

Research in Latin America is mainly focused on individual land use transition phases. In many cases we only analyze conservation and management strategies separately and not as a landscape mosaic, forgetting that tropical landscapes are made up of forest land use transition phases. For example, researchers have evaluated the role of protected areas and incentive-based forest conservation in the individual maintenance of ecosystem services (Jones et al., 2020; Mohebalian and Aguilar, 2018; Nepstad et al., 2006; Porter-Bolland et al., 2011; Sánchez-Azofeita et al., 2007); others have studied the reduction or loss of ecosystem services due to timber extraction or deforestation (Armenteras et al., 2009; Gerwing, 2002; Rutishauser et al., 2015; Sierra et al., 2021; West et al., 2014). In the case of ecosystem services recovery, studies commonly analyze one restoration strategy at a time. Moreover, studies usually evaluate one or few ecosystem services (Alamgir et al., 2016; Anderson-Teixeira et al., 2012; Bertzky et al., 2010; Brancalion et al., 2014; Dauber et al., 2005; Gerwing, 2002; Lara et al., 2009; Pearson et al., 2017; Sist and Nascimiento, 2007; Suryatmojo et al., 2011; Sutherland et al., 2016; Uriarte et al., 2011), paying little attention to the ecosystem service multifunctionality.

To the author’s knowledge, the assessment of the ecosystem service multifunctionality across the land use transition phases has not been conducted before in Latin America, as most studies comprise temperate forests (Cruz-Alonso et al., 2018; Funk et al., 2019; Strobl et al., 2019). As mentioned before, several studies concentrate on single ecosystem services and one land use transition phase at a time, overlooking the landscape perspective. This calls for more comprehensive studies that contribute to the understanding of ecosystem services conservation, decline, and recovery within the landscape. Moreover, little is known about the impact of incentive-based forest conservation programs on the preservation of ecosystem services. There is a lack of studies that evaluate the influence of this program in the provision of ecosystem services in the logged and successional forest near the incentive-based forest conservation and if the program could affect deforestation decisions in communities close by.

This dissertation aims to reduce this knowledge gap by incorporating ecological and human perspectives into the analysis. This research adopts the multifunctional landscape approach which includes a variety of land use transition phases and a diverse provision of ecosystem services to explore and analyze human-environmental challenges in an integrated way (Fischer et al., 2017). Hence, this dissertation contributes by showing new results of how a variety of land use transition phases can supply different ecosystem service multifunctionality, how incentive-based forest conservation can influence ecosystem services provided within and outside the program across the land use transition phases and the influence of this program of deforestation rates and the household decision to deforest beyond the conserved area. For this purpose, I have the following aims and associated research questions:

1. Assess the ecosystem service multifunctionality and ecosystem services provision across the land use transition phases.
    - What synergies and trade-offs are observed between ecosystem service multifunctionality and ecosystem services across the land use transition phases?
    - What is the effect of timber extraction on the ecosystem service multifunctionality and ecosystem services provision?
    - Which recovery phase provides the highest ecosystem service multifunctionality and provision of ecosystem services?
2. Evaluate the influence of incentive-based forest conservation on ecosystem services provision across the land use transition phases.
    - Is there a difference in the provision of ecosystem services in old-growth forests under incentive-based forest conservation when compared with old-growth forests with no incentive-based forest conservation?
    - Does incentive-based forest conservation influence the provision of ecosystem services in neighboring logged forests?
    - Does incentive-based forest conservation influence the recovery of ecosystem services in the neighboring successional forests?
3. Understand the contribution of incentive-based forest conservation to halt deforestation beyond the limits of the conservation program.
    - Does the annual rate of net deforestation at the parish level change after the implementation of incentive-based forest conservation?
    - Does incentive-based forest conservation influence household deforestation decisions in farms close to the program?

The first aim addressed in Publication I (appendix 2), explores the synergies and trade-offs among ecosystem services and ecosystem service multifunctionality across land use transition phases (Figure 2). In addition, I quantify and compare the level of ecosystem services supply and the ecosystem service multifunctionality in old-growth forests versus logged forests to evaluate the effects of selective timber harvesting in the decline of ecosystem services and ecosystem service multifunctionality. I also assess the potential to recover individual ecosystem services and the ecosystem service multifunctionality by evaluating three recovery phases (successional forest, plantation, and agroforestry systems).

The second aim is addressed in Publication II (appendix 3); it analyses the effect of the incentive-based forest conservation program (Socio Bosque), in maintaining ecosystem services within the forest under conservation. Besides, I also evaluate whether the presence of this program influences the reduction in ecosystem services in the adjacent logged area, or if it enhances the recovery of ecosystem services in successional forests.

Finally, the third aim is addressed in Publication II and III (appendix 3 and 4). On one hand, I evaluate whether the presence of the incentive-based forest conservation program influenced the annual rate of net deforestation at a parish level (Publication II). On the other hand, I evaluate whether this program influences households´ deforestation decisions while controlling household characteristics (Publication III).

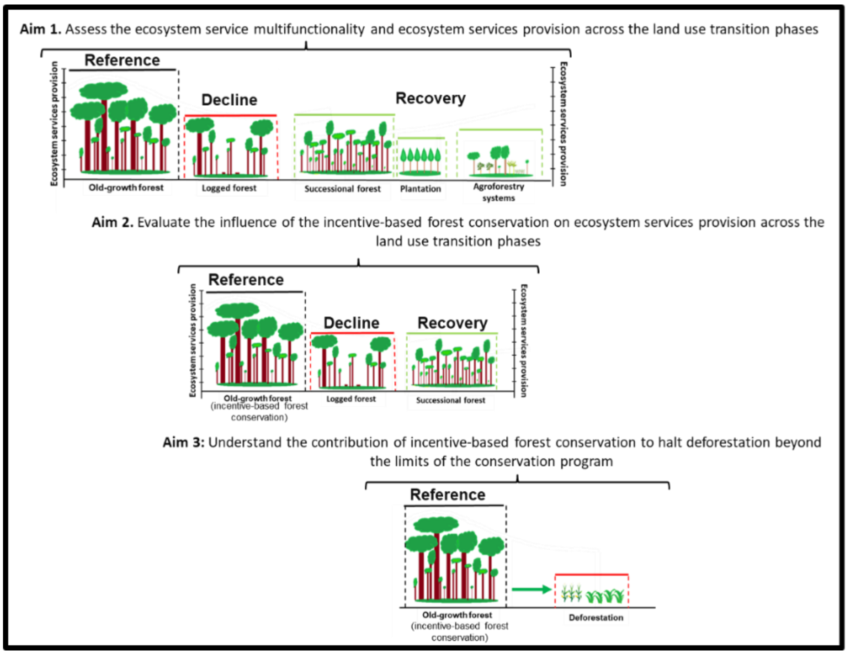

Figure 2. Schematic representation of the dissertation aims

4. Paper Contributions

The dissertation is based on three peer-reviewed publications (Appendix 2 to Appendix 4) which are listed in the following table.

Table 1. List of publications on which this dissertation is based

List of publications	Summary	Author´s contributions
Eguiguren, P., Ojeda Luna, T., Torres, B., Lippe, M., Günter, S., 2020. Ecosystem Service Multifunctionality: Decline and Recovery Pathways in the Amazon and Chocó Lowland Rainforests. *Sustainability* 12, 7786.	This publication explores the ecosystem services and ecosystem service multifunctionality synergies and trade-offs across the land use transition phases. Besides, it evaluates the decline in the provision of ecosystem services and ecosystem service multifunctionality due to timber extraction and the potential recovery of the ecosystem services and ecosystem service multifunctionality in three common recovery pathways.	Conceptualization, P.E., and S.G.; methodology, P.E. and S.G.; Formal analysis, P.E.; Investigation, P.E.; Data curation, P.E.; Writing—original draft preparation, P.E. and T.O.L.; Writing—review and editing, P.E., T.O.L., B.T., M.L., and S.G.; Project administration, P.E., T.O.L., and B.T.; funding acquisition, P.E., T.O.L., and S.G., Supervision, S.G.
Eguiguren, P., Fischer, R., Günter, S., 2019. Degradation of Ecosystem Services and Deforestation in Landscapes with and without Incentive-Based Forest Conservation in the Ecuadorian Amazon. *Forests* 10, 442.	The publication analyzes the influence of incentive-based forest conservation on ecosystem services maintenance within the area of the programs. It also explores if the program can influence timer extraction activities and the recovery of the successional forest, outside the intervention area of the program. Finally, explores if the program can have a positive effect on	Conceptualization: P.E. and S.G.; Methodology: P.E. and S.G.; Formal analysis: P.E.; Investigation: P.E.; Data curation: P.E.; Writing—original draft preparation: P.E.; Writing—review and editing; P.E., R.F., and S.G; Visualization: P.E.; Supervision: S.G.; Project administration: S.G., P.E., and R.F.; Funding acquisition: S.G., R.F., and P.E.

	deforestation rates at the parish level	
Ojeda, T., Eguiguren, P., Torres, B., Günter S. and Dieter M. 2020. What Drives Households Deforestation Decisions? Insights from the Ecuadorian Lowland Rainforests. *Forests*, 11, 1131	Households are the principal land use decision-makers in the tropical frontiers. Therefore, their actions have a direct impact on the deforestation process. This publication analyses the determinants of household deforestation decisions, with an emphasis on the influence of incentive-based forest conservation.	Conceptualization, T.O.L, P.E., S.G., B.T., and M.D.; Methodology, T.O.L., and P.E.; Validation, T.O.L, P.E., S.G., B.T., and M.D.; Formal analysis, T.O.L., and P.E.; Investigation T.O.L. and P.E.; resources, T.O.L, P.E., and S.G; data curation, T.O.L., and P.E.; writing—original draft preparation, T.O.L. and P.E.; writing—review and editing, T.O.L, P.E., S.G., B.T. and M.D.; supervision, S.G., and M.D.; Project administration, T.O.L, P.E., S.G., and B.T.; Funding acquisition, T.O.L, P.E., and S.G.

P.E.: Paul Eguiguren; S.G.: Sven Günter; R.F.: Richard Fisher; T.O.L.: Tatiana Ojeda Luna; B.T., Bolier Torres, M.L.: Melvin Lippe; and, M.D.: Matthias Dieter.

3. State of Research

This chapter presents the theoretical concepts and research that oriented this dissertation. This section begins with the multifunctionality approach, then it presents the current research related to the land use transition phases, followed by the influence of conservation strategies, the main drivers of deforestation and ecosystem services degradation, as well as the motivations of households' behavior to deforest, and recovery options to maintain and promote the enhancement of ecosystem services.

3.1. Multifunctionality approach

Human-modified landscapes are common in the tropics, reducing forest-related livelihoods and benefits people depend on, emphasizing the need to adopt frameworks capable to address environmental and human needs. The multifunctional approach has been proposed as a suitable framework that could help find a balance between the ecological, production, and cultural functions and halt the loss of ecosystem services (Lovell and Johnston, 2009). Multifunctionality is commonly linked to the provision of ecosystem services (Stürck and Verburg, 2017); thus a multifunctional landscape can simultaneously supply different livelihoods opportunities, and food security, conserve species and ecosystems and provide aesthetics recreationally (Hölting et al., 2019; Knoke et al., 2016; Lovell and Johnston, 2009; O'Farrell and Anderson, 2010). Complementarily, ecosystem service multifunctionality refers to the capacity of a forest to co-supply simultaneously multiple ecosystem services to society (Manning et al., 2018).

Multifunctionality studies have been mainly executed in Europe on temperate forests and have analyzed the capacity to supply multiple ecosystem services (Cruz-Alonso et al., 2018; Mouillot et al., 2011; Rodríguez-Loinaz et al., 2015; Van der Plas et al., 2018), the effects of land use change (Schindler et al., 2014; Stürck and Verburg, 2017), to evaluate ecosystem´s conservation status (Maes et al., 2012) and on forest management to evaluate the capacity of forests (e.g. *Quercus* or *Fagus*) or mixed forests to fulfill different ecosystem services to society (Borrass et al., 2017; Hausler and Scherer-Lorenzen, 2001; Paletto et al., 2012). In tropical areas like the Amazon Basin and the Chocó region, there is still a lack of studies that include the multifunctionality approach; this could be related to a high diversity of forest types and ecosystems in the tropics, as well as more complex landscape dynamics when compared to Europe.

Tropical forests must fulfill different demands from society by providing several ecosystem services. Therefore, it is crucial to understand the amount in which each ecosystem service is provided, especially to anticipate any shortages; however, quantifying single and multiple ecosystem services is still a challenge. In recent years, ecologists have increased their interest to evaluate ecosystem multifunctionality (Hölting et al., 2019; Manning et al., 2018), recognizing that single ecosystem services evaluations are not enough to understand the landscape dynamics. Multifunctional evaluations were first applied as a relative source use index (Hooper and Vitousek, 1998) and later applied to assess how ecological attributes (e.g. biodiversity) are related to ecosystem functions in several studies (Fanin et al., 2018; Gross et al., 2017; Hooper and Vitousek, 1998; Maestre et al., 2012b; Mouillot et al., 2011).

Recently, multifunctionality has been applied to understand how forests can supply several benefits to society (Manning et al., 2018; Schuldt et al., 2018). To quantify the ecosystem service multifunctionality the averaging approach has been commonly used, this approach uses the mean of the standardized values of the services to obtain one single metric providing an easy interpretation of the multifunctionality (Hölting et al., 2019; Mouillot et al., 2011; Schuldt et al., 2018). High values show that an ecosystem is achieving high levels of multifunctionality. On the other hand, low values suggest a reduction in the multifunctionality levels, capturing the effects of management decisions. Ecosystem service multifunctionality could help to improve management decisions and complemented with the evaluation of ecosystem services synergies could contribute to identifying umbrella ecosystem services.

3.2. Ecosystem services

The provision of ecosystem services within a forest depends on the ecosystem's functions. The ecosystem functions are an ecosystem-centered concept and are the product of the interactions between the ecosystem structures and processes (Brockerhoff et al., 2017; De Groot et al., 2002; Mace et al., 2012). In consequence, ecosystem services are a result of complex ecological processes and functions within forests. The ecosystem services concept incorporates a human-centered perspective, in which ecosystem services are defined as the benefits that people obtain directly or indirectly from ecosystems (Alcamo, 2003; Brockerhoff et al., 2017; Haines-Young and Potschin, 2018; MEA, 2005).

Ecosystem services evaluation and discussions gained attention during the 1990s (Costanza et al., 1997; Daily, 1997) and different frameworks have emerged since then (De Groot et al., 2002; Haines-Young and Potschin, 2018; MEA, 2005; TEEB, 2008). Among these frameworks, the Millennium Ecosystem Assessment (MEA) (2005) has been positioned as one of the most recognized and commonly used around the world. The MEA highlights the importance of ecosystem services and human welfare, making this framework suitable for studies that evaluate the influence of human actions on ecosystem services provision.

The MEA classifies the ecosystem services into four categories: i) supporting services, defined as those services that influence the generation and maintenance of other groups of services as they are related to ecosystem processes (e.g. nutrient cycling, soil formation, pollination, or primary productivity) (Alcamo, 2003; Costanza et al., 2017; MEA, 2005). ii) provisioning services which comprise those products used at the local level, with a tangible benefit for human welfare; these services are obtained directly from ecosystems (e.g. food, timber, non-timber forest products, medicine). iii) regulating services refer to any benefit from the regulation processes occurring within the ecosystem (e.g.: climate regulation). iv) cultural services which are those non-tangible benefits people obtain from the ecosystem (e.g. cultural diversity, aesthetic values, spiritual or religious) (MEA, 2005) (Figure 3).

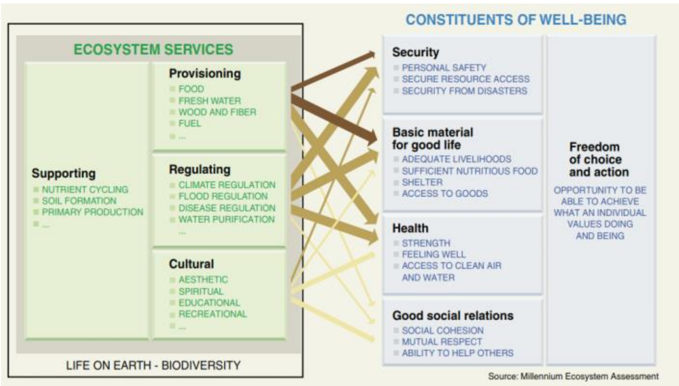

Figure 3.The link between ecosystem services and human well-being. Source: MEA, (2005)

Given the difficulty of directly measuring ecosystem services, a common practice is to identify indicators that are used as proxies to quantify the level of provision of the ecosystem services. When identifying these indicators, the researcher needs to consider the facility to easily measure them, indicators should reflect most accurately the selected ecosystem service. Adequate identification of such indicators allows a better understanding of the influence of natural resources management and land use transitions on ecosystem services provision, given more reliable information for decision-making (Bunker et al., 2005; De Groot et al., 2010; Don et al., 2011; Pearson et al., 2017; Roopsind et al., 2017; Van Oudenhoven et al., 2012).

3.2.1. Ecosystem services synergies and trade-offs

People depend on ecosystems and their services, in the last decades the unsustainable use of natural resources has influenced ecosystems, affecting and reducing their capacity to provide ecosystem services (MEA, 2005). Because of this, the ecosystem services concept is used to evaluate the interactions between natural systems and human well-being. Ecosystem services interact with each other, these interactions are called synergies and trade-offs. Synergies are expressed as win-win situations or positive correlations generating the simultaneous enhancement in various ecosystem services; whereas, trade-offs results and win-lose conditions or negative correlations where the increment of one ecosystem service will result in a deplete of another (Bennett et al., 2009; Raudsepp-Hearne et al., 2010).

Any management decision on an ecosystem will have a positive or negative effect on different services (Balvanera et al., 2012). For example, Turner et al., (2014) and Raudsepp-Hearne et al., (2010), observed trade-offs in land use/cover between provisioning services (e.g. crops and animals) and regulating services (e.g. carbon), here the increment of area for food supply lead to a decrease on forest and therefore the carbon pools. Land use change from forest to pastures also causes a loss of soil supporting services indicators (e.g. carbon, nitrogen, and potassium) (Mainville et al., 2006). Within an ecosystem, trade-offs are also observed, in forests under logging interventions the increase in the use of provisional services mainly timber, creates a reduction in the above-ground carbon stocks. Slash pine (*Pinus elliottii*) ecosystems for timber production could also result in a reduction of water yield (Cademus et al., 2014). Meanwhile, above-ground biomass in plantations which is related to above-ground carbon, is a major source of soil carbon through litter fall in the first stages (Lu et al., 2014), meaning that a great reduction of above-ground carbon due to timber harvesting will affect later in soil carbon (Montagnini and Jordan, 2005). On the other hand, Yang et al., (2015) noted synergies in tea plantations with a positive relationship among the provisioning services (e.g. food) and between regulating services (soil regulation with water regulation), meanwhile, Foley et al., (2005) and Turner et al., (2014) mentioned synergies between carbon and water supply and Raudsepp-Hearne et al., (2010) among carbon and forest recreation.

Biodiversity influences the provision of ecosystem services (Balvanera et al., 2016, 2006; Chopra et al., 2005; Harrison et al., 2014; Mace et al., 2012). According to Potts et al., (2010) a decline in the abundance and diversity of birds or insects can reduce animal pollination for wild plant populations. Balvanera et al., (2006), observe a positive relationship between biodiversity and services related to nutrient cycling, in this study plant and mycorrhizal diversity increase the nutrients in plant pools. Meanwhile, Gamfeldt et al., (2013) found that biomass production and soil carbon storage are greater when the number of species increases in production forest. These studies show the strong relationship between biodiversity and ecosystem services and the importance of biodiversity to underpin different services.

3.3. Land use transition phases: the conservation, decline, and recovery of forests ecosystem services

The land use transition phases are defined as changes in the land use system and are associated with changes in the supply of ecosystem services (Bremer and Farley, 2010; Lambin and Meyfroidt, 2010; Wilson et al., 2017). These land use transition phases consider the forest quality, allowing the understating of how different driving forces like conservation strategies (reference forest), timber extraction activities (decline phase), and restoration or reforestation actions (recovery phase) influence forest integrity and ecosystem services. This concept goes beyond the common definition of the forest transition that only observes the changes in forest cover.

The most typical land use transition phases observed in the tropics start from the reference forest to an overexploited or managed forest (Bremer and Farley, 2010; Lambin and Meyfroidt, 2010; Wilson et al., 2017). In the case of Ecuador, some of the reference forests or old-growth forests are under conservation strategies like protected areas or incentive-based forest conservation. Despite 25% of the country's territory is under any of these strategies, there is still a large amount of forest without any protection (Añazco et al., 2010; MAE, 2018). It is precisely in these forests without protection status where logging activities are implemented, resulting in the decline phase of the forest ecosystem services. The land use transition phases at the landscape level are also represented by the recovery processes of the forest cover and ecosystem services (Lambin and Meyfroidt, 2010; Wilson et al., 2017). Forest landscape restoration including natural regeneration, activities of reforestation for commercial or conservation purposes, and the implementation of agroforestry systems are proposed as alternatives to halt ecosystem degradation and enhance forest recovery (Sabogal et al., 2015).

3.3.1. Actions to conserve forest ecosystems and their ecosystem services

Almost 31% (4.06 billion ha) of the world is covered by forests, the tropical forest represents 45% of the worlds forest area (FAO, 2020a), meanwhile, the Amazon Basin and the

Chocó-Darien together contribute more than 20% of the world forest (0.81 billion ha) (FAO, 2011; WWF, 2015). The Amazon and Chocó forest contribute with several direct and indirect benefits to human well-being, and also act as habitat shelter for flora and fauna, becoming two of the most important biodiversity hotspots in the world (Basthlott et al., 2007; Marchese, 2015; Myers et al., 2000). Locally, provisioning services such as timber for construction or furniture and non-timber forest products related to food, medicine, and materials are important (de la Torre et al., 2008; Mejia and Pacheco, 2014), especially in local communities where their income depend on natural resources (Ojeda et al., 2020). In addition, cultural services play a central role in the forest frontiers, mainly due to the high diversity of ethnic groups that are settled here, evidencing the links between nature with the traditional knowledge and beliefs of indigenous people (Angarita-Baéz et al., 2017; Elwell et al., 2020). On a global scale, forests offer great potential for climate regulation by storing and sequestering large stocks of carbon (Baker and Spracklen, 2019; Saatchi et al., 2011).

Despite the importance of forests to human well-being, unsustainable practices have led to deforestation and forest degradation process, affecting biodiversity, and resulting in a reduction of several ecosystem services. Different policy instruments have been designed to halt deforestation and forest degradation, one of the most common instruments is command and control (e.g. protected areas) (Armenteras et al., 2009; Nagendra, 2008; Naughton-Treves et al., 2005) and more recently the introduction of incentive-based instruments to promote forest conservation or payment of ecosystem services supply (e.g. water) (Bond et al., 2009; Jones et al., 2016; Sánchez-Azofeita et al., 2007; Wunder, 2015).

Command and control as protected areas have been usually adopted for biodiversity conservation and safeguarding remaining habitats, having positive results to stop deforestation and promote species conservation (FAO and UNEP, 2020; Naughton-Treves et al., 2005). Command and control strategies have expanded in recent years, accounting for more than 240,000 protected areas and protecting 18% (726 million ha) of the world's forest area. 31% of the forest in protected areas in the world is in South America, becoming the region with the highest percentage of forest under protection on the planet (FAO, 2020a; FAO and UNEP, 2020), meanwhile, in the case of Ecuador 19% of the country territory is conserved through protected areas (Cuesta et al., 2015; MAE, 2018).

Research on command and control strategies has been mainly oriented to protecting and discovering the biodiversity potential of the conserved areas. Also, research suggests the hypothesis that protected areas are the most effective barrier to halting deforestation in the core and buffer zones (Bray et al., 2008; Bruner et al., 2000). Nevertheless, research focused on protected areas presents contrasting results. Bray et al., (2008), observed that deforestation rates in protected areas were higher than the community-based conservation in the Maya forest, although with no statistical difference between them. On the other hand, Bruner et al., (2000) and Naughton-Treves et al., (2005), evaluate deforestation in protected areas across the tropics, founding that protected areas have a good performance to reduce deforestation. Nepstad et al., (2006), found that forest clearing in the Amazon was twenty times higher outside the park than inside, meanwhile, Armenteras et al., (2009), observed in Colombia that deforestation was four times higher in the buffer zone than in the protected areas. As deforestation in the buffer zones increase, protected areas will become more isolated, calling for additional initiatives linked to local socio-economic development (Naughton-Treves et al., 2005).

Use restrictions do not always attract the attention of forest owners, in this sense in the last decades' incentive-based forest conservation or payment for ecosystem services are considered as an alternative (Bond et al., 2009). In these programs, the forest owners must be compensated by the beneficiaries of the service for the forgone benefits from alternative land use (Bond et al., 2009). These instruments are defined as voluntary agreements between the forest owners and buyers, where the forest owners accept to conserve a forest area or a selected ecosystem service (Bond et al., 2009; Wunder, 2015). Besides, is been hypothesized that these instruments can contribute to generating environmental and socio-economic benefits, e.g., biodiversity conservation, reducing forest clearing, alleviating poverty, and improving rural livelihoods (Blundo-Canto et al., 2018; Bond et al., 2009; Eliasch, 2008; Grieg-Gran et al., 2005; Wunder, 2015; Wunder and Wertz-Kanounnikoff, 2009). Studies related to incentives or payments for ecosystem services are mainly focused on the socio-economic aspects (Arriagada et al., 2018; Bartels et al., 2010; Bremer et al., 2014; Cole, 2010; Locatelli et al., 2008; Raes et al., 2014; Rodríguez et al., 2013), whereas environmental assessments are mostly aimed to assess the influence of the instruments to reduce deforestation from the spatial perspective (Armenteras et al., 2009; Cuenca et al., 2018; Jones et al., 2016; Pfaff et al., 2008). But there is still a lack of evidence from the ecological performance taking into consideration the ecosystem services perspective (Börner et al., 2017).

Sánchez-Azofeita et al., (2007) and Robalino and Pfaff, (2013), evaluated Costa Rica´s payment for ecosystem services program for the period 1997-200. They found that the program during this period had little impact to halt deforestation at the national level, arguing that the effect of former conservation policies could influence this evaluation. At the regional level in the Osa Peninsula, Sierra and Russman, (2006), contrast the percentage of forest cover in farms with and without payments, finding no statistical difference between them. But suggests that payment could contribute to agricultural land abandonment promoting forest recovery, which was supported by Morse et al., (2009) who found that payments influence forest expansion in the San Juan la Selva corridor.

In contrast, for Ecuador, the Socio Bosque program[1] (MAE, 2016) has shown positive results. Jones et al., (2016) and Mohebalian and Aguilar, (2018) at the regional level in the Northern Ecuadorian Amazon, contrast areas within and without the Socio Bosque program, observing that incentivized forest conservation reduces deforestation rates. Meanwhile, at the national level, Cuenca et al., (2018) observed that the Socio Bosque program helps to avoid deforestation by 1.5% on average in the areas that are involved in the program. The Socio Bosque program in Ecuador is an incentive-based forest conservation program that began in 2008 intending to conserve the forest, reduce deforestation, and improve living conditions in local communities (MAE, 2019, 2016). Communities that voluntarily join the program sign a 20-year contract with the Ecuadorian government to protect their forest (no extractive use is allowed in the forest). In turn, communities receive cash compensations ranging from USD 0.70 to USD 35.00 per ha yr^{-1}, depending on the total size of the conserved forest (MAE, 2019, 2016, 2012). The program holds almost 1.6 M ha (6% of the country's territory) under long-term protection, becoming an important conservation strategy in Ecuador (MAE, 2016). 64% of the conservation area of the program is concentrated in the provinces of Pastaza, Napo, and Orellana. Within the three provinces, 98.4% of the area conserved by the Socio Bosque program is covered by collective contracts (linked to communities) (MAE, 2019, 2012), which is related to the large extension of community forests in the country (5 to 7.5 M ha of total forest) (Añazco et al., 2010; Palacios and Freire, 2004).

3.3.2. Forest loss and the ecosystem services decline

Despite the importance of forest ecosystems, anthropogenic activities had negatively influenced ecosystems' integrity. The way humans use the forests and their services can create economic and social benefits in the short term, but intensive use can trigger long-term forest

[1] *Socio Bosque*: governmental program that offers economic incentives for forest conservation. Forest owners voluntarily accept and participate in the program for a twenty years period (MAE, 2016).

degradation with the decline in ecosystem services (Foley et al., 2007; MEA, 2005). Which can also be the first step toward deforestation (Asner et al., 2005). In Latin America, as well as, the Amazon and the Chocó region, 70% of the forest degradation is related to timber extraction, becoming the main driver of forest degradation in these regions. Whereas the remaining 30% is related to fuelwood, charcoal, and uncontrolled fires (Gerwing, 2002; Hososuma et al., 2012; Kissinger et al., 2012). Forest degradation is defined as a reduction in the provision or benefits obtained from an ecosystem service, due to natural or anthropogenic pressures that do not trigger a land use change (FAO and UNEP, 2020; Hososuma et al., 2012; IPBES, 2018; MEA, 2005).

In areas under timber extraction, the increase in the intensities and frequency of logging could result in highly degraded forests (Putz et al., 2012), affecting several ecosystem services. Locally, the provisioning services could be highly affected, for example, with a reduction of 31% of the merchantable species volume by logging, only 72% of their volume is recovered after 20 years (Vidal et al., 2016). Also, communities perceive changes and a decline in the availability of non-timber forest products like fruits or nuts, and even a decrease in hunting rates by 62% after timber extraction (Menton, 2003; Rist et al., 2011). Supporting services related to the soil can be impacted also, mainly by changes in the nutrient cycles due to a loss in the above-ground pools (Montagnini and Jordan, 2005), but also with a decrease in soil properties (e.g. nitrogen, carbon, sodium, potassium or calcium), especially in roads and desks as a consequence of litter removal (McNabb et al., 1997; Olander et al., 2005). Biodiversity supports ecosystem services as it has a positive link to support functions, therefore the loss of biodiversity could lead to the decline of ecosystem services (Balvanera et al., 2016; Harrison et al., 2014; Mace et al., 2012). Logging could influence tree species composition, tree survival, and recruitment rates (Shenkin et al., 2015). Meanwhile, richness in fauna (invertebrates, amphibians, and mammals) can decrease when high logging intensities take place (Burivalova et al., 2014).

Regulating services can be impacted by logging activities as well. In this case, as tree cover can influence local temperature and rainfall (Makarieva et al., 2014), the reduction in tree canopy cover and the degree of forest fragmentation could result in a decrease in the number of rain days (Webb et al., 2005). In this regard, Mei and Wang, (2010) suggest that a decrease in precipitation could happen as the intensity of logging increase and a transition phase to high deforestation occur. Moreover, logging can result in a decline of approx. 25% of the above-ground carbon stocks require between 16 and 43 years to recover the values before the harvesting activity (Blanc et al., 2009; Rutishauser et al., 2015; West et al., 2014). Logging may have a global influence on the carbon cycle. According to Pearson et al., (2014), in the tropics logging can contribute 12% of the emissions equivalent to those from deforestation, meanwhile for the Amazon Basin, Asner et al., (2005), suggest that 25% of the carbon emissions could be related to logging interventions.

Deforestation can trigger the loss of ecosystem services, also, forest conversion to other land uses can result in forest fragmentation or have a negative impact on rainfall (Mei and Wang, 2010; Webb et al., 2005). Besides, the loss of biomass pools by forest clearing provokes the loss of carbon stocks, timber volumes, non-timber products, and the decline of soil nutrients, as well as, the reduction and possible extinction of plants and animals diversity (Foley et al., 2007; Haddad et al., 2015; Mainville et al., 2006; Spracklen and Garcia-Carreras, 2015). Deforestation results from complex processes related to socio-economics, demographic, and political factors. These are determined by proximate causes such as the agricultural expansion and infrastructure and the underlying causes linked to social-economic, government policies, and technological factors (Geist and Lambin, 2001; Kaimowitz and Angelsen, 1998; Lambin and Geist, 2008; Montagnini and Jordan, 2005).

According to Armenteras et al., (2017), in Latin America, Chile and Argentina present the highest deforestation rates (1990 – 2014) followed by Ecuador, being the Ecuadorian lowland forest one of the most affected. From the spatial perspective, the most important drivers in Latin America are related to forest cover change to agriculture and infrastructure (Armenteras et al., 2017). In the case of Ecuador, the main deforestation drivers are the forest cover change to crops and pastures, road opening also has a great impact especially in the first 10 km from the road, followed by oil extraction and mining, especially in some areas of the Ecuadorian Amazon (Castro et al., 2013; Sierra, 2013; Sierra et al., 2021; Wasserstrom and Southgate, 2013). Deforestation studies based on remote sensing have increased lately, contributing to mapping deforestation trends and the understanding of land use change patterns.

In a context where production and consumption decisions are interrelated and households are surrounded by imperfect markets, land use decisions are given on household internal and external factors (Kaimowitz and Angelsen, 1998). The main factors in Latin America that increase deforestation at the household level are related to the absence of a land title, socio-economic factors such as family size, farm size, and assets, and infrastructure like distances to the market (Angelsen and Kaimowitz, 1999; Brondízio et al., 2009; Godoy et al., 1996; Walker et al., 2000). In Ecuador studies focused on deforestation at the household level date from the 1990s. These studies show that the main factors of deforestation are education level, age of household head, number of males of working age, and land title (Mena et al., 2006; Pichón, 1996). In this sense, to slow down the deforestation rates it is important to assess the forest cover change under a spatial approach, but also it is imperative to understand the internal and external factors that influence household deforestation behavior.

3.3.3. The recovery pathways toward the enhancement of ecosystem services

The last decade's unsustainable use of natural resources has negatively impacted the maintenance of several ecosystem services, resulting in a decline or even a loss of them. As a response, different initiatives have emerged, for example, the 20×20 Initiative which intends to restore 20 M ha (Initiative 20x20, 2020), the Bonn Challenge aims to restore 350 M ha (Bonn Challenge, 2020), or REDD+. Besides, other approaches like the Forest Landscape Restoration (FLR) initiative, have been proposed as alternatives to halt ecosystem degradation and enhance forest recovery. With the Manila Declaration (2019), FLR is gaining more recognition among policymakers and governments, as it could suitable approach to restoring ecosystem services, improving the community's well-being (Chazdon et al., 2020b), and balancing environmental and socio-economic needs (Chazdon et al., 2020b). FLR is a long-term process with a special focus on people´s needs, the recovery of ecological integrity, and improving landscape functions while addressing deforestation and forest degradation drivers (Chazdon et al., 2020a, 2020b; Mansourian et al., 2005). Among the most common actions to recover ecosystem integrity and services are ecological restoration (Morrison and Lindell, 2011), reforestation with native and exotic species for timber production purposes or conservation; and the implementation of more friendly agricultural systems with a mix of crops and trees as the agroforestry systems (Sabogal et al., 2015).

Ecological restoration is applied in ecosystems that have been heavily degraded or transformed into another land use. The main aim is to recover the ecosystem´s functions and processes which in turn contributes to the recovery of ecosystem services (Aronson et al., 2007). Restoration actions can be active or passive, aiming to assist the recovery of an ecosystem and return it to reference levels or its historic trajectory (ecosystem with no signs of degradation that restoration actions intend to redress) (Benayas et al., 2009; SER, 2004). Passive restoration or natural regeneration (second-growth forest) is the spontaneous recovery in abandoned areas after a forest clearing, it involves native trees and can be facilitated by human intervention only with fencing or fire control (Aronson et al., 2007; Chazdzon and Guariguata, 2016; Crouzeilles et al., 2017; Gann et al., 2019). It is also considered a good low-cost strategy from the long-term perspective (Chazdon, 2014). In contrast, active or assisted restoration is characterized by active human intervention to speed up the recovery process, in this case usually planting a seedling, direct seeding, the management of disturbance regimen (e.g. thinning and burning), and the control of invasive species is executed (Aronson et al., 2007; Crouzeilles et al., 2017; Morrison and Lindell, 2011).

