Improving estimates of CO_2 emissions under REDD+ in the Colombian Amazon: Better understanding for climate change mitigation

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I certify that all material in this thesis which is not my own work has been identified and that no material has previously been submitted and approved for the award of a degree by this or any other University.

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Abstract

Land-cover change is the second most important source of anthropogenic greenhouse gases (GHG) emissions, generating around 7-14% of the total carbon dioxide (CO_2) emissions around the world. More than one million km^2 of tropical forests were lost during the period 2000-2012 around the world, from which forests-to-pasture conversion was the most common land-use change in key regions such as the Amazon. Strategies to mitigate climate change by reducing deforestation and forest degradation (e.g. REDD+) require country- or region-specific information on carbon (C) stocks in forests and their dynamics with land-cover change, in order to develop accurate Forest Reference Emission Levels (FRELs) to be submitted to the UNFCCC as benchmarks for assessing the performance of countries participating in REDD+ activities. Nevertheless, FREL development is incipient and their elaboration is mostly based on highly uncertain Tier 1 information from IPCC. In this research I present the first region-specific Tier 3 information and emission factors on soil, dead wood and below-ground biomass C pools and their dynamics during 20 years of forest-to-pasture conversion under different management practices in the Colombian Amazon. Based on these region-specific Tier 3 emission factors on C stocks in forests and their change after pasture establishment, I report for the first time the net CO_2 emissions from forest-to-pasture conversion in the Colombian Amazon. The results also demonstrate that Tier 3 region-specific information is 70% higher and is substantially more accurate than estimates based on using IPCC Tier 1 information, which emphasizes the urgency for countries implementing REDD+ to develop improved data and methodologies. The information reported here will contribute to strengthening the REDD+ National Strategy of Colombia, by

supplying accurate data and models that can be included within the next Colombian FREL.

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Abbreviations

AFOLU	Agriculture, Forestry and Other Land Use
AGB	Above-ground biomass
BGB	Below-ground biomass
BGN	Below-ground necromass
C	Carbon
CO_2	Carbon dioxide
CP	Conference of the Parties
CWD	Coarse woody debris
C3-C	C derived from C3-plants
C4-C	C derived from C4-plants
DW	Dead wood
$\delta^{13}C$	Soil isotopic composition of ^{13}C
$\delta^{15}N$	Soil isotopic composition of ^{15}N
FAO	Food and Agriculture Organization of the United Nations
FREL	Forest reference emission level
GHG	Greenhouse gas
Gt	Gigatonnes (1 Gt = 10^9 tonnes)
HFFL	Heads of forage-fed livestock
HG	High-grazing intensity sub-region
IDEAM	Institute of Hydrology, Meteorology and Environmental Studies of Colombia
IPCC	Intergovernmental Panel on Climate Change
LG	Low-grazing intensity sub-region

LULC	Land-use and land-cover
MADS	Ministry of Environment and Sustainable Development of Colombia
Pg	Petagrams (1 Pg = 10^{15} g)
REDD+	Reducing emissions from deforestation and forest degradation (REDD), plus the conservation and sustainable management of forests and enhancement of forest carbon stock (+)
SOC	Soil organic carbon
t CO_2e	Tonnes of carbon dioxide equivalent
UNFCCC	United Nations Framework Convention on Climate Change

Chapter 1: Introduction

1. Introduction

1.1. General context

The Paris Agreement on Climate Change (UNFCCC, 2015), recently adopted by the developed and developing countries during the 21st Conference of the Parties to the United Nations Framework Convention on Climate Change in Paris, gives new support to implement actions aimed to reduce greenhouse gases (GHG) emissions from deforestation and forest degradation and to improve conservation, sustainable management of forests and enhancement of forests carbon (C) stocks (REDD+). According to the Agreement, developing countries should undertake reductions in their GHG emissions based on "*the best available science*", which should be reflected in a significant improvement of their efforts to mitigate climate change over time (UNFCCC, 2015).

Strategies for avoiding deforestation such as REDD+ could help to reduce anthropogenic GHG emissions by around 2.9-4.8 billion tonnes of CO_2 equivalent (Gt CO_2e) per year (Don et al., 2011). Nevertheless, countries willing to access result-based payments through REDD+ activities require the development of forest reference emission levels (FREL) as benchmarks for assessing country's performance (UNFCCC, 2010; UNFCCC, 2011; FAO, 2014; UNFCCC, 2014). In general, FRELs are based on estimates of emission factors, which represent the emissions or removals of all important GHGs associated with land-cover conversion in all relevant C pools per hectare (i.e. total changes in C stocks), and the activity data, referring to the total extent of a deforested, degraded or regenerated area during an established period (Verchot et al., 2012).

By the end of 2014 Brazil, Colombia, Ecuador, Guyana, Malaysia and Mexico submitted their FREL to the UNFCCC as part of the requisites to participate in

REDD+ activities. All FRELs, however, only included the above- and below-ground biomass C pools in forests, and conservatively excluded the other C pools and their changes related to CO_2 emissions or removals after conversion from forest to any post-deforestation land-use category (Conafor, 2014; GFC, 2014; MADS, 2014; MAE, 2014; MMA, 2014; MNRE, 2014). In fact, under the IPCC Tier 1 approach it could be assumed that 100% of C stored in the dead wood and below-ground biomass C pools is emitted immediately after deforestation, conservatively assuming no C in these pools in a post-deforestation land categories such as pastures (IPCC, 2006). Similarly, changes in C stocks in soils under a Tier 1 approach might be estimated assuming a linear transition to a new equilibrium in a post-deforestation land category, based on default information about climate, land-use and management practices supplied by IPCC (2006). However, even though the IPCC provides the default information on these parameters, the Tier 1 methodology also allows to conservatively exclude the soil organic C pool when the country submitting the FREL reports a lack of accurate information to estimate GHG emissions and removals from this C pool. Nevertheless, under the UNFCCC Stepwise Approach (UNFCCC, 2012), countries have the option to improve their initial FREL by incorporating high-quality data, improved methodologies and additional C pools developed from country- or region-specific information and field measurements following an IPCC Tier 3 approach.

In recent years Colombia has strengthened its capacity to support and implement REDD+ projects in the country, by improving its information on deforestation rates, drivers of deforestation, and C stocks stored in the above-ground biomass in natural forests (MADS, 2014). According to the latest official available information, more than one million hectares of forest were lost in the

Colombian Amazon over the period from 2000 to 2012, and conversion from forests to pastures has become the most common land-use change in the country with pastures occupying ~83% of the converted area during this period (Ideam, 2014a). Nevertheless, estimates of net CO_2 emissions due to deforestation, or particularly to forest-to-pasture conversion, in the Colombian Amazon are uncertain due to a lack of reliable information, especially related to the emission factors of the relevant C pools and their change with land-use conversion.

1.2. Aim of the research

Considering the urgency for countries implementing REDD+ to develop improved data and methodologies that meet the principles of accuracy, completeness, integrity and transparency established in the Paris Agreement (UNFCCC, 2015), the aim of this research was to better quantify CO_2 emissions associated with the conversion from forests to pastures in the Colombian Amazon in support of the REDD+ National Strategy of Colombia.

1.3. Specific objectives of the research

1. To quantify the soil organic C stocks and changes over 20 years post forest-to-pasture conversion in the Colombian Amazon;
2. To quantify the dead wood C stocks and changes over 20 years post forest-to-pasture conversion in the Colombian Amazon;
3. To quantify the below-ground necromass C stocks and changes over 20 years post forest-to-pasture conversion in the Colombian Amazon;
4. To estimate the Tier 3 net CO_2 emissions due to forest-to-pasture conversion in the Colombian Amazon for the period 2000-2012.

1.4. Thesis structure and main results

The research presented in this thesis is divided into eight chapters. **Chapter 1** presents an overview of the carbon emissions in the context of REDD+, and includes a review of: land-cover changes and CO_2 emissions associated with deforestation, legacy fluxes from land-cover change, REDD+, the IPCC guidelines to quantify GHG emissions/removals within the Agriculture, Forestry and Other Land Use (AFOLU) sector, and a description of the guidelines to estimate and report the GHG inventory for the category ‘Forest lands converted to Grasslands’. This chapter also presents the general aim and specific objectives of the research. **Chapter 2** describes the relevant characteristics of the site where the research took place, and **Chapter 3** presents the Chronosequence Approach as the method used to monitor changes in C stocks in soils, dead wood and below-ground biomass C pools to establish the long-term CO_2 emissions associated with forest-to-pasture conversion in the Colombian Amazon.

Chapters 4 to 7 present the main results of the research. Each one of these chapters contains, in turn, six sections: abstract, introduction, materials and methods, results, discussion and conclusions. The general aim and specific objectives of each chapter are presented at the end of the introduction section. **Chapter 4** focuses on establishing the influence of land-cover change from forest to pasture and subsequent land management practices on soil organic C stocks and dynamics in the Colombian Amazon. Results presented in this chapter show the contrasting dynamics of soil organic C between pasture areas with high- and low-grazing intensity, and highlight the importance of low-grazing practices to reduce CO_2 emissions. **Chapter 5** aims to determine the effect of forest-to-pasture conversion and subsequent land management practices on the dead wood C pool in the Colombian Amazon. This chapter shows how high-grazing intensity practices led to a higher reduction of the dead wood C pool after 20 years of pasture establishment, and also emphasize the importance of low-grazing practices for reducing CO_2 emissions. **Chapter 6** focuses on determining the C stocks in dead coarse roots (below-ground necromass) in forests of the Colombian Amazon and its dynamics with forest-to-pasture conversion and subsequent land management practices. As with the dead wood C pool, high-grazing intensity practices led to a larger reduction in below-ground necromass 20 years following pasture establishment compared with low-grazing intensity management. Finally, **Chapter 7** presents for the first time a Tier 3 assessment of the net CO_2 emissions from forest-to-pasture conversion over the Colombian Amazon region based on the results obtained in Chapters 4, 5 and 6, and compare these results with estimates based on IPCC Tier 1 default information. Results presented in this chapter indicate that Tier 3 region-specific information are 70% higher and substantially more accurate than estimates based on using IPCC Tier 1 information, and highlight the urgency for countries implementing REDD+ to develop improved data and methodologies that meets the principles of accuracy, completeness, integrity and transparency established in the Paris Agreement of the UNFCCC.

Finally, **Chapter 8** presents the general conclusions and recommendations derived from the research.

1.5. Carbon emissions in the context of REDD+

1.5.1. Land-cover change

Land-cover conversion is the second most important source of anthropogenic greenhouse gases (GHG) emissions (Don et al., 2011), generating around 7-14% of the total carbon dioxide (CO_2) emissions around the world (Harris et al., 2012). Deforestation in the tropics has been estimated at ~1.1 million km^2 during the period 2000-2012 (Hansen et al., 2013), and conversion from forests to pasture has become the most common land-use change globally (Elmore and Asner, 2006; IPCC, 2006). Most of the deforested area in the Amazon has been occupied by cattle pasture (Fearnside and Barbosa, 1998; Asner et al., 2004; Desjardins et al., 2004), although since the beginning of the 21st century the area for the agricultural production has increased, mainly focused on soybean crops (Nepstad et al., 2008; Pacheco et al., 2012).

After cutting the forest, management practices to establish pasture areas in the Amazon may include: the use of fire to eliminate the maximum amount of plant material left by deforestation or to control the expansion of secondary vegetation (Fearnside et al., 1993; Kauffman et al., 1995; Kauffman et al., 1998; IPCC, 2006; Aragão and Shimabukuro, 2010), implemented once or more times depending on burning efficiency of dead wood (Fearnside et al., 1999; Fearnside et al., 2001); the use of machinery to remove unburned woody debris (Murty et al., 2002; Marin-Spiotta et al., 2009); the introduction of improved pastures and legume species (Alarcón and Tabares, 2007; Mosquera et al., 2012), or the use of fertilizers and lime to improve pasture productivity (Jiménez and Lal, 2006; Fisher et al., 2007). Management practices implemented during pasture establishment are as important as land-cover changes in determining carbon (C) dynamics and GHG emissions/removals (Fearnside and Barbosa, 1998; Dias-Filho et al., 2000; Berenguer et al., 2014; Luyssaert et al., 2014), and among them, grazing intensity can significantly contribute to increase or reduce C stocks (Uhl et al., 1988).

Nevertheless, there are some difficulties in quantifying the global deforested area because of the large number of definitions found in the literature for 'forest' and 'deforestation'. According to Lepers et al. (2005), more than 90 different definitions of forest are used on a global scale, complicating the standardization of information and results on land-use and land-cover (LULC) changes. The UNFCCC (2001) proposed a definition of forest for countries implementing the Clean Development Mechanism (CDM), that includes a range of minimum area (i.e. 0.05-1.0 ha), tree crown cover (i.e. 10-30%), and minimum height at maturity in situ (i.e. 2-5 m), and that should be selected according to the characteristics of every country. Depending on the selected values, forest area, as well as deforested area and associated GHG emissions, could increase or decrease (FAO, 2006).

Most of the definitions, however, agree that deforestation occurs when a forest land is converted to a different category of land use, reaching levels below 10% of the original tree crown cover (FAO, 2006; Lepers et al., 2005). Some of the definitions on deforestation found in the literature include: the conversion from forest lands, including open, closed, fragments, plantations or secondary forest, to non-forest lands such as savannahs, agriculture areas or, non-vegetated areas (Ramankutty et al., 2007); an area of forest land that has been converted by humans or nature as consequence of any disturbance (Harris et al., 2012); or simply, the conversion from forests to non-forests (Houghton et al., 2012).

After deforestation, every portion of land will remain in a conversion category (e.g., Forest to Grasslands or Forest to Croplands) during a particular period, after which the landscape as a whole will approach to an 'equilibrium state' (IPCC, 2006; Fearnside, 1997). The IPCC (2006) default period in which a portion of land remains in any conversion category after land-cover change is 20 years, although each country might propose its own conversion period. According to Fearnside and Barbosa (1998), a post-deforested landscape in equilibrium in the Brazilian Amazonia would consist of around 43.0% productive pasture, 6.0% degraded pasture, 4.0% farmland, 2.0% secondary forest converted from agriculture and 50.0% secondary forest converted from pasture.

Deforestation in Colombia represents a threat to the forest ecosystems and for the environmental services they offer. The total area of forest lost in the country from 1990 to 2010 was 5,566,041 hectares (Cabrera et al., 2011), mainly concentrated in the Amazon and Andes regions of Colombia. The increasing pressure of colonization of the Colombian Amazon, mainly driven by farmers looking for areas to develop cattle ranching activities, has been facilitated by its proximity to the densely populated Andean region (Etter et al., 2006a; Etter et al., 2006b; Etter et al., 2006c).

1.5.2. Carbon emissions due to deforestation

Land-cover change from forest lands to other land categories generally results in high loss of C, mainly from the above-ground biomass, and elevated CO_2 emissions to the atmosphere (Morton et al., 2008; IPCC, 2006). Nevertheless, C emissions from LULC changes are the component of global C cycle with the highest uncertainties (Harries et al., 2012; De Fries et al., 2008; Ramankutty et al., 2007). The IPCC AR5 report (2013) on climate change accounts for emissions from fossil fuel combustion of 8.3 ± 0.7 Gt C yr^{-1} during the period 2002-2012, compared to the 0.9 ± 0.8 Gt C yr^{-1} for emissions from anthropogenic LULC changes for the same period. Table 1.1 presents a summary of estimates of C emissions from land-cover change with values from different sources, showed by Ramankutty et al. (2007) and Houghton et al. (2012).

One important factor for improving the accuracy of C flux estimates from land-cover change is to better understand the land-use dynamics after deforestation, including the fate of the land following clearance (e.g. grassland, croplands, or secondary forest), and the management practices implemented after deforestation (e.g. slash-and-burn system or the use of machinery; Ramankutty et al., 2007; Morton et al., 2008). Estimations of GHG emissions based only on gross carbon flux (i.e. taking into account only the carbon emissions due to loss of the forest) overestimate the net flux of C from sources and sinks, like land abandonment, regrowth or reforestation, and delayed emissions from previous land uses (i.e. legacies), degradation or clearing of secondary forests (Houghton et al., 2012; see Figure 1.1).

Besides its contribution to better understand whether tropical forests will continue to be a sink for atmospheric carbon into the future, increasing the accuracy in the estimation of CO_2 emissions also has significant repercussions for the application of policies focused on avoiding deforestation (De Fries et al., 2008), as a mitigation alternative against climate change. As a first attempt to estimate net fluxes of carbon from land-cover change (i.e. including deforestation of primary forest and regrowth of secondary forest), Yepes et al. (2011b) recently published the results of the CO_2 emissions in Colombia for the period 2005 – 2010, as part of the Colombian strategy for avoiding deforestation and GHG emission.

Table 1.1 Global carbon emission estimates (Gt C yr^{-1}) from LULC changes for the periods 1980-1989, 1990-1999, and 2000-2009

Study	Period			Presented by
	1980's	1990's	2000's	
Houghton (2003)	2	2.2	-	Ramankutty et al. (2007)
Fearnside (2000)	2.4	-	-	Ramankutty et al. (2007)
McGuire et al. (2001)	0.9-1.6	-	-	Ramankutty et al. (2007)
DeFries et al. (2002)	0.6 (0.3-0.8)	0.9 (0.5-1.4)	-	Ramankutty et al. (2007)
Achard et al. (2004)	-	1.1 ± 0.3	-	Ramankutty et al. (2007)
Denman et al. (2007)	1.4 (0.4-2.3)	1.6 (0.5-2.7)	-	Houghton et al. (2012)
Friedlingstein et al. (2010)	-	-	1.1 ± 0.7	Houghton et al. (2012)

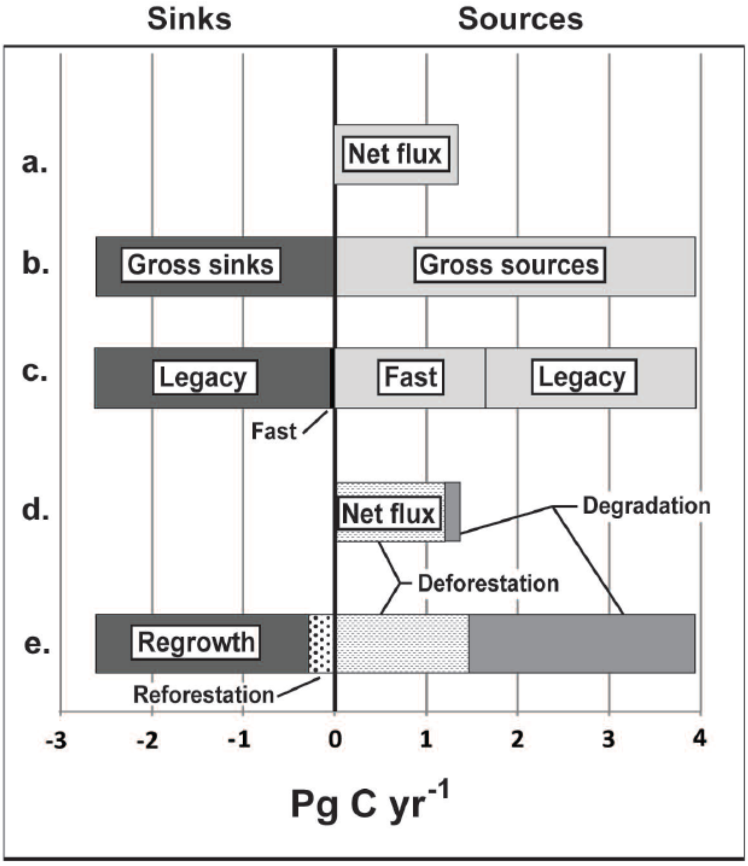

Figure 1.1 Mean annual net (a) and gross (b) sources and sinks of carbon 2000-2009 attributable to LULC changes (from Houghton's analysis as reported in Friedlingstein et al., 2010). "Legacy" in 2c refers to the sinks (regrowth) and sources (decomposition) from activities carried out before 2000; "Fast" in 2c refers to sinks and sources resulting from the current year's activity. Most of the net flux (2d) is attributable to deforestation, with a smaller fraction attributable to forest degradation. The reverse is true for gross emissions 2e): degradation accounts for more of the gross emissions than deforestation. Most of the gross annual sink (2e) is attributable to regrowth (in logged forests or the fallows of shifting cultivation), with a smaller sink attributable to reforestation (an increase in forest area following abandonment of agricultural land. Taken from Houghton et al. (2012).

1.5.3. Legacy fluxes from land-cover change

Land-cover change events in one particular year will still influence CO_2 emissions in subsequent years, due to the decomposition of the remaining dead matter after deforestation (Ramankutty et al., 2007; Sierra et al., 2012). While instantaneous emissions occur during the same year of the event of deforestation due to fire or rapid decomposition of the labile material (e.g. leaves or twigs), inherited or legacy fluxes can persist for months to decades resulting in a long-term C loss and related GHG emissions (Houghton et al., 2012; IPCC, 2006).

Decomposition of dead organic matter and its associated CO_2 emission is controlled by three major factors: substrate quality, microbial community composition and physical environment (Chapin et al., 2002; Swift et al., 1979). Changes in land-cover directly affect these factors, with repercussions on the input of fresh detritus and variations on decomposition rates (IPCC, 2006). An accurate quantification of the fraction and fate of biomass removed as a result of deforestation, and the mechanism used for removing above- and below-ground biomass for the establishment of the new land use, will improve C flux estimates caused by historical land-cover change activities (Morton et al., 2008).

Some of the key elements to estimate C flux from deforestation in the tropics include the quantification of rates and dynamics of land-cover change, the estimation of C stocks in the original vegetation (i.e. forest), the pattern of delayed C loss from historical land-cover change (i.e. legacy fluxes), and the integration of these factors into models that represent the dynamics of C change (Ramankutty et al., 2007). The contribution of instantaneous and legacy fluxes to the global net land use flux during 2000 to 2009 is presented by Houghton et al. (2012) (reproduced as Figure 1.1 in this document). This results show that the contribution of legacy fluxes to the gross emissions was larger than the instantaneous emissions attributable to land-cover change.

Instantaneous and delayed CO_2 emissions caused by LULC changes have constantly increased from the second half of the 19th century, with a large input from deforestation in the tropics. In this context, reducing emissions from deforestation and forest degradation, plus the conservation and sustainable management of forests, and enhancement of forest carbon stock (REDD+), has appeared as a novel alternative to mitigate CO_2 emissions from LULC change, loss of biodiversity, and loss of resources for people who live from forest products.

1.5.4. REDD+

Reducing emissions from deforestation (RED) and forest degradation (REDD), plus the conservation and sustainable management of forests, and enhancement of forest carbon stock (REDD+), was initially proposed in 2005 during the 11th Conference of the Parties to the UNFCCC (CP/11), and finally defined in 2007 during the CP/13 (Angelsen and McNeil, 2012). REDD+ was designed to provide economic incentives to forest users that reflect the value of carbon stored in live trees (Angelsen et al., 2012).

The success of the REDD+ idea as one of the most effective and efficient mitigation strategies against climate change, relies on its potential to be a win-to-win policy that combines GHG emissions reduction with poverty reduction and the protection of biodiversity (Angelsen and McNeil, 2012). REDD+ could redefine the role of forest-rich developing countries around the world, from aid dependency to providers of a global public good (i.e. climate mitigation) to all countries (Brockhaus and Angelsen, 2012).

Article 5 of the Paris Agreement (UNFCCC, 2015) now provides the support for countries to take actions to conserve and enhance the reservoir of GHG, including forests, and encourage them to implement and support activities related to REDD+. Nevertheless, the complexity of international negotiations to reach agreements among countries on how to tackle the risk generated by climate change has modified the original ends and means of REDD+, driven by two main streams: business-as-usual groups, represented by people receiving benefits from land-cover change activities (e.g. cattle ranchers or oil palm farmers), and supporters of REDD+ comprising groups with different interests (Brockhaus and Angelsen, 2012).

An important characteristic of REDD+ that makes it different from previous initiatives of conservation, is that it shall be evaluated based on its performance, which requires that results will have to be accurately measured (Angelsen et al., 2012). Good quality information will give governments and institutions the power to accurately measure REDD+ performance in order to receive fair incentives according to their avoided emissions (Korhonen et al., 2012).

Countries willing to access result-based payments through REDD+ activities require the development of forest reference emission levels (FREL) as benchmarks for assessing country's performance (UNFCCC, 2010; UNFCCC, 2011; FAO, 2014; UNFCCC, 2014). In general, FREL are obtained from estimates using emission factors, which represent the emissions or removals of all important GHGs associated with land-cover conversion in all relevant C pools per hectare (i.e. total changes in C stocks), and the activity data, referring to the total extent of a deforested, degraded or regenerated area during an established period (Verchot et al., 2012).