Research in ecological restoration has been focused on evaluating how active or passive restoration can contribute to the recovery of ecosystem services and the ecosystem´s integrity to reference levels, Restoration actions can be affected by the proximity and percentage of other forest areas in the landscapes, the severity of the last land use and by a climatic condition like precipitation (Aronson et al., 2007; Chazdon and Guariguata, 2016; Crouzeilles et al., 2017). In this regard, natural regeneration has promising results in terms of biodiversity (e.g. plants, birds, and invertebrates), biomass, litter, density, tree height, and specific regulating ecosystem services like carbon sequestration and supporting services related to nutrient cycling (Amazonas et al., 2011; Benayas et al., 2009; Chazdon, 2008; Crouzeilles et al., 2017; Martin et al., 2013; Poorter et al., 2016; Shimamoto et al., 2014). According to Chazdon and Guariguata (2016), naturally regenerated forest or second-growth forest in the first and mid-stage has high rates of above-ground carbon accumulation, in Latin America, if only 40% of the pastures have a land use change to the nature-regenerated forest, 2.0 Pg C could be stored in 40 years. But the recovery of carbon stocks in below-ground biomass could take longer periods (Martin et al., 2013). This shows the worldwide potential of this restoration action in climate regulation. Besides, restoration can contribute to recovering supporting services, influencing nutrient cycling (Amazonas et al., 2011; Heneghan et al., 2008). Nevertheless, when intensive disturbance that leads to high degradation takes place or in the case of isolated areas with high deforestation rates, assisted restoration can be necessary (Crouzeilles et al., 2017).

Although increasing efforts are being made to assess the recovery of ecosystem integrity, functions, and ecosystem services, information regarding multifunctionality is still scarce in Latin America and is concentrated mainly in temperate ecosystems (Cruz-Alonso et al., 2018; Funk et al., 2019; Strobl et al., 2019). Raising the necessity for more research on the tropics, especially for the Amazon Basin and the Chocó region.

4. Materials and Methods

This chapter describes the study area, the sampling design, the statistical analysis, the methods applied for the estimations of the ecosystem services and biodiversity indicators, the ecosystem service multifunctionality calculation, and the procedure applied for the analysis of the annual rate of net deforestation and the household´s decision to deforest.

4.1. Study area and sampling design

This study was carried out in the Central Amazon (Pastaza, Napo, and Orellana provinces) and the Chocó (Esmeraldas province) lowland rainforests (Figure 3) of Ecuador, which represent 54% of the remnant forest of the country and are two important biodiversity hotspots of the world (Basthlott et al., 2007; Marchese, 2015; Myers et al., 2000). These two areas have a considerable potential to supply different ecosystem services. At the local level, these forests are the habitat shelter for biodiversity and provide important timber and non-timber forest products (e.g. food or medicine) for the local populations. At the global level, these forests have great potential for climate regulation through the conservation of carbon stocks.

Within the study areas, 12 landscapes of 10 km × 10 km were randomly selected covering a total area of 162,000 ha. In the Central Amazon, eight landscapes were selected (Figure 4). In this region, the annual precipitation ranges between 2,800 mm and 4,000 mm and the mean annual temperature is between 22 °C and 27 °C (INAMHI, 2015). In the Chocó region, four landscapes were selected. This region has annual precipitation between 728 mm and 3681 mm, and a mean annual temperature between 22 °C and 27 °C (INAMHI, 2015).

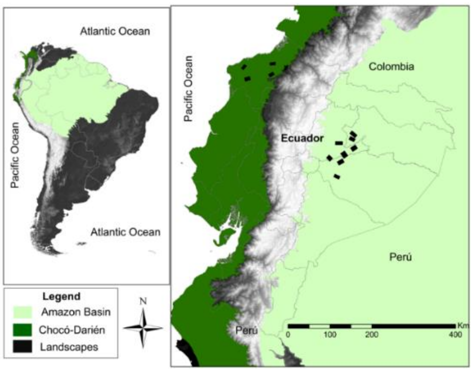

Figure 4. Landscapes selected in the Ecuadorian Central Amazon and Chocó lowland rainforest.

In each landscape, one participatory community workshop was conducted to explain to the local population the purpose of the study and ask for previous consent before starting the field campaign. In total, 12 workshops were conducted involving 73 communities. Participants included men and women, knowledgeable about their territory, with the capacity to answer land use-related questions. During each workshop, participants were asked to delineate on A3 printed satellite images, the different land uses they identify within the landscape. The information from this exercise was used as preliminary forest zoning where participants identified areas with undisturbed forests, logged forest, successional forest, agroforestry systems, and agricultural areas. The initial forest zoning was then contrasted with field visits and with land cover maps available from the Ministry of Environment, Water and Ecological Transition from Ecuador (MAATE). As a result, the following land use transition phases were identified (Figure 5):

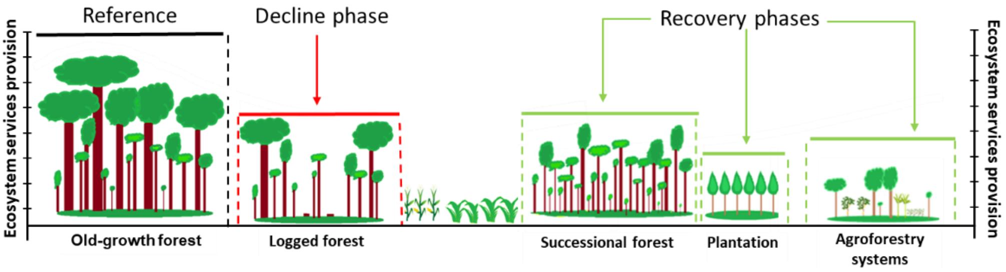

Figure 5. Schematic representation of the land use transition phases. Source: Eguiguren et al., (2020). The recovery phases do not imply a process, these are options for ecosystem services recovery at a landscape level.

- Old-growth forest is defined as a mature forest with unknown human disturbance (Chazdon, 2014; FAO, 2020b).
- Logged forests are forest areas where forest interventions (timber extraction) have been executed. Timber extraction took place in the last two to five years. According to the Ecuadorian forest regulations, there are two main harvesting programs: i) simplified harvesting programs (PAFSI, for its Spanish acronym), which are small-scale interventions with non-mechanized drag and with a five-year cutting cycle. ii) sustainable harvesting programs (PAFSU, for its Spanish acronym), which are medium to large-scale programs, characterized by mechanized hauling and a fifteen-year cutting cycle. In the Central Amazon, logged forests are under PAFSI programs, and in the Chocó under PAFSU programs.
- Successional forests correspond to naturally regenerated forests or second-growth forests which are the result of abandoned lands (e.g. pastures) (Brown and Lugo, 1990;

Chazdon, 2014). The ages of the successional forest in both study areas are between 11 and 28 years of succession.

- Agroforestry systems integrate trees and crops; in the Central, Amazon are related to traditional systems of diversified production called chakras (Torres et al., 2015). In the agroforestry systems selected for this study, it is possible to find tree species like *Cedrela odorata*, *Cordia alliodora*, *Inga edulis*, *Citrus reticulata*, *Citrus sinensis*, *Citrus limon*, or *Mangifera indica*. Commonly these tree species are mixed with palms and crops like *Theobroma cacao*, *Musa paradisiaca*, and *Manihot esculenta*.
- Plantations are planted forests (FAO, 2020b). In the Central Amazon, correspond to *Ochroma pyramidale* plantations (common name: balsa) with ages between two and three years old. In the Chocó, correspond to *Tectona grandis* plantations (common name: teak) with ages among four to 18 years old.

Across the land use transition phases three plots of 40 m × 40 m (1600 m^2) were randomly selected within each landscape. In total, 156 plots were established in the Central Amazon and Chocó (Table 2). For logged forests in the Chocó, I established only nine plots since it was not possible to obtain authorization to conduct the forest inventory in other sites under logging interventions. For plantations, I established six plots in the Central Amazon and nine plots in the Chocó since there were no plantations in all landscapes.

Table 2. Number of plots installed for the Central Amazon and the Chocó.

Land use transition phases	Central Amazon (# plots)	Chocó (# plots)
Old-growth forest	24 [2]	12 [3]
Logged forest	24	9
Successional forest	24	12
Agroforestry systems	24	12
Plantation	6	9
Total	156 plots	

During the community workshops, general information on the socio-economic activities that characterize the landscape and the list of the households settled in each landscape were also obtained. Detailed socio-economic data was obtained through surveys (486) conducted to households randomly selected from the list provided during the workshops. Surveys were

[2] Central Amazon old-growth forest plots: 12 plots in areas under the Socio Bosque Program and 12 plots in old-growth forest without the program and with unknown human disturbance.
[3] Chocó old-growth forest plots: 3 plots in the Mache Chindul Ecological Reserve, 3 plots in the El Pambilar Wildlife Refuge, and 6 plots in old-growth forest without any kind of formal protection and with unknown human disturbance.

applied face-to-face; they included aspects related to household socio-demographic characteristics, land uses and forest clearing, assets, and policy instruments.

4.2. Ecosystem services and biodiversity indicators

Several indicators were estimated to evaluate the provision of ecosystem services and biodiversity in the study areas (Table 3). Provisioning, regulating, and supporting services were selected due to their importance at the local and global levels and based on the MEA (2005) classification. Also, indicators related to plant diversity were considered due to their positive link with ecosystem services (Balvanera et al., 2016; Cardinale et al., 2011; Chopra et al., 2005; Harrison et al., 2014).

Table 3. Selected ecosystem services and the indicators used for the assessment

Category	Indicator	Units
Provisioning services	Total timber volume	TV: m^3 ha^{-1}
	Timber volume potential	TVP: m^3 ha^{-1}
	Non-timber forest products	NTFP: # of sp. per plot
Regulating services	Above-ground carbon stocks	AGC: Mg ha^{-1}
	Soil carbon stocks	SOC: Mg ha^{-1}
Supporting services	Nitrogen in soil	N: %
	Phosphorus in soil	P: mg kg^{-1}
	Potassium in soil	K: meq/100 mL
Biodiversity	Richness	R: # sp. per plot
	Tree and palm diversity	D: Index per plot
	Endemism	E: % per plot

For the provisioning services, total timber volume (TV), timber volume potential (TVP), and non-timber forest products (NTFP) were selected as indicators. TV corresponds to the total volume for each tree within the plots, meanwhile the TVP account only for those trees and species that can be harvested according to the Ecuadorian forest regulation and the minimum diameter cuts (MAE, 2015, 2010). Timber volume values were estimated considering the diameter at breast height, the tree height, and the form factor (Native species: 0.7; *Tectona grandis*: 0.55 and *Ochroma pyramidale*: 0.73) (MAE and FAO, 2014; Murillo, 2012; Rodríguez et al., 2017). The total tree height was estimated based on four equations which are presented in Table 4. The NTFP is the number of species with medicine, material, or food use, the identification of this species was based on secondary information (de la Torre et al., 2008).

Table 4. Equations for tree height estimations

The equation for tree height	Eq.
$\ln Ht = 0.786 + 0.5956(\ln DBH)$ ($p < 0.0001$; $R^2 = 0.69$; $n = 668$)	1)
$Ht = -4.0692 + 5.1391(\ln DBH)$ ($p < 0.0001$; $R^2 = 0.81$; $n = 110$)	2)
$Ht = -11.6292 + 8.9439(\ln DBH)$ ($p < 0.0001$; $R^2 = 0.69$; $n = 298$)	3)
$Ht = -3.9687 + 5.9616(\ln DBH)$ ($p < 0.0001$; $R^2 = 0.70$; $n = 56$)	4)

Ht: tree height. DHB: diameter at breast height. 1) Tree height equation for Amazonia trees. 2) Tree height equation for balsa. 3) Tree height equation for Chocó trees. 4) Tree height equation for teak

Above-ground carbon stocks (AGC) and soil carbon stocks (SOC) were considered indicators for regulating services. For AGC the above-ground carbon stocks of the trees (AGC_{trees}) and the palms (AGC_{palms}) were estimated. AGC_{trees} were calculated based on the Chave et al., (2014) equation (Eq. 5, Table 5), for this purpose the DBH (trees ≥ 10 cm), species wood density, and the environmental stress variable were used. The wood density for the trees was extracted from the Global Wood Density Database, Aguirre et al., (2015); Chave et al., (2009); MAE, (2014); Zanne et al., (2009) databases. The genus, family, or plot mean was applied for those species that were not found in either of these databases. AGC_{palms} was estimated with the Goodman et al., (2013) equation (Eq. 6, Table 5), using the DBH. In the case of agroforestry systems and plantations, a variety of specific equations were applied (Eq. 7 – 13, Table 5). To convert from above-ground biomass a conversion factor of 0.47 was used (IPCC, 2013). Finally, SOC was estimated at a depth of 30 cm with the bulk density and the percentage of organic carbon content (Eq. 14, Table 5).

Table 5. Equations for the calculations of above-ground carbon stocks and soil carbon stocks

Equation	Reference	Eq.
$AGB_{tree} = exp\begin{bmatrix} -1.803 - 0.976E + 0.9676\ln(\rho) \\ +2.673\ln(DBH) - 0.0299\,[\ln(DBH)^2] \end{bmatrix}$	(Chave et al., 2014)	5)
$\mathrm{AGB_{palm}} = \exp(-3.3488 + (2.7483 \times \ln(DBH))$	(Goodman et al., 2013)	6)
$\mathrm{AGB_{cassava}} = -0.67 + 0.44 \times d_{30}$	(Jadan et al., 2012)	7)
$\mathrm{AGB_{cocoa}} = 1.040 \times \exp^{0.0736 \times d}$	(Ordóñez et al., 2011)	8)
$\mathrm{AGB_{coffea\ arabica}} = 93.424 \times \exp^{0.208 \times d}$	(Ordóñez et al., 2011)	9)
$\mathrm{AGB_{coffea\ robusta}} = 242.6 \times \exp^{0.1264 \times d}$	(Ordóñez et al., 2011)	10)
$\mathrm{AGB_{musa}} = 185.1209 + (881.9471 \times \ln H/H^2$	(Anacafe, 2008)	11)
$\mathrm{AGB_{balsa}} = \exp(-2.45 + 2.30 \times \ln(DBH))$	(Douterlungne et al., 2013)	12)
$\log_{10}\mathrm{AGB_{teak}} = -0.815 + 2.382 \times \log_{10} DBH$	(Perez-Cordero and Kanninen, 2003)	13)
$\mathrm{C_{soil}} = BD \times \%CO \times D$	(Pearson et al., 2005); (Ravindranath and Ostwald, 2008)	14)

AGB: above-ground biomass. DBH: diameter at breast height. ρ: wood density. E: environmental stress variable. H: height. d: diameter. BD: bulk density. %CO: percentage of organic carbon content. D: depth.

Three indicators for supporting services were estimated based on two mixed samples a at depth of 30 cm per plot. The Kjeldahl method was used to calculate the total nitrogen (%) in the soil, meanwhile, the Olsen's methodology was applied to determine the potassium (K meq/100 mL) and the content of phosphorus (P mg kg^{1}) in the soil.

Biodiversity has an important role in the ecosystem processes and functions, underpinning the ecosystem services supply (Balvanera et al., 2016; Cardinale et al., 2011; Chopra et al., 2005; Harrison et al., 2014). For this study richness, diversity in trees and palms, and endemism were selected. Species richness was calculated by the number of species of trees and palms per unit of area. Diversity was estimated with the Shannon Index considering the trees and palms in each plot (Eq. 15). Meanwhile, endemism estimations were based on the red book of endemics plants of Ecuador (León-Yánez et al., 2012).

$$H = \sum \rho_i \times Ln\ \rho_i \qquad (15)$$

where ρ_i is the species relative proportion

4.3. Ecosystem service multifunctionality

Ecosystem service multifunctionality was estimated following the multifunctional average approach (Mouillot et al., 2011; Schuldt et al., 2018). The average approach has been previously used in various studies focused on the evaluations of bundles of ecosystem services, functions, biodiversity, functional trait diversity, and landscape multifunctionality (e.g. Fanin et al., 2018; Finney and Kaye, 2017; Gross et al., 2017; Hooper and Vitousek, 1998; Maestre et al., 2012a, 2012b; Mouillot et al., 2011; Rodríguez-Loinaz et al., 2015; Schuldt et al., 2018; Stürck and Verburg, 2017; Tresch et al., 2019). For this purpose, the ecosystem services and biodiversity indicators were standardized considering minimum and maximum values among the land use transition phases, and the mean of the standardized values was used as an indicator of the ecosystem service multifunctionality. Multicollinearity was evaluated by the variance inflation factor – VIF (Hair et al., 2006; Midi and Bagheri, 2010; Wooldridge, 2016); which resulted in a VIF value of 3.21 [4].

4.4. Estimation of the deforestation rates and the assessment of the household´s decision to deforest

Deforestation was evaluated from the spatial and household perspective. The influence of the incentive-based forest conservation program on the annual rate of net deforestation was evaluated at a parish level using land use/cover maps provided by the Ministry of Environmental of Ecuador, for the periods 2000 to 2008 and 2008 to 2016. For estimations of the annual rate of net deforestation, Equation 16 (MAE, 2017) was applied.

$$\text{Deforestation rate} = \left(\frac{\text{A2}}{\text{A1}}\right)^{1/(\text{t2 - t1})} - 1 \qquad (16)$$

where:

A1 = Forest area at the beginning of the period;

A2 = Forest area at the end of the period.

The agriculture household theory was used as a basis for the evaluation of the household´s decision to be deforested. This theory has been applied before to evaluate deforestation or forest clearing at a household level (Babigumira et al., 2014; Caldas et al., 2007, 2002). In the study areas, households operate in a context of imperfect markets, and the decision of production and consumption are interrelated. In this context besides profit maximization, the internal characteristics of the households are relevant in the land use decisions. Therefore, to evaluate the influence of incentive-based forest conservation in household decisions to deforest, the household characteristics (age of the household head, ethnicity, education, number of males, commercialization rate, access to credit, physical assets, and land endowments as farm size and forest area within the farm), the quality of forest resources (timber volume potential), the natural resources governance (land tenure and governmental grants); and the infrastructure (distance to

[4] Variance inflation factor (VIF): values between 5 and 10 correspond to moderate collinearity and greater than 10 suggest the presence of high collinearity (Hair et al., 2006; Midi and Bagheri, 2010; Wooldridge, 2016).

forest and distance to markets) were used as control variables (Brondízio et al., 2009; Pichón, 1997a, 1997b; Vasco et al., 2018; Walker et al., 2000).

4.5. Statistical analysis

To assess the ecosystem service multifunctionality and ecosystem services provision across the land use transition phases (aim one), an ANOVA (LSD Fisher $p \leq 0.05$) using general mixed models was executed to understand the potential decline and recovery of ecosystem services provision and ecosystem service multifunctionality in the Central Amazon and the Chocó. The land use transition phases within the Central Amazon and the Chocó were considered as fixed effects and landscapes as random effects. In the case of timber volume potential, above-ground carbon stocks, phosphorus, and potassium a logarithmic transformation were applied to meet the ANOVA assumptions. Normality and homoscedasticity were evaluated. Only, for endemism, a non-parametric analysis (Kruskal Wallis test) was required. Plantations were excluded when non-timber forest products, diversity, and endemism were evaluated as they did not present values for these indicators. The synergies and trade-offs of ecosystem services and ecosystem service multifunctionality across land use transition phases (aim one) were assessed by principal component analysis (PCA) for the Central Amazon and the Chocó. Provisioning services (timber volume potential and non-timber forest products), regulating services (above-ground carbon stocks and soil carbon stocks), supporting services (nitrogen, phosphorus, and potassium), biodiversity (plants diversity and endemism), and ecosystem services multifunctionality were used for the PCA analysis, explaining 62.2% of the variance for the Central Amazon and 58.7% for the Chocó.

Aim two, related to evaluating the influence of incentive-based forest conservation on ecosystem services provision across the land use transition phases was assessed under a randomized block design. Analyses of variance (ANOVA LSD Fisher $p \leq 0.05$) with general mixed models were used to evaluate if the incentive-based forest conservation program can contribute to maintaining the ecosystem services (timber volume, above-ground carbon stocks, and soil carbon stocks) and richness within the program area, and to know any influence of the program in the decline and recovery of selected ecosystem services and richness. The presence or absence of incentive-based forest conservation, the land use transition phases like reference (old-growth), decline phase (logged forest), and the recovery phase (succession forest), and their interactions were used as fixed effects, meanwhile, as random effects block and landscape were specified. Assumptions of normality and homoscedasticity were evaluated, and criteria penalized likelihood (AIC - BIC) was applied to select the best model. Blocked landscapes were randomly selected across the Central Amazon in Ecuador, each block takes into consideration one landscape where incentive-based forest conservation was established (Socio Bosque program) and one landscape without this program. In the landscapes where the Socio Bosque program was implemented, only the old-growth forests are under the protection of the program. To control for potential confounding variables like forest cover, percentage of agricultural land, ecosystem type, altitude, soil type, and demographic characteristics (population density, distance from landscape to large cities, and distance from households within landscapes to forest) an ANOVA was executed. ANOVA confirmed that confounding variables were similar and comparable between landscapes within blocks.

Finally, to know if incentive-based forest conservation influences the deforestation rates and households´ deforestation decisions beyond the limits of the conservation areas (aim three), two analyses were applied. On the one hand, an ANOVA (LSD Fisher $p \leq 0.05$) was executed two know if there is any statistical difference between the annual rates of net deforestation during the period 2000 – 2008 versus the period 2008 – 2016 for parishes with the presence of the incentive-based forest conservation program and parishes without the presence of the program. Normality and homoscedasticity assumptions were evaluated. On the other hand, to assess if a household's decision to deforest is influenced by incentive-based forest conservation a logistic regression was executed. The dependent variable was binary response taking the value of 1 when the household cleared the forest or 0 otherwise; meanwhile, the independent variable (incentive-based forest conservation) takes the value of 1 if the household is settled landscapes with the presence of the incentive-based forest conservation program, and 0 if otherwise. Control variables were related to household characteristics, quality of forest resources, resources governance, and infrastructure. Multicollinearity was tested by correlation matrix among the independent and control variables and throughout the VIF values of the model (1.36).

5. Results and Discussion

This chapter is divided into three sections related to each aim and presents the main findings published in the three peer-reviewed papers that comprise this dissertation.

5.1. Ecosystem service multifunctionality and ecosystem services provision across the land use transition phases.

5.1.1. The synergies and trade-offs among ecosystem services, ecosystem service multifunctionality, and the land use transition phases

The principal component analysis (PCA) showed the synergies and trade-offs among ecosystem services and how anthropogenic activities manifested through land use transition phases influence the ecosystems' integrity and their capacity to provide multiple ecosystem services simultaneously (Figure 6). The first two components of the PCA explained between 59% and 62% of the variability for the Chocó and Central Amazon, respectively. For both study areas, the first component showed a strong and positive correlation to the ecosystem service multifunctionality index (M). Synergies are identified by positive correlations; they were observed for ecosystem services bundles comprised of regulating services, provisioning services, and biodiversity indicators. More specifically, above-ground carbon stocks, timber volume potential, non-timber forest products, plant diversity, plant endemism, and ecosystem service multifunctionality, interact in a synergetic way. Positive correlations were also found for soil carbon stocks and nitrogen and phosphorus and potassium in the soil. Synergies indicate that any activity oriented to increase one of those ecosystem services will simultaneously increase services with which there is a synergetic relation; the same will occur with activities that decrease the supply of such services.

From the selected services, above-ground carbon stocks had synergies with various services and with the ecosystem service multifunctionality index pointing out that above-ground carbon stocks could be a conspicuous service to promote multifunctional landscapes. However, above-ground carbon stocks had contrasting patterns with soil-related services. The above-ground carbon stocks had a strong to moderate synergy with the timber volume potential in the Central Amazon and Chocó respectively. This implies that high intensities of timber extraction will reduce the carbon stocks as most of the harvested trees have the largest diameters, heights, and densities, affecting negatively the above-ground carbon stocks (Bunker et al., 2005). Sustainable forest management practices, tree damage reduction, and lower harvest intensities can contribute to reducing the decline of the above-ground carbon stocks (Putz et al., 2008; Triviño et al., 2017). The PCA also showed the synergetic relation between plant diversity and non-timber forest products. Plant diversity can underpin ecosystem services, with a high diversity of trees and palms a higher presence of plant species with the potential to supply non-timber forest products can be found.

a) b)

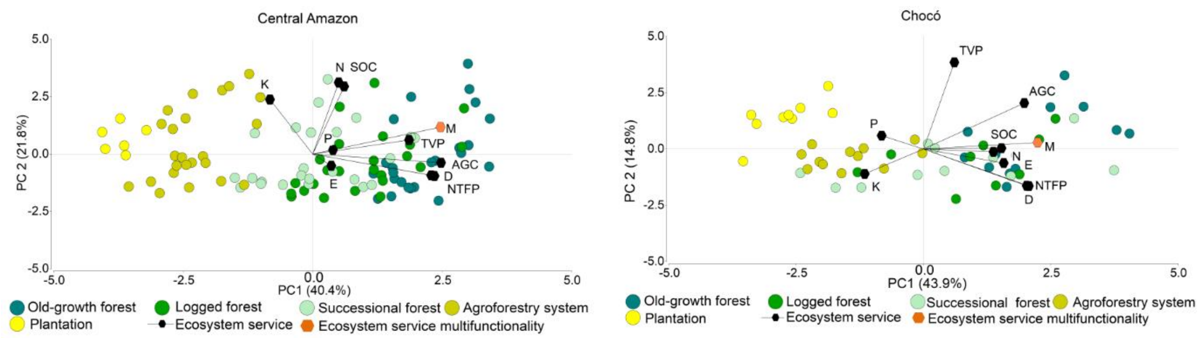

Figure 6. Principal component analysis for the evaluation of synergies and trade-offs. TVP: timber volume potential (m^3 ha^{-1}). NTFP: non-timber forest products (# sp per plot). AGC: above-ground carbon stocks (Mg ha^{-1}). SOC: soil carbon stocks (Mg ha^{-1}). N: nitrogen (%). P: phosphorus (mg kg^{-1}). K: potassium (meq/100 ml), D: Shannon index. E: endemism (% of species per plot). M: ecosystem service multifunctionality. Source: Eguiguren et al., (2020)

The PCA showed that the old-growth forest supplies higher levels of single ecosystem services and the ecosystem service multifunctionality index compared to the other land uses assessed. On the other hand, agroforestry systems and plantations had lower levels of above-ground carbon stocks, timber volume potential, plant diversity, non-timber fortes products, and ecosystem service multifunctionality index. Despite the reduction of provisioning, regulating, and biodiversity indicators, soil-related services were high in the agroforestry systems and plantations, especially for potassium and phosphorus. This finding shows the importance of agroforestry systems and plantations to recover soil properties and nutrient cycling, where the presence of specific species can contribute to enhancing soil-supporting services (Boley et al., 2009; Montagnini, 2000; Wartenberg et al., 2019).

The synergies and trade-offs observed in this study contribute to understanding the interactions between the ecosystem services and the ecosystem service multifunctionality index, identifying the trends and detecting drivers of multiple ecosystem services (Cademus et al., 2014; Lu et al., 2014; Xiangzheng et al., 2016). Based on this analysis it is possible to identify bundles of ecosystem services interacting synergistically, this can be useful to design or support strategies oriented to avoid the potential depletion of other ecosystem services and to improve the management and conservation of natural resources at landscape level (Bennett et al., 2009).

Given that ecosystem services and biodiversity monitoring can be costly and time-consuming, conservation policies should target key services that could be considered umbrella services. Adapting the concept of umbrella species[5] to the ecosystem services framework (Caro, 2010; Siddig et al., 2016; Simberloff, 1998). Umbrella services can be understood as those services with the capacity to generate multiple synergies with other ecosystem services; moreover, umbrella services should be easy to measure and sensitive to forest degradation and land use change. Once an umbrella service is identified, its conservation will help to conserve other services simultaneously. The results of this study reveal that the above-ground carbon stocks can be considered as an umbrella service since it is a sensitive indicator of forest integrity and it had strong to moderate synergies with the ecosystem service multifunctionality index and several ecosystem services. Nevertheless, the above-ground carbon stocks showed contrasting relations with soil indicators, suggesting that above-ground carbon stocks may not be suitable for soil-related services. These findings support the feasibility of strategies such as REDD+ to maintain, recover multiple ecosystem services and conserve biodiversity through the conservation of carbon stocks (UNFCCC, 2020).

5.1.2. Ecosystem services provision and ecosystem service multifunctionality between the land use transition phases

Regarding the capacity of the land use transition phases to supply multiple ecosystem services, the ANOVA showed significant changes across the land use transition phases for the selected services and the ecosystem service multifunctionality index (Table 6). The old-growth forest in both regions provides ecosystem services in similar quantities to other studies conducted in the region in forests with no signal of anthropogenic disturbance (Baker et al., 2004; FAO, 2011; Huang and Asner, 2010; Keith et al., 2009; Saatchi et al., 2011; Valencia et al., 2009). The old-growth forest included in our study has a good status of conservation; therefore, it can be used as a reference line to evaluate the level of ecosystem services provision

[5] The umbrella species approach is oriented to conserve important species within an ecosystem; according to this approach, the protection of specific umbrella species will protect another species as they commonly require large areas to maintain their population (Simberloff, 1998; Caro, 2010; Siddig et al., 2016).

in other forest types and to evidence the effect of anthropogenic activities under decline and recovery phases. The old-growth forest of the Central Amazon and the Chocó have statistically similar quantities of timber volume potential, above-ground carbon stocks, soil carbon stocks, nitrogen and phosphorus in soils, and ecosystem service multifunctionality index (Figure 7 and Figure 8). However, differences between the old-growth forest of the Central Amazon and the Chocó were observed for the non-timber forest products and plant endemism.

The findings evidence that despite having high deforestation rates, old-growth forest remnants in the Chocó, still have the potential to offer a bundle of ecosystem services comparable to those provided in the Central Amazon, where deforestation rates are lower. The old-growth forest supports a wide range of services that provide multiple benefits to society, yet, it is highly threatened by fragmentation and land conversion. The Chocó area is especially threatened due to unsustainable practices, characterized by large-scale timber extraction and the implementation of monoculture farming systems such as oil palm. Therefore, the implementation of strategies oriented to conserve the last remnants of Chocó's old-growth forest in Ecuador is a priority (Fagua and Ramsey, 2019).

Table 6. Analysis of variance of ecosystem services and ecosystem service multifunctionality index

Ecosystem services indicator	p-value	R^2	n
Ecosystem service multifunctionality index – M	<0.0001	0.82	152
Timber volume potential – ln TVP (m^3 ha^{-1})	<0.0001	0.33	140
Non-timber forest product – NTFP (# sp per plot)	<0.0001	0.86	141
Above-ground carbon stocks – ln AGC (Mg ha^{-1})	<0.0001	0.81	153
Soil carbon stocks – SOC (Mg ha^{-1})	0.0187	0.52	156
Nitrogen – N (%)	0.2686	0.82	148
Phosphorus – ln P (mg kg^{-1})	0.0093	0.61	156
Potassium – ln K (meq/100 ml)	<0.0001	0.68	153
Diversity – D (Shannon index)	<0.0001	0.78	141
Endemism – E (% sp per plot)	<0.0001	-	141

Source: Eguiguren et al. (2020)

The decline of ecosystem services and ecosystem services multifunctionality index was evaluated by comparing the old-growth forest and the logged forest (Figure 7). This study found that in the Central Amazon, timber extraction caused a decline of 16% of the ecosystem service multifunctionality index; meanwhile, in the Chocó it decreased by 18%. Some ecosystem services were more impacted than others. The timber volume potential in the old-growth forest was 190.5 m^3 ha^{-1} in the Central Amazon and 149.9 m^3 ha^{-1} in the Chocó, whereas in the logged forest the average timber volume potential was 113.3 m^3 ha^{-1} for the Central Amazon and 75.9 m^3 ha^{-1} for the Chocó (Figure 8). Above-ground carbon stocks also showed an evident decrease, the old-growth forests in the Central Amazon had 167.3 Mg ha^{-1} and 146.9 Mg ha^{-1} in the Chocó; whereas, the logged forests had 113.3 Mg ha^{-1} for the Central Amazon and 102.5 Mg ha^{-1} for the Chocó.

Logging activities had a high impact on these two ecosystem services with a reduction of the timber volume potential by 40% for the Central Amazon and 49% for the Chocó and a decline in the above-ground carbon stocks between 30% to 32%in the study areas. Forests under harvest interventions are experiencing a reduction of these ecosystem services, even though logging is executed following the forestry regulation approved by the Ecuadorian environmental authority. This highlights the need to reevaluate the current regulations to avoid a higher depletion of important services and to give time to the forest to get recovered. Studies with permanent plots in similar ecosystems found that logged forest will need more than 32 years after a reduction of 15 to 23 m^3 ha^{-1} or even more than 43 years when the AGB decrease by 26% (Huang and Asner, 2010; Roopsind et al., 2017; Rutishauser et al., 2015; West et al., 2014). Considering the cutting cycles in the study areas (5 to 15 years) and the high decline in the timber volume potential and the above-ground carbon stocks, the findings suggest that these ecosystem services (timber and carbon stocks) could be reaching critical thresholds. According to Putz et al., (2012), with high logging intensities and more frequent harvesting, logged forests could lead to a permanently degraded forest with lower timber yields for the second or third cut.

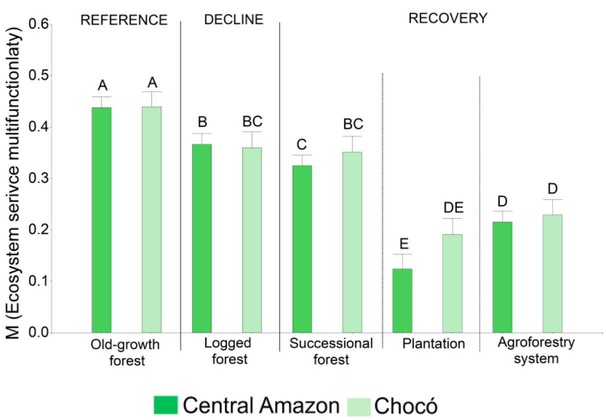

Figure 7. Analysis of variance for the evaluation of the ecosystem service multifunctionality. Different letters indicate a significant difference from each other ($p \leq 0.05$). Source: Eguiguren et al. (2020).

The non-timber forest products species also showed a decline from 39 species per plot in the old-growth forest to 35 species per plot in the logged forest. Nevertheless, this reduction was statistically different only for the Central Amazon. The reduction is explained by changes in forest structure and composition, which can trigger not only a potential loss of these species but could have a conflict of use for both timber and non-timber products in local communities, affecting also rural livelihoods (Rist et al., 2011). Plant endemism decreased only in the Chocó, from 6.4% per plot in the old-growth forest to 3.9% per plot in the logged forest. The Chocó region is a very important hot-spot of endemism as it hosts 20% of the vascular plants of Ecuador (Dodson and Gentry, 1991; Palacios and Jaramillo, 2016). The overexploitation of timber species and tree damage during logging activities can lead to the depletion of these endemic species and their possible extinction (Palacios and Jaramillo, 2016).

Plant diversity showed non-statistical differences when comparing the old-growth forest to logged forest in both study areas (Figure 8). Tree diversity may increase as a result of the disturbances from the logging activities (intermediate disturbance hypothesis), with a higher presence of new pioneer species (Bongers et al., 2009; Connell, 1978; Magnusson et al., 1999; Molino and Sabatier, 2001; Verburg and Van-Eijk-Bos, 2003). But according to Connell, (1978) and Bongers et al., (2009), when disturbance frequency and intensity increase reaching critical thresholds, tree diversity can be affected by the decline of shade-tolerant species.

Logging impacted soil-related services (soil carbon stock, nitrogen, and phosphorus), but no statistical differences were found, suggesting a low effect in the logging gaps. Other studies also found a low impact on logging gaps for soil carbon stocks, nitrogen, and phosphorus in the soil, but with high reductions of these ecosystem services in roads and decks (McNabb et al., 1997; Olander et al., 2005). Soil-related services are negatively influenced by the reduction of nutrient pools from big trees and litter inputs, these reductions can be exacerbated by high logging intensities, more frequent interventions, and a bigger size gap (Dam, 2001; Martinelli et al., 2000; Olander et al., 2005).

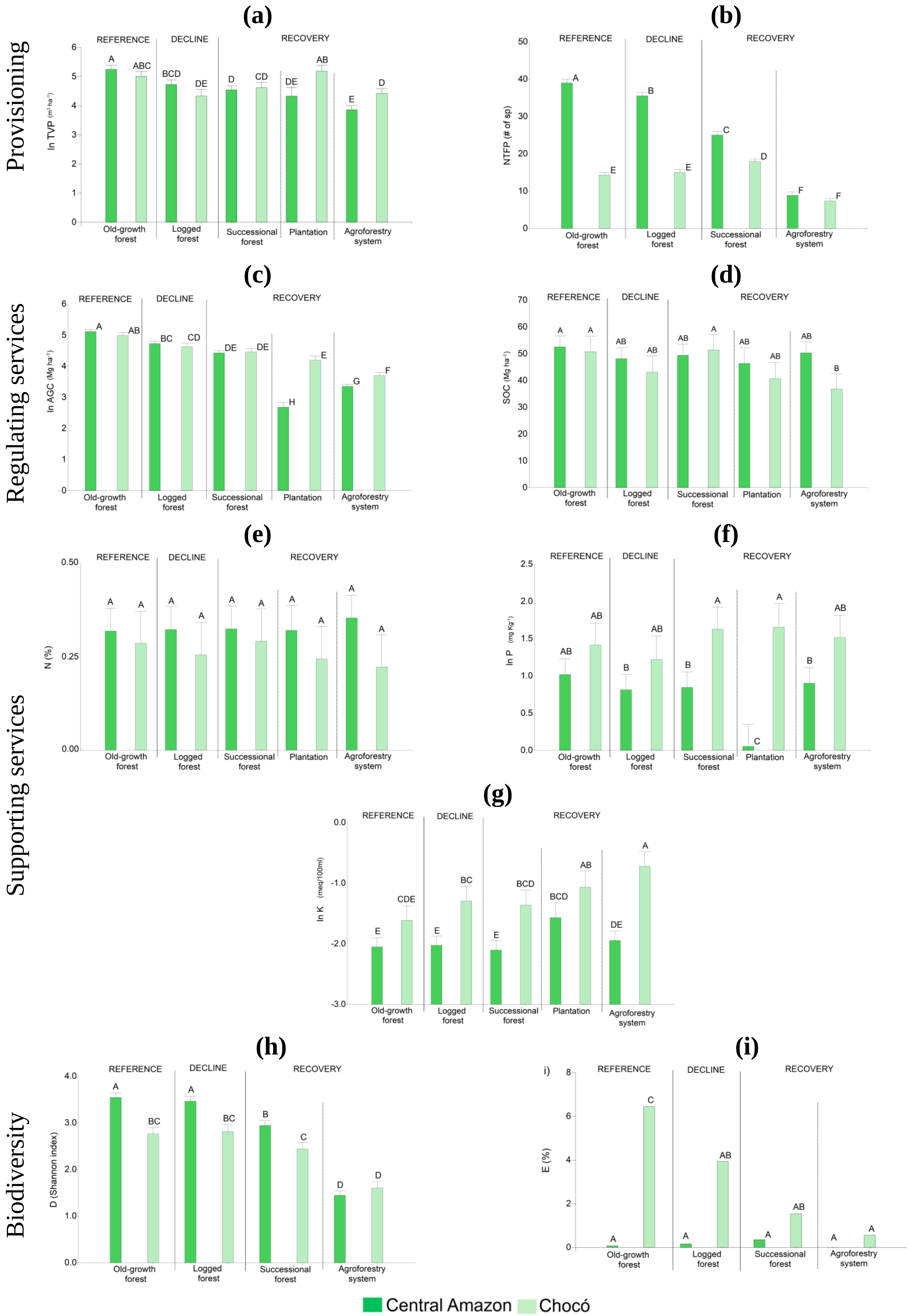

Figure 8. Analysis of variance for the ecosystem services. (a) ln TVP: timber volume potential, (b) NTFP: non-timber forest products, (c) ln AGC: above-ground carbon stocks, (d) SOC: soil carbon stocks, (e) N: nitrogen, (f) ln P: phosphorus, (g) ln K: potassium, (h) D: Shannon index, (i) E: endemism. Different letters indicate a significant difference from each other (p ≤ 0.05). Source: Eguiguren et al. (2020).

In Latin America as well as in the Ecuadorian Central Amazon and Chocó, timber extraction is the main driver for forest degradation (Hososuma et al., 2012). In the study areas, logging activities impacted ecosystem services to different extents. The results evidenced high reductions in timber volume potential and the above-ground carbon stocks; a moderate decrease of non-timber forest products, plant endemism, and plant diversity; and low impacts to soil-related services; altogether these reductions decrease the ecosystem service multifunctionality index. Under this panorama, sustainable forest management practices (SFM) and reduced impact logging techniques (RIL) must be conducted to maintain the ecosystem´s capacity to supply multiple ecosystem services that fulfill society's demands. RIL can contribute to an adequate recovery of ecosystem services; according to different studies, when RIL is applied it reduces tree damage on the above-ground carbon stocks and the commercial timber volume decrease by 17% and 21% respectively; likewise, these services can be recovered to reference level after 16 to 20 years (Putz et al., 2008; Vidal et al., 2016; West et al., 2014).

In Ecuador, 68% of the harvestable timber with good quality is located in the study areas (MAE and FAO, 2014); therefore, SFM and RIL should be implemented to safeguard the good forest potential to offer valuable timber species and to ensure an adequate provision of multiple ecosystem services. This implementation needs to consider an adjustment of the logging cycles and logging intensities considering the forest recovery potential.

Regarding the assessment of ecosystem services and ecosystem service multifunctionality index recovery, this research considered the successional forests, agroforestry systems, and forest plantations. The successional forest offers the highest ecosystem service multifunctionality index (0.32 to 0.35) when compared to the other recovery phases evaluated (Figure 7). The successional forest also offers high values of provisioning services: the timber volume potential was among 93.6 m^3 ha^{-1} and 101.4 m^3 ha^{-1}, meanwhile, the non-timber forest products were between 18 and 25 species per plot, in the study areas. The above-ground carbon stocks and plant diversity in the successional forest were also higher than in agroforestry systems and plantations; values ranged from 83.9 Mg ha^{-1} to 86.4 Mg ha^{-1} for the above-ground carbon stocks and the diversity index was among 2.9 and 2.4 in the Central Amazon and the Chocó respectively (Figure 8). The successional forest in the study areas comprises young and mid-stages of natural succession, these stages are characterized by high rates of biomass (Chazdon et al., 2016), which contribute to increase timber volume and carbon stocks. Soil-related services in the successional forest were also high and close to the old-growth forest. The results presented in this study show the potential of the successional forest to recover provisioning services, regulating services, supporting services, and biodiversity. However, the recovery occurs with different species composition compared to the old-growth forest and logged forests (Chazdon et al., 2010; Martin et al., 2013).

The agroforestry systems and the plantations offer a lower ecosystem service multifunctionality index when compared to the successional forest; nevertheless, these recovery phases prove to have a great potential to recover soil-related services like soil carbon stocks, nitrogen, phosphorus, and potassium, which is closely related to the presence of tree species that contribute in nitrogen fixation and foliar biomass, helping to enhance soil nutrients through soil organic material (Ferrari and Wall, 2004; Lojka et al., 2012). The recovery of soil-related services is critical, especially in the Central Amazon, where there is low soil fertility (Bravo et al., 2017). In the Central Amazon, Mainville et al., (2006) found that land use change from forest to pastures can reduce soil supporting services such as carbon by -68%, and nitrogen and potassium by -50% each. The Central Amazon is characterized by high annual precipitations (average annual precipitation 4,118 mm) which drives nutrient leaching after forest clearing and the consequent loss of soil fertility. Negative impacts of forest clearing were also found by Murty et al (2002), Lal (2004), and Don et al. (2011), for the soil carbon stocks in other latitudes; they found that forest conversion into crops reduces between 25% and 75% of soil carbon stocks.

The high values of soil-related services in the successional forest, agroforestry systems, and forest plantations show the capacity of these forest uses to restore regulating (soil carbon stocks) and supporting services (nitrogen, phosphorus, and potassium in soil) to reference levels. Nevertheless, it is worth noting that in the long term, such services could decrease especially in plantations due to a reduction of above-ground biomass provoked by timber extraction over different rotation cycles (Fernández-Moya et al., 2014; Montagnini and Jordan, 2005). The assessed recovery phases help to recover the ecosystem service multifunctionality as well as individual ecosystem services and biodiversity. Additionally, successional forests, agroforestry systems, and plantations could contribute to improving ecosystem connectivity in highly degraded landscapes (Barlow et al., 2007; Chazdon et al., 2017; Chazdon and Guariguata, 2016; Crouzeilles et al., 2017; Guariguata and Ostertag, 2001; Holl and Aide, 2010) and enhance livelihood strategies as in the study areas 50% to 60% of household income depends on natural resources (Ojeda et al., 2020).

From the ecological perspective, the successional forest is probably the suitable option to restore degraded landscapes, as it offers the highest multifunctionality, even close to the logged forest. However, the conservation of successional forests may not be attractive enough for landowners as they will need land for a living. The quantitative evidence presented in this study can inform the implementation of strategies such as forest landscape restoration or REDD+; by understanding the potential that recovery phases have in human-modified landscapes, decision-makers can orient restoration and conservation efforts in a more efficient way to satisfy multiple demands. As land is the main productive asset in the study areas, monetary incentives are needed to promote the conservation of successional forests and to compensate landowners for the forgone benefits of not using these areas for another purpose (e.g. crops). Besides, to reduce the pressure on the old-growth forest and logged forests, it is necessary to integrate agroforestry systems and forest plantations in landscape planning, as they represent productive activities that help to improve local welfare.

5.2. The influence of incentive-based forest conservation on ecosystem services provision across the land use transition phases

This section presents the findings on the influence of the incentive-based forest conservation program (Socio Bosque) on the land use transition phases considered in this study (Table 7). Results showed that the incentive-based forest conservation program is contributing to the maintenance of the evaluated ecosystem services. Forests under the conservation program presented similar levels of timber volume (227 m^3 ha^{-1} and 246 m^3 ha^{-1}), above-ground carbon stocks (159 Mg $C \cdot ha^{-1}$ to 172 Mg $C \cdot ha^{-1}$), and soil carbon stocks (50 Mg $C \cdot ha^{-1}$ to 54 Mg $C \cdot ha^{-1}$) to undisturbed forests in other areas of the Amazon Basin (Baker et al., 2004; FAO, 2015, 2011; Saatchi et al., 2011; Sist and Nascimiento, 2007; Valencia et al., 2009). However, there was no significant difference between the old-growth forest with and without the incentive program in the study areas. Similarly, Mohebalian and Aguilar, (2018) found no statistical difference in areas with and without the incentive when they evaluated degradation effects. In this study, a higher presence of timber species was observed in areas under the incentive-based forest conservation program when compared to areas without the program. This suggests that in non-Socio Bosque lands a higher intensity of timber extraction could be occurring.

The Socio Bosque program currently accounts for 1.6 M ha; 64% of the area under the program is in the Central Amazon (Pastaza, Napo, and Orellana) (MAE, 2019), which highlights the importance of these results to inform about the effect of this program on ecosystem services. The Central Amazon still has a low population density compared to other areas of the country (INEC, 2010) and households still manage a considerable amount of land, therefore, the implementation of Socio Bosque has been crucial to promoting forest conservation at the farm level. Households resort to their own forests to satisfy the demand for forest products and to perform their livelihood strategies. When forest resources deplete and soil fertility decrease in the cultivated areas, higher pressure on the remnant forest occurs, especially towards those without any protected status. This fact highlights the importance of implementing incentive-based forest conservation programs for maintaining ecosystem services that local populations rely on.

Table 7. Analysis of variance for carbon stocks, timber volume, and tree species richness

Dependent Variable	Incentive-based forest conservation vs non- Incentive-based forest conservation	Land use transition phases (old-growth, logged, and successional forest)	Interaction: Conservation program and Land use transition phases	R^2	n
	p-value	p-value	p-value		
TV (m^3 ha^{-1})	0.4330	<0.0001***	0.3498	0.57	68
AGC (Mg C·ha^{-1})	0.4624	<0.0001***	0.0696*	0.61	70
SOC (Mg C·ha^{-1})	0.3738	0.2755	0.1753	0.53	72
Richness (#sp./plot)	0.8933	<0.0001***	0.0570*	0.65	72

TV: Timber volume. AGC: Above-ground carbon. SOC: Soil organic carbon. * p-value ≤ 0.10. **p-value ≤ 0.05. *** p-value ≤ 0.0001. Source: Eguiguren et al., (2019)

The main differences between ecosystem services provision and the influence of incentive-based forest conservation were found in the declining phase related to timber extraction activities (Figure 9 and Figure 10). When old-growth forests were compared to logged forests, the results evidenced statistically significant high impacts of timber extraction on above-ground carbon stocks and richness. The main goal of the incentive-based forest conservation program is to protect the old-growth forest; however, this research shows that the Socio Bosque program could influence timber extraction in areas close to forests under conservation. The logged forest close to the old-growth forest under the conservation program has higher above-ground carbon stocks (125 Mg C·ha^{-1}) than logged forests in landscapes without the conservation program (101 Mg C·ha^{-1}). Likewise, above-ground carbon stocks were reduced by 21% in logged forests close to areas under the incentives, whereas logged forests in landscapes with no conservation program had 41% less above-ground carbon stocks. Besides, a decline in species richness was found in logged forests when compared with the old-growth forest under the conservation program, highlighting the importance of this strategy for the protection of plant species. The adequate conservation of old-growth forests and a lower decline of ecosystem services in logged forests close to the conservation program can be the result of a higher presence of government representatives who are monitoring and controlling Socio Bosque areas (Jones et al., 2016, 2020). Jones et al., (2016), found that people living near Socio Bosque report more restrictions related to the use of natural resources and a quick response from the national environmental authorities regarding illegal extraction.

These results show that it is possible to balance and integrate forest conservation and timber production (Runting et al., 2019). On one hand, the incentive-based forest conservation program is contributing to the maintenance of the ecosystem services, as no reduction of ecosystem services was observed within the Socio Bosque areas. On the other hand, the logged forest close to Socio Bosque had less decline in the provision of ecosystem services than the logged forest in landscapes without the program. Nevertheless, the reduction of 21% of the above-ground carbon stocks in the logged forests close to Socio Bosque is still high, according to other studies, and considering that a reduction of above-carbon stocks between 10% to 25% will require between 12 to 43 years to recover (Piponiot et al., 2016; Rutishauser et al., 2015).

The recovery phase was evaluated with the successional forest. The evaluated successional forest has lower values of timber volume, above-ground carbon stocks (Figure 9), and different species composition, than old-growth and logged forests. No statistical differences were found between the successional forest close to the incentive-based forest conservation and the successional forest in landscapes without the program (Figure 10). The results showed that there is no influence of the program in the evaluated recovery phase. In contrast, Sierra and Russman, (2006), who compared farms with and without payments for ecosystem services in Costa Rica, they found that payments may contribute to agricultural abandonment, allowing natural regenerating processes and implying that these programs can be suitable to promote landscape restoration (Daniels et al., 2010). Given that successional forests play a relevant role within the landscape, they should be included in conservation strategies or incentive programs such as Socio Bosque. Also, these forests can contribute to landscape restoration and ecosystem services provision, plus help to support biodiversity conservation in old-growth forests (Gibson et al., 2011; Morse et al., 2009; Thanichanon et al., 2013).

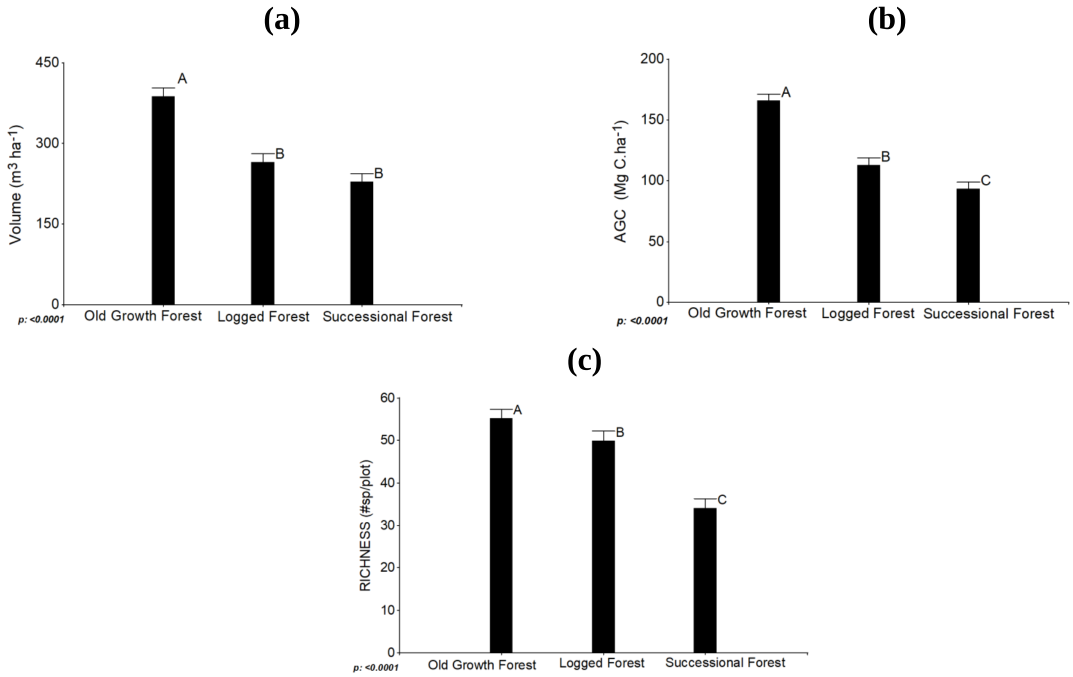

Figure 9. Analysis of variance of ecosystem services provision between land use transition phases (maintenance, decline, and recovery). a) Timber volume. b) AGC: Above-ground carbon stocks. c) Tree species richness. Different letters indicate a significant difference from each other ($p \leq 0.05$). Source: Eguiguren et al 2019.

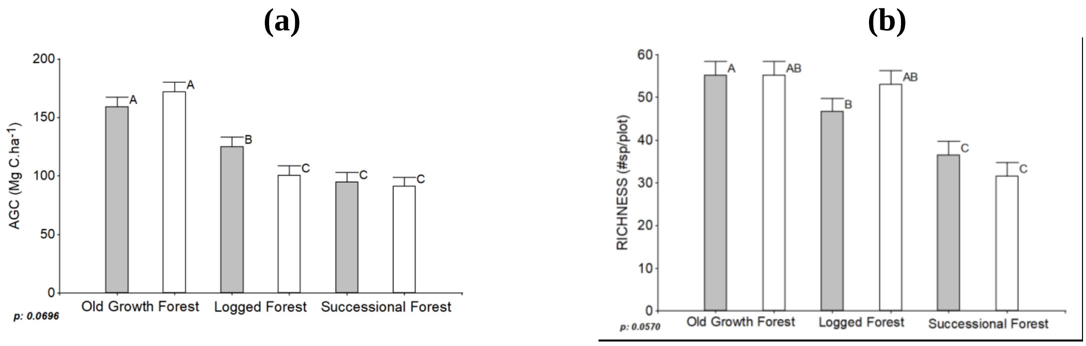

Figure 10. Analysis of variance of ecosystem services provided for the interaction between the conservation program and land use transition phases (maintenance, decline, and recovery). a) AGC: above-ground carbon stocks. b) Tree species richness. Different letters indicate significant differences from each other ($p \leq 0.10$). Grey bars represent landscapes with incentive-based forest conservation and white bars non- incentive-based forest conservation landscapes. Source: Eguiguren et al 2019.

5.3. The contribution of incentive-based forest conservation to halt deforestation beyond the limits of the conservation program

The incentive-based forest conservation in the study areas is contributing to halting deforestation processes. The study showed statistical differences in the annual net deforestation rate before and after the implementation of incentive-based forest conservation at the parish level. Before the establishment of the incentive program, the annual net deforestation rate was

-1.09% between 2000 and 2008, and after the implementation of the program decreased to -0.18% between 2008 and 2016. In contrast, for the parishes with no presence of the incentives program, the annual net deforestation rate increased from -0.41% between 2000 and 2008 to -0.61% between 2008 and 2016 (Figure 11).

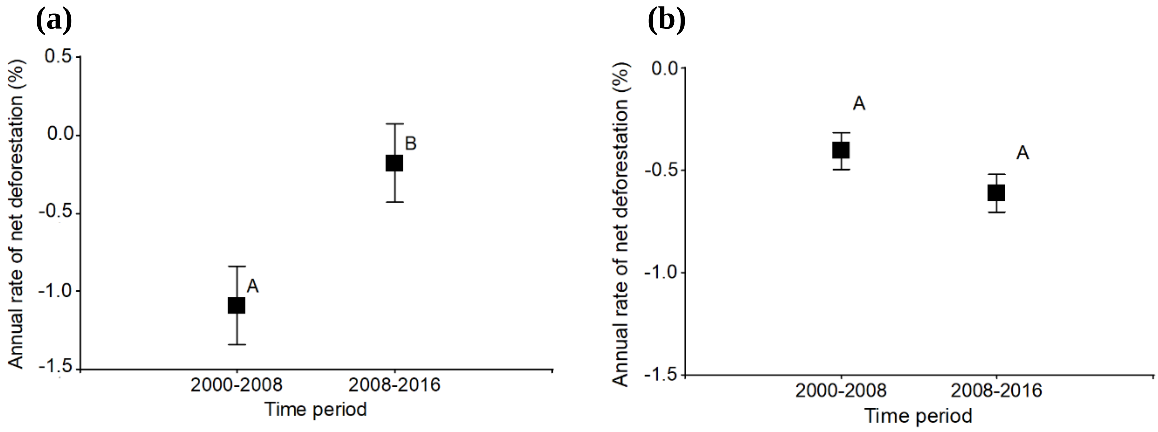

Figure 11.Analysis of variance of annual net deforestation rate (%) at the parish level. a) incentive-based forest conservation landscapes ($p = 0.04$). b) Non-incentive-based forest conservation landscapes ($p = 0.15$). Different letters indicate significant differences from each other ($p \leq 0.05$). Source: Eguiguren et al 2019.

Although studies conducted in the tropics found that command and control conservation programs, such as protected areas, can reduce deforestation inside and outside the areas under protection (Bruner et al., 2000; Nagendra, 2008), mixed results have been reported for studies concerning incentive-based conservation programs. For the latter, evaluations are concentrated in Costa Rica, where incentives showed a small impact in decreasing deforestation rates between 1997 – 2000. However, it is possible that previous conservation policies implemented in this country may have influenced deforestation rates or that the areas selected under the incentives program were not deforestation hotspots (Pfaff et al., 2008; Sánchez-Azofeita et al., 2007). For Ecuador, Mohebalian and Aguilar, (2018), found that areas under the Socio Bosque program are less likely to be deforested. Likewise, Jones et al., (2016) compared areas under the incentive and without the incentive and reported that the average annual deforestation rate reduced between 0.4% to 0.5% with the presence of this program. Similar findings were obtained by Cuenca et al., (2018) in a nationwide study where the authors found that the implementation of the incentives program contributes to avoiding deforestation by 1.5% to 3.4% of the forest.

Studies on the influence of Socio Bosque to stop deforestation in Ecuador show a clear positive trend towards reduction. Although these findings are important, little is known about whether this program has any influence on households' decision to deforest. When analyzing the role of the incentive-based forest conservation program at the household level, the statistical analysis also showed that this program has the potential to generate positive effects beyond the limits of the forests under this program. Controlling for household characteristics, land endowments, the quality of forest resources, and infrastructure, the results showed that households living close to the areas under the incentive-based forest conservation program have lower odds (56% less) to deforest in contrast to those households located in landscapes with no influence of the incentives program.

Table 8. Logit regression results of deforestation at the household level for the Central Amazon

Variable of interest	Coef.	RobustStd. Err.	*p*	Odds Ratio
Incentives for forest conservation (0/1)	-0.810	0.271	***	0.445
Control variables				
Household characteristics				
Age of the household head (years)	-0.087	0.050	*	0.917
Age squared	0.001	0.000	n.s.	1.001
Indigenous group (0/1)	-0.055	0.415	n.s.	0.946
Education of the household head (0/1)	-0.181	0.209	n.s.	0.834
Number of males of working age	-0.157	0.056	***	0.855
Commercialization rate (%)	0.001	0.002	n.s.	1.001
Credit (0/1)	0.467	0.301	n.s.	1.595
Physical asset index	0.289	0.359	n.s.	1.335
Land endowments				
Farm >5ha (0/1)	-2.225	0.358	***	0.108
Forest area within the farm (%)	0.043	0.006	***	1.044
Quality of forest resources				
Timber volume potential (m^3/ha)	0.009	0.002	***	1.009
Institutional environment				
Land titling (0/1)	-0.171	0.486	n.s.	0.843
Governmental grants (01)	-0.985	0.297	***	0.373
Infrastructure				
ln Distance to forest patch (km)	-0.200	0.078	**	0.819
ln Distance to market (km)	0.093	0.111	n.s.	1.098
Intercept	-1.075	1.420	n.s.	
Number of observations	486			
Hosmer Lemeshow goodness of fit				
x^2	7.83			
p	0.45			

*p<0.10, **p<0.05, ***p<0.01. n.s. no significant. Source: Ojeda Luna et al. (2020)

The results showed that incentive-based conservation has a promising potential for combating deforestation beyond the limits of the conservation area and could help to build up multifunctional landscapes. It seems that the Socio Bosque program is raising conservation awareness even in areas outside the program. Some studies show that local people enrolled in the Socio Bosque program conduct frequent surveillance and when they detect illegal practices, within and around the program area, they report it to the environmental authority (Jones et al., 2016, 2020). Moreover, in zones with an incentive-based conservation area, there is a higher presence of governmental staff that constantly monitors the compliance of conservation contracts (Jones et al., 2016; Krause et al., 2013). Perhaps these facts restrain neighboring households to deforest as they may perceive a higher probability to get caught in illegal activities.

6. Synthesis

This study was conducted in two regions of Ecuador: The Central Amazon and the Chocó which are two important areas of biodiversity conservation and for the ecosystem services supply on a local and global scale. These regions are suitable areas to evaluate how modified forest landscapes are providing ecosystem services through a mosaic of forest uses called land-use transition phases, and to assess the effect of a conservation policy on achieving deforestation reduction aims at the parish and household level. In these areas, as well as in many tropical regions, the current demands from a growing population are transforming previously forested and intact landscapes into human-modified systems where several land uses coexist as a manifestation of multiple needs. People need land for shelter, food production, and leisure; at the same time, humanity is worried because of the effects of forest conversion on climate change exacerbation. Landscape transformation creates deforestation and forest degradation which also triggers negative consequences to ecosystem services and human welfare. To decrease the impacts of landscape transformation, several international initiatives have been created to pursue conservation and development goals. Likewise, governments are implementing policies to control deforestation and degradation. Although well-intentioned, such initiatives pursue somewhat conflicting goals, evidencing that stakeholders and decision-makers must adopt an approach that promotes territories capable of satisfying human demands while keeping ecological functions.

In recent years, the multifunctional approach has gained momentum in the scientific community and among land-use planners. This approach considers the forest as an integral part of the landscape and intends to build landscapes able to provide several ecosystem services for people while maintaining the landscape's resiliency. Multifunctional landscapes integrate production and conservation actions, which are essential for human well-being. Hence, multifunctional landscapes can contribute to protecting forests, their biodiversity, and ecosystem services, as well as mitigate climate change impacts. Incorporating the multifunctional landscape approach implies recognizing that natural ecosystems and anthropogenic production systems coexist and interact in complex ways. This is also an acknowledgment that a landscape comprises several components (natural resources, people, and policies) that shape the system and need to be understood to promote the sustainable use of resources.

This dissertation contributes to the multifunctional landscape approach by complementing the existing literature in the tropics. In publication 1 and publication 2, I address the effects of land use transition phases on ecosystem services provision; additionally, I assess the influence of conservation policies i.e. incentive-based forest conservation on ecosystem services supply, deforestation reduction, and household decisions to deforest. Most studies consider only one land use transition phase and individual ecosystem services, overlooking that landscapes show transition phases that represent the productive, economic, and conservation dynamics occurring in tropical landscapes. The key for land use planners is understanding how land use transition phases could enhance a set of diverse ecosystem services. In this study I link the multifunctional approach and the land use transition phases, acknowledging that within landscapes we observe the transition from undisturbed old-growth forests to logged forests with a reduction of ecosystem services, but also the recovery of ecosystem services associated with successional forests, plantations, and agroforestry systems. In this way, this study delves into the relationship between forests, people, and policies as integral components of human-modified landscapes.

This study, furthermore, complements the scientific literature on the multifunctional landscape approach and the ecosystem services framework by identifying ecosystem services that could be used as an umbrella service. The results reveal that above-ground carbon stocks can be considered as an umbrella service as they presented high synergetic relations with the ecosystem service multifunctionality index and various ecosystem services. The results have practical implications in the monitoring process of international initiatives such as REDD+, as synergies between carbon pools with provisioning services, and plant diversity show that above-ground carbon stocks are essential for bundles of ecosystem services. Therefore, actions oriented to improve or conserve the carbon pools can benefit the conservation of other ecosystem services in parallel.

The study additionally contributes to the scientific debates on what strategies should be implemented to supply a more diverse set of ecosystem services and to reduce the impacts of anthropogenic activities. The analyzed forest landscapes are a mix of land use transition phases that include conservation strategies, timber extraction activities, and forest landscape restoration. To maintain ecosystem services for future generations, stakeholders must promote a multifunctional landscape approach that allows for reconciling conservation, the sustainable use of natural resources, and the development objectives founded upon common concerns and agreements. Multifunctional landscapes that include incentive-based conservation, forest management, and forest landscape restoration strategies, can contribute to dealing with the environmental challenges of deforestation and forest degradation. The results from publication 1 and publication 2 highlight that strategies such as Socio Bosque can contribute to building multifunctional landscapes, as this program can help to maintain forest ecosystem services, reduce logging impacts and decrease deforestation rates in the surrounding areas.

From the three forest landscape restoration strategies analyzed, the successional forest has the highest potential to provide ecosystem service multifunctionality. However, it should be noted that households in forest frontiers have urgent economic needs, and use the forest as a source of immediate cash through timber extraction or as land for cultivation. Therefore, to promote the conservation of successional forests, it is necessary to create policies that recognize the importance of successional forests for rural livelihoods. In the Ecuadorian context, successional forests are not used to their full potential. One possible explanation could be related to the fact that successional forests still represent a low proportion of area compared to old-growth forests in the Ecuadorian Amazon, forcing local people to turn to old-growth forests when they want to extract valuable timber species. Moreover, timber species of high commercial value are less abundant in secondary forests, creating an additional incentive to convert these forests to more profitable uses. The lack of infrastructure and knowledge about the best way to use successional forests species, whether for timber or non-timber purposes, is another aspect that limits the use and conservation of successional forests. More research is needed to identify more species that can be used by the forest industrial sector. One practical implication result is that the Ecuadorian government must implement monetary and non-monetary incentives to promote natural regeneration as a restoration strategy. Incentives could be focused on forest management or to facilitate access to credits. They can help to acquire the required technology for forest products harvesting, and help to create added value in products from the successional forest.

Agroforestry systems are also an important production system to recover ecosystem services, though not at the same level as successional forests; however, agroforestry systems are attractive to households because they can turn into alternative income sources. In Ecuador, the chakras (a traditional type of agroforestry system) are a source of food for the household’s subsistence but are also a source of cash as some of the products from the chakra are sold in local markets. Currently, the Ecuadorian government has a National Forest Restoration Plan, which considers agroforestry systems as restoration under productive systems; under this context, this study highlights the need to promote agroforestry systems to satisfy multiple demands at a landscape level.

6.1.Study limitations and recommendations for future research

This study was conducted with data from in situ measurements and provides quantitative information on ecosystem services supply across different land use transition phases. The selected study areas are representative of the land use dynamics and the lowland rainforest ecosystems of Ecuador; therefore, the information provided here could serve as a proxy to understand the dynamics of land uses and the effects on ecosystem services, as well as, to improve the knowledge on the effects of incentive-based programs, nevertheless, future research should take into consideration different tropical countries with similar incentive-based programs, in order to have solid conclusions about the positive influence of this program on ecosystem services provision and deforestation reduction.

In addition, even though the analysis was not focused on the spatial approach, it gives important information about the ecosystem services provided for the Ecuadorian landscapes. The ecological characteristics and the environmental benefits of forests cannot be fully measured by remote sensing. A further step should be to incorporate spatial analyzes using the detailed information from this study to model the ecosystem services supply in different land uses and to incorporate the effect of conservation programs into the analysis.

7. Conclusions

This study presents quantitative evidence on ecosystem service multifunctionality across the land use transition phases which has been scarcely analyzed in Latin America. This study addresses how the provision of single and bundles of ecosystem services can be affected by land-use transition phases occurring at the landscape level and the role of incentive-based forest conservation programs on landscape dynamics and household behavior. Landscapes are mosaics of several land uses that reflect ecological decline and recovery phases due to anthropogenic activities. These natural-human interactions have important implications for ecosystem services provision and require a deep understanding to orient landscape management decisions. By analyzing how land-use transition phases affect ecosystem services supply, it is possible to detect which land uses have a strong impact on the depletion of one or more services and to identify the most effective land uses for ecosystem services' recovery. This information has important implications for practice, as it orients stakeholders to promote land uses that sustain local livelihoods while reducing as many ecological impacts as possible, building sustainable and resilient landscapes. In this study, results showed that successional forests are a suitable option to restore ecosystem service multifunctionality; therefore, this type of forest should be better promoted in national and international restoration programs. Nevertheless, for successional forests to be attractive as a landscape restoration strategy, it is necessary to compensate owners for the foregone benefits from alternative land uses.

Regarding the ecosystem service multifunctionality across the land use transition phases, the study gives important insights into ecosystem services provision and supports environmental decision-making, to guide ecosystem management and conservation, especially in the current decade where natural resources are under increasing pressure. Despite that Ecuador is one of the Latin American countries with the highest deforestation rates, in the Ecuadorian Central Amazon, there are still old-growth forest areas with a high capacity to supply multiple ecosystem services, but human actions like logging activities are reducing this capacity. Provisioning and regulating services, and plant diversity, have a high to moderate impact and could be reaching critical thresholds. This is a complex problem that calls for multiple tools that could include inter alia, the following actions: i) old-growth forests under sustainable forest management must be considered as a strategy to ensure ecological, social, and economic benefits to landowners, especially in areas where deforestation hotspots occur. Sustainable forest management could prevent forest cover loss due to land use change to other economically attractive land uses (e.g. pastures); ii) promote sustainable forest management at the farm level through monetary and non-monetary incentives; iii) reevaluate the timber volumes that the environmental authority permits to extract in old-growth forests; iv) implement reduced impact logging techniques; v) reevaluate the cutting cycles considering the time that tropical forests need to recover affected ecosystem services; vi) establish a strict monitoring and post-harvesting control measures.