Most developing countries lack this information which increases the difficulty of implementing a reliable measurement reporting and verification system for assessing the performance of REDD+, according to IPCC guidelines (Romijn et al., 2012). In fact, the majority of countries rely on default values and conversion factors proposed by IPCC for accounting their carbon emissions and obtaining carbon credits, achieving results with high uncertainties that are subject to 'discounts' (i.e. less benefits for GHG emission reduction) according to the requirements of some robust standards and methodologies (Verchot et al., 2012).

1.5.5. IPCC Guidelines for quantifying GHG in the agriculture, forestry and other land use (AFOLU) sector

The Intergovernmental Panel on Climate Change (IPCC) assesses the scientific, technical and socio-economic information relevant for the understanding of the risk of human-induced climate change, and from the last decade of the 20th century it has published and revised the guidelines for national greenhouse gas inventories for the agriculture, forestry and other land use (AFOLU) sector (IPCC 2006). The IPCC sets the foundation for the development of standards that establish the requirements for estimating GHG emission reduction and their associated carbon credits (Estrada and Joseph, 2012).

Land-cover changes generally result in the transfer of carbon from one pool to another, as for example from above-ground biomass to dead wood C pools, and then to the soil organic C pool or to the atmosphere through decomposition. The IPCC (2006) has defined five carbon pools for which carbon stocks changes and GHG emissions and removals can be estimated:

- Biomass

    - Above-ground biomass (AGB): all biomass of living vegetation, both woody and herbaceous, above the soil including stems, stumps, branches, bark, seeds, and foliage.
    - Below-ground biomass (BGB): all biomass of all live roots. Fine roots of less than (suggested) 2 mm diameter are often excluded because these often cannot be distinguish empirically from soil organic matter or litter.
- Dead organic matter
    - Dead wood (DW): Includes all non-living woody biomass not contained in the litter, either standing, lying on the ground, or in the soil. Dead wood includes wood lying on the surface, dead roots, and stumps, larger than or equal to 10 cm in diameter.
    - Litter (LT): Includes all non-living biomass with a size greater than the limit for soil organic matter (suggested 2 mm) and less than the minimum diameter chosen for dead wood (e.g. 10 cm), lying dead, in various states of decomposition above or within the mineral or organic soil. This includes the litter layer as usually defined in soil typologies. Live fine roots above the mineral or organic soil (of less than the minimum diameter limit chosen for below-ground biomass) are included in litter where they cannot be distinguished from it empirically.
- Soils
    - Soil organic carbon (SOC): Includes organic carbon in mineral soils to a specific depth chosen by the country. The default depth is 30 cm. SOM includes organic material (living and non-living) within the

soil matrix, operational defined as a specific size fraction (e.g. all matter passing through a 2 mm sieve).

The IPCC (2006) also defines six broad categories that form the basis to estimate and report GHG emissions and removals from the land use and land-cover change: i) Forest lands, ii) Cropland, iii) Grassland, iv) Wetlands, v) Settlements, and vi) Other lands. Each land-use category is further subdivided into land remaining in that category (e.g. Forest Land Remaining Forest Land) and land converted from one category to another (e.g. Forest Land converted to Grassland).

Estimation of GHG emissions and removals should also be classified in tiers, according to the methodological complexity that a country or project followed during the inventory, that range from Tier 1 (less complex) to Tier 3 (most complex). Moving from lower to higher tiers improves the inventory and reduces uncertainty, but at the same time increases the resources required for conducting the inventories (IPCC 2006).

The accessibility of Tier 1 methodologies allows GHG inventory developers to produce an almost complete accounting of emissions and removals with little investment in data collection and analysis (Verchot et al. 2012). Tier 1 methods are simple to apply and are based on default parameter values (e.g. emission and stock change factors) provided by IPCC (2006). The uncertainty associated to default values supplied by IPCC, however, is extremely high, especially for tropical regions (Don et al., 2011).

For practicality, when Tier 1 methods are used, GHG emissions due to LULC changes are assumed to occur during the same year of the disturbance event, excluding the harvested wood products removed throughout the activity. For instance, instead of estimating the decay of the remaining dead organic matter produced after a disturbance event over a period of several years, Tier 1 methodologies assume that all carbon contained in the dead organic matter pool is emitted in the same year of the land-cover change event (IPCC, 2006). Tier 2 methodology approaches are similar to Tier 1, but instead of using default values, Tier 2 is based on country- or region-specific emission and stock change factors.

Finally, under a Tier 3 approach, high-order methods are used, including models (e.g. decaying models for dead organic matter remaining after a deforestation event) and accurate measurements adjusted to national or regional characteristics (IPCC, 2006). With Tier 3 methodologies, areas where land-cover change events occurred can be statistically monitored over time by using robust methods such as chronosequences. The main constraints related to the implementation of Tier 3 methodologies rely on the cost and effort involved in the production of quality datasets and site specific measurement (Verchot et al. 2012).

1.5.6. Forest lands converted to grasslands (IPCC, 2006)

This section summarizes the IPCC (2006) guidelines for estimating and reporting the GHG inventory for the category Forest lands converted to Grasslands, specifically Forest converted to Grassland, for the main carbon pools under the three tiers of complexity.

Conversion from forests to grassland always involves the transfer of carbon from one pool to another, such as from the above-ground biomass of trees to the dead wood pool or from the below-ground biomass to the soil. Estimating CO_2 emissions from forest-to-pasture conversion requires a two-phase approach, in which there is an abrupt change in the land cover generally associated with clearing and burning activities (i.e. Phase 1), and one gradual loss of carbon occurring during a transition period to a new ‘equilibrium’ state (i.e. Phase 2). The default value for the transition period following the conversion to the new steady-state category is 20 years, within which Phase 1 methods should be implemented during the year of conversion and Phase 2 methods for the subsequent 19 years.

During the transition period in Phase 2 C pools can show a non-linear pattern of C loss that can be applied to accurately estimate the annual emission or removal by the specific pool. Under the IPCC Tier 1 approach, it is assumed that all biomass (i.e. above- and below-ground) and dead organic matter are completely removed during deforestation, and that the C stored in these pools is emitted immediately. There will be no transfer from the above-ground biomass to the dead organic matter pools or from the below-ground biomass to the soil.

Under Tier 2 and Tier 3 approaches, however, it is a good practice to establish the pattern of C loss – or accumulation – occurred during the transition period (Phase 2), using country-specific estimates of biomass and emission/removals factors due to land conversion and including the gradual decomposition of below-ground forest biomass and forest dead organic matter. Specifically for the Tier 3 approach, the inventory will include statistically-based models that shall be validated with independent information from country- or region-specific field locations that are representatives of the site.

1.6. Perspectives for Colombia within REDD+ initiatives

Among the countries that currently are developing its REDD+ strategy, Colombia has made a significant progress by publishing methodologies and results concerned with the modelling of deforestation (Cabrera et al., 2011), the characterization of agents and drivers of deforestation (González et al., 2014), and emission factors that include estimates on carbon content in the natural forests of the country and protocols for its measurement (Navarrete et al., 2011; Phillips et al., 2014; Yepes et al., 2011a), as part of the improvements that the country has done to strengthen its capacity to implementing REDD+. The generation and flow of new and high-quality information to estimate CO_2 emissions due to deforestation within the frame of REDD+, will represent better incentives received by the country (Herold et al., 2012) and, in turn, more opportunities to reduce GHG emissions (Gibbs et al., 2007).

Nevertheless, Colombia needs to continue building its capacities to support REDD+ initiatives by improving the accuracy of the information required to estimate GHG emissions from land-use conversion across the country. The use of region-specific, highly-accurate information to estimate CO_2 emissions associated with LULC changes is crucial, because of its direct implications on the estimates of the opportunity costs of REDD+ projects as “the difference in net earnings from conserving or enhancing forests versus converting them to other, typically more valuable, land-uses” (World Bank, 2011).

CO_2 emissions from below-ground biomass, dead wood and soil C pools in deforested areas are poorly quantified, and the development of highly-accurate information and methodology describing the C dynamics after land-use conversion is required to reduce the uncertainty in CO_2 emission estimations and to improve the accuracy in REDD+ accounting.

Chapter 2: Site description

2. Site description

The study took place in two sub-regions of the Colombian Amazon with differences in terms of deforestation risk (i.e. high- and low-deforestation risk sub-regions, as defined by the Colombian government; González et al., 2014), and management practices related to grazing intensity. According to Mahecha et al. (2002), the carrying capacity of pastures in the Colombian Amazon is 0.8-1.0 heads of forage-fed livestock per hectare (HFFL). Therefore, for this study I defined the high- and low-grazing intensity areas (hereafter HG and LG, respectively) as those pastures in which the number of HFFL per hectare is above and below 1.0 head of livestock ha^{-1}, respectively. As Figure 2.1 shows, HG and LG sub-regions coincide with the high- and low-deforestation risk sub-regions of the Colombian Amazon. High-grazing intensity management practices are evident in HG, where pastures cover ~900,000 ha from a total area of 18,237,519 ha (Ideam, 2014) and cattle density by 2013 was 1,777,549 HFFL, averaging 2.00 HFFL ha^{-1}, according to the National Livestock Inventory of Colombia (Fedegan, 2013). By contrast, pastures in LG in 2013 covered ~150,000 ha from a total area of 23,387,251 ha (Ideam, 2014), and cattle density by the same year was 5,328 HFFL (Fedegan, 2013), averaging 0.03 HFFL ha^{-1}.

HG is located in the west of the Colombian Amazon where the major land forms are low-gradient foot slopes and dissected plains, extending eastward between 800-200 m above sea level, and the predominant soils are Haplic Ferralsols and Haplic Acrisols, respectively. However, Ferralsols occur only in a relatively small portion of the western side of HG, in the transition of the Andean and Amazon regions of Colombia. Mosquera et al. (2012) found no significant differences in total soil organic C and the content of other elements in pastures of the same age located in Ferralsols and Acrisols sites within HG. LG is located in the east of the Colombian Amazon where the land forms are dominated by dissected plains between 90-80 m above sea level, and the soils are mainly Haplic Acrisols (van Engelend and Dijkshoorn, 2013). For this study mean annual precipitation and temperature were calculated as the average between 1970 and 2013 in HG, and between 1972 and 2012 in LG. Mean annual precipitation and mean annual temperature in HG are 3723.9 ± 408.6 mm and 26.6 ± 0.6 °C, respectively, and in LG are 3351.1 ± 341.7 mm and 25.9 ± 0.4 °C, respectively. On average evapotranspiration never exceeds precipitation in either sub-regions. The dominant forest within both sub-regions is the Tropical Moist Forest, which extends over ~39 million hectares, and stores 136.6 ± 4.9 Mg C ha^{-1} and 27.5 ± 0.9 Mg C ha^{-1} in the above- and below-ground biomass, respectively (Phillips et al., 2014).

Pastures are the predominant post-deforestation land cover across the whole Colombian Amazon and are mostly located in HG (Cabrera et al., 2011). According to Bowman et al. (2012), up to 80% of the pasture area in the Colombian Amazon is occupied by farms implementing the extensive cattle ranching system. Farmers from both sub-regions include the use of fire to eliminate the plant material left by deforestation and, in the case of HG, to control the expansion of secondary vegetation. However, whereas in HG the use of machinery to remove the unburned woody debris is a common management practice, it is rarely implemented in LG due to limitations to move heavy equipment to remote areas within the forest. Management in HG also include the use of introduced grasses such as *Brachiaria humidicola* or *B. decumbens*, or the combination of these species with legume species such as *Arachis pintoi* or *Desmodium ovalifolium*, occasionally combined with the application of fertilizers, as a practice to increase cattle liveweight production in up to 500 kg ha^{-1} yr^{-1}

(Fisher et al., 1994; Alarcón and Tabares, 2007; Mosquera et al., 2012). As Fisher et al. (1994) pointed out, the combination of introduced grasses with a nitrogen-fixing legume and the sporadic use of fertilizers, can contribute to increase nutrient cycling, improve beef production, and increase soil biological activity in the pasture areas. Evidences of pasture degradation, however, can be detected in most of the pasture areas in HG after 8 to11 years of establishment, as a result of overgrazing, mechanized tillage, and, in general, the low maintenance of the introduced grasses (Martínez and Zinck, 2004). During the pasture degradation, organic matter input and pasture productivity decline, while pasture cover is reduced and soil erosion and compaction increase (Fonte et al., 2014). In LG, on the other hand, the use of introduced grasses is uncommon, and it is usual to find pasture areas where grasses (C4 vegetation) are mixed with shrubs and trees (C3 vegetation). Figure 2.2 presents a comparison between two 20-year-old pastures in HG and LG that highlight the different landscapes resulting from the implementation of particular management practices in each sub-region following deforestation.