Identifying the synergies and trade-offs among ecosystem services and the ecosystem service multifunctionality index helps to recognize important services that could have a prominent role in the landscape multifunctionality. This is another contribution of this work, as I introduce the concept of umbrella services which, to my knowledge, has not been used under the multifunctional landscape approach and the ecosystem services framework. An umbrella service could be understood as one that dominates synergies with as many ecosystem services as possible and as the one that drives much of the multifunctionality in the landscape. In this study, I found that above-ground carbon stocks have a synergetic relationship with the ecosystem service multifunctionality, provisioning services, and plant diversity, but not for soil-related supporting services; despite this, above-ground carbon stocks can be considered an umbrella service. Identifying umbrella services could contribute to initiatives or projects aimed at recovering simultaneous ecosystem services; for example, if an intervention targets the recovery of an umbrella service, this could benefit additional services with which the umbrella service has synergies. This is the case with the REDD+ mechanism, where the conservation of umbrella services, such as carbon stocks, could have positive effects on timber volume and NTFP. Likewise, umbrella services can contribute to monitoring activities within REDD+ to verify the achievement of conservation objectives. The concept of umbrella services could be a starting point to begin identifying key ecosystem services for landscape multifunctionality, however, it needs to be tested with different ecosystem services than the ones evaluated here.

Landscapes are not only shaped by anthropogenic land uses. Institutions play an essential role in the landscape dynamics and the ecosystem services provided. In this study, I evaluated how an incentive-based forest conservation program impacts ecosystem services, deforestation rates, and deforestation decisions at the household level. The incorporation of institutional elements into the analysis helps to expand the multifunctional landscape approach, as it recognizes that landscape dynamics are influenced by contextual factors, such as conservation policies, which should not be overlooked. The results from this study, have central implications for policy implementation as they highlight the positive effects of incentive-based forest conservation (Socio Bosque) in Ecuador and provide information that supports the maintenance of such policies in the long term. The results of publication 2 and publication 3 indicate that the incentive-based forest conservation program (Socio Bosque) is helping to maintain good levels of ecosystem services within the forest under the program. Moreover, Socio Bosque positively influences ecosystem services provision beyond the limits of the conserved forest, showing lower ecosystem services decrease in logging areas close to the program. Results also showed positive effects on deforestation reduction at the parish level and positive effects on households' decision to deforest, where households living close to conservation areas had lower odds to clear the forest. This contradicts the hypothesis that forest use restrictions in one area might provoke a higher pressure in neighboring forest areas. The findings from this research evidence that it is possible to implement conservation policies and to perform logging activities with lower pressure to adjacent logged forests, as long as such policies consider local people's needs. The results are important for the actual conservation debates in Ecuador since the Socio Bosque program is the main step to articulate forest conservation with REDD+ schemes.

Finally, to harmonize development and conservation goals, we need to look beyond single initiatives; instead, complex landscape dynamics call for the implementation of multiple strategies such as incentive-based forest conservation, sustainable forest management, and forest landscape restoration. Only the integration of all these strategies will allow to secure the continuous ecosystem services provision at the landscape level and will facilitate the construction of multifunctional and resilient landscapes. However, the implementation of multifunctional landscapes is not an easy and short-term task; it requires the engagement of academia, politicians, and land-use decision-makers to happen. Ecuador needs better integration between the Ministry of Agriculture (MAG) and the Ministry of Environment, Water, and Ecological Transition (MAATE). Inter-ministerial policies must be implemented at the landscape level, with the MAG working towards the improvement of agricultural production and the MAATE fostering conservation and sustainable forest management.

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9. Appendix

Appendix 1. Further publication of the author (not part of the thesis). The following publications were completed during the time of the Ph.D. (2016–2022):

Peer-reviewed publications

Peters F., Lippe M., **Eguiguren P.** and Günter S. (2023). Forest Ecosystem Services at Landscape Level– Why Forest Transition Matters?. Forest Ecology and Management (534, 19).

Gordillo F., **Eguiguren P.,** Köthke M., Ferrer Velasco R. and Elsasser P. (2021). Additionality and Leakage Resulting from PES Implementation? Evidence from the Ecuadorian Amazonia. Forests 2021, 12(7), 906.

Eguiguren P., Ojeda-Luna T., Maita J., Samaniego N. y Aguirre N. (2022). Vulnerabilidad al cambio climático en microcuencas de alta montaña abastecedoras de agua en la Región Sur del Ecuador. Revista Bosques Latitud Cero. Vol 12. Num. 1. 43 – 53.

Aguirre N., **Eguiguren P.,** Maita J., Ojeda T., Samaniego N., Furniss M., Aguirre Z. (2017). Potential impacts to Dry Forest species distribution under two climate change scenarios in Southern Ecuador. 3:1, 18-29. Journal of Neotropical Biodiversity.

Eguiguren P., Maita J., Aguirre N., Ojeda T., Samaniego N., Furniss M., Howe C., Aguirre Z. (2016). Tropical ecosystems vulnerability to climate change in southern Ecuador. *Journal of Tropical Conservation Science*.

Chapters book (English):

Eguiguren P., Ojeda T., Lozano P y Günter S. (2021). Estimating carbon stocks across forest types in the Ecuadorian lowland forest. Waldbau weltweit 2.0. 163 – 178.

Chapters book (Spanish):

Eguiguren P., Ojeda T., Lozano P. y Günter S. (2020). Contenidos de carbono en paisajes forestales de la Amazonia Central y el Noroccidente del Ecuador. Pp. 94-108. En: Torres, B., Fischer, R., Vargas J.C. y Günter S. (Eds). Deforestación en paisajes forestales tropicales del Ecuador: bases científicas para perspectivas políticas. Ecuador. 172 pp.

Bravo C., Torres B., Cervantes R., **Eguiguren P.,** Paguay D. y Reyes Hector. (2020). Fertilidad del recurso suelo en paisajes forestales de la Amazonía Central y Noroccidente del Ecuador. Pp. 109 – 126. En: Torres, B., Fischer, R., Vargas J.C. y Günter S. (Eds). Deforestación en paisajes forestales tropicales del Ecuador: bases científicas para perspectivas políticas. Ecuador. 172 pp

Ojeda T., **Eguiguren P.** y Torres B. (2020). Ingresos rurales, dependencia de los recursos naturales y medios de vida en paisajes forestales tropicales del Ecuador. Pp. 127 – 142. En: Torres, B., Fischer, R., Vargas J.C. y Günter S. (Eds). Deforestación en paisajes forestales tropicales del Ecuador: bases científicas para perspectivas políticas. Ecuador. 172 pp.

Fischer R., Tamayo F., De Decker M., Ojeda T., Zhunusova E., **Eguiguren P.,** Ferrer R., Torres B., Gissen L. y Günter S. (2020). Uso de la tierra y gobernanza en paisajes forestales tropicales del Ecuador. 152 – 172. En: Torres, B., Fischer, R., Vargas J.C. y Günter S. (Eds). Deforestación en paisajes forestales tropicales del Ecuador: bases científicas para perspectivas políticas. Ecuador. 172 pp

Appendix 2. Publication 1.

Eguiguren, P., Ojeda Luna, T., Torres, B., Lippe, M., & Günter, S. (2020). Ecosystem service multifunctionality: Decline and recovery pathways in the Amazon and Chocó Lowland rainforests. Sustainability, 12(18), 7786.

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MDPI

Article

Ecosystem Service Multifunctionality: Decline and Recovery Pathways in the Amazon and Chocó Lowland Rainforests

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Abstract: The balance between the supply of multiple ecosystem services (ES) and the fulfillment of society demands is a challenge, especially in the tropics where different land use transition phases emerge. These phases are characterized by either a decline (from intact old-growth to logged forests) or a recovery of ES (successional forests, plantations, and agroforestry systems). This highlights the importance of ecosystem service multifunctionality (M) assessments across these land use transition phases as a basis for forest management and conservation. We analyzed synergies and trade-offs of ES to identify potential umbrella ES. We also evaluated the impact of logging activities in the decline of ES and M, and the influence of three recovery phases in the supply of ES and M. We installed 156 inventory plots (1600 m^2) in the Ecuadorian Central Amazon and the Chocó. We estimated indicators for provisioning, regulating, supporting services and biodiversity. M indicator was estimated using the multifunctional average approach. Our results show that above-ground carbon stocks can be considered as an umbrella service as it presented high synergetic relations with M and various ES. We observed that logging activities caused a decline of 16–18% on M, with high impacts for timber volume and above-ground carbon stocks, calling for more sustainable practices with stricter post-harvesting control to avoid a higher depletion of ES and M. From the recovery phases it is evident that, successional forests offer the highest level of M, evidencing high potential to recover multiple ES after human disturbance.

Keywords: synergies; trade-offs; carbon stocks; timber; non-timber forest products; diversity; umbrella ecosystem services; logging; restoration; Amazon; Chocó

1. Introduction

Tropical forests cover 45% of the global forest area [1,2] and are characterized by an outstanding biodiversity [3]. They exhibit a great potential to supply several ecosystem services (ES) such as provisioning, regulating, supporting, and cultural services which are important for people's well-being at a global and local scale [4,5]. Locally, forests can provide timber and non-timber forest products, maintain soil fertility, help with biological connectivity in fragmented areas, and sustain different levels of biodiversity [6–10]. Globally, forests have attracted attention in the climate change mitigation agenda due to their capacity for carbon sequestration and the reduction of greenhouse emissions [11–13]. Despite the importance of tropical forests, it is estimated that only 20% of the remaining forests are undisturbed [14], with anthropogenic activities as the main threats of an ecosystem's capacity to provide multiple ES [8]. These negative influences on tropical forest have been particularly observed in the Amazon Basin and the Chocó region, which are two important biodiversity hotspots in the world [15–17]. The principal drivers of deforestation and forest degradation (Forest degradation refers to changes or decline in ecosystem services due to anthropogenic disturbances [8,18–20]) in these areas are related to forest conversion to agricultural lands, road opening, population growth, and timber extraction [18,21–24]. Deforestation and degradation processes result in land use transition phases which in turn are associated with changes in ES [6,25,26]. These phases can be represented by natural forests (old-growth forest, logged forest and successional forest), planted forests, and agroforestry systems [19,26]. On the one hand, the transition from undisturbed old-growth forests to logged forests is frequently accompanied by a decline in ES and ecosystem service multifunctionality (M) [8,20,27,28]. On the other hand, successional forests (restoration), plantations (reforestation), and agroforestry systems, can emerge as an effort to recover the ecosystem's integrity and functionality [7,29].

ES are defined as the benefits that people obtain directly or indirectly from ecosystems [8]. The capacity of an ecosystem to co-supply multiple ES is understood as the ecosystem service multifunctionality and translates into various environmental and economic benefits for the society [30–32]. Given that ES are influenced by human activities, ecosystem service multifunctionality is not necessarily provided at the same level. Moreover, international forest policy initiatives and their monitoring systems are rather focused on individual services (timber or carbon), but there is still a lack of knowledge and tools to evaluate the ecosystem service multifunctionality. This is particularly important for REDD+ safeguards or forest landscape restoration initiatives both aiming to preserve forests and biodiversity, combat deforestation and ecosystem degradation, and promote ecosystem restoration. As there is an increasing awareness that forest landscapes have to fulfill multiple functions and services, a deeper understanding of synergies and trade-offs between ES and ecosystem service multifunctionality is required to satisfy the multiple demands of society, for example, food, livelihood opportunities, climate regulation, and conservation of biodiversity [33–35].

Even though ES assessments have gained momentum in the last decades, studies have focused on evaluating selected services such as timber provision, water supply, climate regulation, erosion regulation, or cultural services without considering interactions with other services [36–46]. Other studies have assessed ES either only in mature forest, logged forest, successional forest, or plantations [47–53], showing that more empirical studies integrating different land use transition phases are needed to understand the potential decline and recovery of ES supply. Moreover, little attention has been devoted to the study of ecosystem service multifunctionality. For the Amazon and Chocó lowland rainforests there is still a lack of assessments that comprehensively evaluate synergies and trade-offs on ES supply and ecosystem service multifunctionality throughout different land use transition phases. On the one hand, it is important to understand whether timber extraction impacts different ES and ecosystem service multifunctionality to the same extent. On the other hand, given the accelerated loss of ES, it is necessary to quantify the recovery potential of ES and ecosystem service multifunctionality within successional forests, plantations, and agroforestry systems. Detailed quantitative information on the interactions, decline, and recovery of ES and ecosystem service multifunctionality in ecological systems is needed to efficiently monitor and preserve the remaining forest areas while building on the sustainable use of bundles of ES [33].

Our study contributes to closing this research gap by integrating the analysis of various ES (provisioning, regulating, and supporting) and biodiversity indicators. We include natural forests (old-growth forest, logged forest, and successional forest), planted forest, and agroforestry systems [19,54], which altogether represent the land use transitions phases [25,26] observed in two biodiversity hotspots, the Amazon and the Chocó. We address the fact that ES are interrelated and produce complex dynamics within an ecosystem, which need to be disentangled to make better management decisions. Additionally, we acknowledge that the selected phases supply ES at different levels that need to be quantified with in situ information to identify which one can have a greater potential to sustain ecosystem service multifunctionality (M). In this context, our study aims to: (1) understand which synergies and trade-offs occur between ES and M across different land use transition phases, and to assess whether a single ES could be identified as a potential "umbrella ES"; (2) assess the influence of logging activities on the decline of ES and M; and (3) to identify the potential of successional forests, plantations, and agroforestry systems for the recovery of ES and M. For this purpose a total of 156 inventory plots of 40 m × 40 m were installed among old-growth forests, logged forests, successional forests, plantations, and agroforestry systems in the study areas. Seven indicators were considered for provisioning, regulating, and supporting services [8]. Two indicators related to plant diversity were also included due to the positive link of biodiversity to support ES [55–57]. Finally, with the selected indicators, we estimated M based on the multifunctional average approach [58,59].

2. Materials and Methods

2.1. Study Area

Our study was carried out in the Central Amazon and Chocó lowland rainforests of Ecuador (Figure 1). The Central Amazon is characterized by low fertility and acid soils with a pH of 4.5 and high contents of iron (Fe). The annual precipitation in this region is between 2800 and 4000 mm, with an annual temperature of 22 and 27 °C. In the Chocó soils are more fertile and less acidic with a pH of 5.5; the annual precipitation in the Chocó ranges from 728 to 3681 mm, with an annual temperature of 22 and 27 °C [60,61]. These regions represent 54% of the remnant forest in the country and are strategic areas for biodiversity conservation and for the provision of timber and non-timber forest products [62,63]. We randomly selected 12 landscapes (eight in the Central Amazon and four in the Chocó) of approximately 10 km × 10 km each, covering a total area of 163,000 ha. Although our landscapes are in the lowland rainforests, they are influenced by different climatic conditions that could affect the ES and M supply and the synergies and trade-offs observed between them. Hence, we conducted a cluster analysis as the preliminary analysis step to identify whether a grouping pattern of plots due to climatic influence could be detected. First, we executed a principal component analysis (PCA) with 15 climatic variables obtained from WorldClim [64] for each plot. The first two components of the PCA explained 96.6% of the variability and were used for the cluster analysis considering Ward's methods and Bray–Curtis distance. Cluster analysis showed two large groups, "Central Amazon and Chocó", which were statistically different according to the MANOVA test (Figure A1, Figure A2, and Table A1 in Appendix A) [65,66].

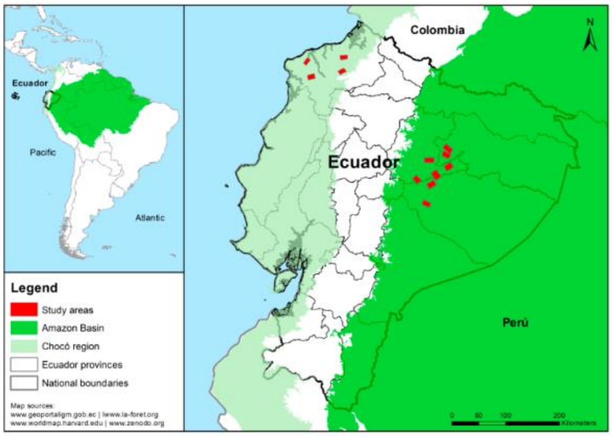

Figure 1. Landscapes selected in the Ecuadorian Central Amazon and Chocó lowland rainforests.

2.2. Sampling Design

Based on available land cover maps, field visits, and participatory mapping exercises performed in 12 community workshops (one per landscape), we obtained a zoning map with the different land use transition phases [6,25,26] (Figure 2). Mapping exercises were conducted with local stakeholders using printouts of high-resolution satellite images obtained from Google Earth. Consequently, we distinguished: (i) Old-growth forests: mature forest with unknown human disturbance [7,19,67]. (ii) Logged forests: forests where timber extraction interventions have been carried out in the last two to five years. In the Central Amazon, logged forests were under simplified harvesting programs (PAFSI, for its Spanish acronym). PAFSIs are small-scale non-mechanized drag programs with a five-year cutting cycle. In the Chocó, logged areas are under the so-called sustainable harvesting programs (PAFSU, for its Spanish acronym). PAFSUs are medium to large scale programs with a fifteen-year cutting cycle, characterized by mechanized hauling. Both, PAFSI and PAFSU, are selective logging systems [68,69]. In one landscape in the Chocó, landholders did not report formal harvesting programs; nevertheless, the owners declared forest intervention in the last two to five years. (iii) Successional forests: second-growth forest resulted from abandoning previously cleared forest [7,70]; random selection resulted in forests ranging between 11 and 28 years of succession. (iv) Agroforestry systems: They are a mixed production system with trees and crops [71]. The most common tree species found in these systems are *Cordia alliodora*, *Inga edulis*, and *Cedrela odorata* mixed with fruit tress like *Citrus reticulata*, *Citrus sinensis*, *Citrus limon*, or *Mangifera indica*. Among the most frequent crops we can cite *Theobroma cacao*, *Musa paradisiaca*, and *Manihot esculenta*. In the Central Amazon, agroforestry systems are called Chakras and correspond to traditional agroforestry systems of diversified production [71]). (v) Plantations: in the Central Amazon these correspond to two to three-years old balsa (*Ochroma pyramidale*) and in the Chocó to four to 18-years old teak (*Tectona grandis*). Balsa usually has a six-year cutting cycle, it grows between 0 and 1000 m a.s.l., with annual precipitation of 1500–3000 mm and temperatures between 22 and 27 °C; balsa reaches a height of 30 m and diameter of 70 cm at breast height. Teak has an 18-year cutting cycle, it grows between 0 and 800 m a.s.l., with an annual precipitation of 1000–2200 mm and temperatures from 22 to 28 °C; teak reaches a height of 30 m and a diameter at breast height of 80 cm.

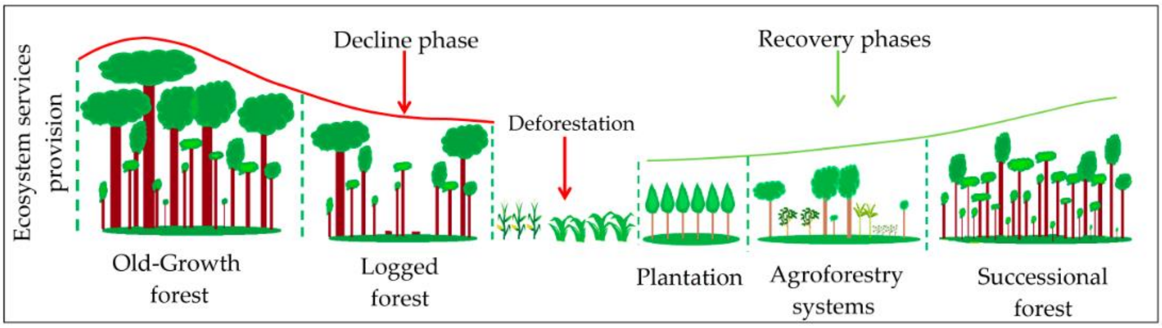

Figure 2. Schematic representation of the land use transition phases based on the ecosystem services provision decline and recovery.

Building on this information, we randomly selected three plots of 40 m × 40 m (1600 m^2) for the old-growth forest, logged forest, successional forest, plantations, and agroforestry systems within each landscape. A total of 156 plots were established across the two study areas (Table 1). For logged forests in the Chocó, we established only nine plots since it was not possible to obtain authorization to conduct the forest inventory in other sites under logging interventions. Meanwhile, for plantations, we only established six plots in the Central Amazon and nine plots in the Chocó since there were no plantations in all the sites.

Table 1. Number of plots installed for the Central Amazon and Chocó.

	Central Amazon (# Plots)	Chocó (# Plots)
Old-growth forest	24 [1]	12 [2]
Logged forest	24	9
Successional forest	24	12
Agroforestry systems	24	12
Plantation	6	9
Total	156 plots	

[1] Central Amazon old-growth forest plots: 12 plots in areas under the Socio Bosque Program and 12 plots in old-growth forest without the program and with unknown human disturbance. [2] Chocó old-growth forest plots: 3 plots in the Mache Chindul Ecological Reserve, 3 plots in the El Pambilar Wildlife Refuge, and 6 plots in old-growth forest without any kind of formal protection and with unknown human disturbance.

2.3. Ecosystem Services Quantification

ES are the result of complex ecological processes and functions performed within forests. Seven indicators were used as proxies for provisioning, regulating, and supporting services, and two for biodiversity [8,72–75] (Table 2). These indicators were chosen based on the Millennium Ecosystem Assessment [8] and their importance in the local and global contexts. The relations observed between indicators provide insights of synergies and trade-offs among ES and reflect the influence of natural resources management and land cover change [73,76]. Ecosystem service multifunctionality (M) estimation was based on the multifunctional average approach [58,59], described in more detail in Section 2.4.

Table 2. Selected ecosystem services and the indicators used for the assessment.

Ecosystem Service	Indicator
Provisioning services	Timber volume potential (TVP, m^3 ha^{-1}) Non-timber forest products (NTFP, # of species per plot)
Regulating services	Above-ground carbon stocks (AGC, Mg ha^{-1}) Soil carbon stocks (SOC, Mg ha^{-1})
Supporting services	Nitrogen in soil (N, %) Phosphorus in soil (P, mg kg^{-1}) Potassium in soil (K, meq/100 mL)
Biodiversity	Tree and palm diversity (D, per plot) Endemism (E, % per plot)

2.3.1. Provisioning Services

Provisioning services refer to products obtained from ecosystems (e.g., food, timber) which are mainly important for human well-being at the local level [8]. We selected timber volume potential (TVP) and non-timber forest products (NTFP) as provisioning services indicators. In the Ecuadorian Amazon and the Chocó, timber provision plays a key role since both areas are important timber suppliers within the country [77]. These areas contain 68% of the national harvestable high-value timber [63]. TVP accounts for the total tree volume of timber species that can be harvested according to the respective species minimum cutting diameter specified in the Ecuadorian forest regulations [69,78]. For TVP, we used diameter at breast height (DBH), total tree height, and a form factor of 0.7 for native species [63]. In the case of *Tectona grandis* plantations, the form factor was 0.55, and 0.73 for the *Ochroma pyramidale* [79–81]. Tree heights were estimated based on four specific equations considering the total height of 1132 trees measured in the Central Amazon and the Chocó. These equations were generated based on own data. Tree height equation for Amazonian trees (Equation (1)) was estimated using 668 trees ($p < 0.0001$; $R^2 = 0.69$). Tree height equation for balsa (Equation (2)) used 110 trees

($p < 0.0001$; $R^2 = 0.81$). Tree height equation for Chocó trees (Equation (3)) was based on 298 trees ($p < 0.0001$; $R^2 = 0.69$). Tree height equation for teak (Equation (4)) was estimated with 56 trees ($p < 0.0001$; $R^2 = 0.70$).

$$\ln \text{Ht} = 0.786 + 0.5956(\ln DBH) \tag{1}$$

where Ht is the total height of the tree and DBH is the diameter at breast height.

$$\text{Ht} = -4.0692 + 5.1391(\ln DBH) \tag{2}$$

where Ht is the total height of the tree and DBH is the diameter at breast height.

$$\text{Ht} = -11.6292 + 8.9439(\ln DBH) \tag{3}$$

where Ht is the total height of the tree and DBH is the diameter at breast height.

$$\text{Ht} = -3.9687 + 5.9616(\ln DBH) \tag{4}$$

where Ht is the total height of the tree and DBH is the diameter at breast height.

NTFPs are paramount for Ecuadorian local communities. According to de la Torre et al. [82], of the 5172 species analyzed in their study in Ecuador, food use corresponds to 30%, medicine 60%, and materials 55%. NTFP represent the number of species per plot with potential use for medicine, food, and materials. Species-use identification was based on secondary information available for the country [82–84] and household surveys from the studied landscapes.

2.3.2. Regulating Services

Regulating services are benefits related to the regulation of ecosystem processes (e.g., air regulation, climate regulation) [8]. Forest ecosystems play an important role in the global carbon cycle [11–13,85]. It is estimated that 652 Gt C (above-ground biomass, below-ground biomass, necromass, and soil) are stored in the world's forests [86]. Our analysis considers the above-ground carbon (AGC) and soil carbon (SOC), both contribute up to 90% of the total carbon pool within tropical forests [12]. We estimated AGC as the sum of above-ground carbon in trees (AGC_{tree}) and above-ground carbon in palms (AGC_{palm}). For AGC_{tree}, we used Chave, et al. [87] equation (Equation (5)), considering DBH greater than 10 cm and species wood density. Tree species wood density was obtained from the Global Wood Density Database [88,89], MAE [90], and Aguirre, et al. [91]. For tree species not included in any of these databases, we used the genus, family, or plot average depending on the case. AGC_{palm} was calculated based on the Goodman, et al. [92] equation (Equation (6)).

$$AGB_{tree} = \exp\left[\begin{array}{c} -1.803 - 0.976E + 0.976\ln(\rho) \\ +2.673\ln(DBH) - 0.0299(\ln DBH)^2 \end{array}\right] \tag{5}$$

where DBH is the diameter at breast height, ρ is the tree species wood density and E is the environmental stress variable.

$$AGB_{palm} = \exp(-3.3488 + (2.7483 \times \ln DBH) \tag{6}$$

where DBH is the diameter at breast height.

In the case of agroforestry systems, we used specific equations for *Manihot esculenta* (Equation (7) [93]), *Theobroma cacao* (Equation (8) [94]), *Coffea arabica* (Equation (9) [94]), *Coffea robusta* (Equation (9) [94]) and *Musa paradisiaca* (Equation (11) [95]).

$$AGB_{Manihot\ esculenta} = -0.67 + 0.44 \times d_{30} \tag{7}$$

where d is the diameter.

$$AGB_{Theobroma\ cacao} = 1.040 \times \exp^{0.0736 \times d} \tag{8}$$

where d is the diameter.

$$\mathrm{AGB}_{coffea\ arabica} = 93.424 \times \exp^{0.208 \times d} \tag{9}$$

where d is the diameter.

$$\mathrm{AGB}_{coffea\ robusta} = 242.6 \times \exp^{0.1264 \times d} \tag{10}$$

where d is the diameter.

$$\mathrm{AGB}_{Musa\ paradisiaca} = 185.1209 + (881.9471 \times \ln H/H^2) \tag{11}$$

where H is the height.

For *Ochroma pyramidale* and *Tectona grandis* we used the equations of Douterlungne et al. [96] (Equation (12)) and Perez-Cordero et al. [97] (Equation (13)) respectively. A conversion factor of 0.47 was used to convert above-ground biomass (AGB) to carbon [85].

$$\mathrm{AGB}_{Ochroma\ pyramidale} = \exp(-2.45 + 2.30 \times \ln DBH) \tag{12}$$

where DBH is the diameter at breast height.

$$\log_{10} \mathrm{AGB}_{Tectona\ grandis} = -0.815 + 2.382 \times \log_{10} DBH \tag{13}$$

where DBH is the diameter at breast height.

SOC was estimated from bulk density, organic carbon content concentration (%), and soil depth (0–30 cm). Bulk density was calculated using the oven-dried weight of soil from a known volume of sampled material at 105 °C until reaching a constant weight; two samples were taken from each plot for this purpose. For the concentration of organic carbon content (0–30 cm horizon), two mixed samples per plot were taken to the laboratory and the wet digestion method of Walkley and Black was used [12,98].

$$\mathrm{SOC} = BD \times \%CO \times D \tag{14}$$

where BD is the bulk density, %CO is the percentage of organic carbon content and D is the depth.

2.3.3. Supporting Services

Supporting services are related to basic ecosystem processes such as soil formation or primary productivity, contributing indirectly to human well-being, and helping to maintain processes and functions for provisioning, regulating, and cultural services [8,99]. In our study, we considered soil quality indicators related to the ecosystem's health. We selected phosphorus (P), total nitrogen (N), and potassium (K) because of their wide application for soil quality assessments [100]. These indicators are crucial in several of the ecosystem's processes as they intervene in biomass development which directly influences plant growth and net primary productivity, and therefore in the supply of other ES [100–102]. In each plot, two mixed soil samples per plot (0–30 cm horizon) were taken to the laboratory. Total nitrogen (N%) was determined by the Kjeldahl method whereas potassium (K meq/100 mL) and the content of phosphorus (P mg kg^{-1}) were estimated with Olsen's methodology [103,104].

2.3.4. Biodiversity

Biodiversity underpins ES provision since it has an important role in the ecosystem´s processes and functions [55–57,105–107]. Quantifying biodiversity is a complex task as it is a multifaceted concept that can be measured or documented in different ways [108]. For this reason, researchers usually focus on certain components of biodiversity. In this regard, plant diversity is a fundamental component of biodiversity [108,109] as it is closely linked with other components of biological diversity, for example, soil fauna, birds, or arthropods [108,110–113]. Besides, plant diversity has a positive effect on ES supply [57,107,114,115] and the interactions occurring among plant species play an important role in the ecosystem functioning. Thus, plant diversity is an important indicator for conservation purposes. Moreover, Ecuador is an important hotspot of biodiversity [15–17] and one of the countries in the tropics with more tree species [116,117]; therefore, for this study, we used plant diversity (D) as a biodiversity indicator. D was estimated with the Shannon index (H) and considers trees and palms (Equation (15)) [108]. We additionally used endemism (E), as a proxy to evaluate the capacity of an ecosystem to offer habitat for species with limited distribution [118]. The regions included in this study are important areas of endemism; of the 4500 endemic species registered for the country, around 78% are threatened to some degree of extinction [119]. E is expressed as the percentage of species per plot whose distributional range is restricted to Ecuador. Estimations of E were based on the red book of endemic plants for Ecuador [119].

$$H = \sum \rho_i \times Ln\ \rho_i \tag{15}$$

where ρ_i is the species' relative proportion.

2.4. Ecosystem Service Multifunctionality (M)

Ecosystem service multifunctionality (M) is defined as the capacity of an ecosystem to simultaneously provide multiple ES to society [31]. Human activities have a direct influence on ecosystem service multifunctionality and can reduce or increase the supply of several benefits from forests. M estimations were based on the multifunctional average approach [58,59]. The approach has been applied in various studies to analyze bundles of ecosystem services, functions, biodiversity, functional trait diversity, and landscape multifunctionality (e.g., [58,59,120–127]). The multifunctional average approach is based on the mean of the standardized values of each ES. For the standardization, minimum and maximum values for the land use transition phases were used. Given that ES could be correlated to each other, we tested for multicollinearity among the selected indicators using the variance inflation factor (VIF) [128–130]; which resulted in a VIF value of 3.21. VIF values between 5 and 10 correspond to moderate collinearity, and greater than 10 suggest the presence of high collinearity [128–130].

2.5. Statistical Analysis

Synergies and trade-offs among the land use transition phases (old-growth forest, logged forest, successional forest, agroforestry systems, and plantations), M, selected ES and biodiversity indicators (TVP, NTFP, AGC, SOC, N, P, K, D, and E) were evaluated with a principal component analysis (PCA) for each study region. Furthermore, we performed an analysis of variance (ANOVA) using general mixed models to evaluate the potential decline and recovery of ES and M. Land use transition phases within the Central Amazon and the Chocó were considered as fixed factors and landscapes as a random factor. Outliers' identification was based on the standardized Pearson residual. Normality and homoscedasticity were evaluated through the Shapiro and Levene tests. TVP, AGC, P, and K were natural log-transformed to meet ANOVA assumptions, and in the case of E, a non-parametric analysis (Kruskal–Wallis test) was performed. For NTFP, D, and E ANOVA, we excluded plantations as they did not have values for these indicators.

3. Results

3.1. Ecosystem Services Synergies and Trade-Offs

The first two components of the PCA explained 62.2% of the variance for the Central Amazon and 58.7% for the Chocó. The first component in both regions showed a grouping pattern that exhibits the change in ES and M across different land use transition phases from old-growth forest to logged forest and then to more transformed systems such as successional forests, agroforestry systems, and plantations. M was closely related to the first component indicating that it explains between 40% and 44% of the variability. As expected, old-growth forests offer higher values of M, demonstrating a good capacity to provide multiple ES. Interestingly, soil parameters showed different patterns between both study areas, while AGC, TVP, NTFP, D, E, and M behaved in a relatively similar way (Figure 3).

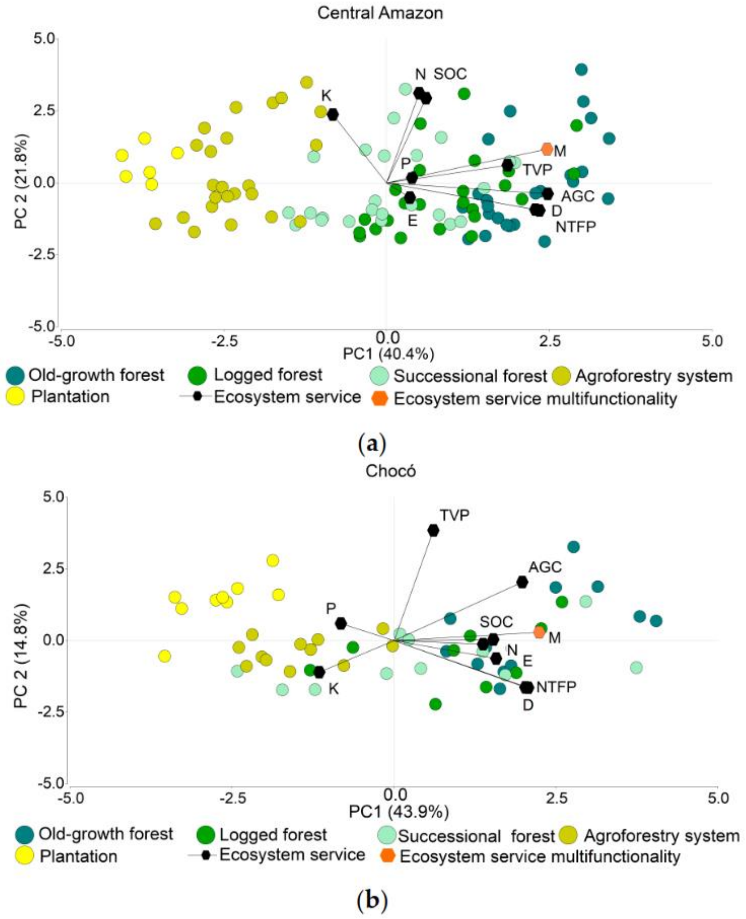

Figure 3. Principal component analysis. (**a**) Central Amazon synergies and trade-offs. (**b**) Chocó synergies and trade-offs. TVP: timber volume potential (m^3 ha^{-1}). NTFP: non-timber forest products (# sp. per plot). AGC: above-ground carbon stocks (Mg ha^{-1}). SOC: soil carbon stocks (Mg ha^{-1}). N: nitrogen (%). P: phosphorus (mg kg^{-1}). K: potassium (meq/100 mL), D: Shannon index. E: endemism (% of species per plot). M: ecosystem service multifunctionality.

We observed that a bundle of ES related to regulating and provisioning services, and biodiversity, are interacting in a synergetic way in both regions. We identified positive correlations between AGC, TVP, NTFP, D, E, and M with higher values in old-growth forest and lower values in agroforestry systems and plantations. Other positive relations were found among SOC and N, and between AGC and TVP, the latter one with a higher correlation in the Central Amazon than in the Chocó. In contrast, negative correlations, which indicate trade-offs, were also detected for both regions between supporting services indicators (K for Central Amazon; P and K for Chocó) with regulating and provisioning services indicators, and biodiversity. Besides, the PCA showed lower AGC, TVP, NTFP, and D values for agroforestry systems and plantations, but revealed an increase of K (Figure 3).

3.2. The Provision of Ecosystem Services and Ecosystem Service Multifunctionality across the Land Use Transition Phases

Analysis of variance showed that ES and M provision varies significantly across land use transition phases (Table 3, Figures 4 and 5), except for N which did not show significant differences (p: 0.2686). TVP, AGC, SOC, N, K, P, and M for old-growth forests in both the Central Amazon vs. Chocó were statistically similar. Nonetheless, NTFP, and E showed statistical differences between the old-growth forests in both regions. E was higher in the Chocó's old-growth forest where 6.4% of species were endemic (Table A2 in Appendix A). In the Chocó region, endemic species were also found in the logged forest, successional forest, and agroforestry systems. In contrast, endemic species in the Central Amazon were circumscribed in a lower percentage to the old-growth, logged, and successional forest only.

Table 3. Analysis of variance of ecosystem services and ecosystem service multifunctionality.

Ecosystem Services Indicator	*p*-Value	R^2	n
Ecosystem service multifunctionality—M	<0.0001	0.82	152
Timber volume potential—ln TVP (m^3 ha^{-1})	<0.0001	0.33	140
Non-timber forest product—NTFP (# sp. per plot)	<0.0001	0.86	141
Above-ground carbon stocks—ln AGC (Mg ha^{-1})	<0.0001	0.81	153
Soil carbon stocks—SOC (Mg ha^{-1})	0.0187	0.52	156
Nitrogen—N (%)	0.2686	0.82	148
Phosphorus—ln P (mg kg^{-1})	0.0093	0.61	156
Potassium—ln K (meq/100 mL)	<0.0001	0.68	153
Diversity—D (Shannon index)	<0.0001	0.78	141
Endemism—E (% sp. per plot)	<0.0001	-	141

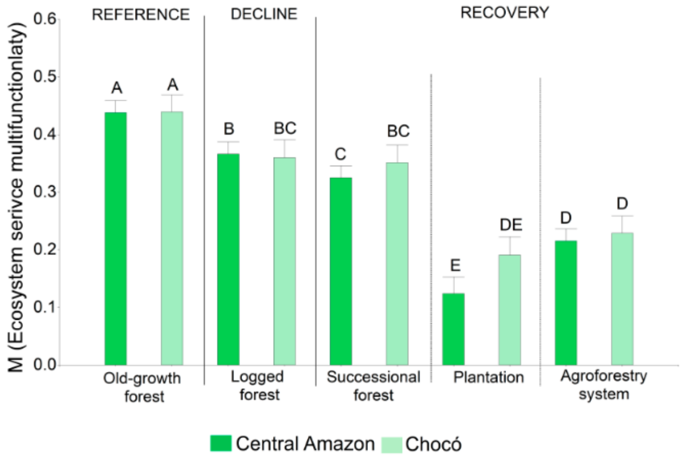

Figure 4. Analysis of variance for the evaluation of the ecosystem service multifunctionality. Different letters indicate a significant difference from each other ($p \leq 0.05$).

The decline in the provision of ES and M as a result of logging interventions was assessed by comparing old-growth and logged forests. We observed a clear reduction in M (Figure 4) by 16% in the Central Amazon and 18% in the Chocó due to timber extraction. Likewise, a decline in specific ES was evidenced (Figure 5). TVP values in the old-growth forest were 190.5 m^3 ha^{-1} in the Central Amazon and 149.9 m^3 ha^{-1} in the Chocó. Compared with old-growth forests, logged forests showed low values for TVP; we estimated 113.3 m^3 ha^{-1} for the Central Amazon and 75.9 m^3 ha^{-1} for the Chocó (Table A2 in Appendix A). We also observed significant differences for AGC (Figure 5). Old-growth forests accounted for 167.3 Mg ha^{-1} in the Central Amazon and 146.9 Mg ha^{-1} in the Chocó; in contrast, logged forests had 113.3 Mg ha^{-1} for the Central Amazon and 102.5 Mg ha^{-1} for the Chocó (Table A2 in Appendix A). The level of SOC, N, and P also decreased due to logging interventions, although no statistical differences were found for these services (Figure 5). Logging activities caused a significant reduction of NTFP species only in the Central Amazon, with 39 species per plot in the old-growth forest in comparison to 35 species per plot in the logged forest. A significant decline in the percentage o endemic species was observed for the Chocó; in the old-growth forest we found 6.4% of E; meanwhil in the logged forest this value decreased to 3.9%. Regarding D, similar levels were found among th logged and old-growth forest.

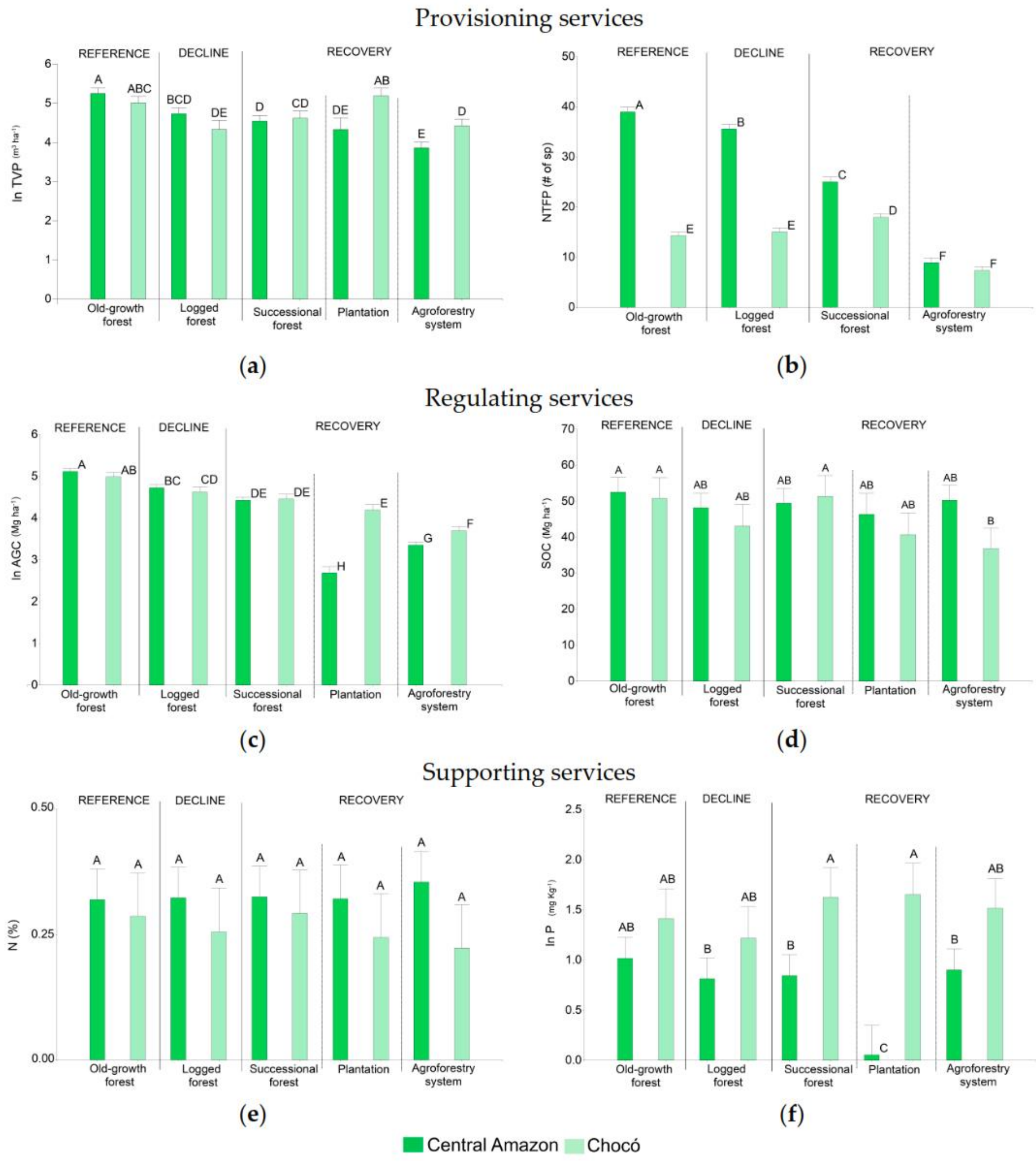

Figure 5. *Cont.*

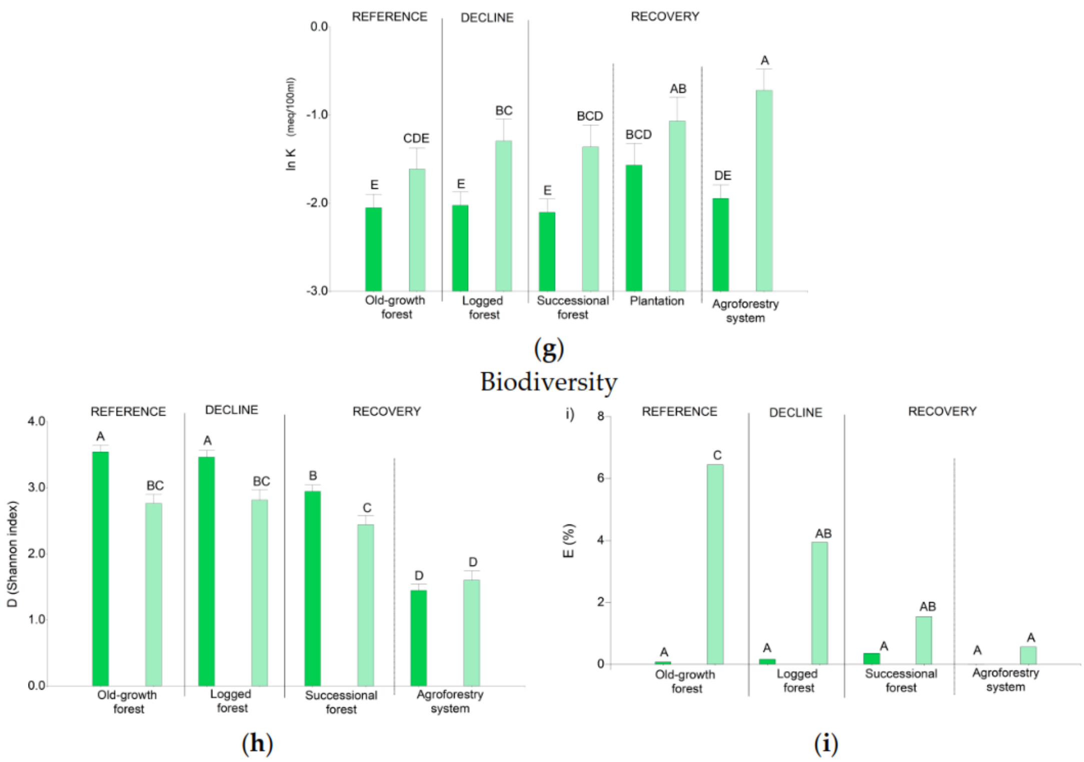

Figure 5. Analysis of variance for the ecosystem services. (**a**) ln TVP: timber volume potential, (**b**) NTFP: non-timber forest products, (**c**) ln AGC: above-ground carbon stocks, (**d**) SOC: soil carbon stocks, (**e**) N: nitrogen, (**f**) ln P: phosphorus, (**g**) ln K: potassium, (**h**) D: Shannon index, (**i**) E: endemism. Different letters indicate a significant difference from each other ($p \leq 0.05$).

The recovery of ES supply and M was evaluated for successional forests, agroforestry systems, and plantations. Our results show an important potential of these recovery phases to recuperate the M (Figure 4) and particular ES (Figure 5). Indicators regarding soil-related services (N, P, K, and SOC) presented similar values than the reference old-growth forest. For example, N values in soil ranged from 0.22% to 0.33% across the three recovery phases and were statistically similar to the old-growth forest with values from 0.29% to 0.33% in both regions. As expected, the M, provisioning (TVP and NTFP) and regulating (AGC) services, and biodiversity (D and E) did not reach similar levels than the old-growth forest. Nonetheless, we observed that successional forests had similar values of ES than logged forests in both study areas. M reached values of 0.32 and 0.35 (Figure 4), TVP ranged between 93.6 and 101.4 m^3 ha^{-1}, AGC was between 83.9 and 86.4 Mg ha^{-1}, and D was between 2.9 and 2.4 in the Central Amazon and the Chocó, respectively (Figure 5 and Table A2 in Appendix A). These results indicate the high capacity of successional forest to recover ES bundles. Plantations and agroforestry systems had low M; though, teak plantations had high values of TVP (179.4 m^3 ha^{-1}) similar to the old-growth forest.

4. Discussion

4.1. Ecosystem Services Synergies and Trade-Offs

The first component of the PCA showed a strong and positive correlation of M and how anthropogenic activities influence the ecosystems' integrity and their capacity to provide multiple ES simultaneously. In the Central Amazon and the Chocó, M, AGC, TVP, NTFP, D, and E have a synergetic relation; whereas SOC, N, P, and K have rather diverging patterns. Interestingly, AGC showed positive relations with various ES similar to those observed for M, which indicates that AGC could be key for ecosystem service multifunctionality. Despite this, between both study areas M and

AGC did not show similar patterns of correlations with soil-ES indicators (P and K), which can be the result of specific edaphoclimatic conditions, such as pH and precipitation intensities that influence soil indicators [131–133]. In our study areas, the average annual precipitation is 1812 mm in the Chocó and 4118 mm in the Central Amazon; whereas the pH is 5.5 in the Chocó and 4.5 for the Central Amazon. The information on average annual precipitation for these two regions was extracted from WorldClim data [62] for each plot installed for this study; likewise, the pH information is based on measurements in each of those plots. In consequence, the lower levels of P and K found in the Central Amazon could be explained by acid soils and leaching effects produced due to higher annual precipitation [61,131].

The observed interrelations between ES indicate that any human activity oriented to modify an ecosystem will have a positive or negative influence in various ES. For example, AGC and TVP had strong synergy in the Central Amazon and a moderate one in the Chocó, meaning that timber yields maximization most likely influence carbon pools directly, especially if the number of timber species and related harvestable timber volumes increase. In this context, the implementation of sustainable forest management practices considering less damage to remaining trees and the harvest of lower timber volumes will help to maintain AGC [134,135]. Additionally, we observed high to moderate positive correlations in the Central Amazon and Chocó between AGC and D. In this respect, Bunker, et al. [136] suggest that selective logging of species with big diameter and high wood density will contribute to a reduction of carbon stocks. There are also strong correlations between D and NTFP which means that a greater diversity of trees and palms holds a higher potential for forests to supply provisioning services, other than timber, which at the same time can promote alternative forest uses. On the other hand, we observed trade-offs mainly in AGC, TVP, D, NTFP, and M primarily for agroforestry systems and plantations. Despite this reduction, the values observed in soil indicators are still high, especially for K and P. Trees within agroforestry systems and plantations can influence soil properties and nutrient cycling [137]. Therefore, specific species can either cause a decrease or will help to enhance soil-related services [53,138].

Since monitoring ES and biodiversity is time-consuming and expensive, the implementation of policies aimed at the conservation of key services will generate simultaneous benefits to other ES. In this context, the umbrella species approach could provide the background for the identification of potential "umbrella ES". This approach aims to conserve important species that require large areas or particular habitats to maintain their population. Therefore, the protection of umbrella species simultaneously contributes to the conservation of other species [139–141]. Based on this approach, conservation policies could target specific umbrella ES considering their capacity to generate multiple synergies with other ES. Umbrella ES should be easily measurable and sensitive to forest degradation and land use change. In this regard, our results showed that AGC can be considered as an umbrella ES since it is a sensitive indicator of forest integrity and it had strong to moderate synergies with M and several ES; however, AGC may not be a good proxy for soil indicators since it showed variable patterns. Thus, our findings support the feasibility of strategies such as REDD+ as it confirms synergies between carbon pools, provisioning services, and biodiversity [142], but also shows some limitations concerning soil ES.

4.2. Assessing the Decline of Ecosystem Services and Ecosystem Service Multifunctionality

The intensity of timber extraction, tree damage during forest intervention, and unplanned logging activities result in temporal or permanent decline of ES and therefore a potential forest degradation [37]. Based on the selected ES, the Central Amazon and the Chocó had a reduction of 16%–18% of M values respectively. AGC was highly affected by a reduction of 30%–32% for both regions, whereas TVP depleted 40% in the Central Amazon and 49% in the Chocó. This reduction occurred even though forest interventions in our study areas followed the technical guidelines from the Ministry of Environment of Ecuador, with cutting cycles between 5 years for PAFSI and 15 years for PAFSU. According to Roopsind, et al. [143], after 32 years the recovery probability of carbon and timber stocks with harvest intensities of 15–23 m^3 ha^{-1} ranges between 45% and 80%. Other studies suggest that a logged forest will need between 12 to 43 years to recover after an AGC reduction of 10%–26% [144–146]. Based on these studies and our estimations, there is evidence that suggests that the logged areas included in our study may not completely reach their initial or reference levels as undisturbed forest if the current cutting cycles are maintained. If logging activities in these areas increase, it will lead to a permanently degraded forest where timber yields could be lower for the second or third cut [147].

Our findings highlight the importance of implementing sustainable forest management practices, such as reduced impact logging (RIL). RIL techniques will allow an adequate recovery of timber and carbon stocks, reducing C emission and supporting climate change mitigation. If RIL is implemented adequately, the damage to the remaining trees can be reduced [135]. In this regard, West, et al. [145] and Vidal, et al. [148] found that after the application of RIL techniques, with a reduction of 17% of AGB and 21% of commercial timber volume, AGB and commercial volume (merchantable species) can be recovered within 16–20 years, respectively.

Regarding NTFP, a decrease was only found in the Central Amazon which suggests a potential loss of species important for local use due to a change in forest structure and composition. The decrease in NTFP species can have implications on livelihood strategies since some species can be important for both timber and NTFP, creating a conflict of use for local people [149]. The logging effect on E was only observed in the Chocó, in this particular environment, continuous timber extraction interventions could lead to a high risk of endemic species extinction due to their restricted habitat distribution. The non-statistical differences of D between old-growth and logged forest in the Central Amazon and Chocó could be explained by the intermediate disturbance hypothesis, wherein the diversity of species can be maximized by the presence of disturbances [150–152]. In our case, probably the timber extraction (disturbance) stimulated new pioneer species contributing to species diversity [153,154]. Nevertheless, high intensity and frequency of disturbances could deplete tree diversity due to the decline of shade-tolerant species [150,151].

Logging intensities, size gap, and frequency influence nutrient cycling through the loss of nutrient pools and the reduction of litter inputs [155,156]. In our study areas, soil-related ES presented a reduction in the logged forest for SOC, N, and P, but with non-statistical differences to the old-growth forest in both study areas, suggesting a low effect of timber extraction. Our results are in line with Olander, et al. [157], who did not find an influence of logging gaps on SOC, N, and P. These authors observed a reduction of these indicators particularly in roads and decks during logging which can be mainly attributed to litter movement. Concomitantly, McNabb, et al. [158], found a reduction in C, N, P, and, K primarily in skid trails, and argued that soil impacts were a combination of logging effects and indirect influence of pioneer species such as *Cecropia* sp. during secondary succession. The removal of litter and nutrient pools from big trees in the above-ground biomass as a result of high logging intensities affect nutrient cycles [155] and therefore soil-related ES. Our results suggest relatively low evidence for degradation effects in soil-related services. However, these results should be interpreted with caution as we did not evaluate the effects of logging interventions on roads or decks.

Timber extraction is an important driver of degradation in the Amazon and Chocó [18]. In our study areas, logging mainly affected TVP and AGC, and to a lower level, the soil-related services, leading to a decrease of M. Since our study regions account for 68% of Ecuador´s harvestable timber with high commercial value [63], our results imply that to maintain ecosystem service multifunctionality and forest integrity, it is essential to implement sustainable use strategies that involve RIL techniques. It is also required to adjust the logging cycles and logging intensities considering the forest recovery potential. Moreover, the establishment of a permanent plot-network for the long-term monitoring of logged forests should be part of the national forestry inventory efforts.

4.3. Identifying the Potential Recovery of Ecosystem Services and Ecosystem Service Multifunctionality

Among the three analyzed recovery phases, the successional forest was the most effective in recovering single ES and M. These results show the importance of the successional forest to recover carbon pools, to provide timber and non-timber products, and to sustain plant diversity. However, successional forests have different species composition than undisturbed forests and are mainly characterized by fast-growing trees and a low abundance of slow-growing trees [159,160]. High values of AGC and TVP in the successional forest are explained by the high rates of biomass during the young and mid-stages of natural succession [48]. Overall, plantations and agroforestry systems had low levels of M. For example, balsa plantations in the Central Amazon had the lowest M values, low stocks of TVP and AGC, which is also expected due to their age structure (two to three-year of age; cutting cycle: 6 years) in contrast to older teak plantations in the Chocó (four to eighteen years; cutting cycle: 18 years). A relevant result was that agroforestry systems in the Chocó provide habitat shelter for endemic plant species (0.55% plot-average) and maintain an important agro-diversity that contributes to the livelihoods and income of local communities. On the other hand, plantations are an important source for timber provision. If smartly allocated, they could alleviate anthropogenic pressures in old-growth forests and contribute to AGC accumulation in the short and mid-term.

Land use change affects soil-related ES indicators by increasing soil erosion and by reducing the nutrient pools in above-ground biomass and litter. A study on soil attributes comparing pastures and forest in the Napo River valley, found that forest clearing caused a soil fertility loss for C (up to −68%), N (up to −50%), and K (up to −50%). This can be explained by a reduction in organic matter and high precipitation associated with nutrient leaching [161]. Other studies suggest that land use conversion from the forest into agricultural land can reduce SOC stocks by 25–75% [76,162,163]. The evaluated recovery phases offer promising values of soil indicators (SOC, N, P, and K). Plantations and agroforestry systems can influence soil properties positively or negatively depending on the species incorporated. Plantations of fast-growing species like *Jacaranda copaia* and *Vochysia guatemalensis* can deplete soil K and P [138], while teak plantations can maintain soil K and P [53]. In this case, the process is linked to leaf litter decomposition as older teak plantations are often more efficient in nutrient up taking from deeper soil horizons [164]. On the other hand, our agroforestry systems contain species such as *Inga edulis*, *Myroxylon balsamum*, and *Erytrina poeppigiana* which facilitate nitrogen fixation and foliar biomass, contributing to soil organic material and recovery of soil nutrients [165,166]. Plantations and agroforestry systems can help to recover soil-related services. However, it is important to highlight that a decline in nutrient pools can occur by a reduction of above-ground biomass after forest clearing or through timber extraction during several rotations cycles for the case of plantations. In the long-term this would lead to a permanent loss of soil fertility [167,168].

The recovery of the ecosystem's integrity in deforested and degraded lands received more attention after the 20 × 20 Initiative, the Bonn Challenge, the FLR initiative, and REDD+. In this sense, our assessment provides empirical evidence of the effect of three common recovery phases on ecosystem service multifunctionality. Our study shows how natural systems that were almost completely modified have the potential to recover M values and reverse anthropogenic degradation. Agroforestry systems and plantations presented the lowest M, but they are important elements at the landscape level as they represent productive activities that help to improve local welfare and reduce pressures on the old-growth forest. This is especially important in our study areas, where 50–60% of household income depends on natural resources [169] and for indigenous communities who culturally rely on traditional agroforestry systems (chakras) to develop their livelihoods, harmonizing ES (mainly provision and regulation) for their well-being and biodiversity preservation.

From the three recovery phases, successional forests reach the highest levels of M. Although the successional forests evaluated here are not fully comparable to old-growth or even logged forests, the levels of M achieved so far were between 73% for the Central Amazon and 80% for the Chocó compared to the old-growth forest and had similar values to the logged forest, but with a different structure and composition. This underpins the importance of successional forests for recovery of ES with a low associated cost; considering these forests in strategies like payment for ecosystem services or sustainable forest management may generate economic benefits to their owners while supporting conservation and avoid a new clearing of these successional forests. Agroforestry systems and plantations might not reach high M values as old-growth forest, but they are of crucial importance for food and timber provision. If both strategies are combined with successional forest they can improve ES provision at the landscape level.

5. Conclusions

We present new evidence on the capacity of different land use transition phases to simultaneously provide bundles of ES as a result of complex interactions. The interrelations among the selected ES show that human-induced alterations on ecological systems have a profound effect on ES bundles. Since AGC was positively correlated with M and presented synergies with multiple ES (e.g., TVP, NTFP, D, and E), we consider that AGC could serve as an umbrella ES as it confirms a synergetic relation with provisioning services and biodiversity indicators. Actions focused to improve or conserve the carbon pools can benefit other ES in parallel. However, for soil-related services, the conservation of AGC may not be entirely effective.

Our result shows that logging activities provoke a decline in ES and M, reaching critical thresholds. M and particular ES such as TVP and AGC are highly affected, whereas soil-related services showed lower impacts due to timber extraction. More intense and frequent logging interventions can lead to long-term degradation or even trigger a drastic forest cover change with high negative effects on the ecosystem service multifunctionality. Sustainable management practices are required to reduce tree damage. It is also necessary to harmonize cutting cycles considering the forest recovery potential, accompanied by strict monitoring and post-harvesting control measures. These actions will help to avoid forest conversion into alternative land uses, maintaining ecosystem service multifunctionality, especially in landscapes under high deforestation pressure, such as the Amazon and the Chocó.

Our results highlight how successional forest, plantations, and agroforestry systems have the potential to recover ecosystem service multifunctionality and reverse human disturbance after forest clearing. From the evaluated recovery phases, the successional forest achieves the highest levels of M values and is the best option for recovering an ecosystem´s integrity. Nevertheless, as more human-modified landscapes emerge and the need for provisioning services increases (e.g., food, medicine, timber), it is also important to envisage options towards agroforestry systems and plantations. The incorporation of different alternatives to recover ES and M will help to reduce the gap between conservation and economic activities in local communities.

This study evidences that there is an urgent need for a broad range of interventions to integrally tackle environmental problems in tropical landscapes. On the one hand, actions that conserve the standing intact forests and reduce the ongoing deforestation are required; on the other hand, better forest management practices aimed at reducing ES depletion have to be implemented. This shows that there is no unique strategy and calls for integrated landscape approaches and initiatives aiming to harmonize climate change mitigation with other forest functions, such as REDD+.

Author Contributions: Conceptualization, P.E., T.O.L., and S.G.; methodology, P.E. and S.G.; formal analysis, P.E. and T.O.L.; investigation, P.E.; data curation, P.E.; writing—original draft preparation, P.E. and T.O.L.; writing—review and editing, P.E., T.O.L., B.T., M.L., and S.G.; project administration, P.E., T.O.L., and B.T.; funding acquisition, P.E., T.O.L., and S.G., supervision, S.G. All authors have read and agreed to the published version of the manuscript.

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Conflicts of Interest: The authors declare no conflict of interest.

Appendix A

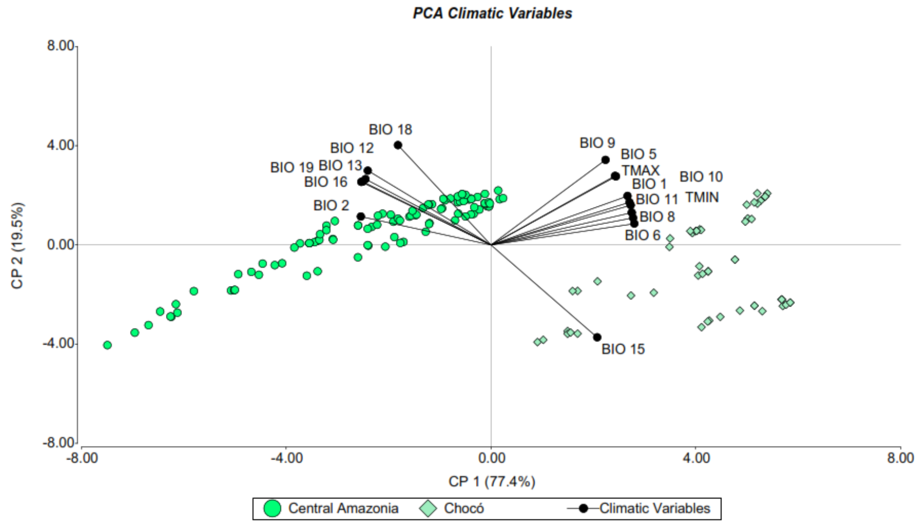

Figure A1. PCA based in climatic variables BIO 1: annual mean temperature, TMIN: annual mean minimum temperature, TMAX: annual mean maximum temperature, BIO 2: mean diurnal range, BIO 5: max temperature of the warmest month, BIO 6: min temperature of the coldest month, BIO 8: mean temperature of the wettest month, BIO 9: mean temperature of driest quarter, BIO 10: mean temperature of warmest quarter, BIO 11: mean temperature of coldest quarter, BIO 12: annual precipitation, BIO 13: precipitation of wettest month, BIO 15: precipitation seasonality (coefficient of variation), BIO 16: precipitation of wettest quarter, BIO 18: precipitation of warmest quarter and BIO 19: precipitation of coldest quarter.

Table A1. MANOVA for the two main groups identified from cluster analysis.

Cluster	PC 1	PC 2	n	
Cluster 1 = Chocó	4.16	−0.97	54	A
Cluster 2 = Central Amazon	−2.2	0.51	102	B

Wilks: p-value ≤ 0.0001
Pillai: p-value ≤ 0.0001
Lawley–Hotelling: p-value ≤ 0.0001
Roy: p-value ≤ 0.0001
Different letters (A and B) indicate significant difference from each other

Table A2. Descriptive statistics for the selected ecosystem services.

	Region	TVP (m^3 ha^{-1})	NTFP (# sp. $plot^{-1}$)	AGC (Mg ha^{-1})	SOC (Mg ha^{-1})	N (%)	P (mg kg^{-1})	K (meq/100 mL)	D (index)	E (%)	M (index)
Old-growth forest	Central Amazon	190.57	38.98	167.34	52.57	0.32	2.77	0.13	3.54	0.07	0.44
	Chocó	149.90	14.26	146.94	50.79	0.29	4.10	0.20	2.76	6.44	0.44
Logged forest	Central Amazon	113.30	35.55	113.30	48.19	0.32	2.25	0.13	3.46	0.16	0.37
	Chocó	75.94	15.01	102.51	43.13	0.26	3.39	0.27	2.81	3.94	0.36
Successional forest	Central Amazon	93.69	25.03	83.93	49.45	0.33	2.34	0.12	2.94	0.36	0.32
	Chocó	101.49	17.94	86.49	51.42	0.29	5.10	0.26	2.44	1.54	0.35
Plantations	Central Amazon	75.94	-	14.73	46.39	0.32	1.06	0.21	-	-	0.12
	Chocó	179.47	-	66.02	40.68	0.24	5.21	0.34	-	-	0.19
Agroforestry systems	Central Amazon	47.47	8.87	28.50	50.34	0.35	2.46	0.14	1.45	0.00	0.22
	Chocó	83.10	7.34	40.45	36.88	0.22	4.57	0.49	1.60	0.56	0.23

TVP: timber volume potential. NTFP: Non-timber forest products. AGC: above-ground carbon stocks. SOC: soil carbon stocks. N: nitrogen. P: phosphorus. K: potassium, D: Shannon index. E: endemism. M: ecosystem service multifunctionality.

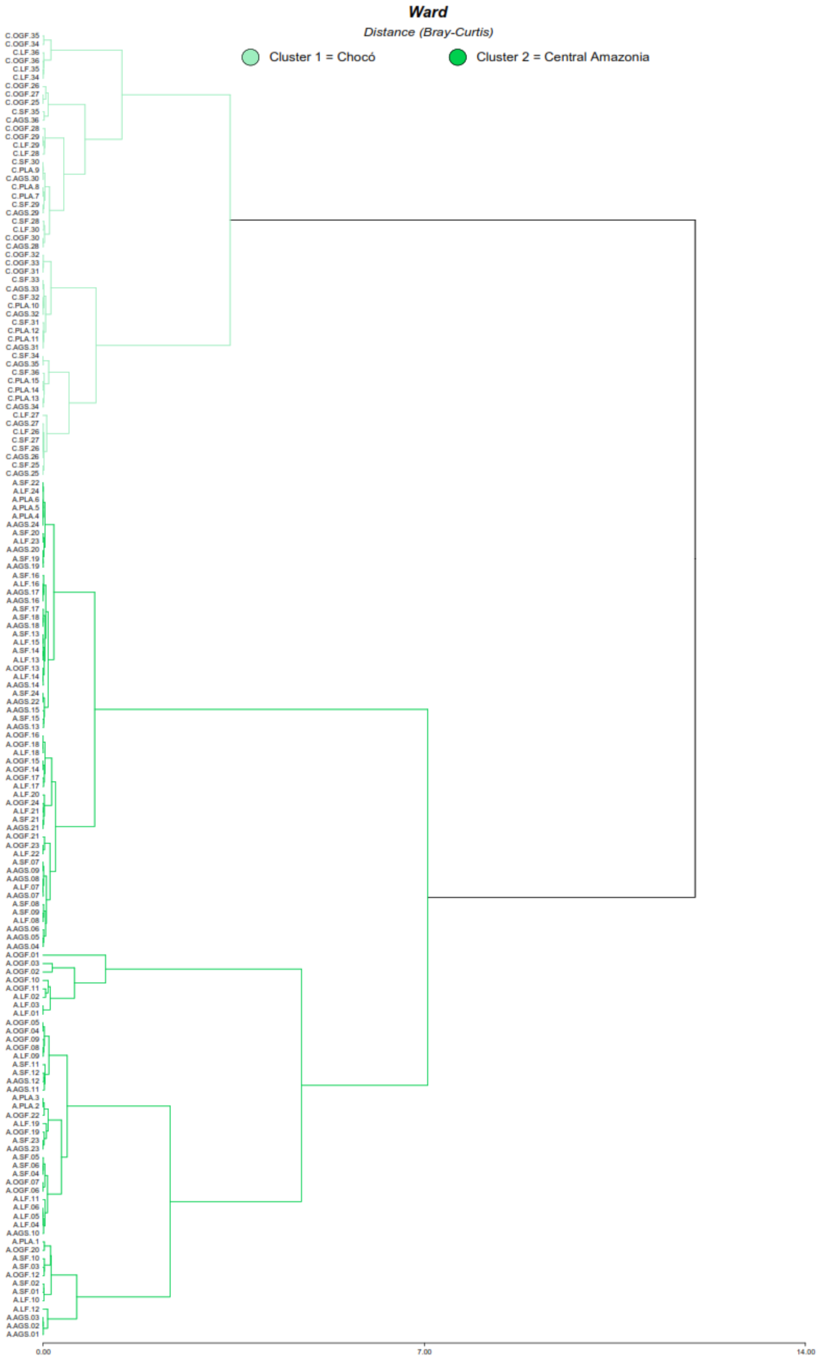

Figure A2. Cluster analysis based on the first two components resultant from the PCA. OGL: old-growth forest. LF: logged forest. SF: successional forest. AGS: agroforestry systems. PLA: plantation.