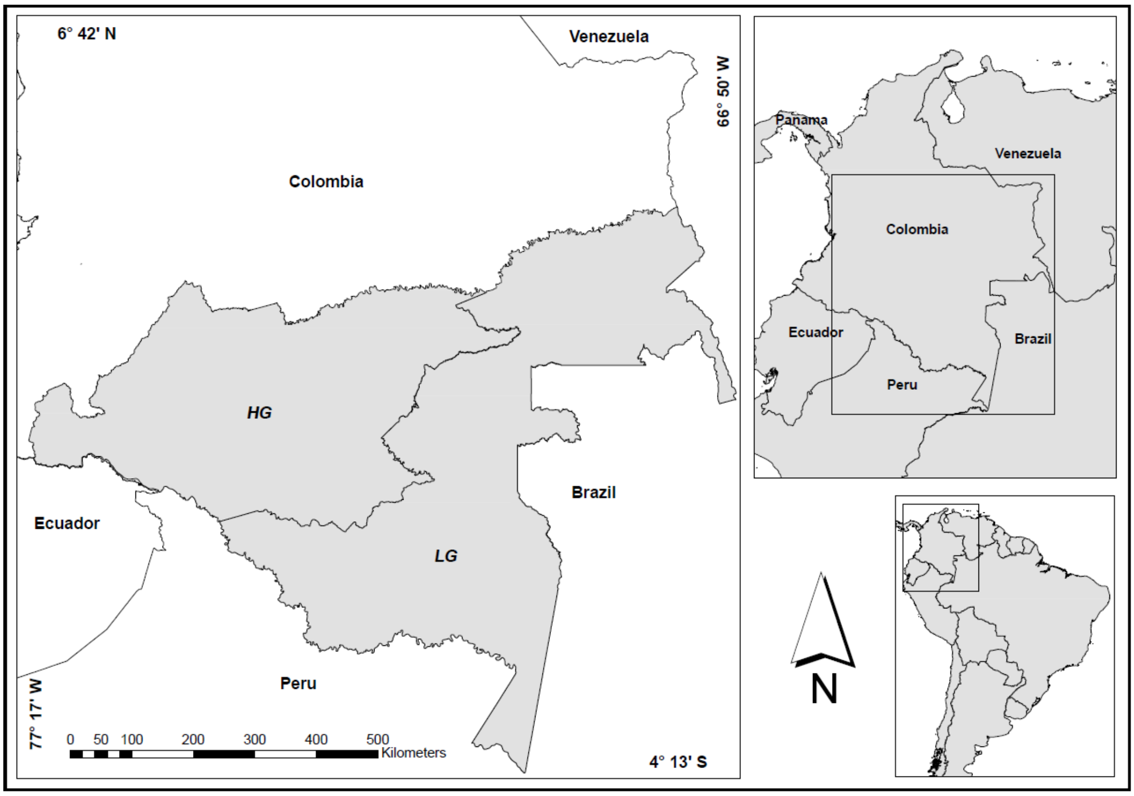

Figure 2.1 Location of the high- and low-grazing intensity sub-regions in the Colombian Amazon (HG and LG, respectively). Panels on the right side show the location of Colombia within South America.

Figure 2.2 20-year-old pastures in the a) high-grazing intensity (HG) and b) low-grazing intensity (LG) sub-regions of the Colombian Amazon.

Chapter 3: The chronosequence approach

3. The chronosequence approach

The chronosequence approach was implemented to monitor changes in C stocks in soils, dead wood and below-ground biomass C pools to improve the information required to establish the long-term CO_2 emissions associated with conversion from forests to pasture in both sub-regions of the Colombian Amazon. Thus, one chronosequence of six sites representing forest-to-pasture conversion was identified in each of the two sub-regions of the Colombian Amazon, through the use of satellite images, official maps of deforestation in Colombia, and interviews with landowners and local people, who also provided information about the land-use history and management.

Both chronosequences covered a period of 20 years of forest-to-pasture conversion, with the primary forest as the reference point (i.e. 0 years since deforestation; Figure 3.1). The chronosequences were established in areas of pasture that previously had forest cover and have been kept as pastures ever since the deforestation event. In order to have a more representative sequence of the forest-to-pasture conversion, the chronosequences included the first stages of the pasture establishment, corresponding to areas recently deforested (i.e. around one year since deforestation) and areas recently burned to clean the biomass remaining from deforestation (i.e. one to two years after deforestation). The rest of the chronosequence was established in pastures of around 5, 12 and 20 years old since deforestation in both sub-regions (Table 3.1).

The landscape of the area devoted to cattle ranching in HG is mainly dominated by extensive pastures established during the last decades, with the sporadic occurrence of small (< 1.0 ha), scattered patches of remnant forest. The distance from the selected pasture areas to the closest forest in HG was < 1.0

km in the case of the 1- and 2-year-old pastures, ~4.0 km in the case of the 5-year-old pasture and > 30 km in the case of the 12- and 20-year-old pastures. In contrast, the areas dedicated to cattle ranching in LG are embedded within the primary forest matrix, so selected pasture areas from all stages of the chronosequence in this sub-region are surrounded by the adjacent forest.

Table 3.1 Location and time after deforestation of the identified sites comprising both chronosequences at the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions in the Colombian Amazon.

Site	Post-deforestation time (yr)	Location (Lat., Long.)
HG		
Primary forest	0.0	01°39’41.8’’N, 75°37’41.8’’W
1-yr-old pasture	0.6	01°39’53.8’’N, 75°37’30.0’’W
2-yr-old pasture	1.7	01°39’38.7’’N, 75°37’28.1’’W
5-yr-old pasture	5.4	01°44’02.8’’N, 75°38’03.1’’W
12-yr-old pasture	12.0	01°27’30.4’’N, 75°37’34.8’’W
20-yr-old pasture	20.0	01°27’22.5’’N, 75°37’57.9’’W
LG		
Primary forest	0.0	04°10’10.1’’S, 69°55’17.0’’W
1-yr-old pasture	0.7	04°09’53.6’’S, 69°54’50.0’’W
2-yr-old pasture	1.5	04°09’49.3’’S, 69°55’02.8’’W
5-yr-old pasture	5.0	04°09’51.5’’S, 69°54’41.2’’W
12-yr-old pasture	12.0	04°09’58.8’’S, 69°54’10.5’’W
20-yr-old pasture	20.0	04°10’02.2’’S, 69°55’17.9’’W

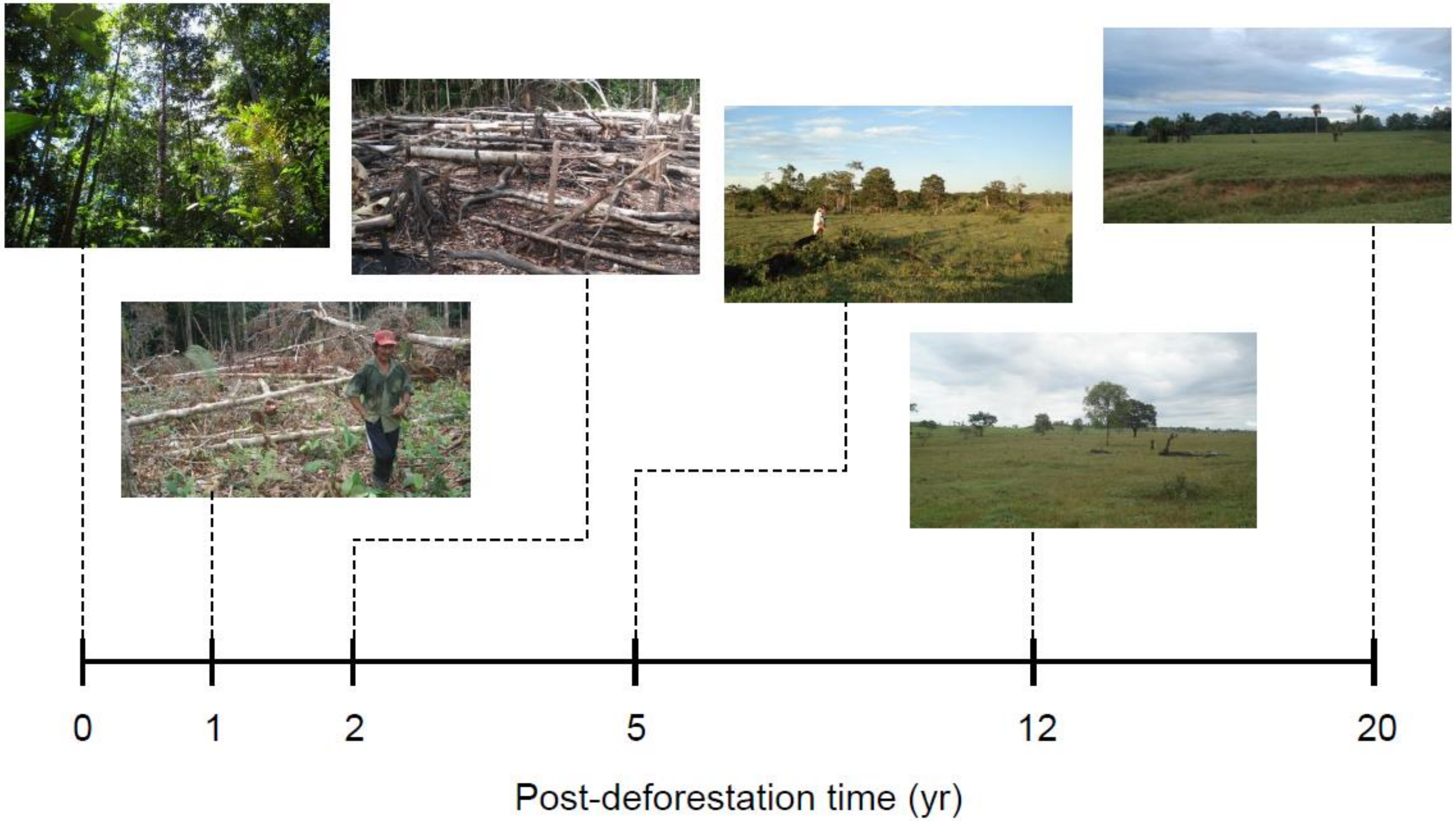

Figure 3.1 Chronosequence approach representing 20 years of forest-to-pasture conversion in the Colombian Amazon.

Chapter 4: Conversion from forests to pastures in the Colombian Amazon leads to contrasting soil carbon dynamics depending on land management practices

4. Conversion from forests to pastures in the Colombian Amazon leads to contrasting soil carbon dynamics depending on land management practices

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4.1. Abstract

Strategies to mitigate climate change by reducing deforestation and forest degradation (e.g. REDD+) require country- or region-specific information on temporal changes in forest carbon (C) pools to develop accurate emission factors. The soil C pool is one of the most important C reservoirs, but is rarely included in national forest reference emission levels due to a lack of data. Here I present the soil organic C (SOC) dynamics along 20 years of forest-to-pasture conversion in two sub-regions with different management practices during pasture establishment in the Colombian Amazon: high-grazing intensity (HG) and low-grazing intensity (LG) sub-regions. I determined the pattern of SOC change resulting from the conversion from forest (C3 plants) to pasture (C4 plants) by analysing total SOC stocks and the natural abundance of the stable isotopes ^{13}C along two 20-year chronosequences identified in each sub-region. I also analysed soil N stocks and the natural abundance of ^{15}N during pasture establishment. In general, total SOC stocks at 30 cm depth in the forest were

similar for both sub-regions, with an average of 47.1 ± 1.8 Mg C ha^{-1} in HG and 48.7 ± 3.1 Mg C ha^{-1} in LG. However, 20 years after forest-to-pasture conversion SOC in HG decreased by 20%, whereas in LG SOC increased by 41%. This net SOC decrease in HG was due to a larger reduction in C3-derived input and to a comparatively smaller increase in C4-derived C input. In LG both C3- and C4-derived C input increased along the chronosequence. N stocks were generally similar in both sub-regions and soil N stock changes during pasture establishment were correlated with SOC changes. These results emphasize the importance of management practices involving low-grazing intensity in cattle activities to preserve SOC stocks and to reduce C emissions after land-cover change from forest to pasture in the Colombian Amazon.