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Appendix 3. Publication 2

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Article

Degradation of Ecosystem Services and Deforestation in Landscapes With and Without Incentive-Based Forest Conservation in the Ecuadorian Amazon

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Abstract: Anthropogenic activities such as logging or forest conversion into agricultural lands are affecting Ecuadorian Amazon forests. To foster private and communal conservation activities an economic incentive-based conservation program (IFC) called Socio Bosque was established. Existing analyses related to conservation strategies are mainly focused on deforestation; while degradation and the role of IFC to safeguard ecosystem services are still scarce. Further on, there is a lack of landscape-level studies taking into account potential side effects of IFC on different forest types. Therefore we assessed ecosystem services (carbon stocks, timber volume) and species richness in landscapes with and without IFC. Additionally, we evaluated potential side-effects of IFC in adjacent forest types; hypothesizing potential leakage effects of IFC. Finally, we tested if deforestation rates decreased after IFC implementation. Forest inventories were conducted in 72 plots across eight landscapes in the Ecuadorian Central Amazon with and without IFC. Plots were randomly selected within three forest types (old-growth, logged and successional forests). In each plot all individuals with a diameter at breast height greater than 10 cm were measured. Old-growth forests in general showed higher carbon stocks, timber volume and species richness, and no significant differences between old-growth forests in IFC and non-IFC landscapes were found. Logged forests had 32% less above-ground carbon (AGC) and timber volume in comparison to old-growth forests. Surprisingly, logged forests near IFC presented higher AGC stocks than logged forests in non-IFC landscapes, indicating positive side-effects of IFC. Successional forests contain 56% to 64% of AGC, total carbon and timber volume, in comparison to old-growth forests, and 82% to 87% in comparison to logged forests. Therefore, successional forests could play an important role for restoration and should receive more attention in national climate change policies. Finally, after IFC implementation deforestation rate decreased on parish level. Our study presents scientific evidence of IFC contribution to conserving ecosystem services and species richness. In addition IFC could help indirectly to reduce degradation effects attributed to logging, indicating potential compatibility of conservation aims with forest activities at a landscape level.

Keywords: carbon stocks; timber volume; species richness; forest integrity; Socio Bosque program

1. Introduction

Tropical forests provide a wide range of ecosystem services, including supporting, provisioning, regulating and cultural services [1]. Whereas at local level provisioning services such as food, timber and medicine are of high importance for single land users and communities, regulating services related to climate regulation and carbon sequestration play a central role at the global level [1]. Tropical ecosystems can store between 208 Gt C and 288 Gt C of carbon in above and below ground biomass [2–4]. Latin America's forests hold 49% of the total carbon biomass in the tropics [4]. Additionally, tropical forest ecosystems are considered hotspots of biodiversity; it is estimated that between 50–80% of all terrestrial species are located in these ecosystems [5,6].

Anthropogenic activities have affected tropical forests in terms of degradation, forest cover loss and fragmentation [7,8]. Deforestation and forest degradation contributed between 10% and 20% of global anthropogenic greenhouse emissions between 1990–2005 [4,9]. Globally the annual rate of net forest loss has declined from 7.3 M ha year^{-1} in the 1990s to 3.3 M ha year^{-1} between 2010 and 2015. This decreasing tendency has been also observed in tropical forests, with a decline from 9.5 (1990s) M ha year^{-1} to 5.5 M ha year^{-1} (2010–2015) [10]. Despite a recent trend of reduced deforestation rates in the tropics, the annual loss is still remarkably high, and an unknown amount of carbon is possibly being emitted by degradation rather than by deforestation processes [11,12].

The Amazon Basin and Ecuadorian forest ecosystems have experienced similar dynamics since 1990. Deforestation in the Amazon Basin was specifically high with an annual rate of net deforestation of −0.45% for the 1990s, whereas annual deforestation for Ecuador was −0.65% during the same period [2,13]. Although for Ecuador the annual rate of net deforestation has decreased to −0.48% for the period 2014–2016, it remains high in contrast to other countries of the region [13,14]. The main drivers that lead to deforestation are road construction, oil extraction, land use change from forests to agricultural crops and grasslands, and finally a growing population density in formerly forested areas [15–18].

Besides the negative effect of deforestation, it is important to highlight the impact of forest degradation on ecosystem integrity by reducing the availability of goods and services; such as carbon or water regulation [19–22] and biodiversity [23–28], due to species extinction, loss of natural habitats and changes in species distribution [29]. In Latin America, around 70% of forest degradation is driven by logging activities, followed by firewood, charcoal extraction and forest fires [30,31]. The intensity of harvesting activities has a direct influence on forest structure, which is related to the reduction of carbon stocks, timber volume and species composition through vegetation loss and tree damage [32–34]. A logging intensity of 4.5 to 5.7 trees ha^{-1} could result in a reduction of aboveground stand biomass by 17% to 26%; meanwhile, a high logging intensity of 10 trees ha^{-1} reduced biomass by 48% [33,35]. According to Rutishauser et al. [36], when forests lose between 10% to 25% of above carbon stocks due to logging, forest recovery will take among 12 to 43 years, which shows the significant impact of logging activities.

Due to the anthropogenic impact on forest ecosystems in recent decades, there has been a continuous debate on what kind of conservation strategies would provide a better solution for ecosystem services' conservation. It has been shown that governmental conservation strategies (e.g., protected areas) can contribute to the protection of tropical ecosystems by reducing deforestation [37–39]. These areas have expanded around the tropics and today represent almost 27% of the forest area [40,41]. In Ecuador, protected areas represent almost 20% of the territory [42]. However, due to use restrictions, protected areas do not always generate benefits for local communities [43–45]. Therefore, strategies based on incentives for forest conservation (IFC) can be more attractive to local people as they are assumed to contribute to alternative income and poverty alleviation. Such strategies could also be an adequate way of combining ecosystem conservation and the provision of natural resources [46–48]. This is key since local communities play an important role in the management and conservation of natural resources. For example, in Ecuador it is estimated that between 5 and 7.5 M ha of forest belongs to indigenous communities [49,50]. This emphasizes how important these actors are for land use decisions. On the other hand, community land forest traditionally aiming to preserve forest lands for future generations, has as well shown promising results for conservation. Some studies indicate that forest loss can be lower in these cases or even similar to protected areas [51–53]. In Ecuador, Jones et al. [54] and Mohebalian and Aguilar [55], found that areas with IFC (Socio Bosque program) in the Amazon region are less deforested than areas that are not part of the program. However, Sánchez-Azofeifa et al. [56] and Pfaff et al. [57] found that areas with IFC do not have lower deforestation rates than other areas.

Many studies analyzing conservation measures such as IFC or protected areas have been focused on assessing effects of deforestation [51–56,58,59]. However information on forest cover change does not take into account forest integrity parameters in order to measure potential degradation effects of ecosystem services. In our study we incorporated both into our assessment since we evaluated deforestation after the implementation of IFC and the potential degradation effects. Forest degradation is defined as the reduction of goods and ecosystems services (e.g., timber or carbon) due to anthropogenic disturbances [60–62]. As indicators of forest degradation we used carbon stocks, total tree volume and species richness. Species richness was selected as a basic indicator related to forest ecosystem composition and function [23,25–28]. Due to the fact that IFC poses restrictions of access and use of protected old-growth forests, it is possible that anthropogenic pressures could be deviated towards adjacent forests, leading to higher pressure on logged or successional forests. Under this scenario, IFC could create leakage effects at a landscape level, which to our knowledge is an important research gap, and of particular importance for the Ecuadorian Central Amazon region. Therefore, we assessed if IFC has an influence on ecosystem services degradation beyond the areas which are under conservation. This is an important scientific contribution to the land sparing/land sharing debate for tropical lowland ecosystems [63–67]. The results could improve the understanding of compatibility or incompatibility of conservation measures and timber production at a landscape level as basis for multifunctional landscapes.

In our study area logging activities are an important potential driver of degradation, especially when sustainability criteria are not applied in practice. Despite the considerable logging activities that are taking place in the lowland forests of Ecuador, scientific information related to the impacts of timber extraction on ecosystems services is still scarce. For this reason, we analyzed how logging activities affect ecosystem services in landscapes with and without the presence of IFC. While logging is an important driver of degradation in many areas of the world, agricultural use and pasture management are important drivers of deforestation. These activities can lead to the establishment of mosaics of successional forest stages, with rather unknown potential to contribute to forest ecosystem services. In the Central Amazon region, many successional forests result mainly from abandoned pastures; nevertheless, the potential contribution of this forest type has been overlooked. We took into consideration this knowledge gap and we evaluated the contribution of these forests types to the maintenance of ecosystem services in landscapes with and without IFC. The comprehensive consideration of all dominant forest types (old-growth, logged and successional forests) in combination with IFC, allows for a better decision-making process in the context of forest landscape conservation, sustainable use and restoration.

We focused on the Socio Bosque program in Ecuador which is an IFC program that began in 2008 with the aim of conserving forest, reducing deforestation and improving living conditions in local communities [68,69]. Communities that voluntarily join the program sign a 20-year contract with the Ecuadorian government in order to protect their forest (no extractive use being allowed in the forest). In turn, communities receive cash compensations ranging from USD 0.70 to USD 35.00 per ha year^{-1}, depending on the total size of the conserved forest [68–70]. Since the program holds almost 1.6 M ha under long-term protection, Socio Bosque has become an important conservation strategy in Ecuador [69].

We tested five research questions and hypotheses related to forest degradation and deforestation. (1) The IFC-regime has a general positive effect on carbon stocks, timber volume and species richness in forest landscapes across all forest types (old-growth, logged and successional forest). The second and third hypotheses are related to assessing interactions between forest types and landscapes with and without IFC. (2) Old-growth forests under IFC have higher amounts of carbon stocks, timber volume and species richness than old-growth forest in non-IFC landscapes. (3) Logged and successional forests adjacent to old-growth forest under IFC have lower carbon stocks, timber volumes and species richness than logged and successional forest in non-IFC landscapes, hypothesizing potential leakage effects. We assumed that the presence of IFC regimes on old-growth forest might have a negative side-effect causing higher pressure on nearby forest types. (4) Further on, we aim to show how logged and successional forests differ in amounts of carbon, timber volume and species richness in comparison to old-growth forests in order to provide estimates of potential degradation effects. Finally, we hypothesize that: (5) the annual rate of net deforestation before and after the establishment of the Socio Bosque program was different in IFC and non-IFC landscapes at a parish level, assuming that the IFC program contributed to reducing deforestation rates on the landscape level. In IFC landscapes use restrictions only applied to old-growth forests under the Socio Bosque program; while logged and successional forests were not part of the program and had no land use restrictions.

The following sections of the manuscript are divided in four parts: in Section 2 we provide contextual information related to the study area, to the IFC program and to logging procedures in the Central Amazon region of Ecuador; in addition, we describe our data collection methods and statistical analyses. In Sections 3 and 4 we present the results and we discuss the implications of the influence of IFC on the degradation of ecosystem services and species richness; as well as, deforestation. Finally, in Section 5 we present the relevant conclusions of our study.

2. Materials and Methods

2.1. Study Area

The study was conducted in the Central Amazon region of Ecuador, in the provinces of Pastaza, Napo and Orellana (Figure 1). The region is still characterized by high forest cover, representing almost 43% (5.5 M ha) of the total forest area of the country [71]. In addition, it hosts a high level of biodiversity and a species richness of up to 200 tree species per hectare [72,73]. However, the selected provinces also have interesting land use change dynamics which influence forest cover loss, showing an annual net deforestation rate up to −0.87% at a parish level for the period 2014–2016 [13,17,18].

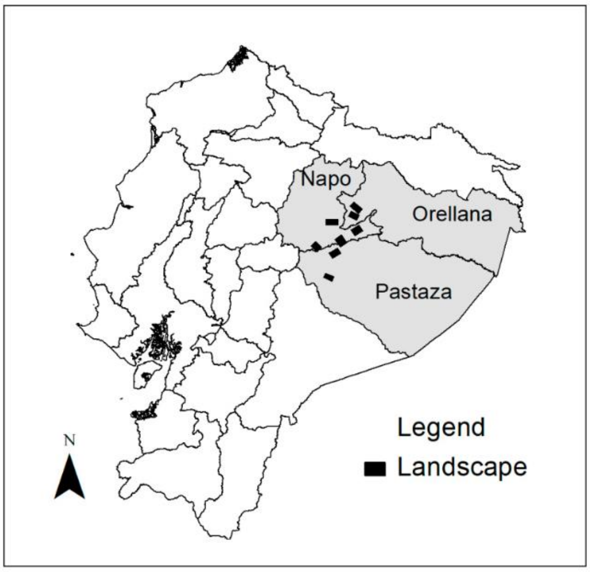


Figure 1. Selected landscapes in Ecuadorian Amazon region.

2.2. Data Collection

2.2.1. Plot Design

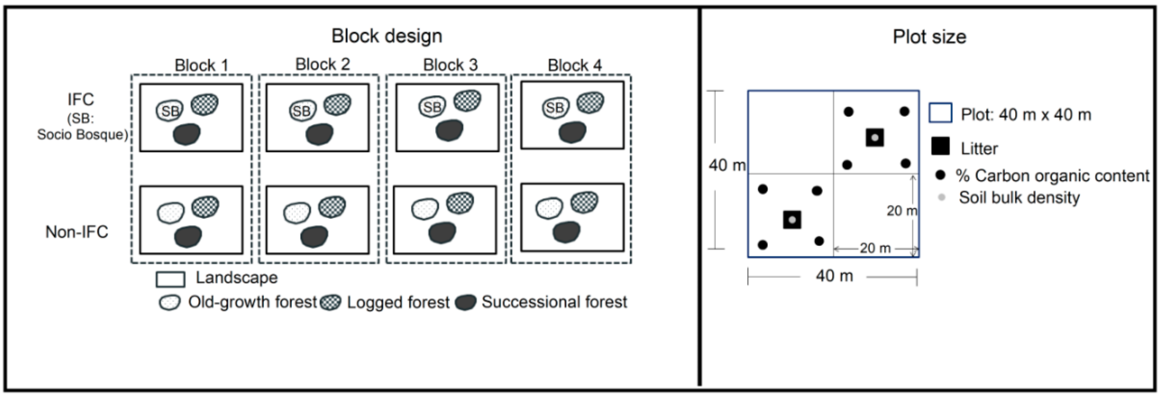
Throughout the three provinces, eight landscapes (sites) of approximately 10 × 10 km were selected (four IFC and four non-IFC landscapes); they were characterized by lowland forests between 335 and 1100 m above sea level and were allocated in different parishes which represent independent administrative units. Since our aim was to evaluate whether IFC have an influence on forest degradation and deforestation, we used a randomized block design. Each block was composed of one landscape where Socio Bosque was stablished (hereafter referred to as IFC landscape) and one landscape without this conservation program (hereafter called non-IFC landscape) (Figure 2a). The Socio Bosque program protects approximately 1.6 M ha through collective and individual contracts, 64% of the conservation area of the program is concentrated in the provinces of Pastaza, Napo and Orellana. Within the three provinces 98.4% of the area conserved by Socio Bosque program is covered by collective contracts (linked to communities) [68–70], which is related to the large extension of community forests in the country (5 to 7.5 M ha of total forest) [49,50]. It is important to highlight that in IFC landscapes, Socio Bosque has been implemented only in old-growth forests. Due to the fact that this is a voluntary program there may be confounding variables that could influence our results. To address this issue our blocked landscapes were randomly selected across the Central Amazon region in Ecuador. Moreover, we controlled potential confounding variables (Tables A1 and A2) such as forest cover, percentage of agricultural land, ecosystem type, altitude, soil type and demographic characteristics (population density, distance from landscape to large cities and distance from households within landscapes to forest). Through analysis of variance we confirmed that all those variables were similar and comparable between landscapes within blocks (Table A3). As a result, the randomized block design used in this study ensured that confounding variables were balanced across the treatment (IFC landscapes/with Socio Bosque) and control (non-IFC landscapes/without Socio Bosque). Consequently, the main difference of degradation effects on ecosystem services and species richness can be attributed to the presence or absence of the IFC program and not to confounding variables.

Each landscape included three forest types. The first type was old-growth forest, which is a mature forest with low or not known human disturbance [74,75]. Secondly, logged forests were considered, which were defined as areas with forest interventions carried out in the last two to four years mainly under the Simplified Forest Harvesting Program (Spanish acronym PAFSI). The PAFSI program is characterized by non-mechanized activities where timber is harvested under a selective logging procedure; after the forest intervention, the area cannot be logged for the following five years. According to records from the Ministry of Environment of Ecuador (MAE), this program is performed in relatively small areas (11 to 24 ha on average) with average intensity of 11–24 m^3 ha^{-1} (standing volume approved by MAE) [76–79]. Logged forests with interventions older than four years were not included in our study, this means that logged forests in our study are comparable in regard to time and that intervention and differences in our target variables can therefore be attributed to differences in logging intensity. The third type of forest was successional forest, resulting from abandoning previously cleared forest (second growth forest) [74,80]. We randomly selected successional areas; this resulted in areas of abandoned pastures between 12 and 28 years of recovery. In order to ensure that the comparison of the successional forests in landscapes with IFC and without IFC is adequate, an analysis of variance was carried out considering the age of the successional forests as a dependent variable, resulting in no statistical differences between the ages of the successional forests near IFC and successional forest in landscapes without the program (Table A4).

In our IFC landscapes, only old-growth forests were under the Socio Bosque program and extractive use is not allowed in these areas. In IFC landscapes, logged and successional forests are located outside the Socio Bosque program and have no land use restrictions; therefore, can be used by their owners. For simplicity and to facilitate the interpretation from now on, logged and successional forests will be called as logged forests near IFC and successional forests near IFC. Old-growth forests in non-IFC landscapes were communal areas with no current use that communities set aside for future needs (e.g., to satisfy land demands from upcoming generations). These were mature forests reported by local people during mapping exercises. However, the absence of previous human interventions in these sites cannot be assured. In order to map the aforementioned forest types, we used secondary information (e.g., land cover maps and PAFSIs records) available from the Ministry of Environment of

Ecuador (MAE), and mapping exercises during participatory community workshops performed in each landscape.

Within landscapes, for each of the three forest strata, three plots were randomly selected using a grid of 200 m × 200 m in order to secure an adequate distribution [81,82]. This resulted in a total of 72 inventory plots: 24 for old-growth forest (12 in IFC and 12 in non-IFC landscapes), 24 in logged areas (12 near IFC old growth forest, 12 in non-IFC landscapes) and 24 in successional forest (12 near IFC old growth forest, 12 in non-IFC landscapes). Plots had a size of 40 m × 40 m (1600 m^2) (Figure 2b). We kept a minimum distance of 100 m between plots and the forest border to avoid edge effects due to anthropogenic activities like agriculture and roads, as well as natural influences like rivers [83,84].



(a) **(b)**

Figure 2. (**a**) Randomized block design for the selection of the study areas. (**b**) Plot design implemented to assess information on carbon stocks, total timber volume and species richness.

We assessed a variety of carbon pools for each plot, including above-ground carbon in trees (AGC_{tree}), above-ground carbon in palms (AGC_{palm}), soil organic carbon (SOC) at a depth of 30 cm [85–87] and dead organic matter (DOM). DOM included downed dead wood (DDW), standing dead wood (SDW) and litter (L). A carbon conversion factor of 0.47 was used [88].

In order to estimate total carbon stocks, we first calculated biomass (Table 1). Above-ground carbon (AGC) was obtained as the sum of AGC_{tree} and AGC_{palm}. AGC_{tree} was calculated applying the equation from Chave et al. [89] (Table 1, Equation (1)). Diameter at breast height (or above the buttresses, if present) of individuals greater than 10 cm and species wood density were used as input parameter for this case. For the wood density of tree species, we used datasets from the Global Wood Density Database [90,91], MAE [92] and Aguirre et al. [93]. For tree species not included in either of these databases, we used the genus average. For a genus represented by a single individual, the average per family was used, and for those representing one family, mean wood density by plot was considered. AGC_{palm} was determined based on Goodman et al. [94] taking into consideration the diameter at breast height (Table 1).

DDW was estimated by volume and three wood density classes (sound: 0.45 gr/cm^3, intermediate: 0.34 gr/cm^3 and rotten: 0.25 gr/cm^3) (Table 1, Equation (3)). Core samples for these classes were extracted in the forest and oven dried in the laboratory to a constant mass at 105 °C, following Chave [95] and Williamson and Wiemann [96]. In the case of SDW, volume and wood density was also considered (Table 1, Equation (4)). For volume, a taper function was used to estimate the minimum diameter at a known height (Table 1, Equation (9)). Mean wood density of all living trees within the plot was used as proxy. All DDW and SDW pieces on the plots (40 × 40 m) were recorded. Litter samples were collected from two subplots of 0.5 × 0.5 m each (Figure 2b). Dry mass was calculated in the laboratory [85–87].

Table 1. Equations for biomass and carbon stock calculation.

Equation		Reference
$AGB_{tree} = \exp\left[\begin{array}{c} -1.803-0.976E+0.9676\ln(\rho) \\ +2.673\ln(DBH)-0.0299\left[\ln(DBH)^2\right] \end{array}\right]$	(1)	Chave et al. [89]
$AGB_{palm} = \exp(-3.3488 + (2.7483 \times \ln(DBH)))$	(2)	Goodman et al. [94]
$DDW_{biomass} = V \times \rho_{class}$	(3)	Pearson et al. [86] Ravindranath and Ostwald [87]
$SDW_{biomass} = V \times \rho$	(4)	Pearson et al. [86] Ravindranath and Ostwald [87]
$L = \text{dry matter}$	(5)	Pearson et al. [86]
$DOM = DDW_{biomass} + SDW_{biomass} + L$	(6)	Pearson et al. [86] Ravindranath and Ostwald [87]
$SOC = BD \times \%CO \times Deep$	(7)	Pearson et al. [86] Ravindranath and Ostwald [87]
$V = \frac{\pi}{4} \times \frac{(D_1^2+D_2^2)}{2} x l$	(8)	FAO [97]
$d_h = 1.59 \times D \times h^{-0.091}$	(9)	Chamber et al. [98]
$\text{Total Biomass} = TAGB + DOM + SOC$	(10)	Pearson et al. [86] Ravindranath and Ostwald [87]

AGB_{tree}: Above-ground carbon in trees. AGB_{palm}: Above-ground carbon in palms. $DDW_{biomass}$: Downed dead wood. $SDW_{biomass}$: Standing dead wood. L: Litter. DOM: Dead organic matter. SOC: Soil organic carbon. TAGB: Total above ground biomass. V: Volume. WD: Wood density. D1: Diameter minimum. D2: Diameter maximum. l: Length. BD: Bulk density. %CO: Carbon organic content. DBH: Diameter at breast height. ρ: Wood density. *E*: environmental stress. h: height. Carbon conversion factor: 0.47 [88].

SOC was estimated from bulk density, the concentration of organic carbon content (%), and the soil depth (0–30 cm). Bulk density was calculated from the oven-dried weight of soil from a known volume of sampled material at 105 °C until reaching a constant weight. In this case, two samples were taken from each of the plots (40 m × 40 m). For concentration of organic carbon content (0–30 cm horizon), samples were taken at the corners of two subplots (Figure 2b). The wet digestion method of Walkley and Black was used for this purpose [86,87,99].

Total tree volume was calculated considering the DBH (≥10 cm), the total tree height and the form factor (0.56) [97]. A total of 668 trees total heights were measured. Log–log linear regression considering total height as the dependent variable and DBH as the independent variable provided the best fit to estimate the height of the remaining trees (Equation (11)).

$$\ln Ht = 0.786 + 0.5956(\ln DBH)\ \left(p < 0.0001;\ R^2 = 0.69\right) \tag{11}$$

where:

Ht = Total tree height;
DBH = Diameter at breast height.

We also determined the species richness (number of species of trees and palms per plot) [100,101] and the importance value index (IVI) (Equation (12)) [100,102] since these variables can be interpreted as an indicator of conservation, ecosystem function and, therefore, diversity-related ecosystem services [23,25–28,103].

$$IVI = DoR + DR + FR \tag{12}$$

where:

IVI = Importance index value;
DoR = Relative dominance;

DR = Relative density;
FR = Relative frequency.

The annual rate of net deforestation was calculated following the procedure specified by MAE [13] comparing the forest area at parish level between 2000–2008 and 2008–2016 and using Equation (13). We used land use and land cover maps from 2000, 2008 and 2016, which can be downloaded from the platform of the interactive environmental map provided by the Ministry of Environment of Ecuador (http://mapainteractivo.ambiente.gob.ec/portal/).

$$\text{Deforestation rate} = \left(\frac{\text{A2}}{\text{A1}}\right)^{1/(\text{t2}-\text{t1})} - 1 \quad (13)$$

where:

A1 = Forest area at the beginning of the period;
A2 = Forest area at the end of the period.

2.2.2. Statistical Analysis

General mixed models were used to test the degradation effects. Analyses of variance (ANOVA LSD Fisher $p \leq 0.05$) were performed considering the fixed effects conservation regime (IFC and non-IFC), forest type (old-growth, logged and succession forest) and their interaction. As random effects, we considered block and landscape. Assumptions of normality and homoscedasticity were evaluated, and criteria penalized likelihood (AIC–BIC) was applied to select the best model for AGC stocks (Mg C·ha^{-1}), DOM (Mg C·ha^{-1}), SOC (Mg C·ha^{-1}), total carbon stocks (Mg C·ha^{-1}), timber volume (m^3 ha^{-1}) and species richness. For AGC, DOM and timber volume outliers were excluded. We considered those plots as outliers which had a value greater than twice the standard deviation from the mean of each one of the forest types.

With regard to the hypothesis of deforestation, ANOVA was performed to check for statistical differences in annual rates of net deforestation between the periods 2000–2008 and 2008–2016 for IFC and non-IFC, respectively. Normality and homoscedasticity assumptions were evaluated.

In addition, cluster analysis taking into account species importance index value (IVI, a measure of ecological importance) [100,102] was applied in order to evaluate the similarity of the forest types between IFC and non-IFC landscapes.

3. Results

3.1. Degradation

Regarding the hypothesis of ecosystem services' degradation, we present analyses of variance for carbon stocks, total tree volume and species richness comparing IFC vs. non-IFC landscapes; as well as forest types and their interaction with and without IFC (Table 2, Figures 3 and 4). In addition, we conducted a similarity analysis considering the IVI per species for the different forest types under IFC and non-IFC (Figure 5).

IFC vs. non-IFC landscapes did not show statistical effects on ecosystem services independently from the forest type in terms of carbon stocks, timber volume and species richness (Table 2). This implies that IFC-regime does not have a general effect on carbon stocks, timber volume and species richness in forest landscapes across all forest types (old-growth, logged and successional forest).

Table 2. Analysis of variance for carbon stocks, timber volume and species richness.

Dependent Variable	IFC vs. non-IFC	Forest Type	Interaction: Forest Types Influenced by IFC and Non-IFC	R^2	*n*
	p-Value	*p*-Value	*p*-Value		
AGC (Mg C·ha^{-1})	0.4624	<0.0001 ***	0.0696 *	0.61	70
DOM (Mg C·ha^{-1})	0.6044	0.0974 *	0.1619	0.12	67
SOC (Mg C·ha^{-1})	0.3738	0.2755	0.1753	0.53	72
Total carbon (Mg C·ha^{-1})	0.6208	<0.0001 ***	0.0459 **	0.66	72
Timber volume (m^3 ha^{-1})	0.4330	<0.0001 ***	0.3498	0.57	68
Richness (#sp./plot)	0.8933	<0.0001 ***	0.0570 *	0.65	72

IFC: incentive-based conservation program AGC: Above ground carbon. DOM: Dead organic matter. SOC: Soil organic carbon. * *p*-value ≤ 0.10. ** *p*-value ≤ 0.05. *** *p*-value ≤ 0.0001.

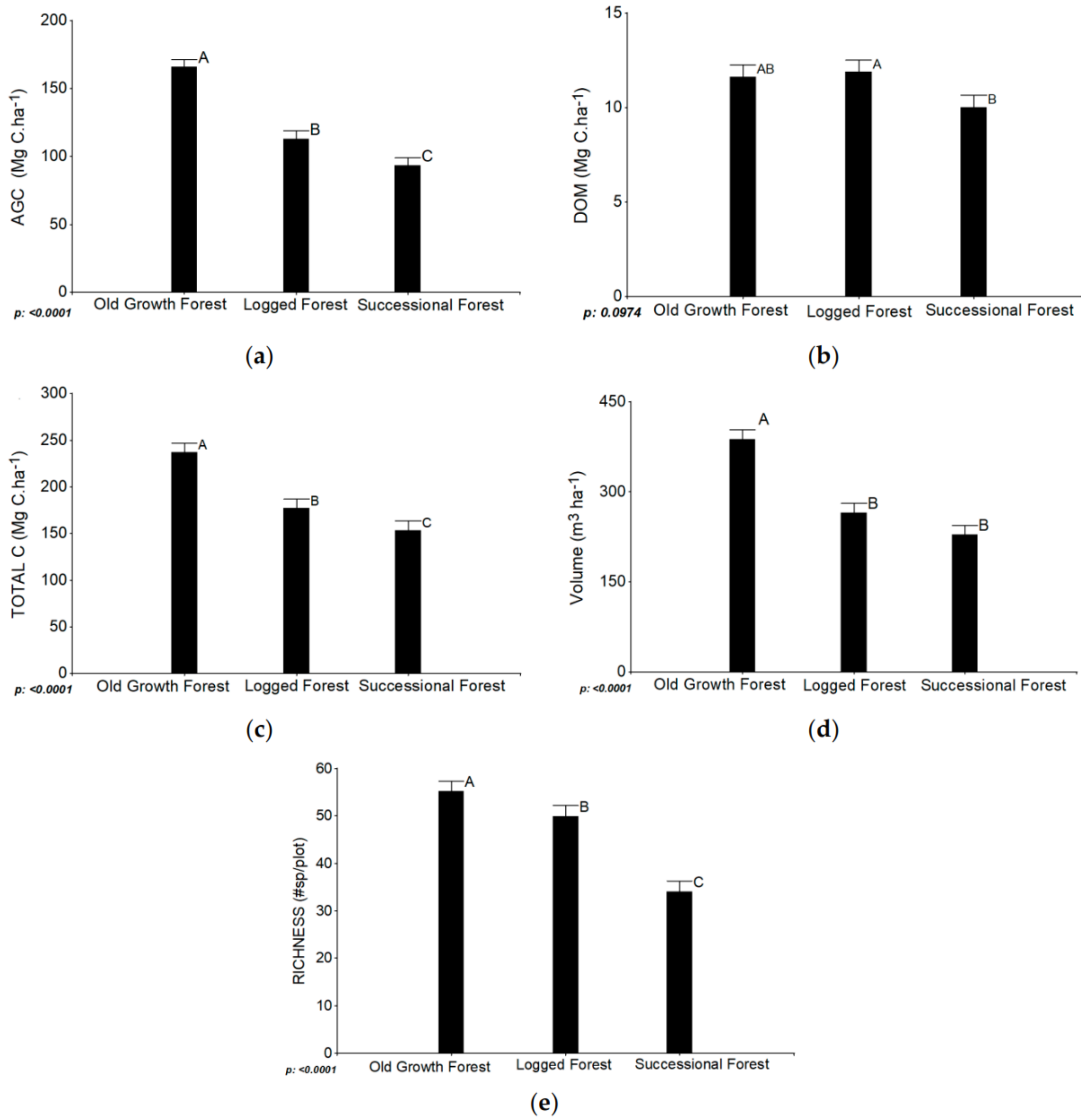


Figure 3. Analysis of variance for carbon stocks, timber volume and species richness between forest types. (**a**) AGC: Above-ground carbon stocks. (**b**) DOM: Dead organic matter. (**c**) Total C: Total carbon stocks. (**d**) Total tree volume. (**e**) Species richness. Different letters (A, B and C) indicate a significant difference from each other ($p \leq 0.05$ or $p \leq 0.10$).

3.1.1. Influence of Forest Types on Ecosystem Services Degradation

In this section, we analyze potential degradation effects of logging activities and the restoration potential of successional forests in comparison to old-growth forests. The following results show means by forest types independent of whether plots were located in IFC or non-IFC landscapes.

There were statistical differences ($p \leq 0.05$) between old-growth, logged and successional forest with respect to AGC, total carbon, timber volume and species richness (Table 2). These results show that there is a degradation of ecosystem services (carbon and volume) and species richness in the logged forest when compared to old-growth forest (Figure 3). In the case of DOM, there is a significant difference at $p \leq 0.10$. SOC did not show significant differences (Table 2). Old-growth forest had the highest mean of AGC with 166 Mg C·ha^{-1}, followed by logged forest with 113 Mg C·ha^{-1} and successional forest with 93 Mg C·ha^{-1}.

Total carbon stocks were about 237 Mg C·ha^{-1} for old-growth forest, 177 Mg C·ha^{-1}, for logged forest and 153 Mg C·ha^{-1} in the successional forest. As expected, timber volume was higher in the old-growth forest with 387 m^3 ha^{-1} as well as statistically different from logged (265 m^3 ha^{-1}) and successional forest (228 m^3 ha^{-1}). DOM across the forest types was between 10–12 Mg C·ha^{-1}, whereas SOC for all forest types ranged between 48–53 Mg C·ha^{-1} (Table A6).

Species richness differed between the evaluated forest types. We found an average of 55 species per plot in old-growth forests, 50 species in logged forests and 34 species in successional forests (Figure 3, Table A6).

3.1.2. Influence of IFC, Non-IFC and Forest Types on Ecosystem Services Degradation

Here we present results of interactions between forest types and landscapes with and without IFC. We analyze whether old-growth forests under IFC have higher amounts of carbon stocks, timber volume and species richness than old-growth forest in non-IFC landscapes. Moreover, we present information on logged and successional forests adjacent to old-growth forest under IFC and logged and successional forest in non-IFC landscapes.

Statistical differences were found for interactions between IFC landscapes, non-IFC landscapes and forest types, namely for AGC ($p \leq 0.10$), total carbon ($p \leq 0.05$) and richness ($p \leq 0.10$) (Table 2, Figure 4). The mean AGC for old-growth forest under IFC was 159 Mg C·ha^{-1}, whereas for old-growth forest with non-IFC it was 172 Mg C·ha^{-1}, showing no statistical difference between them. However, when comparing AGC from different forest types, we found significant differences. An interesting result was that logged forests near areas under IFC (Socio Bosque) had higher amounts of AGC (125 Mg C·ha^{-1}) than logged forest in non-IFC landscapes (101 Mg C·ha^{-1}). Successional forest near IFC and in non-IFC landscapes had the lowest AGC of between 95–91 Mg C·ha^{-1}. Old-growth forest total carbon stocks were similar, having 228 Mg C·ha^{-1} for IFC and 246 Mg C·ha^{-1} for non-IFC landscapes; but, were much higher compared to other forest types. Logged forest near IFC stored 190 Mg C·ha^{-1} and 163 Mg C·ha^{-1} in non-IFC areas; the successional forest had 157 Mg C·ha^{-1} and 149 Mg C·ha^{-1} respectively (Figure 4; Table A7).