4.2. Introduction

In the tropics, soil organic carbon (SOC) represents 30-60% of the total C stored in forest ecosystems (Don et al., 2011), and some studies have reported soil C emissions of 0.2 Pg C yr^{-1} due to deforestation in the tropics (Houghton, 1999; Achard et al., 2004). Most studies suggest that C content in forest soils decrease with depth. For instance, Malhi et al. (2009) reported SOC stocks at three sites of the Brazilian Amazon of 74-127 Mg C ha^{-1} in the top 1 meter depth, 28-63 Mg C ha^{-1} in the 1-2 m layer and 19-37 Mg C ha^{-1} in the 2-3 m layer. Conant et al. (2001) found that 64% of the SOC is located in the top 50 cm, and Moraes et al. (1996) reported a total SOC content of 47 Pg C in the Brazilian Amazon, of which 21 Pg C are stored in the top 20 cm. Despite of the importance of soils as one of the largest organic C pools in tropical forests, the impact of deforestation on this reservoir has been barely quantified (Don et al., 2011) and more information is required.

Management practices implemented in pastures are as important as land-cover changes in determining SOC dynamics and C emissions from human-modified tropical ecosystems (Berenguer et al., 2014), and among them, grazing intensity can significantly contribute to preserve or reduce SOC stocks (Uhl, et al., 1988). Depending on management, soils under grassland cover may act either as a C sink or source (Fearnside and Barbosa 1998; Dias-Filho et al., 2000). For example, Veldkamp (1994) found a net decrease in soil C content of 2-18% in degraded pastures 25 years after deforestation, and Fearnside (1997) found net average emissions of 3.9 Mg C ha^{-1} in 10-11 year-old pastures in poorly managed grasslands. On the other hand, Fisher et al. (1994) found that C content is higher in soils where management practices include the combination of introduced grasses with a nitrogen-fixing legume compared to local grasses in Colombia, and Moraes et al. (1996) found an increase of 17% and 19% in SOC content in two 20-year-old chronosequences established in well managed pastures in Brazil.

Strategies for avoiding deforestation such as REDD+ could help to reduce anthropogenic C emissions by around 0.8-1.3 Pg C yr-1 (Don et al., 2011). According to the United Nations Framework Convention on Climate Change (UNFCCC) decisions (UNFCCC, 2010; UNFCCC, 2011; UNFCCC, 2014), countries willing to access result-based payments through REDD+ activities require developing forest reference emission levels (FRELs) as benchmarks for assessing country's performance (FAO, 2014). In general, FRELs comprise the emission factors, representing the emissions/removals of all important GHG associated with land-cover conversion in all relevant C pools (i.e. total changes in C stocks), and the activity data, referring to the size of a deforested or degraded area (Verchot et al., 2012). Despite its importance as a C reservoir (Marin-Spiotta et al. 2009; Powers et al., 2011), the soil C pool was not included in any of the FRELs submitted to the UNFCCC in 2014 by Brazil, Colombia, Ecuador, Guyana, Malaysia and Mexico due to data scarcity (Conafor, 2014; GFC, 2014; MAE, 2014; MMA, 2014; MNRE, 2014). Colombia, in particular, included the above- and below-ground biomass C pool in forests in its FREL, but neither the SOC stock nor its change after conversion from forest to any post-deforestation land-use category were included (MADS, 2014). However, under the UNFCCC Stepwise Approach (UNFCCC, 2012), countries have the option to improve their initial FRELs by incorporating high-quality data, improved methodologies and additional C pools developed from country- or region-specific information and field measurements following an IPCC Tier 3 approach.

Development of emission factors to describe changes in SOC due to land-cover conversion can be done by using robust methods such as chronosequences (IPCC, 2006) combined with soil analysis of total C and nitrogen (N), and the natural abundance of the stable isotopes ^{13}C and ^{15}N (Marin-Spiotta et al., 2009; Elmore and Asner, 2006; López-Ulloa et al., 2005; Desjardins et al., 2004; Camargo et al., 1999; Moraes et al., 1996; Piccolo et al., 1996). Differences in discrimination against ^{13}C between C3 and C4 plants due to variations in their photosynthetic pathways (Dawson et al., 2002; Balesdent and Mariotti, 1996), as well as differences in the soil isotope composition of ^{15}N ($\delta^{15}N$) associated with the rate of atmospheric N_2 fixation (Piccolo et al., 1994), can be used to establish the pattern of change in SOC and N stocks resulting from the conversion from forest to pasture and the impact of the management practices (e.g. grazing intensity) on C emissions or removals.

In this chapter I present new Tier 3 information and emission factors on SOC pool and its dynamics during 20 years of forest-to-pasture conversion under different management practices in the Colombian Amazon. In my study I addressed the following general question: to what extent land-cover change from forest to pasture and subsequent land management practices affect SOC dynamics in the Colombian Amazon? Therefore, I aimed to better quantify SOC stocks and changes with forest-to-pasture conversion in support of REDD+ initiatives. Specifically, my objectives were to:

1. Quantify and describe the SOC dynamics in two sub-regions of the Colombian Amazon with high- and low-grazing intensity after 20 years of forest-to-pasture conversion.
2. Quantify the relative input of C derived from C3 and C4 plants to the total SOC, and its variation with soil depth and pasture age in both sub-regions.
3. Quantify the N stocks during 20 years of forest-to-pasture conversion in both sub-regions of the Colombian Amazon.
4. Determine the emission factors of the SOC pool in both sub-regions according to IPCC (2006), by applying region-specific equations developed in this study describing SOC dynamics.

4.3. Materials and methods

4.3.1. Soil sampling and laboratory analysis

One 200 m transect was established at every stage of each chronosequence in both sub-regions. The starting point of each transect was randomly selected and 11 sampling points were established every 20 m. At every sample point the litter was removed and soil samples were collected at 0-10 cm, 10-20 cm and 20-30 cm depth using an AMS Soil Core Sampler. A total of 33 samples were collected at every stage of each chronosequence, for a grand total of 198 samples per sub-region. Additionally, six samples of leaf litter were collected from both forests and another six samples of plant material were collected from the 20-yr pastures at both sub-regions, in order to establish reference values for isotope analyses. Soil samples were used to determine dry weight, bulk density, and C and N content in the Analytical Services Laboratory at the International Center of Tropical Agriculture (CIAT/CGIAR). Soil samples were oven-dried at 60 °C, and sub-samples of 18.0 mg were taken after soils were ground and passed through a 2 mm sieve. Total organic C and N contents were determined by the dry combustion method (at 900 °C), using a PE 2400 Series II CHNS/O Analyzer calibrated with certified acetanilide (C_8H_9NO).

In order to compare SOC and N stocks from forest and pasture areas in an equivalent weight (Fearnside and Barbosa, 1998), corrections for soil compaction due to cattle trampling were applied according to the methodology proposed by Ellert and Bettany (1995). Total SOC stocks (SOC_T; Mg C ha^{-1}) to 30 cm depth were thus calculated as the sum of the SOC stocks at 0-10 cm, 10-20 cm and 20-30 cm depth, obtained as the product of the C content, bulk density and soil thickness at every depth j, as follows:

$$SOC_T = \sum\left(C_j \times \rho_j \times L_j\right) \qquad (4.1)$$

where C_j (kg C kg^{-1} soil) is the C content, ρ_j (g cm^{-3}) is the soil bulk density, and L_j (cm) is the soil thickness adjusted for compaction at the depth j. Total N stocks (N_T; Mg N ha^{-1}) to 30 cm depth were calculated similar to SOC_T, but replacing C_j by N_j in Eqn. (4.1) as the N content at the depth j (kg N kg^{-1} soil).

4.3.2. Isotope analysis

Soil sub-samples and plant material were used to establish the stable isotope composition of ^{13}C ($\delta^{13}C$) and ^{15}N ($\delta^{15}N$) at forests and pastures of all stages in both sub-regions. The analyses of $\delta^{13}C$ and $\delta^{15}N$ were carried out in the Stable Isotope Facility at the University of California–Davis, using an Elementar Vario EL Cube elemental analyser, interfaced to a PDZ Europa 20-20 isotope ratio mass spectrometer (http://stableisotopefacility.ucdavis.edu/index.html). Soil $\delta^{13}C$ (‰) results are relative to the VPDB standard, and are expressed as:

$$\delta^{13}C = \left(\frac{R_{sample} - R_{std}}{R_{std}}\right) \times 1000 \quad (4.2)$$

where R_{std} is the standard ^{13}C:^{12}C ratio. Variation in $\delta^{13}C$ composition due to differences in the photosynthetic pathway between C3 vegetation (mostly trees and shrubs) and C4 vegetation (mostly tropical grasses), and the resulting differentiated discrimination against ^{13}C, can be used to determine the fraction of soil C derived from C3-plants (C3-C) and C4-plants (C4-C) to total SOC as a result of the forest-to-pasture conversion. The fraction of C4-C and C3-C at each depth j at every stage of the forest-to-pasture conversion was calculated using a two-compartment mixing model (Balesdent and Mariotti, 1996):

$$F_{C4-Cj} = \left[\frac{(\delta_{Sj} - \delta_F)}{(\delta_{Vp} - \delta_{Vf})}\right] \times SOC_J \quad (4.3)$$

where $F_{C4\text{-}Cj}$ (Mg C ha^{-1}) is the fraction of C4-C, δ_{Sj} is the $\delta^{13}C$ of the soil sample, δ_F is the average $\delta^{13}C$ of the soil at forests (-27.5 ± 0.7‰, in this study), δ_{Vp} is the average $\delta^{13}C$ of C4 plant material from pastures (-17.4 ± 2.0‰, in this study) and δ_{Vf} is the average $\delta^{13}C$ of the forest litterfall (-31.2 ± 0.7‰, in this study). Moreover, the fraction of C3-C at the same depth j ($F_{C3\text{-}Cj}$) was calculated as:

$$F_{C3-Cj} = SOC_J - F_{C4-Cj} \quad (4.4)$$

Total C4-C_T and C3-C_T fractions to 30 cm depth were thus calculated as the sum of $F_{C4\text{-}Cj}$ and $F_{C3\text{-}Cj}$, respectively, at each depth j. Soil isotope composition of $\delta^{15}N$ (‰) was calculated as:

$$\delta^{15}N = \left[\left(R_{sample} - R_{standard}\right) - 1\right] \times 1000 \tag{4.5}$$

where R represents the sample and standard ^{15}N:^{14}N ratio. Atmospheric $R_{standard}$ = 0.0036765 (Piccolo et al., 1994).

4.3.3. Statistical analysis

All data were tested for normality and homogeneity of variance. Analysis of variance (ANOVA) were performed to establish differences in mean soil bulk density, SOC_T and N_T stocks, C:N ratio, $\delta^{13}C$, $\delta^{15}N$, C3-C and C4-C among the six stages of each chronosequence, and between similar stages of the forest-to-pasture conversion between the two sub-regions. Additional ANOVA tests were performed to determine differences of these variables with increasing depth within every stage of each chronosequence. Post hoc Tukey-HSD tests were made when significant differences were found. Linear, exponential and logarithmic regression analyses were tested to establish the pattern of variation of SOC_T stocks, C3-C, and C4-C to a depth of 30 cm, and within every depth (i.e. 0-10 cm, 10-20 cm and 20-30 cm), during the forest-to-pasture conversion at both sub-regions, and the model which best fitted the data in every case was selected. Selection of the best model depended on the resulted r^2 and P-value of each model. Pearson's correlation analyses between SOC and N stocks were performed to assess similarities in the pattern of variation along the forest-to-pasture conversion at both sub-regions. All the analyses were evaluated with a significance level of $P < 0.05$. The equations selected to describe the pattern of variation of SOC_T to 30 cm depth were used to establish the emission factors for the total change in SOC stocks due to the forest-to-pasture conversion after 20 years of deforestation at both sub-regions, according to the IPCC (2006) Tier 3 approach. Thus, the results expressed in Mg C ha^{-1} were converted to tonnes of CO_2 equivalent per hectare (t CO_2e ha^{-1}) by using the C-to-CO_2e conversion factor of 44/12 (UNFCCC, 2012), and were assessed at a 95% confidence interval.

4.4. Results

4.4.1. Bulk density

Soil bulk density (ρ) was significantly different ($P < 0.05$) during the forests-to-pasture conversion in each sub-region, as well as between pastures with similar age from the different sub-regions. While in HG ρ progressively increased with time within the 30 cm profile, in LG it remained almost constant at the same depth (Table 4.1). A total increase in soil ρ in the top 0-10 cm layer of 27% in HG and 14% in LG with respect to the original forest were registered at the end of both chronosequences. Values of ρ at the 0-10 cm layer were significantly different from those in the deeper 10-20 cm and 20-30 cm layers at the forest and at the 1- and 2-year-old pastures in LG. In contrast, there were no differences with depth along the chronosequence in HG, except for the 20-year-old pasture (Table 4.2). Soil ρ showed a similar increasing pattern across the soil profile and along the chronosequence in HG, with a peak during the 12-year-old pasture at the 10-20 cm and 20-30 cm layers, and during the 20-year-old pasture at the 0-10 cm layer. After 12 years of pasture establishment in HG there was a pronounced decline in ρ at the 20-30 cm layer. For LG, while soil ρ increased along the whole chronosequence in the top 0-10 cm layer, it tended to decrease with time since disturbance in the 10-20 cm and the 20-30 cm layers (Figure 4.1a).

4.4.2. Soil organic carbon (SOC) stocks

Total soil organic carbon (SOC_T) stocks down to 30 cm depth in the primary forest were similar between both sub-regions, with an average of 47.1 ± 1.8 Mg C ha^{-1} in HG and 48.7 ± 3.1 Mg C ha^{-1} in LG. After deforestation SOC_T stocks significantly varied with pasture age in both sub-regions, although the patterns of soil C change were notably opposite between HG and LG (Table 4.1). During the first year of the forest-to-pasture conversion, SOC_T increased by the same magnitude in HG and LG (53.0 ± 2.7 Mg C ha^{-1} and 55.9 ± 3.6 Mg C ha^{-1}, respectively), but after the use of fire the variation of SOC_T at pastures was significantly different between HG and LG (Table 4.1). In HG, SOC_T was 37% higher at the 2-year-old pasture, and then 28%, 20% and 20% lower at the 5-, 12- and 20-year-old pastures, respectively, compared to the original forest. Oppositely, SOC_T in LG was 11% lower at the 2-year-old pasture with respect to the forest, and 18%, 13% and 41% higher at the 5-, 12- and 20-year-old pastures, respectively.

The vertical distribution of SOC showed significant differences among forests and pastures at most stages in both sub-regions, except for the 1- and 2-year-old pastures in LG (Table 4.2). As expected, the largest proportion of SOC was found in the top 0-10 cm layer in all stages of the forest-to-pasture conversion in both sub-regions, followed by a reduction towards the 10-20 cm and 20-30 cm layers, with the exception of the 1-year-old pasture in LG (Figure 4.1b). After five years of forest-to-pasture conversion in HG, the proportion of SOC in the top 0-10 cm soil layer increased to the detriment of the proportion in the deeper layers. This pattern was not registered in LG, where the proportions of SOC were similar across the soil profile during the post-deforestation time (Figure 4.1b). In both sub-regions SOC was higher at the end of the 20 years of pasture establishment in the top 0-10 cm layer compared to the forest.

4.4.3. Nitrogen (N) stocks and C:N ratio

Total soil nitrogen (N_T) stocks at 30 cm depth were only significantly different between HG and LG at the 2-year-old pastures. The average N_T content in the soil of forests at this depth was 4.19 ± 0.12 Mg N ha^{-1} in HG, and 3.72 ± 0.14 Mg N ha^{-1} in LG, but the values increased to 5.76 ± 0.42 Mg N ha^{-1} and 4.76 ± 0.23 Mg N ha^{-1} after two and one years since deforestation, respectively (Table 4.1). While N_T stocks at 30 cm depth were significantly different in almost all stages of the chronosequence in LG, it was only significantly different at the 2-year-old pasture in HG. There were also significant differences in N stocks among the soil layers in all stages of the forest-to-pasture conversion in HG, and in the forest, 5- and 20-year-old pastures in LG (Table 4.2). As in SOC, the largest proportion of soil N was found in the top 0-10 cm layer in both sub-regions, although in HG the difference between the 0-10 cm layer and the deeper layers increased to ~50% after the fifth year of pasture establishment (Figure 4.1c).

Variation in soil N_T at 30 cm depth and in N stocks at 0-10 cm, 10-20 cm and 20-30 cm depth were highly correlated with variation in SOC_T at 30 cm depth and with SOC stocks at every soil layer, respectively (Figure 4.2). These results indicate both that losses in C stocks occurring during the forest-to-pasture conversion are concomitant with losses in N stocks, as well as gains in soil N are associated with gains in soil C. C:N ratios ranged from 9.4 to 11.4 in HG and from 11.5 to 19.2 in LG, and were significantly higher in the forest and the 5-, 12- and 20-year-old pastures in LG compared to the same stages in HG (Table 4.1). There were also significant differences in C:N ratios with depth in most of the stages of the forest-to-pasture conversion in both sub-regions (Table 4.2), but whereas in HG C:N tended to increase with depth and keep around constant along the forest-to-pasture conversion, in LG it tended to decrease with depth and to increase with time (Figure 4.1d).

Table 4.1 Mean soil bulk density (g cm^{-3}), total soil organic carbon (SOC_T; Mg C ha^{-1}) and total nitrogen (N_T; Mg N ha^{-1}) stocks, C:N ratio, $\delta^{13}C$ (‰) and $\delta^{15}N$ (‰) to a depth of 30 cm. Values in parenthesis represent the standard error of the mean (n = 11). Letters indicate significant differences among stages within each sub-region ($P < 0.0001$; $P = 0.02$ for differences in ρ in LG), and the asterisk symbol represents differences between the two sub-regions at the same stage of the forest-to-pasture conversion ($P < 0.0001$). HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

Forest-to-pasture conversion	Bulk density		SOC_T		N_T		C:N ratio		$\delta^{13}C$		$\delta^{15}N$	
HG												
Primary forest	0.75	(0.03)^{a*}	47.1	(1.8)^{bc}	4.19	(0.12)^{a}	11.2	(0.2)^{a*}	-27.6	(0.1)^{a*}	6.8	(0.2)^{a*}
1-yr pasture	0.75	(0.02)^{a*}	53.0	(2.7)^{c}	4.65	(0.19)^{a}	11.4	(0.2)^{a}	-27.7	(0.1)^{a*}	6.5	(0.1)^{ab*}
2-yr pasture	0.82	(0.03)^{ab*}	64.6	(3.7)^{d*}	5.76	(0.42)^{b*}	11.4	(0.3)^{a}	-27.6	(0.3)^{a*}	5.8	(0.2)^{b*}
5-yr pasture	0.89	(0.04)^{bc}	33.9	(2.4)^{a*}	3.62	(0.28)^{a}	9.4	(0.2)^{c*}	-24.2	(0.3)^{b}	7.5	(0.2)^{c*}
12-yr pasture	1.01	(0.02)^{c}	37.5	(1.8)^{ab*}	3.80	(0.14)^{a}	9.9	(0.3)^{bc*}	-24.3	(0.3)^{b}	8.4	(0.2)^{d*}
20-yr pasture	0.96	(0.02)^{c}	37.9	(1.6)^{ab*}	3.69	(0.17)^{a}	10.3	(0.1)^{b*}	-21.5	(0.2)^{c*}	8.9	(0.1)^{d*}
LG												
Primary forest	1.03	(0.03)^{ab*}	48.7	(3.1)^{a}	3.72	(0.14)^{ab}	13.0	(0.5)^{a*}	-24.8	(0.1)^{a*}	11.8	(0.3)^{ab*}
1-yr pasture	0.96	(0.02)^{ab*}	55.9	(3.6)^{ab}	4.76	(0.23)^{c}	11.7	(0.4)^{a}	-25.1	(0.2)^{a*}	12.0	(0.2)^{ab*}
2-yr pasture	1.06	(0.03)^{b*}	43.5	(3.6)^{a*}	3.73	(0.19)^{ab*}	11.5	(0.4)^{a}	-25.3	(0.1)^{a*}	11.8	(0.2)^{ab*}
5-yr pasture	0.95	(0.03)^{a}	57.4	(3.7)^{ab*}	4.51	(0.20)^{bc}	12.6	(0.3)^{a*}	-24.2	(0.3)^{a}	11.8	(0.3)^{ab*}
12-yr pasture	1.04	(0.02)^{ab}	55.0	(1.7)^{ab*}	4.30	(0.11)^{abc}	12.9	(0.5)^{a*}	-25.1	(0.4)^{a}	12.5	(0.3)^{a*}
20-yr pasture	1.01	(0.03)^{ab}	68.8	(5.7)^{b*}	3.58	(0.24)^{a}	19.2	(0.7)^{b*}	-24.7	(0.2)^{b*}	11.1	(0.3)^{b*}

Table 4.2 Mean soil bulk density (g cm^{-3}), soil organic carbon (SOC; Mg C ha^{-1}) and nitrogen (N; Mg N ha^{-1}) stocks, and C:N ratio to 0-10 cm, 10-20 cm and 20-30 cm depth. Values in parenthesis represent the standard error of the mean (n = 11). Letters indicate significant differences among depths within each stage of the forest-to-pasture conversion (P < 0.05).

	Forest-to-pasture conversion					
	Primary forest	1-yr pasture	2-yr pasture	5-yr pasture	12-yr pasture	20-yr pasture
High-grazing intensity						
Bulk density						
0-10 cm	0.77 (0.04)	0.80 (0.02)	0.85 (0.03)	0.98 (0.05)	0.96 (0.02)	1.05 (0.01)^{a}
10-20 cm	0.79 (0.04)	0.74 (0.03)	0.81 (0.04)	0.91 (0.10)	1.03 (0.03)	1.00 (0.04)^{a}
20-30 cm	0.70 (0.04)	0.71 (0.03)	0.79 (0.04)	0.79 (0.07)	1.02 (0.05)	0.82 (0.04)^{b}
SOC stock						
0-10 cm	19.8 (1.8)^{a}	22.0 (1.5)^{a}	25.9 (1.3)^{a}	17.6 (1.9)^{a}	17.9 (1.4)^{a}	21.4 (1.3)^{a}
10-20 cm	15.5 (1.1)^{ab}	17.6 (0.9)^{b}	22.2 (2.2)^{ab}	9.4 (0.6)^{b}	11.6 (0.6)^{b}	9.7 (0.3)^{b}
20-30 cm	12.3 (1.0)^{b}	13.7 (0.6)^{c}	16.8 (1.5)^{b}	7.2 (0.5)^{b}	8.3 (0.5)^{c}	7.2 (0.2)^{b}
N stock						
0-10 cm	1.84 (0.16)^{a}	1.98 (0.10)^{a}	2.28 (0.11)^{a}	2.00 (0.23)^{a}	1.85 (0.09)^{a}	2.16 (0.14)^{a}
10-20 cm	1.35 (0.08)^{b}	1.54 (0.06)^{b}	1.88 (0.19)^{ab}	0.94 (0.08)^{b}	1.17 (0.07)^{b}	0.91 (0.03)^{b}
20-30 cm	1.04 (0.08)^{b}	1.16 (0.05)^{c}	1.50 (0.19)^{b}	0.71 (0.06)^{b}	0.81 (0.06)^{c}	0.67 (0.02)^{b}
C:N ratio						
0-10 cm	10.8 (0.3)^{a}	11.1 (0.2)^{a}	11.3 (0.1)	8.9 (0.2)^{a}	9.6 (0.3)	10.0 (0.1)^{a}
10-20 cm	11.4 (0.2)^{ab}	11.4 (0.2)^{ab}	11.8 (0.2)	10.1 (0.3)^{b}	10.1 (0.3)	10.7 (0.2)^{b}
20-30 cm	11.7 (0.3)^{b}	11.8 (0.2)^{b}	11.6 (0.4)	10.2 (0.3)^{b}	10.4 (0.4)	10.8 (0.2)^{b}

Table 4.2 (continuation) Mean soil bulk density (g cm^{-3}), soil organic carbon (SOC; Mg C ha^{-1}) and nitrogen (N; Mg N ha^{-1}) stocks, and C:N ratio to 0-10 cm, 10-20 cm and 20-30 cm depth. Values in parenthesis represent the standard error of the mean (n = 11). Letters indicate significant differences among depths within each stage of the forest-to-pasture conversion (P < 0.05).