No significant differences were found for species richness of old-growth forest under IFC and non-IFC (55 species each per plot), and logged forests in non-IFC landscapes (53 species). In contrast, species richness of logged forests near IFC (46 species) were statistically different from old-growth forests nearby. Surprisingly, these significances did not occur between logged forests and old-growth forests in non-IFC landscapes. Successional forests had the lowest number of species per plot with 36 species in IFC and 31 species in non-IFC landscapes (Figure 4).

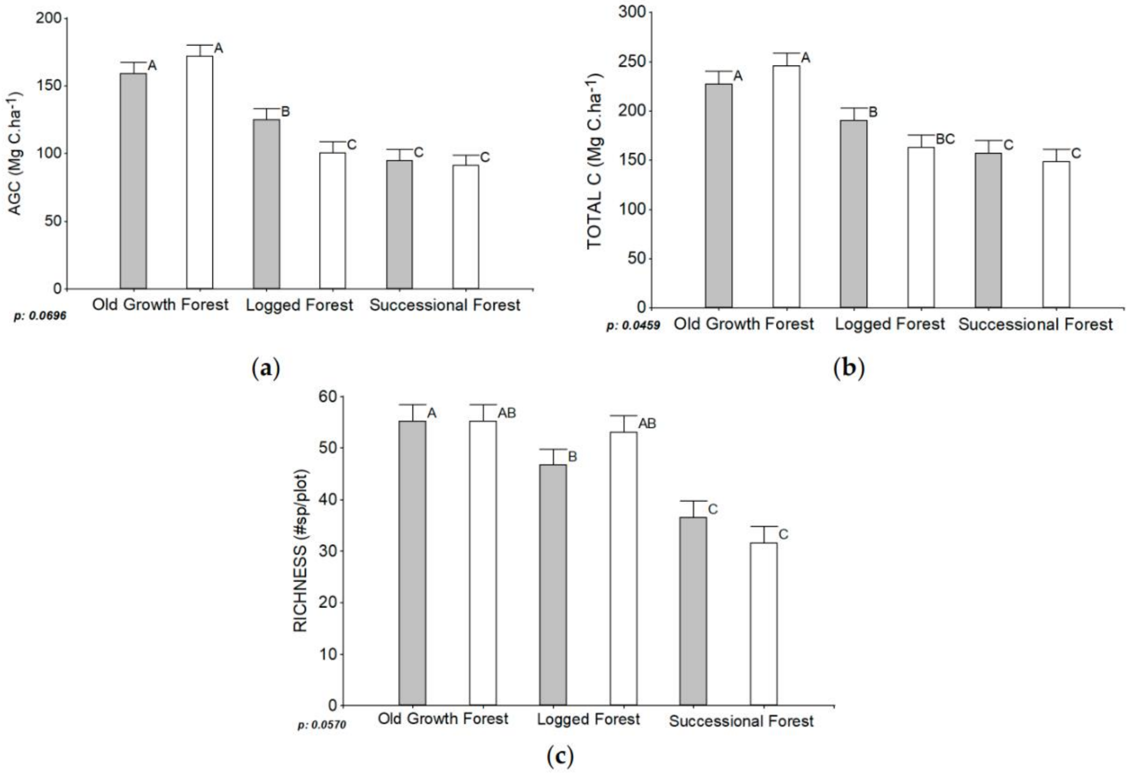


Figure 4. Analysis of variance of carbon stocks and richness considering the interaction between IFC, non-IFC landscapes and forest types. (**a**) AGC: Above-ground carbon stocks. (**b**) Total C: Total carbon stocks. (**c**) Species richness. Different letters indicate significant differences from each other ($p \leq 0.05$ or $p \leq 0.10$). Grey bars represent landscapes with IFC and white bars non-IFC landscapes. Within IFC landscapes only old growth forest is under Socio Bosque program and extractive use is not allowed, in the case of logged and successional forests there are no restrictions and can be used by their owners.

Cluster analysis for species composition separated successional forests (near IFC and non-IFC) from old-growth and logged forests. Old-growth forests under IFC and non-IFC landscapes were grouped together, suggesting that species compositions in these areas are comparable. Species composition in logged forests in areas near IFC forests was more similar to old-growth forests than to logged forests in non-IFC landscapes, implying that there may be less degradation if logging is carried out near areas under IFC (Figure 5).

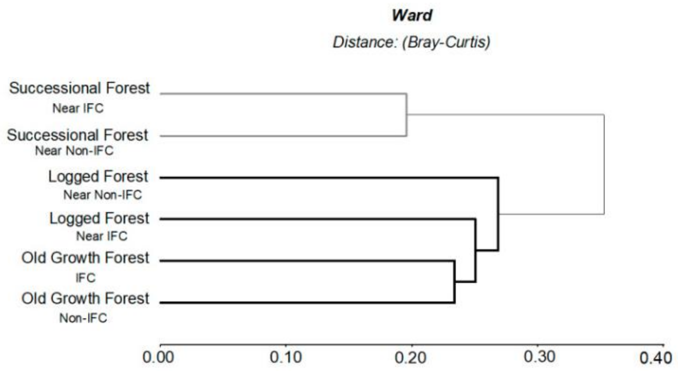
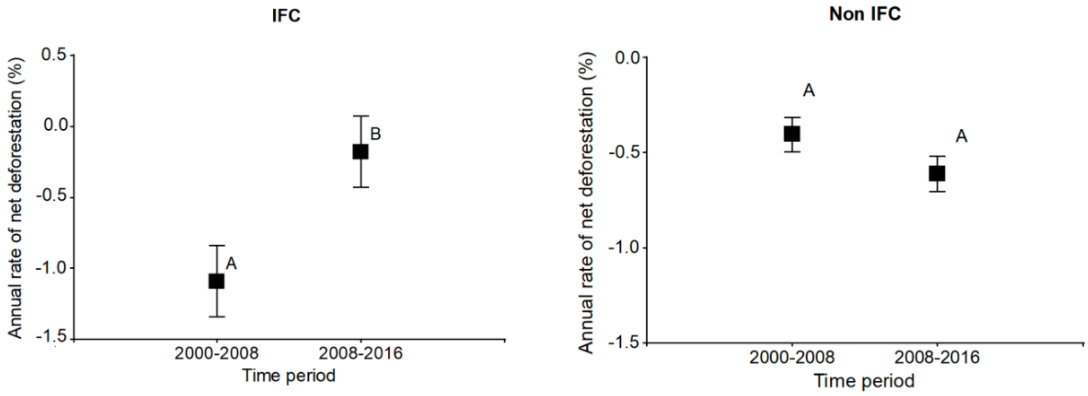


Figure 5. Similarity analysis based on species important index value (IVI) for different forest types in IFC landscapes and non-IFC landscapes.

3.2. Deforestation

The annual rate of net deforestation at the parish level was statistically reduced in parishes where IFC was implemented. Before the implementation of the program, parishes with IFC had an average annual rate of net deforestation of −1.09% (2000–2008). This rate decreased to −0.18% (2008–2016) after the implementation of the IFC program (Figure 6a). In contrast, in parishes without IFC (Figure 6b), annual deforestation was lower in the beginning and increased from −0.41% for the period 2000–2008, to −0.61% for 2008–2016. The increase was, however, not significant.



(a) **(b)**

Figure 6. Analysis of variance of annual net deforestation rate (%) at the parish level. (**a**) IFC landscapes ($p = 0.04$). (**b**) Non-IFC landscapes ($p = 0.15$). Different letters indicate significant differences from each other ($p \leq 0.05$).

4. Discussion

4.1. Degradation

4.1.1. Influence of Forest Types on Ecosystem Services Degradation

The main difference found in our study can be attributed to forest types. Mean total carbon stocks (AGC + DOM + SOC) in old-growth forests were 237 Mg C·ha^{-1}, and for AGC they were 166 Mg C·ha^{-1}. These results are consistent with other studies of tropical lowland forests in the Pan-Amazon region which ranged between 175 Mg C·ha^{-1} and 238 Mg C·ha^{-1} for total carbon stocks [2,4,104] and among 120 Mg C·ha^{-1} and 165 Mg C·ha^{-1} for AGC [2,4,105,106]. With regard to provisioning services, we estimate an average timber volume in old-growth forests of 387 m^3 ha^{-1}, which is within the range of other studies from the Pan-Amazon region reporting between 240 m^3 ha^{-1} and 425 m^3 ha^{-1} [14,34]. Therefore, our results for carbon stocks and volume suggest that the sampled old-growth forests have an adequate conservation status, with carbon and volume levels similar to the ones reported for undisturbed old-growth forest or with minimal impact [104–106].

As reported by Hososuma et al. [30] and Kissinger et al. [31], 70% of the forest degradation effects in Latin America are related to timber extraction, which is consequently the main degradation driver. Our analyses also revealed that logged forests show signs of degradation due to timber extraction, which were reflected in 25% less total carbon stocks, 32% less AGC and timber volume compared with old-growth forests. Logged areas followed mainly the criteria of PAFSI program (selective logging); in this sense, harvest intensity and the extent of damage caused by intervention are important factors that influence the forest's ability to recover and maintain ecological functions [34,107]. According to Sist and Nascimiento [34], only 50% of the commercial stand volume would recover after 30 years, under reduced impact logging (RIL) techniques with an average extraction intensity of 6 trees ha^{-1} and 20% of damage caused by harvest. West et al. [35] compare above-ground biomass (AGB) of conventional logging (CL) and RIL and show that plots under CL lose 26% of AGB while those with RIL lose 17%. They conclude that after 16 years RIL plots recovered by 100% (2.8 Mg ha^{-1} year); whereas CL plots only recovered 77% (0.5 Mg ha^{-1} year). While Rutishauser et al. [36], mention that a forest may require 12, 43 or 75 years of recovery if logging activities reduce 10%, 25% or 50% of the ACG.

Logging activities cause considerable degradation in forest stands and can produce significant changes in the structure. In addition, in the Amazon region logging activities could contribute to land use change [108]. Forest degradation and deforestation are important factors in our study area since in Ecuador 63% of the volume containing high-quality timber that can be logged legally is located in Amazon lowland forests [109]. Therefore, the Amazon lowland forest is attractive for timber harvesting. Considering the substantial degradation of these ecosystems due to logging, it is important to prevent further illegal timber extraction in logged areas. In addition, a cascading use (Cascading use: "is a strategy to use raw materials such as timber or other biomass, in chronologically sequential steps as long, often and efficiently as possible for materials and only to recover energy from them at the end of the product life cycle. It is the intention that the increased cascading use of wood will contribute to more resource efficiency and consequently reduce pressure on the environment" [110]) of timber products can be an adequate strategy for reducing further carbon losses in order to increase forest resource efficiency and reduce pressure on forest ecosystems [110]. Since cascading use is a strategy that requires efficiency within industries for processing timber after harvesting, any planning of large-scale use of Amazon timber potential has to be accompanied by strict control of logging intensities and recovery periods as well as the setting up of an efficient timber processing infrastructure. As a complement thereto, the conservation of old-growth forest areas through IFC programs (Socio Bosque) is necessary. This is particularly relevant to the area that was studied, since Pastaza, Napo and Orellana contain about 5.5 M ha of forest, corresponding to 43% of the total forest area of the country [71,109].

Successional forests in our study areas were between 12 and 28 years. In comparison to the other forest types, they had the lowest values of AGC, total carbon, timber volume and species richness. They were statistically different from old-growth and logged forests (except for volume in logged forest) (Figure 3). According to our results, successional forest could hold between 56–64% of AGC (Mg C·ha^{-1}), total carbon stocks (Mg C·ha^{-1}) and timber volume (m^3 ha^{-1}) in contrast to old-growth forest, and among 82% to 87% in comparison to logged forest. Despite these low amounts (93 Mg C·ha^{-1} average) compared with the other forest types, it is important to highlight the potential of naturally regenerated forests for landscape restoration and carbon sequestration [111], especially in landscapes where logging is carried out in order to compensate carbon losses from forest interventions. For example, if we consider that in the provinces of Pastaza, Napo and Orellana a total of 3597 ha year^{-1} of forest have been regenerated between 2014 and 2016 [13], the potential sequestered carbon for the next three decades would be 334,521 Mg C considering the AGC average of our data (successional forest between 12 to 28 years). It is noteworthy to mention that our successional forests can sequester around 71 Mg C·ha^{-1} in AGC between 12 and 16 years; while after 25 and 28 years of regeneration this amount reaches 140 Mg C·ha^{-1}. Therefore, the potential for carbon stock can be higher if those forests reach a middle age of regeneration, with most likely additional positive side-effects on biodiversity indicators.

Based on our results, successional forests contribute to compensate carbon losses from logging on a landscape scale. Moreover, they can serve as an effective tool for biological corridors to improve connectivity; and in addition, provide goods and services to local communities helping with livelihoods diversification [74,112].

4.1.2. Influence of IFC, Non-IFC and Forest Types on Ecosystem Services Degradation

We found that, in the Ecuadorian Amazon lowland forest, there was no difference in carbon stocks, timber volume and species richness between old-growth forest with IFC and old-growth forest in non-IFC landscapes. Old-growth forests in IFC and non-IFC showed no degradation of ecosystem services since they had carbon stocks and volume contents similar to mature forests with adequate conservation in other Amazonian forests [104–106]. As we note in the previous sections, there is a lack of studies analyzing the effects of conservation policies and degradation in Ecuador. In our extensive search, we found the work of Mohebalian and Aguilar [55] who detected degradation effects in areas not enrolled in the Socio Bosque program, but the differences were statistically not significant. They also found a higher presence of timber species in Socio Bosque areas than in other areas, which indicates harvesting activities in non-Socio Bosque lands. However is important to point out that this study was performed in private landholdings, whereas our study was performed in communal lands. In the Central Amazon region (Pastaza, Napo and Orellana) 98% of Socio Bosque program area is under communal tenure, which highlights the importance of our results.

Other studies in the tropics mention that areas without a conservation strategy could have positive outcomes for community forestry compared to state protection. When people have ownership or management rights over ecosystem resources, they will genuinely support conservation objectives in comparison to a situation of restricted access [52,113,114]. Yet, such community protection will only function if embedded in a context where utilization of resources is possible, e.g., in nearby designated areas [115,116]. Bray et al. [51], Naughton-Treves et al. [117] and Porter et al. [53] found that forest loss on communal lands was lower or equal to that in protected areas. Despite the fact that old-growth forests in non-IFC landscapes do not show evidence of degradation, it is important to consider that the Ecuadorian Central Amazon currently has a low population density [118], and households still manage a considerable amount of land, so they resort to their own forests in order to satisfy the demand for forest products. Nevertheless, as soon as the resources in private lands are depleted, it is likely that the remaining forests, especially those without any protection status, will be more vulnerable. In that context, IFC can become more important for maintaining forest integrity, especially when forest pressure increases in the future. Given the fact that currently there are still areas of old-growth forest which have not been protected, the importance of policies aiming to conserve or otherwise manage forests sustainably is significant. Such policies can take the form of programs like Socio Bosque, governmental or communal protected areas, or any other strategy.

Our study identified an unexpected positive effect for logged forests near IFC forests, as these forests had higher AGC stocks (125 Mg C·ha^{-1}) than logged forests in non-IFC landscapes (101 Mg C·ha^{-1}). When these forests were compared with old-growth forests, we estimated a reduction in AGC of 21% in logged forests near IFC and 41% in logged forests in non-IFC landscapes. Such findings are surprising and suggest that the presence of a conservation program in the area has effects on how logging is performed. The adequate conservation of old-growth forest, as well as less degradation of logged forests on IFC landscapes, can be related to a greater presence of governmental representatives who are negotiating, monitoring and controlling IFC areas. Consequently, people living close to those areas could feel more pressure to follow the forest regulations, since there is a higher probability of being caught breaking the law than in non-IFC landscapes, where government presence could be more scarce. Jones et al. [54] reported that people living around Socio Bosque areas perceive more restrictions and a fast response by environmental authorities when illegal activity occurs. Our results show that is possible to balance and integrate forest conservation and timber production [66]. Nevertheless, it is necessary to implement improved forest management procedures, like reduced impact logging (RIL), which may result in lower forest degradation. Runting et al. [66] show that extreme land use sharing or extreme land use sparing in forest landscapes is not necessarily the optimal solution. At landscape level, mixed land use with both approaches is probably the better option to guarantee conservation of ecosystem services [66,67].

Species richness is an important attribute that can influence ecosystems services and also reflects anthropogenic impact. Studies show a positive relationship between species richness and ecosystem services [23,25–28]. The number of species per plot that we found in old-growth forests under IFC versus non-IFC was the same. Cluster analyses showed that they were also similar in composition, suggesting that there is no species loss when we compare old-growth forests across landscapes. In contrast, Shahabuddin and Rao [114] mentioned that areas without conservation strategies "fall short of the needs of comprehensive biological conservation," implying that state protection is more effective and that communal forest conserves an altered species composition and tends to lose species, often those with the highest conservation value.

Logged forests had a similar number of species per plot in comparison with old-growth forests, but, as expected, they showed a change in species composition due to human intervention. The surprisingly high values of species richness in logged forests located in non-IFC landscapes could thus be attributed to a post-logging stimulation of pioneer species which had been suppressed before and reached the DBH threshold of this study after logging interventions [119–121]. Our results on species composition confirm this finding and are in line with results of stronger logging disturbances in forests without the presence of IFC.

As expected, successional forests near IFC and in non-IFC landscapes showed the lowest carbon stocks, timber volume, and species richness; in addition, they presented a different species composition (Figure 5) in contrast to old-growth forest and logged forest. Nevertheless, due to increase in population growth, deforestation and land abandonment, successional forest can become more important in the tropics [122,123]. Therefore, this forest type will play a relevant role and should be included in conservation strategies or incentives programs since they can contribute to landscape restoration and provision of ecosystems services, and in addition help to support biodiversity of old-growth forest [124–126].

4.2. Deforestation

The parishes in this study showed two trends: (i) in parishes influenced by IFC, the deforestation decreased from −1.09% in 2000–2008 to −0.18% in 2008–2016, implying a significant (p = 0.04) and positive change of deforestation. (ii) The parishes without IFC, on the other hand, showed an increase of forest loss changing from −0.41% in 2000–2008 to −0.61% for 2008–2016, but the change was not statistically different between the periods (p = 0.15); nevertheless, it is worth emphasizing that deforestation has increased in the absence of a formal conservation strategy.

This is in line with studies carried out in other tropical regions; for example, Bruner et al. [38] assessed 93 protected areas across the tropics and found that only 17% of the areas had a net forest clearance since their establishment. Nagendra [39] evaluated 49 locations where protected areas had a lower rate of clearance than in surrounding areas. Similar results were found by Mohebalian and Aguilar [55] regarding areas with Socio Bosque program in Ecuador, according to which these areas are 9% less likely to be deforested. Jones et al. [54] mention that this program can reduce the average annual deforestation rate (forest cover change) in the Ecuadorian Amazon region by 0.4% - 0.5% in contrast to areas without it. At the national level, Cuenca et al. [127] suggest that between 1.5% to 3.4% of the forest might have been deforested in 2014 if Socio Bosque would not have been implemented. Understanding how IFC influences forest conservation with in situ data has important implications for future policies. The relation between IFC and deforestation is assumed to be due to financial compensation and higher governmental presence in controlling illegal activities, which are thus seen as important factors for avoiding deforestation in areas with IFC programs [54]. However, incentives do not always have a major effect on the reduction of forest cover loss; Sanchez-Azofeifa et al. [56] and Pfaff et al. [57] found that payments for ecosystem services in Costa Rica have little effect on declining deforestation rates between 1997–2000. Though, it is possible that there was an influence of conservation policies implemented previously, or that these areas were not under high deforestation pressure [1].

5. Conclusions

The main focus of recent research has been the evaluation of the effect of IFC on forest cover loss. Though this kind of analysis provides important information about the effectiveness of the programs in maintaining forest cover, it does not indicate the condition of ecosystems and their functionality as a result of anthropogenic activities. We reduce this knowledge gap through the incorporation of forest degradation aspects into the assessment of ecosystems services and species richness in landscapes with and without IFC. The adequate conservation of ecosystem services in IFC areas and the unexpected positive side-effect of IFC on adjacent forest types, suggests that conservation strategies and logging activities are getting more compatible at a landscape level, without deviating higher pressure (leakage effect) to adjacent forest types. Additionally, at the parish level, we found a decrease of deforestation rates after the implementation of the IFC program. People living around Socio Bosque areas were probably restrained from illegal logging, overexploitation or land use conversion in adjacent areas due to a greater presence of governmental representatives or higher environmental awareness of the society. Our results are supporting evidence for the importance of IFC as an alternative conservation strategy to maintain forest integrity. Since the IFC program is a main step to articulate forest conservation with REDD+ schemes (Reducing Emissions from Deforestation and Forest Degradation), our results are very pertinent for the actual conservation debates in Ecuador.

In this study, we also considered two important drivers of deforestation and degradation that affect ecosystem services and species richness: (i) timber extraction and (ii) conversion of old-growth forests into pastures or agricultural lands. There is evidence of a significant reduction of carbon stocks and timber volumes as a consequence of logging activities in comparison to old-growth forests. However the carbon storage is significantly higher than successional forests resulting from abandoned pastures or agricultural lands. Sustainable forest management can consequently be considered as important tool for climate change mitigation where the agricultural frontier puts old growth forests under risk of conversion. Long term studies are needed to evaluate the effect of logging intensities and frequency, in order to find an optimal balance between climate change mitigation potential and financial returns of forest interventions. The considerable carbon stocks of successional forests on abandoned lands could make them a promising but overlooked instrument for restoration programs. They could help to offset the negative impacts of degradation (logging) and forest cover loss (deforestation) on a landscape level and therefore should be incorporated into the REDD+ strategies as part of the climate change mitigation efforts.

Finally, considering the enormous potential of timber and other natural resources in the Amazon, there is an urgent need for land use planning. In this sense, the incorporation of landscape approaches can serve as an integral tool to prioritize areas for conservation, sustainable forest management and restoration.

Author Contributions: Conceptualization: P.E. and S.G.; Methodology: P.E. and S.G.; Formal analysis: P.E.; Investigation: P.E.; Data curation: P.E.; Writing—original draft preparation: P.E.; Writing—review and editing; P.E., R.F. and S.G; Visualization: P.E.; Supervision: S.G.; Project administration: S.G., P.E. and R.F.; Funding acquisition: S.G., R.F. and P.E.

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Conflicts of Interest: The authors declare no conflict of interest.

Appendix A

Table A1. Environmental variables considered for the block design.

Landscape	Environmental Variables								
	Forest Cover [1] (%)	Agricultural Land [1] (%)	Ecosystem [2]	Soil Type [3]	Soil Texture [4]	Mean Distance from Plots to Roads (m)	Altitude m a.s.l. (mean)	Min.	Max
IFC-1 (block 1)	71.52	27.31	tropical lowland forest	Inceptisol/Andisol	medium–moderately coarse	3328.86	887.9	560.0	1180.0
Non-IFC-1 (block 1)	57.6	39.32	tropical lowland forest	Inceptisol/Andisol	medium–moderately coarse	1087.61	667.0	479.0	1049.0
IFC-2 (block 2)	72.62	25.77	tropical lowland forest	Inceptisol/Andisol	fine	1312.41	525.6	403.0	758.0
Non-IFC-2 (block 2)	81.84	16.77	tropical lowland forest	Inceptisol/Andisol	fine–medium	1302.99	568.6	403.0	1028.0
IFC-3 (block 3)	65.03	28.59	tropical lowland forest	Inceptisol	fine–medium	1487.63	429.4	348.0	688.0
Non-IFC-3 (block 3)	50.17	43.13	tropical lowland forest	Inceptisol	fine–medium	826.91	372.2	299.0	600.0
IFC-4 (block 4)	61.83	34.21	tropical lowland forest	Inceptisol/Andisol	fine–medium	988.60	552.7	331.0	819.0
Non-IFC-4 (block 4)	49.37	48.16	tropical lowland forest	Inceptisol/Andisol	fine–medium	1181.18	451.3	315.0	654.0

Source: [1] MAE [128]; [2] MAE and FAO [109]; [3] SIGTIERRAS [129]; [4] MAGAP [130].

Table A2. Demographic variables considered for the block design.

Demographic Variables (Socio-Economic)			
Landscape	Population Density (hab. per km^2) [1]	Mean Distance from Households within Landscapes to Forest (km) [2]	Distance from Landscape to Large Cities (km) [2]
IFC-1 (block 1)	12.68	2.82	24.00
Non-IFC-1 (block 1)	9.44	2.65	30.42
IFC-2 (block 2)	9.00	2.38	46.83
Non-IFC-2 (block 2)	9.33	2.03	60.50
IFC-3 (block 3)	18.09	2.05	35.84
Non-IFC-3 (block 3)	21.24	1.43	58.24
IFC-4 (block 4)	24.22	0.83	10.20
Non-IFC-4 (block 4)	29.77	1.12	16.80

[1] INEC [118]; [2] Based on own information gathered through 673 surveys and eight community workshops (one per landscape) within the eight landscapes comprised by the LAFORET Project, Thünen Institute of International Forestry and Forest Economics, Germany.

Table A3. Analysis of variance between IFC and Non-IFC landscapes considering cofounding variables.

Factor	IFC vs. non-IFC		
	IFC Mean (std.err)	Non-IFC Mean (std.err)	*p*-Value
Forest (%)	67.75 (5.67)	59.75 (5.67)	0.3569
Agricultural land (%)	28.97 (5.07)	36.85 (5.07)	0.3143
Mean Altitude m a.s.l.	598.90 (84.21)	514.78 (84.21)	0.5064
Mean Distance from plots to roads (m)	1779.38 (379.25)	1099.67 (379.25)	0.2520
Population Density (hab. per km^2)	16.00 (4.22)	17.45 (4.22)	0.8166
Distance from households within the landscapes to forest (km)	2.02 (0.39)	1.81 (0.39)	0.7099
Distance from landscape to large cities (km)	29.22 (9.39)	41.49 (9.39)	0.3911

Table A4. Analysis of variance of successional forest age between IFC and Non-IFC landscapes.

Variable	IFC (mean/Std. err.)	Non-IFC (mean/Std. err.)	*p*-Value
Successional forest age	20 (1.26)	19 (1.26)	0.6439

Table A5. Mean values and standard error of AGC, DOM, SOC, total carbon, timber volume and species richness for landscapes with IFC and Non-IFC.

Variable	IFC	S.E.	Non-IFC	S.E.
AGC (Mg C·ha^{-1})	126.62	4.81	121.23	4.93
DOM (Mg C·ha^{-1})	11.37	0.54	10.94	0.52
SOC (Mg C·ha^{-1})	51.44	5.31	48.70	5.31
Total carbon (Mg C·ha^{-1})	191.77	10.43	185.97	10.43
Volume (m^3 ha^{-1})	303.39	15.65	283.23	15.94
Richness (#species/plot)	46.11	2.56	46.64	2.56

AGC: Above ground carbon. DOM: Dead organic matter. SOC: Soil organic carbon.

Table A6. Mean values and standard error of AGC, DOM, SOC, total carbon, timber volume and species richness for forest types.

Variable	Old Growth Forest	S.E.	Logged Forest	S.E.	Successional Forest	S.E.
AGC (Mg C·ha^{-1})	165.63	5.87	112.97	5.87	93.17	5.75
DOM (Mg C·ha^{-1})	11.60	0.67	11.86	0,64	10.01	0.64
SOC (Mg C·ha^{-1})	52.57	5.39	48.19	5.39	49.45	5.39
Total carbon (Mg C·ha^{-1})	236.85	10.40	176.75	10.40	153.01	10.40
Volume (m^3 ha^{-1})	387.29	16.09	264.78	16.09	227.85	16.42
Richness (#species/plot)	55.17	2.26	49.92	2.26	34.04	2.26

AGC: Above ground carbon. DOM: Dead organic matter. SOC: Soil organic carbon.

Table A7. Mean values and standard error for AGC, DOM, SOC, total carbon, timber volume and species richness for IFC, non-IFC and forest types.

Variable	Old-Growth Forest with IFC	S.E.	Old-Growth Forest in Non-IFC Landscapes	S.E.	Logged Forest Near IFC	S.E.	Logged Forest in Non-IFC Landscapes	S.E.	Successional Forest Near IFC	S.E.	Successional Forest in Non-IFC Landscapes	S.E.
AGC ($Mg\ C{\cdot}ha^{-1}$)	159.36	7.94	171.90	8.28	125.37	7.94	100.57	8.28	95.13	7.94	91.20	7.94
DOM ($Mg\ C{\cdot}ha^{-1}$)	12.02	0.97	11.18	0.93	12.85	0.93	10.88	0.89	9.26	0.93	10.76	0.89
SOC ($Mg\ C{\cdot}ha^{-1}$)	50.91	5.78	54.24	5.78	51.26	5.78	45.12	5.78	52.16	5.78	46.75	5.78
Total carbon ($Mg\ C{\cdot}ha^{-1}$)	227.60	12.78	246.10	12.78	190.39	12.78	163.10	12.78	157.32	12.78	148.71	12.78
Volume ($m^3\ ha^{-1}$)	380.28	22.35	394.30	23.16	283.62	23.16	245.95	22.3	246.28	22.35	209.42	24.07
Richness (#species/plot)	55.17	3.19	55.17	3.19	46.67	3.19	53.17	3.19	36.50	3.19	31.58	3.19

AGC: Above ground carbon. DOM: Dead organic matter. SOC: Soil organic carbon.

Table A8. Annual rate of deforestation at a parish level for areas influence by IFC and Non-IFC between 2000–2008 and 2008–2016.

IFC–Non-IFC	Annual net Deforestation Rate (%) 2000–2008	Annual net Deforestation Rate (%) 2008–2016	Tendency ↓↑
IFC 1	−0.77	0.78	↓
Non-IFC 1	−0.36	−0.49	↑
IFC 2	−1.23	−0.33	↓
Non-IFC 2	−0.55	−0.68	↑
IFC 3	−1.31	−0.41	↓
Non-IFC 3	−0.53	−0.43	↓
IFC 4	−1.06	−0.76	↓
Non-IFC 4	−0.18	−0.85	↑

↓ Decrease of the annual net deforestation rate between periods 2000–2008 to 2008–2016 at a parish level. ↑ Increase of the annual net deforestation rate between periods 2000–2008 to 2008–2016 at a parish level. Source: SUIA, (2018).

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Appendix 4. Publication 3

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What Drives Household Deforestation Decisions? Insights from the Ecuadorian Lowland Rainforests

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Abstract: Tropical forests, and more concretely, the Amazon Basin and the Chocó-Darién, are highly affected by deforestation activities. Households are the main land-use decision-makers and are key agents for forest conservation and deforestation. Understanding the determinants of deforestation at the household level is critical for conservation policies and sustainable development. We explore the drivers of household deforestation decisions, focusing on the quality of the forest resources (timber volume potential) and the institutional environment (conservation strategies, titling, and governmental grants). Both aspects are hypothesized to influence deforestation, but there is little empirical evidence. We address the following questions: (i) Does timber availability attract more deforestation? (ii) Do conservation strategies (incentive-based programs in the Central Amazon and protected areas in the Chocó-Darién) influence deforestation decisions in household located outside the areas under conservation? (iii) Does the absence of titling increase the odds of a household to deforest? (iv) Can governmental grants for poverty alleviation help in the fight against deforestation? We estimated a logit model, where the dependent variable reflects whether or not a household cleared forest within the farm. As predictors, we included the above variables and controlled by household-specific characteristics. This study was conducted in the Central Amazon and the Chocó-Darién of Ecuador, two major deforestation fronts in the country. We found that timber volume potential is associated with a higher odds of deforesting in the Central Amazon, but with a lower odds in the Chocó-Darién. Although conservation strategies can influence household decisions, the effects are context-dependent. Households near the incentive-based program (Central Amazon) have a lower odds of deforesting, whereas households near a protected area (Chocó-Darién) showed the opposite effect. Titling is also important for deforestation reduction; more attention is needed in the Chocó-Darién where numerous households are living in untitled lands. Finally, governmental grants for poverty alleviation showed the potential to generate positive environmental outcomes.

Keywords: tropical forest; Amazon; Chocó-Darién; Socio Bosque; protected areas; titling; timber; cash transfers

1. Introduction

The world's forest area declined from 4128 M ha in the 1990s to 3999 M ha in 2015 [1], with an annual net forest loss of 4.7 M ha between 2010 and 2020 [2]. South America accounts for 21% of the world's remaining forest [1]. Within South America, the Amazon Basin and the Chocó-Darién are two important ecoregions due to their role in sustaining biodiversity, supplying local and global ecosystem services, and supporting local livelihoods [3–6]. Both regions are, however, highly threatened by deforestation activities. In Ecuador, more than 50% of forests (6.3 M ha) are in the Central Amazon and on the Northern Coast, the latter comprising part of the Chocó-Darién [7,8]. Similar to other tropical areas, deforestation has reduced Ecuadorian forests from 14.6 M ha in 1990 to 12.5 M ha in 2018, with an annual net deforestation rate of −0.46% (58,429 ha) between 2016 and 2018 [7]. Crop expansion, logging activities, cattle ranching, oil palm plantations, mining, and oil concessions are highlighted as the direct drivers of deforestation in the country [9–12]. In Ecuador, the major deforestation hotspots are in the Chocó-Darién and in the Amazon Basin [7,13]. The Chocó-Darién is highly deforested on the Ecuadorian side [13,14] and has the highest deforestation rate in both the entire ecoregion and within South America [7,15]. Compared with the Chocó-Darién, the Central Amazon demonstrates low deforestation rates [7], yet it is nonetheless experiencing a gradual forest decline.

The Ecuadorian government established two major conservation strategies to halt deforestation: protected areas (PAs) and the incentive-based conservation program called Socio Bosque (SBP). PAs dominate conservation strategies around the world [16]. They are a command and control policy based generally on rigorous mechanisms to keep forests and wildlife intact [17,18]. In Ecuador, PAs also stipulate the long-term strict protection of natural ecosystems [19]. They represent the largest conservation strategy in the country, covering around 19% (4.8 M ha) of the continental territory [7,20]. SBP, on the other hand, provides direct monetary transfers to individual and communal landowners who voluntarily agree to conserve their forests under a 20 year contract that is regularly monitored by the government [21,22]. SBP covers 6.3% of the territory (1.6 M ha) [7,23] and is among the ten largest incentive-based conservation programs in the world [24]. Although more than 25% of Ecuadorian forests are under some type of conservation program [7], there is still a significant proportion of forests without protection [25].

In Ecuador, farm households are important agents for the conservation and the conversion of lowland rainforests [26–30]; they are the ones who make most land-use decisions, e.g., area for cultivation or area for conservation [31]. Due to the presence of imperfect markets that characterize developing countries such as Ecuador, land-use decisions adopted by agricultural societies are determined by factors that go beyond the notion of profit maximization, making deforestation not only a market-driven decision [32,33]. In these contexts, land-use decisions reflect the management of the production factors land, labor, and capital in connection with household demographics and exogenous elements that characterize the natural and institutional environment [34].

Generally, there is still a continuing debate on the influences on deforestation decisions at the farm level [33]. Due to the high costs associated with household data collection, few studies have focused on the agents of deforestation [26]. Existing research shows that considerable attention has been devoted to understanding the relationship between household-specific variables and deforestation [33,35–37]. However, the influence of the quality of natural resources and the conservation strategies remain largely understudied [38]. Likewise, little attention has been devoted to the potential relationship between governmental grants aimed at reducing poverty and deforestation [39].