	Forest-to-pasture conversion											
	Primary forest		1-yr pasture		2-yr pasture		5-yr pasture		12-yr pasture		20-yr pasture	
Low-grazing intensity												
Bulk density												
0-10 cm	0.83	(0.03)^{a}	0.80	(0.04)^{a}	0.89	(0.06)^{a}	0.91	(0.03)	1.02	(0.02)	0.96	(0.02)
10-20 cm	1.08	(0.04)^{b}	1.00	(0.02)^{b}	1.11	(0.02)^{b}	0.91	(0.02)	1.03	(0.03)	0.97	(0.04)
20-30 cm	1.19	(0.03)^{b}	1.09	(0.02)^{b}	1.17	(0.02)^{b}	1.02	(0.05)	1.07	(0.06)	1.09	(0.05)
SOC stock												
0-10 cm	20.5	(1.4)^{a}	22.5	(1.9)	17.0	(1.6)	23.5	(1.9)^{a}	22.1	(1.1)^{a}	29.2	(2.1)^{a}
10-20 cm	13.6	(1.0)^{b}	15.0	(0.7)	12.6	(0.7)	16.9	(1.1)^{b}	16.8	(0.6)^{b}	19.6	(2.1)^{b}
20-30 cm	11.5	(0.9)^{b}	15.6	(3.5)	11.8	(2.4)	13.7	(0.7)^{b}	13.2	(0.8)^{c}	15.7	(1.7)^{b}
N stock												
0-10 cm	1.34	(0.06)^{a}	1.70	(0.10)	1.31	(0.09)	1.65	(0.11)^{a}	1.53	(0.07)	1.43	(0.09)^{a}
10-20 cm	1.15	(0.06)^{ab}	1.43	(0.04)	1.16	(0.04)	1.43	(0.06)^{ab}	1.33	(0.03)	1.03	(0.07)^{b}
20-30 cm	1.08	(0.05)^{b}	1.46	(0.20)	1.15	(0.13)	1.25	(0.02)^{b}	1.26	(0.10)	0.93	(0.15)^{b}
C:N ratio												
0-10 cm	15.2	(0.6)^{a}	13.1	(0.6)^{a}	12.8	(0.5)^{a}	14.2	(0.4)^{a}	14.9	(0.2)^{a}	20.5	(0.7)
10-20 cm	11.7	(0.4)^{b}	10.4	(0.4)^{b}	10.9	(0.4)^{b}	11.7	(0.3)^{b}	12.9	(0.2)^{b}	18.7	(0.8)
20-30 cm	10.5	(0.5)^{b}	10.1	(0.5)^{b}	9.6	(0.7)^{b}	11.0	(0.5)^{b}	10.8	(0.7)^{c}	18.2	(1.9)

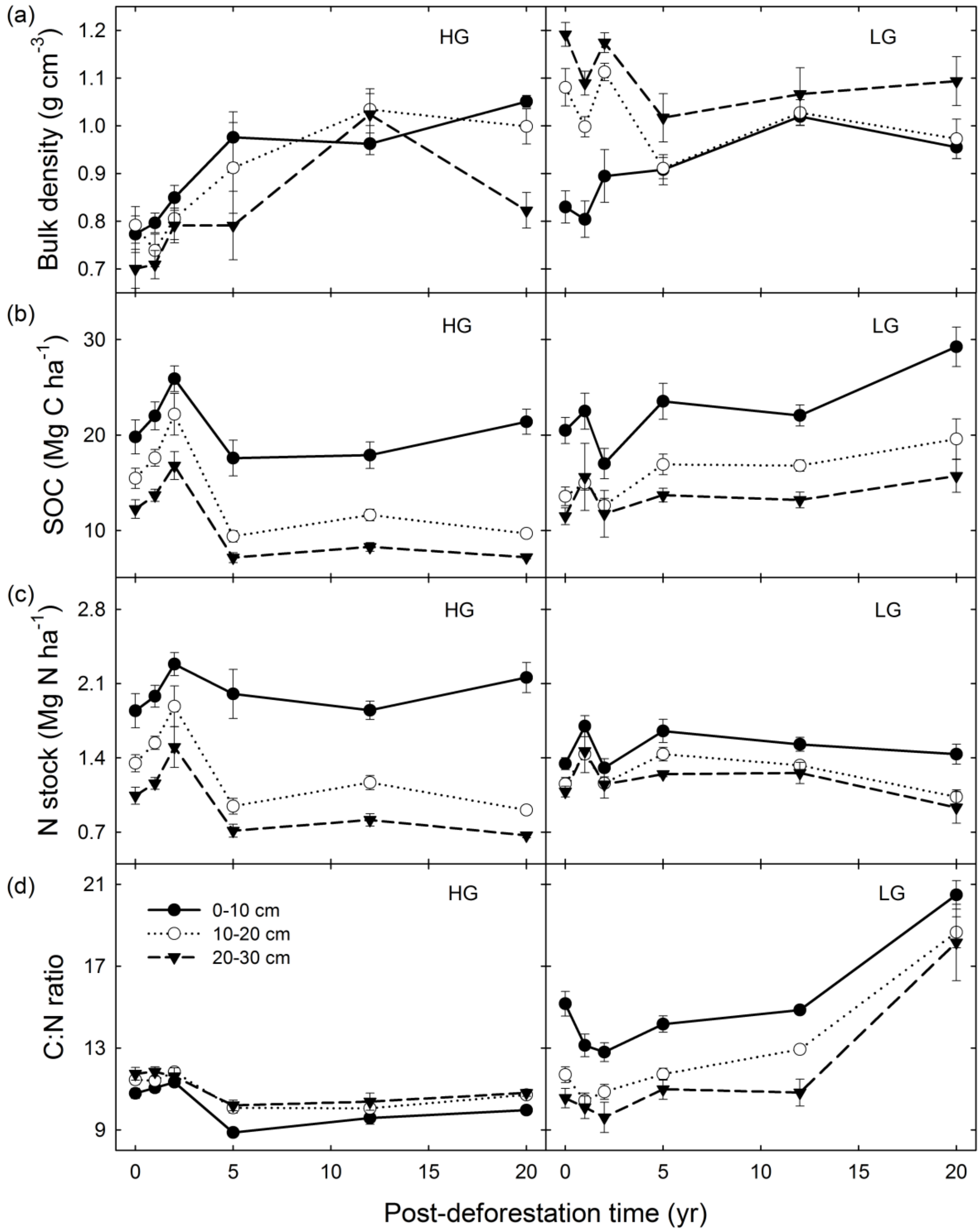

Figure 4.1 Mean a) soil bulk density, b) soil organic carbon (SOC) stocks, c) nitrogen (N) stocks, and d) C:N ratio to 0-10 cm (solid line), 10-20 cm (dotted line), and 20-30 cm (dashed line) depth, during the forest-to-pasture conversion in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Bars represent the standard error of the mean (n = 11).

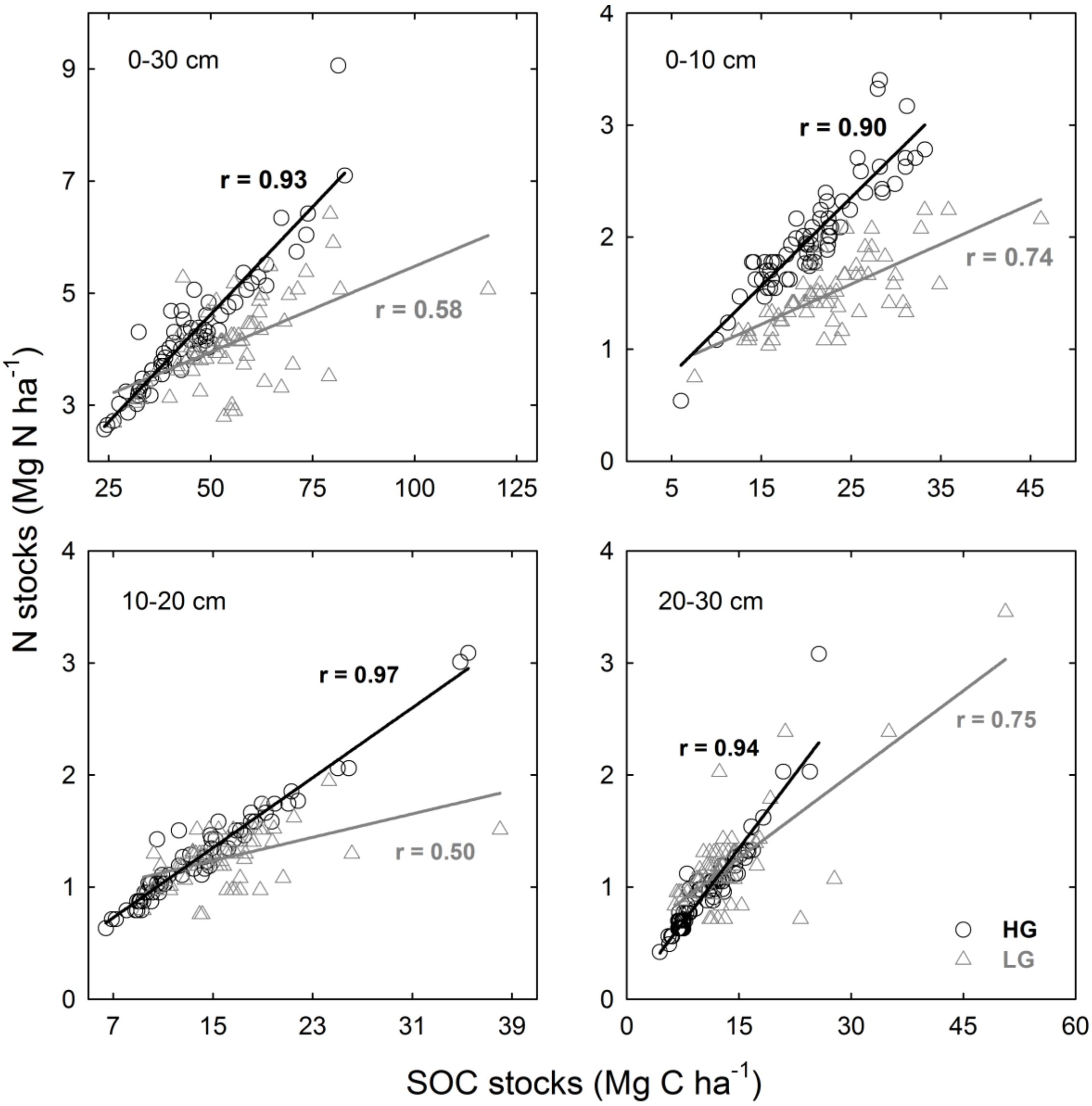

Figure 4.2 Correlation analyses between SOC stocks and soil N stocks at the whole 0-30 cm soil profile and at the 0-10 cm, 10-20 cm and 20-30 cm layers in HG (black circles) and LG (grey triangles) during 20 years of pasture establishment ($P < 0.0001$ in all analyses. r: Pearson's correlation coefficient; the black and grey straight lines represent the best-fit line obtained by linear regression analyses in HG and LG, respectively; n = 66). HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

4.4.4. Natural abundance of ^{13}C and ^{15}N

Soil isotopic composition of ^{13}C ($\delta^{13}C$) at 30 cm depth at the forest, 1-, 2- and 20-year-old pastures was significantly different between both sub-regions (Table 4.1). There was a significant increase of $\delta^{13}C$ during the forest-to-pasture conversion at this depth in HG, ranging from -27.6‰ at the forest stage, to -21.5‰ at the 20-year-old pasture. In LG, although variation of $\delta^{13}C$ was not so evident along the chronosequence (i.e. between -25.3‰ and -24.7‰), there was a significant increase at the 5-year-old pasture which reached a value of -24.2‰. There were also significant differences of $\delta^{13}C$ going down the soil profile in both sub-regions. In LG, $\delta^{13}C$ in the 0-10 cm and 10-20 cm layers were similar along the chronosequence, whereas in HG there was a large, significant increase in soil $\delta^{13}C$ in the top 0-10 cm layer with stage, which decreased in magnitude with depth to 30 cm (Table 4.3).

Natural abundance of ^{15}N in soils at 30 cm depth was significantly different between all stages of the forest-to-pasture conversion in both sub-regions (Table 4.1). Higher values of ^{15}N were found at LG compared to HG, ranging from 11.1‰ to 12.5‰ and from 5.8‰ to 8.9‰, respectively. A decrease in natural abundance of soil ^{15}N occurred during the slash and burn events in HG, after which a gradual increase up to 31% of the original content was recorded at the end of the 20 years of pasture establishment. In LG, on the other hand, soil $\delta^{15}N$ at 30 cm depth remained relatively stable during the entire chronosequence. Values of soil $\delta^{15}N$ were always lower in the top 0-10 cm layer in both sub-regions and significantly different from the values recorded in the deeper layers (Table 4.3).