In Ecuador, most deforestation studies have been conducted from the spatial perspective with aggregated information [7,9–15,29,40]. Despite the substantial contribution of such studies, accounting for factors influencing household deforestation decisions in spatial models is challenging [41]. Most of the few deforestation assessments at the household level in Ecuador are located in the Northern Amazon [34,42–45], evidencing the need to include other regions with different contexts. Moreover, these studies date from the 1990s, calling for new empirical evidence that re-evaluates the relationship between household behavior and deforestation [29,46].

In this study, we explore the determinants of household deforestation decisions in the lowland rainforest frontiers. We focus on exogenous elements that reflect the quality of the forest resources (timber volume potential) and the institutional environment (conservation strategies, titling, and governmental grants for poverty alleviation). These aspects have been hypothesized to influence land-use decisions in the forest frontiers [34,45], but with the exception of titling, results from empirical models are still missing [47,48]. We address the following questions: (i) Does timber availability attract more deforestation? (ii) Do conservation strategies (SBP and PAs) influence deforestation decisions in households located outside the areas under conservation? (iii) Does the absence of titling increase the odds of deforestation at the farm level? (iv) Can governmental grants for poverty alleviation help in the fight against deforestation? We estimated a logit model, where the dependent variable indicates whether or not forest was cleared by a household in the farm. As predictors, we considered the variables previously mentioned, and we controlled by household-specific characteristics. This research was conducted in the Central Amazon and the Chocó-Darién of Ecuador, two regions that host lowland rainforests with high biodiversity and a good capacity to supply multiple ecosystem services [3,4,49–51]. These regions undergo contrasting deforestation processes where rural households maximize their welfare in a context of imperfect markets.

By using in-situ information, we can represent the real conditions of households, local markets, and forest resources, capturing with more accuracy the socioeconomic and environmental contexts of our study regions [52]. Exploring the factors that influence household deforestation decisions is crucial for the design and implementation of effective conservation policies harmonized with the local and global development pathways [9,53].

2. Materials and Methods

2.1. Study Region

Our study was conducted in the lowland rainforest frontiers of the Central Amazon (Napo, Pastaza, and Orellana provinces) and the Chocó-Darién (Esmeraldas province) of Ecuador. These two areas are considered biodiversity hotspots [4,5], hold approximately 6.3 M ha of forests, and account for 68% of the legally harvestable timber volume of suitable quality in Ecuador [8]. Despite the biological importance of the Central Amazon and the Chocó-Darién, these regions are highly susceptible to deforestation.

Based on Angelsen and Rudel [54], the selected study areas illustrate two stages of the forest transition. On the one hand, the Chocó-Darién depicts a stage characterized by high deforestation and low forest cover; it is estimated that in 1970 more than 80% of this region was covered by lowland forests, but now, more than 85% of the original forest cover has been lost [40]. Between 2000 and 2008, the Chocó-Darién of Ecuador had an annual net deforestation rate of −1.43%, and it decreased to −0.61% between 2016 and 2018 (Figure 1), yet this region has 2.5 times the level of deforestation as compared to both the Ecuadorian and South American rates [7,55]. In the Chocó-Darién, the proximate drivers of deforestation are commercial logging followed by agricultural expansion and infrastructure extension [27,40]. The Central Amazon, on the other hand, is in the initial stage of the forest transition, with 82% of forest cover [56] and an annual net deforestation rate of −0.21% between 2016 and 2018 (Figure 1). Despite having low deforestation levels when compared to the Chocó-Darién, deforestation in the Central Amazon is slowly increasing. Small-scale agricultural expansion is the most predominant proximate cause of forest loss in this area [10,11].

The study sites were originally inhabited by indigenous people; now, however, settlers also occupy part of the territory. Together indigenous and non-indigenous groups own and manage the land, with indigenous people occupying the largest share of the territory. In the Central Amazon, more than two thirds of interviewed households belong to the Amazonian Kichwas, the largest indigenous population in the Ecuadorian Amazon [57]. In the Chocó-Darién, around 20% of households belong to the indigenous Chachis. The remaining percentages are comprised of settlers or locally called mestizos (mix of Spaniards and Indigenous descendants) and afro-Ecuadorians (descendants of African slaves), who, partly motivated by the land reforms of 1964 and 1973, migrated from the different parts of the country in search of land. Our study regions are characterized by old and stable settlements, with the presence of elementary schools and, in some cases, with basic primary care facilities. On average, in the Central Amazon, interviewed households were established 23 years ago and, in the Chocó-Darién, 17 years ago.

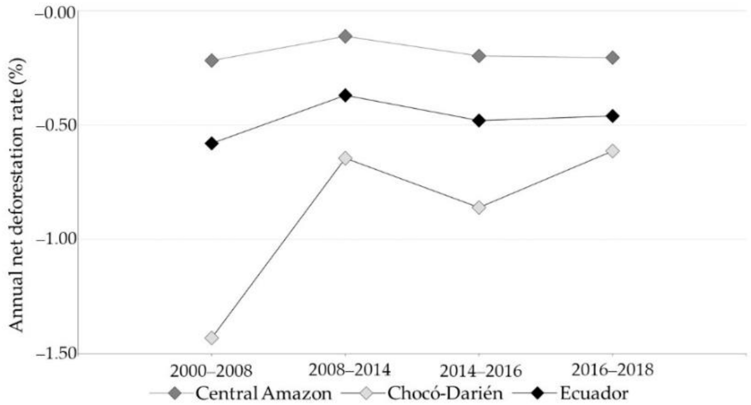

Figure 1. Annual net deforestation rate in the study areas selected. The periods 2000–2008, 2008–2014, 2014–2016, and 2016–2018 correspond to the national deforestation assessments carried out by the Ministry of Environment of Ecuador. Source: authors' own elaboration based on the information available in the Ecuadorian system for environmental information (SUIA by its Spanish acronym) [7].

Most farmers in our study areas are poor or extremely poor. The average annual household income is estimated in USD 4360 in the Central Amazon and USD 6560 in the Chocó-Darién; farm-related activities (mainly crop production) contribute more than 56 and 69% to the household income, off-farm income is between 32 and 25%, and governmental grants among 11 and 6% [58]. Off-farm jobs are often sporadic and many of households from our sample have no off-farm employment opportunities. Governmental grants are probably the only secure cash income a household receives in the course of the year. These grants are monthly cash transfers given by the Ecuadorian government to households under extreme poverty. Households benefiting from this policy are expected to invest the money in health and education in order to reduce chronic malnutrition and preventable diseases in children, and to increase the return-to-school rate and continue education to children and teenagers [59]. Around 68% of interviewed households in the Central Amazon and 56% in the Chocó-Darién benefit from these transfers, which also reflect the high levels of poverty that characterize these areas.

The co-existence of diverse cultures creates a mix between traditional and non-traditional lifestyles. In the past, indigenous people combined crop cultivation, such as cassava and plantain, with fishing, hunting, and gathering wild resources exclusively for domestic consumption. Due to their contact with other groups and their socioeconomic systems, today indigenous' farms also include some cash crops such as cacao [60], while the consumption of wild resources is declining [61]. Settlers also maintain a mix between cash and food crops and some of them own cows [62]. Given the climatic conditions, cultivation in both regions is possible year-round, with family labor as the main input for production [58]. Slash and mulch is a common clearing technique (the felled vegetation is not burned but is left on the ground to decompose); burning is less feasible due to the presence of constant precipitation and humidity [34,63,64]. When clearing the forest, farmers extract trees and sell them to the intermediaries or directly in the local markets [65]. Farm households face several problems with regard to their production and consumption decisions. These include a lack of credit (households that received a credit account for 34% in the Central Amazon and 18% in the Chocó); price and yield fluctuations; inadequate soil management; lack of knowledge; rudimentary technology, and insufficient technical assistance [66,67].

2.2. Sample Selection and Data Collection

We randomly selected 12 sites of approximately 10 × 10 km representing the most characteristic production activities within the regions (Figure 2). In the Central Amazon, we selected eight sites, four of them containing areas under SBP. In the Chocó-Darién, we chose four sites, two of them influenced by a PA. In each site, we conducted face-to-face household surveys to collect socioeconomic data including household socio-demographics, land-use and forest cover change, and the institutional environment. In the 12 sites, we also installed a total of 69 plots of 40 × 40 m in old-growth forests (36 plots) and logged forests (33 plots) to collect data to estimate the timber volume potential used in the regression analysis. Old-growth forests are areas with unknown human disturbance. Logged forests correspond to areas where timber extraction was conducted in the last two to five years; in the Central Amazon, this extraction was under simplified harvesting programs (Programa de Aprovechamiento Forestal Simplificado−PAFSI in Spanish), whereas in the Chocó-Darién was under the so-called sustainable harvesting program (Programa de Aprovechamiento Forestal Sustentable−PAFSU in Spanish).

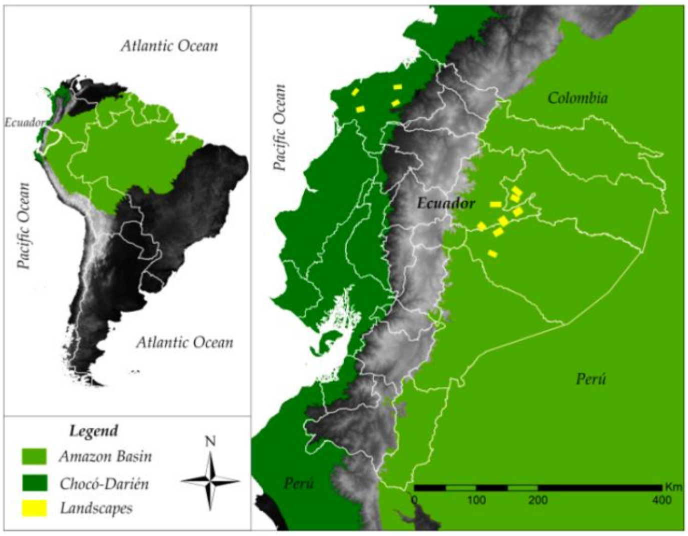

Figure 2. Location of sites selected for our study in the Central Amazon and the Chocó-Darién of Ecuador.

2.3. Econometric Model

Our empirical model departs from the agricultural household theory [68], which has been adapted in several studies to understand household decisions regarding deforestation or forest clearing in rural settings [32,33,36,69]. Farm households in our study sites illustrate what Ellis [70] calls a "peasant economy"; their livelihoods rely highly on agricultural production, and households are partially engaged into imperfect markets [58]. When markets are perfect and well-functioning, production and consumption decisions can be separated and household production decisions, including land-use, can be modeled as a profit-maximizing problem where households maximize their objective function based on net economic gains [68,71,72]. However, in the context of imperfect markets, as is the case of our study sites, households are both producers and consumers of goods, meaning that consumption and production decisions are interdependent, hence non-separable. In the presence of non-separability, the assumption of profit maximization does not entirely hold and the framework of utility maximization serves as a basis to assess households decisions [68,73]. In this context, besides market prices, consumption demands and resource distribution may play a key role in production and land management [73], implying that characteristics related to the household (e.g., age, family size, education) are also relevant to understand land-use decisions and therefore deforestation [74].

Farm households maximize their objective function subject to limited endowments of labor, land, and capital, as well as factors beyond the household control such as the natural capital and the presence of institutions [31,75]. In consequence, under the utility maximization framework, the determinants of deforestation include both household internal and exogenous factors [31,34,45,76,77]. Household-specific factors include land, labor and capital endowments, and demographic characteristics (e.g., family size, age or sex composition, and education). Exogenous factors comprise the natural resource base (e.g., quality of forest resources) and the institutional environment (e.g., policies oriented to improve infrastructure, health and education, and property rights) [34,45]. Studies on deforestation at the household level have focused mainly on the household-specific factors [33]. However, variables related to the natural resource base and the institutional environment are less common in these assessments [78].

With these considerations, the econometric model we used to evaluate the aspects influencing deforestation at household level in the Central Amazon and the Chocó-Darién of Ecuador is a logit model of the following form [79]:

$$\mathrm{logit}(Y) = L_i = \ln\left(\frac{P_i}{1 - P_i}\right) = \alpha + \beta X \tag{1}$$

where Y is the binary response variable, which takes a value of 1 if the household cleared forest and 0 if otherwise (during the survey application, households self-reported whether they had cleared forest within their farm, and this information was used to build our dependent variable); L_i is the log of the odds ratio; Pi is the probability of clearing the forest, and (1 − Pi) is the probability of not converting forest into alternative land-use; α is the Y intercept; β is the regression coefficient, and X is the vector of explanatory variables (Table 1 presents a description of variables used in the model). Our emphasis is to explore whether household decisions to deforest are influenced by the quality of the natural resource base and the institutional environment. These two elements have been previously hypothesized to influence deforestation [41,45,46,72] but have not been explicitly evaluated in the Ecuadorian context.

The quality of the natural resource base can be seen as a "straitjacket" that may exacerbate the constraints (e.g., lack of technology) that farm households deal with. As the quality of natural resources varies from farm to farm, farmers can face distinct constrictions [34,45]. The quality of soils and forest resources are important factors that can motivate farmers to deforest. Since data on soil quality at the appropriate disaggregation level were not available, we used the timber volume potential as a proxy for the quality of the natural resource base. In areas with higher availability of timber, farmers could perceive that forest resources are unlimited and could feel more motivated to deforest [80].

Farmers can also respond in different ways depending on the institutional environment that surrounds them. We focus on three policies within the institutional environment: (i) forest conservation strategies including the incentive-based program SBP and a command and control policy such as PAs; (ii) property rights measured through titling; and (iii) governmental grants for poverty alleviation. In tropical regions, the influence of conservation strategies can vary according to the context and location [81–84]. Negative results have been observed when conservation comprises strict protection or when conservation policies do not contemplate the participation of local people [18,85]. The ongoing deforestation close to conservation areas impacts on the integrity and functionality within conservation areas; therefore, it is necessary to evaluate the role of conservation strategies in the buffer zones [86,87]. More empirical evidence is needed to understand the contexts in which conservation strategies can result in a win–win situation, i.e., less deforestation inside and outside the conservation areas. Policies regarding property rights can also determine the way households perceive forests. Titled lands are associated with long-term decisions, which may include better management of forest resources and more sustainable production systems [88]. Although, no direct connection seems to exist between governmental grants and environmental goals, recent evidence suggests that cash transfers to poor people have the potential to create synergies with deforestation reduction [39]. In developing countries, many poor people live in or near forests and often rely on forest resources to overcome cash and credit limitations and to cope with shocks [89]. Cash transfers for poverty alleviation can influence land-use decisions, for example by reducing the need to extract timber as a means to obtain immediate cash, or by substituting more agricultural expansion to produce food crops [39]. Since, in our study sites, there is a considerable number of households that receive governmental grants (between 56% and 68%), evaluating the relationship with household deforestation decisions is worthwhile.

As control variables, we included household-specific characteristics related to land, labor and capital endowments, and demographic characteristics that have been reported in other studies to determine deforestation decisions. In order to account for the potential effect of households nested within sites, we run our models with robust errors clustered at site level. We examined the presence of multicollinearity through a correlation matrix, and we found that the number of crops, cattle ranching, off-farm jobs, and vehicle access were correlated with important variables in our model. For example, we observed that raising cattle is an activity dominated by non-indigenous families, who on average, have 6.5 cows compared with 0.3 for their indigenous counterparts. The number of crops was positively correlated with timber volume potential, our variable of interest, and with the indigenous group, showing that indigenous households tend to have a more diversified crop production. Off-farm jobs were correlated with distance and farm size; whereas vehicle access was correlated with the conservation strategy. The model presented in the Section 3 has a variance inflation factor (VIF) of 1.29 for the Central Amazon and 1.36 for the Chocó-Darién. According to the literature, VIF values of 5 to 10 suggest moderate multicollinearity, meanwhile when values exceed 10, they correspond to high multicollinearity [90]. The variables included in our model and their expected effects are presented in Table 1.

Table 1. Predictors selected for the logit model to evaluate household deforestation decisions.

Variable	Description	Expected Sign
Household characteristics		
Age of the head of household	Age in years	Older households are less likely to be physically able to clear forest [33], therefore we expect a negative sign.
Indigenous group	0 = the head of household does not belong to an indigenous group 1 = otherwise In the Central Amazon, 1 corresponds to Amazonian Kichwa, whereas in the Chocó-Darién, it corresponds to Chachi.	There are mixed results on this topic. On the one hand, non-indigenous people, "settlers", have been pointed out as agents of deforestation [34,66]; however, some studies mention that indigenous people can also engage in unsustainable practices when they have access to a market economy [31,91,92]. Overall, we expected a negative correlation between indigenous households and deforestation.
Education of the head of household	0 = the head of household completed at least the primary school 1 = the head of household has little or no education	More education increases household consumption and production, leading to more deforestation [34]. In addition, educated people have more opportunities to obtain agricultural loans and access markets, promoting agricultural extension [42].
Number of males in working age	Number of males living in the household between 15 and 65 years.	A higher number of adult males living in the household is related with more deforestation [42,43].
Commercialization rate	Percentage of income coming from the sale of agricultural produce.	Commercially oriented households are more likely to deforest to increase their agricultural lands [33].
Credits	0 = the household received a credit 1 = otherwise	There are mixed results related to the influence of credits; however, literature indicates that people tend to invest credits into activities that provoke more deforestation [41,77].
Physical asset index	Comprise physical assets owned by the household (e.g., car, motorbike, bike, telephone, TV, radio, refrigerator).	Asset-rich households tend to clear more forest since they may have more means to develop expansive agriculture [33,42].
Land endowments		
Farm size	0 = farms with less than 5 ha 1 = farms larger than 5 ha	Having a smaller farm leads to a more intensive use of the soil, which is translated into more forest clearing [34].
Forest area within the farm	Percentage of forest area within the farm boundaries prior to the deforestation occurred in the last five years.	A higher proportion of forest area could give the feeling that forest resources are unlimited, motivating people to consume more forest derived products and to convert more forest area into other uses [80].
Quality of forest resources		
Timber volume potential	Average timber volume between old-growth forests and logged forests in m^3/ha, which can be legally harvested following the minimum cutting diameter specified in the Ecuadorian forest law for each species [93,94]. Its estimation considers the tree height, diameter at breast height, and a form factor of 0.7, as recommended by Segura et al. [95].	There is no empirical evidence on the effect of timber potential on deforestation. Our assumption is that the more timber volume available, the more attractive it is for farmers to deforest.
Natural resources governance		
Land titling	0 = the household does not have land titling 1 = otherwise	Formal tenure is linked with less forest converted to agricultural lands [34,96]
Presence of conservation strategy	In the Central Amazon, four selected sites are influenced by Socio Bosque program (SBP), whereas in the Chocó-Darién, two sites are influenced by a protected area (PA). 0 = no conservation strategy is present in the landscape 1 = otherwise	On the one hand, strict protection with very few participation of local actors, such as PAs, is reported to be insufficient to disincentive deforestation outside the areas under conservation [86]. Conversely, when protection is accompanied by incentives, such as the SBP, people are more engaged in conservation and have higher environmental awareness [22], suggesting less deforestation in neighboring farms. Despite these mixed results, we hypothesized that households near conservation strategies have lower odds to deforest than those with no conservation strategy in their proximities, implying that the presence of a formal conservation instrument, regardless of whether it is command and control or incentive-based, has the potential to reduce pressure on forests beyond the limits of the areas that are under conservation.

Table 1. *Cont.*

Variable	Description	Expected Sign
Governmental grants	0 = no household member receives a cash transfer 1 = at least one person in the house is benefited	Governmental grants tend to relax the constrained budget of poor people, reducing the need to deforest [39].
Infrastructure		
Distance to the forest	Distance in km from the house to the forest plot owned by the household.	Spatial assessments show that more deforestation occurs when forests are close to the house [13]. Our assumption is that higher distances relate to less deforestation.
Distance to market	Distance from the house to the main market in km.	Higher distance to markets reduces the likelihood to deforest [41,43].

3. Results

The results of the econometric analysis showed that the natural resource base and the institutional environment have a significant role on households' decisions to deforest (Table 2).

Table 2. Logit regression results of deforestation at household level for the Central Amazon and the Chocó-Darién of Ecuador.

Variables	Central Amazon				Chocó-Darién			
	Coef.	Robust Std. Err.	*p*	Odds Ratio	Coef.	Robust Std. Err.	*p*	Odds Ratio
Household characteristics								
Age of the head of household (years)	−0.087	0.050	*	0.917	−0.132	0.104		0.876
Age squared	0.001	0.000		1.001	0.001	0.001		1.001
Indigenous group (0/1)	−0.055	0.415		0.946	0.874	0.344	**	2.397
Education of the head of household (0/1)	−0.181	0.209		0.834	−1.503	0.459	***	0.223
Number of males in working age	−0.157	0.056	***	0.855	0.768	0.329	**	2.155
Commercialization rate (%)	0.001	0.002		1.001	0.000	0.007		1.000
Credit (0/1)	0.467	0.301		1.595	0.838	0.446	*	2.311
Physical asset index	0.289	0.359		1.335	−0.040	0.605		0.960
Land endowments								
Farm >5 ha (0/1)	−2.225	0.358	***	0.108	−1.809	0.060	***	0.164
Forest area within the farm (%)	0.043	0.006	***	1.044	0.045	0.021	**	1.046
Quality of forest resources								
Timber volume potential (m^3/ha)	0.009	0.002	***	1.009	−0.048	0.016	***	0.953
Institutional environment								
Conservation strategy [1] (0/1)	−0.810	0.271	***	0.445	2.294	0.620	***	9.918
Land titling (0/1)	−0.171	0.486		0.843	−2.150	0.762	***	0.117
Governmental grants (0/1)	−0.985	0.297	***	0.373	−1.420	0.307	***	0.242
Infrastructure								
ln distance to the forest patch (km)	−0.200	0.078	**	0.819	0.317	0.105	***	1.373
ln distance to market (km)	0.093	0.111		1.098	−0.307	0.193		0.735
Intercept	−1.075	1.420			5.609	3.371		
Number of observations	486				215			
Hosmer Lemeshow goodness of fit								
x^2	7.83				9.43			
p	0.45				0.31			
VIF	1.29				1.36			

* $p < 0.10$, ** $p < 0.05$, *** $p < 0.010$. [1] For the Central Amazon, the conservation strategy refers to Socio Bosque Program (SBP), and for the Chocó-Darién, it corresponds to protected areas (PA).

Timber volume potential, used as an indicator of the quality of forest resources, showed a significant effect for the Central Amazon and the Chocó-Darién although in different directions. The increase of one cubic meter of harvestable timber is associated with about 1% increase in the odds of a household to deforest in the Central Amazon, and with 5% decrease in the Chocó-Darién. Concerning the institutional environment, our results indicate that conservation strategies have the potential to influence household decisions in the buffer areas; however, the magnitude and direction of this effect varies according to the context and the type of strategy implemented. Households in the Central Amazon living close to the incentive-based conservation, SBP, have a 56% lower odds of deforesting than those with no presence of SBP in the proximities. On the other hand, in the Chocó-Darién, the odds of deforestation were 9 times higher for households living around state-controlled PAs than for those with no PA in their surroundings. Titling was only significant for the Chocó-Darién and showed that households with titled land have an 88% lower odds of deforesting than their counterparts. Governmental grants were negatively associated with deforestation; households receiving cash transfers had between a 63 (Central Amazon) and 76% (Chocó-Darién) lower odds of deforesting than those who do not benefit from this policy.

The control variables included in the model indicate that household-specific factors also have an effect on households decisions. Land endowments showed a significant effect in both regions; households with farms larger than five ha have a 89% and 84% lower odds to deforest in the Central Amazon and in the Chocó-Darién, respectively. However, large forest areas have higher odds of being deforested. One additional percentage of forest increased the odds of a household to deforest by 4% in the Central Amazon and 5% in the Chocó-Darién. Household characteristics indicated that, in the Central Amazon, households with more manpower are associated with a 15% lower odds of deforesting; whereas in the Chocó-Darién, households with more males of working age are 2.2 times more likely to deforest. In the Chocó Darién, ethnicity, education, and credits were also significant. Indigenous Chachis are 2.3 times more likely to deforest than their non-indigenous neighbors. Little or no education is linked with lower odds of deforesting, which is in line with other similar studies [34,42,88]. Despite the fact that credits are only significant at 10%, it is important to note the magnitude (2.3 times higher odds) that credits have on deforestation. From infrastructure variables, only distance to forest was significant, showing that one increase in distance reduces the odds of a household to deforest by 0.20% in the Central Amazon but increases by 0.32% in the Chocó-Darién.

4. Discussion

In Ecuador, ancestral populations and indigenous people own 7 M ha of forests, most of them in the Amazon and the Chocó-Darién [25,97,98]. Landholders have the power to decide how to manage their land in compliance with the law. The Environmental Organic Code (Código Orgánico Ambiental—COA in Spanish) allows landholders to use their forests, whether for subsistence or commercial purposes, as long as they follow the regulations established in the law [99]. According to the COA, no management plans are needed when landholders use the forest for subsistence or cultural reasons (Art. 109). In this case, harvesting forest products does not qualify as an infringement of the law, but the environmental authority is in charge to regulate the quantities to be used or extracted (Art. 315). However, for commercialization purposes, landholders need to present a management plan, following the respective guidelines (Art. 107, Art. 109); failing to do so is punished by the law (Art. 317, Art. 318). Deforestation within the farm creates negative environmental externalities (e.g., fragmentation, habitat loss, greenhouse gas emissions, and degradation of ecosystem services) that exceed the farm boundaries and affect human well-being [9,100]. Uncovering the factors related to household decisions to deforest is essential for the design of effective conservation actions.

As previously mentioned, the quality of natural resources that landholders have at their disposal influences their land-use decisions. In Central Amazon, livelihoods are characterized by small-scale agriculture, and farmers are limited by low fertile and acid soils, forcing them to expand agricultural fields to cope with productivity declines [101]. In this region, forests with high timber volumes are associated with a higher odds of deforesting. We estimated, on average, a timber volume potential of 176 m^3/ha for the Central Amazon; however, timber in this region is considered a by-product of agricultural expansion and contributes less than 10% to the total household income [28,58,102]. During forest clearing, farmers extract timber species of high commercial value and sell them to intermediaries or to local markets, obtaining an additional benefit from forest conversion. In areas of high abundance of timber species, farmers sell standing trees to intermediaries and use the cash to finance more agricultural expansion [28], showing that their main interest is land instead of timber. In the Central Amazon, small-scale timber markets satisfy local and regional demands for construction and furniture; however, most of the timber is commercialized in the neighboring cities, while only 8% is locally consumed [28,65]. In the Chocó-Darién, on the other hand, we found an opposite relation between timber potential and deforestation decisions. We estimated that the timber volume potential is 118 m^3/ha for this region and timber extraction for commercialization purposes is an important livelihood strategy and has been noted as the main driver of deforestation [27,42]. Farmers begin by harvesting hard-wood species; when these are depleted, soft-wood species are sold. High intensity and frequent logging activities can result in degraded forests [103]. When tree species of commercial value disappear, farmers convert the already degraded forests into agricultural lands as an alternative livelihood strategy; this conversion does not occur in the short-term and it is not possible to capture with cross-sectional studies. In the Chocó-Darién, precious timber species are declining [104] and logged forests now have only half of the timber volume potential compared to intact forests [51], increasing the likelihood of a total forest clearing in the medium to long-term. With the decline of timber species in the Chocó-Darién, it is probable that timber markets relocate to the Central Amazon, which accounts for 63% of the legally harvestable timber volume [8]. It is urgent to encourage sustainable forest management (SFM) with reduced-impact logging techniques, and to promote fair participation of farmers within value chains. This must be accompanied by more control of illegal logging, which is a serious concern in both regions [58] and more post-harvesting monitoring. More resources are needed to strengthen the capacity of the environmental authority.

Even when the primary goal of conservation strategies is to safeguard forests inside the areas under protection, land-use intensification outside them negatively affects the integrity and functionality of conservation areas [87]. Therefore, it is necessary to identify circumstances under which these policies can disincentivize deforestation beyond the borders of the conservation. Our results evidence that, in the study regions, deforestation is occurring close to conservation areas, calling for the inclusion of more local landholders in present and future conservation strategies. The presence of conservation strategies portrayed mixed results on household deforestation decisions across the regions in our study. Spatially explicit studies also have found contrasting results [105–110], supporting the fact that the effect of conservation policies on deforestation is context-specific. In our study, households in the Central Amazon, living close to the incentive-based conservation SBP had lower odds to deforest compared with their peers with no SBP in their surroundings. Likewise, assessments using ecological indicators quantified less deforestation inside SBP areas and less degradation in forests close to SBP [22,49,108,111]. Our results prove that incentive-based conservation has a promising potential for combatting deforestation beyond the limits of the conservation area. It seems that SBP is raising conservation awareness even in areas outside the program. Some studies show that local people enrolled in SBP conduct frequent surveillance and when they detect illegal activities, within and around the SBP area, they report it to the environmental authority [22,112]. Moreover, in zones with an SBP area, there is a higher presence of SBP staff that constantly monitors the compliance with conservation contracts [22,49,113]. Perhaps, these facts restrain neighboring households to deforest, as they may perceive higher probabilities to get caught in illegal activities.

Conversely, in the Chocó-Darién, households living close to a PA had a higher chance of deforesting than those with no PA in their proximities. This does not imply that PAs are driving more deforestation; it rather evidences that near PAs pressures on forests are high, and households clear their remnant forest despite the potential sanction they could receive if they are caught infringing the law. Our results indicate that in a context of high deforestation and a strong presence of timber markets, people living in the buffer areas need additional alternatives to reduce deforestation within their farms. Since around 69% of the tropical moist forest in surrounding PAs has experienced a decline in the forest cover [107], it is necessary to design mechanisms that enhance the role of such strategies in the buffer areas. The well-being of people living near PAs can be improved in different ways. For example, when tourism is promoted, PAs can help to generate additional income and reduce poverty levels; moreover, positive links have been found on children's health for those living close to PAs [114]. It means that to be effective in preventing forest degradation and deforestation in the long-term, the management of PAs does not have to be disconnected with the people living in the vicinity, and positive benefits can be generated when more holistic management approaches are considered.

The Ecuadorian Constitution of 2008 recognized legal rights over lands to indigenous and ancestral possessors [62], and with the new forest policy reforms, titling increased considerably in the country. Individual tenure rights were granted for individual landholders, and collective tenure rights were recognized for indigenous communities; these comprised land-use rights including forests [28]. From our sample, 86% of households reported having legal titling in the Central Amazon and only 40% in the Chocó-Darién. Precisely, in the latter region, titling was significantly associated with less deforestation. There is important evidence indicating that formal land tenure leads to less forest conversion [34,42,96,115], which was also corroborated with our findings. In this respect, solving land tenure issues is crucial to preventing more deforestation of fragile ecosystems such as the Chocó-Darién. Having land tenure security can bring additional benefits such as less timber harvested illegally [91], the adoption of more sustainable land-use practices [34,116,117], and the possibility of participating in governmental programs such as conservation programs or trainings [115].

The environmental effect of governmental grants is an aspect that has remained understudied. However, we are starting to see studies evidencing that poverty alleviation policies are not necessarily at odds with conservation objectives [39,114]. Results from our study also contribute to supporting this argument. Controlling for other aspects, we found that households receiving governmental grants had lower odds of deforesting. Our findings suggest that cash transfers to poor households can reduce the need of clearing more forest to obtain extra income or to produce food, indicating that governmental policies for poverty alleviation have the potential to support forest conservation. In this sense, these results could motivate more research in this area and the integration of people working in the natural resources conservation sector when social policies are designed.

Besides discussing the results for our main variables of interest, we also briefly discuss some points concerning our control variables. The positive association between indigenous Chachis and deforestation evidences their high dependence on forest resources in a context of lack of technology, informal markets, and historic marginalization. Livelihoods in indigenous households are characterized by a higher crop diversification, mixed with the collection of forest products and a low engagement in cattle ranching; evidencing the role that ethnicity has on land-use decisions. As long as indigenous people are more integrated into markets, subsistence-oriented production systems are likely to transform into a cash-based economy; under the market imperfections, people can be pulled into a lose–lose situation in which environmental degradation comes hand-in-hand with poverty exacerbation [31,40,46,118–121]. More education is needed to provide local people the opportunity to participate in markets under more equitable conditions. So far, education is insufficient in our study sites; people who manage to finish primary school have serious economic limitations to continuing their studies. As suggested by Pichón and Bilsborrow [122], we believe that for the positive effect of education to be translated in less deforestation, there must be a significant increase in education levels in the forest frontiers.

Our results also reflected that the limitations given by a small farm size lead people to use the soil intensively [34]. In the Central Amazon, farmers have on average 26 ha of land, whereas in the Chocó-Darién, their counterparts have 23 ha. Continuous plot subdivisions to satisfy new demands for land have fragmented and reduced the size of many farms [123]. For households whose small farm is the only productive asset, policies restricting the forest use or forest conversion might be more oppressing. In contexts such as the Central Amazon and the Chocó-Darién, characterized by expansive production systems and poor agricultural technology [31,45], households clear the forest to extend their agricultural fields as the only option to compensate productivity declines or to cope with the depletion of forest resources, evidencing the role of land in supporting livelihood strategies. Higher percentages of forest cover within the farm can attract agricultural extension in the Central Amazon and more timber extraction in the Chocó-Darién, putting larger forest areas at a higher risk of conversion.

Finally, households in the Central Amazon are less likely to deforest if their forest plots are located at longer distances from homesteads, which is in line with our previous assumption; however, in the Chocó-Darién, we observed an opposite relation. Ninety percent of forest clearing in the Central

Amazon occurred within 4.4 km and within 5.5 km in the Chocó-Darién from the house to the forest plot owned by the farmer. A previous study analyzing deforestation with spatial techniques also suggests that in the Chocó-Darién deforestation occurs in more remote areas [40]. Forest areas near the house have already been converted to other land uses, and the remaining forests are located at longer distances. Palacios and Jaramillo [104] found that tree species with high commercial value are in severe decline and tree species with big diameters are less abundant for the timber market within short distances, forcing farmers to move longer distances [12,104].

5. Conclusions

Understanding the attributes that lead a household to convert forestlands to other land-uses can help to design better conservation policies. The quality of natural resources and the institutional environment are elements that significantly influence household deforestation decisions. In contexts where farmers depend more on agricultural production, valuable tree species can generate an additional incentive for forest clearing, since timber commercialization can finance agricultural expansion. When timber markets dominate the economy, continuous timber harvesting leads to forest clearing once valuable timber species are depleted; however, this effect is not captured in cross-sectional studies.

Conservation strategies have the potential to influence household decisions outside the areas under protection; however, the effects are context-dependent. On the one hand, SBP is associated with lower odds of deforestation in farms close to the program. This can be attributed to the higher awareness that SBP has created towards conservation. On the other hand, PAs need additional strategies to reduce deforestation in the buffer zones, especially in contexts with high deforestation pressures. This does not mean that PAs are not effective in reducing deforestation within their borders. There is a clear acknowledgment that without PAs in our study areas, deforestation could be much higher. As agricultural lands are getting closer to forests, deforestation risks are posed on new areas including those which are now under strict protection; therefore, lessons learnt from SBP and PAs can help to design or improve conservation strategies aligned to the current social demands.

Titling continues to be an important element for an adequate governance of the territory and for forests in particular. In places where land titles are scant, such as the Chocó-Darién, the effect of having secure property rights on deforestation is evident. The Ecuadorian government needs to facilitate the titling process in order to avoid more deforestation in the last remnants of the Chocó-Darién.

Governmental grants that aimed to alleviate poverty show good signs of helping to reduce deforestation. Undoubtedly, more research is still needed on this topic. However, in times of high uncertainty, high unemployment rates, and a continued demand for land, well-designed cash transfer schemes can help to create positive outcomes for the environment.

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