Table 4.3 Mean soil isotopic composition of $\delta^{13}C$ (‰) and $\delta^{15}N$ (‰) to 0-10 cm, 10-20 cm and 20-30 cm depth. Values in parenthesis represent the standard error of the mean (n = 11). Letters indicate significant differences among stages within each sub-region (P < 0.0001). HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

	Forest-to-pasture conversion											
	Primary forest		1-yr pasture		2-yr pasture		5-yr pasture		12-yr pasture		20-yr pasture	
HG												
$\delta^{13}C$ (‰)												
0-10 cm	-28.1	$(0.2)^{a}$	-28.2	$(0.1)^{a}$	-27.9	(0.3)	-23.5	$(0.3)^{a}$	-23.2	$(0.5)^{a}$	-17.7	$(0.3)^{a}$
10-20 cm	-27.4	$(0.1)^{b}$	-27.5	$(0.1)^{b}$	-27.5	(0.3)	-24.5	$(0.3)^{b}$	-24.7	$(0.3)^{b}$	-23.1	$(0.2)^{b}$
20-30 cm	-27.3	$(0.2)^{b}$	-27.3	$(0.1)^{b}$	-27.3	(0.3)	-24.5	$(0.3)^{b}$	-25.1	$(0.3)^{b}$	-23.8	$(0.2)^{b}$
$\delta^{15}N$ (‰)												
0-10 cm	6.0	$(0.2)^{a}$	5.9	$(0.1)^{a}$	5.3	$(0.2)^{a}$	6.0	$(0.2)^{a}$	7.3	$(0.2)^{a}$	7.6	$(0.2)^{a}$
10-20 cm	7.2	$(0.2)^{b}$	6.8	$(0.2)^{b}$	6.0	$(0.3)^{b}$	8.0	$(0.2)^{b}$	8.9	$(0.2)^{b}$	9.6	$(0.1)^{b}$
20-30 cm	7.2	$(0.2)^{b}$	6.8	$(0.1)^{c}$	6.1	$(0.2)^{ab}$	8.0	$(0.2)^{b}$	9.1	$(0.2)^{b}$	9.5	$(0.1)^{b}$
LG												
$\delta^{13}C$ (‰)												
0-10 cm	-25.5	$(0.3)^{a}$	-26.3	$(0.4)^{a}$	-25.4	(0.3)	-25.7	$(0.4)^{a}$	-25.2	(0.5)	-25.5	$(0.4)^{a}$
10-20 cm	-24.6	$(0.3)^{ab}$	-24.1	$(0.3)^{b}$	-25.1	(0.3)	-23.8	$(0.3)^{b}$	-24.7	(0.4)	-24.4	$(0.3)^{b}$
20-30 cm	-24.4	$(0.3)^{b}$	-25.1	$(0.2)^{b}$	-25.4	(0.3)	-23.0	$(0.2)^{b}$	-25.5	(0.4)	-24.0	$(0.2)^{b}$
$\delta^{15}N$ (‰)												
0-10 cm	10.5	$(0.5)^{a}$	9.8	$(0.5)^{a}$	9.7	$(0.4)^{a}$	9.3	$(0.4)^{a}$	9.9	$(0.4)^{a}$	8.8	$(0.4)^{a}$
10-20 cm	12.3	$(0.4)^{b}$	13.1	$(0.1)^{b}$	12.5	$(0.2)^{b}$	12.6	$(0.4)^{b}$	13.5	$(0.3)^{b}$	11.4	$(0.3)^{b}$
20-30 cm	12.5	$(0.5)^{b}$	13.1	$(0.4)^{b}$	13.3	$(0.2)^{b}$	13.6	$(0.2)^{b}$	14.1	$(0.2)^{b}$	13.0	$(0.4)^{c}$

4.4.5. Soil C turnover

Conversion from forests to pasture modified the original SOC_T stocks in both sub-regions (Figure 4.3a). Whereas SOC_T down to 30 cm depth decreased with time since conversion in HG ($r^2 = 0.24$; $P < 0.0001$), in LG SOC_T tended to increase ($r^2 = 0.20$; $P = 0.0002$). While C3-C_T in HG followed a similar decline over the chronosequence ($r^2 = 0.51$; $P < 0.0001$) with a pronounced decrease after the second year of deforestation (i.e. after the fire event), C4-C_T tended to increase ($r^2 = 0.75$; $P < 0.0001$) (Figure 4.3b,c). There was a clear dominance of C3-C_T over C4-C_T inputs to SOC_T during the forest stage in HG (C3-C_T = 46.4 ± 1.8 Mg C ha^{-1}; C4-C_T = 0.7 ± 0.3 Mg C ha^{-1}), that progressively reduced as C3-C_T declined and C4-C_T increased along the chronosequence (C3-C_T = 21.2 ± 0.8 Mg C ha^{-1}; C4-C_T = 16.7 ± 1.0 Mg C ha^{-1} at the 20-yr pasture). Both sources of soil C input in LG tended to increase after deforestation (C3-C_T: $r^2 = 0.14$; $P = 0.0017$; C4-C_T: $r^2 = 0.16$; $P = 0.0007$), with no large change after the fire event (Figure 4.3b,c). The dominance of C3-C_T over C4-C_T during the forest stage in LG was less pronounced than in HG (C3-C_T = 39.1 ± 2.8 Mg C ha^{-1} and C4-C_T = 9.6 ± 0.5 Mg C ha^{-1}), although C3-C_T increased in 26% at the end of the forest-to-pasture conversion.

No trend was found in SOC in the upper 0-10 cm layer in HG along the chronosequence, with an average of 20.8 ± 1.7 Mg C ha^{-1}, reflecting both a strong decrease in the C3-C content ($r^2 = 0.56$; $P < 0.0001$) and the pronounced increase of C4-C ($r^2 = 0.77$; $P < 0.0001$) at this depth. SOC and C3-C tended to decrease with time in the 10-20 cm layer ($r^2 = 0.29$; $P < 0.0001$, and $r^2 = 0.42$; $P < 0.0001$, respectively) as well as in the 20-30 cm layer ($r^2 = 0.35$; $P < 0.0001$, and $r^2 = 0.43$; $P < 0.0001$, respectively), with a marked reduction after the fire event at both depths. C4-C increased in the 10-20 cm ($r^2 = 0.67$; $P < 0.0001$) and 20-30 cm layers (r^2 = 0.54; $P < 0.0001$), although in a lower proportion compared to the top 0-10 cm layer. Significant differences in C3-C were found among the three depths in all stages of the forest-to-pasture conversion in HG ($P < 0.0001$), and in most of the stages in the case of C4-C, except for the forest and the 2-years-old pasture (Table 4.4).

There was a weak but still significant increase in SOC (r^2 = 0.20; P = 0.0002), C3-C (r^2 = 0.11; P = 0.006) and C4-C (r^2 = 0.09; P = 0.01) in the top 0-10 cm layer in LG, as well as in SOC (r^2 = 0.22; $P < 0.0001$), C3-C (r^2 = 0.21; $P < 0.0001$) and C4-C (r^2 = 0.09; P = 0.01) at the 10-20 cm layer along the forest-to-pasture conversion. However, no trend was found in SOC, C3-C and C4-C in the 20-30 cm layer in this sub-region, where stocks averaged 13.6 ± 1.9, 10.6 ± 1.5, and 3.0 ± 0.6 Mg C ha^{-1}, respectively. There were significant differences in C3-C stocks among soil layers in most stages of LG, except for the 2-year-old pasture, and only the 5-year-old pasture showed differences in C4-C among depths (Table 4.4).

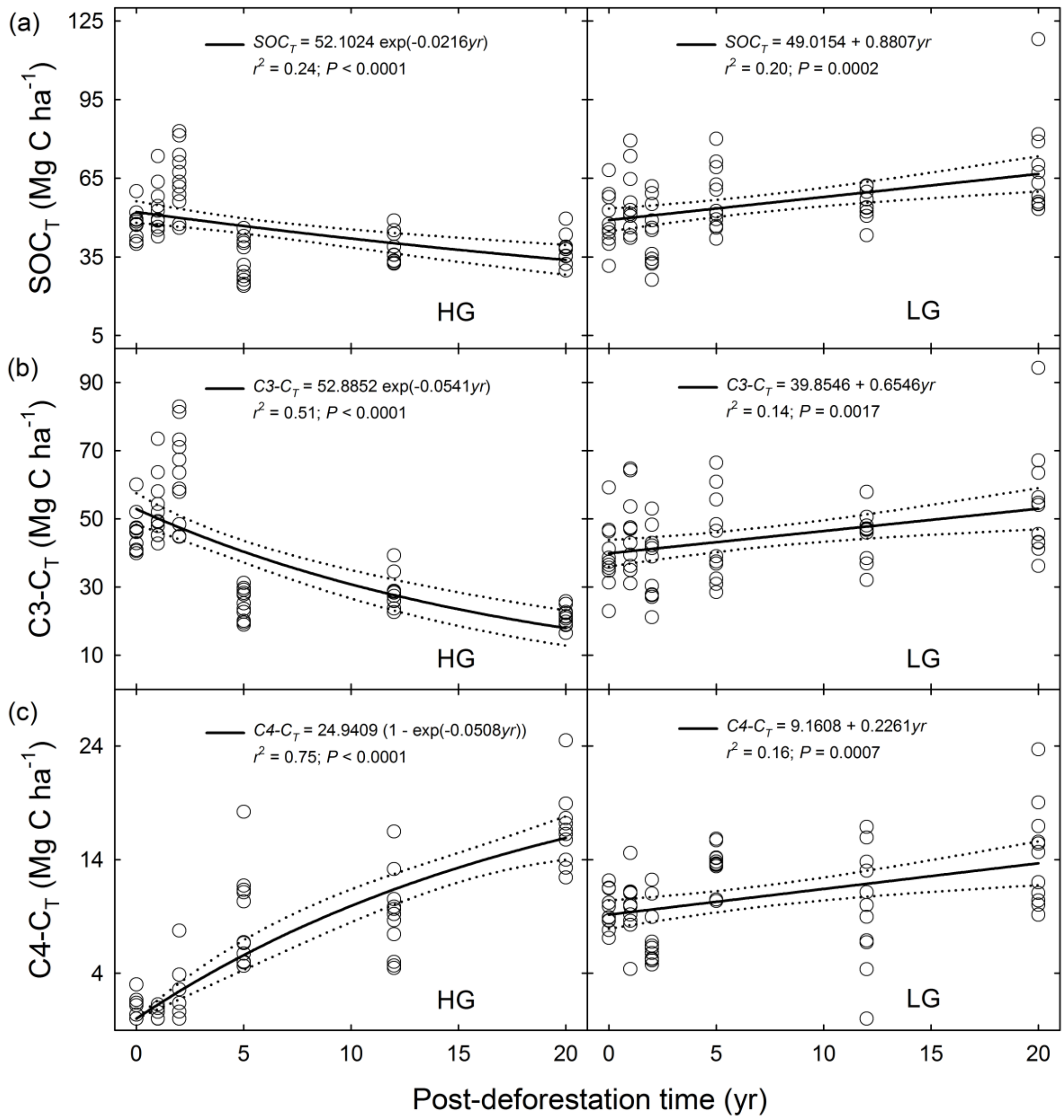

Figure 4.3 Variation of a) total soil organic carbon (SOC_T), b) C derived from C3 plants ($C3\text{-}C_T$), and c) C derived from C4 plants ($C4\text{-}C_T$) to a depth of 30 cm during the forest-to-pasture conversion in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Significance of all regressions was evaluated at P < 0.05. The dotted lines represent the 95% confidence interval.

Table 4.4 Mean soil C derived from C3 plants (C3-C; Mg C ha^{-1}) and C derived from C4 plants (C4-C; Mg C ha^{-1}) to 0-10 cm, 10-20 cm and 20-30 cm depth. Values in parenthesis represent the standard error of the mean (n = 11). Letters indicate significant differences among depths within each stage of the forest-to-pasture conversion (P < 0.05).

	Forest-to-pasture conversion											
	Primary forest		1-yr pasture		2-yr pasture		5-yr pasture		12-yr pasture		20-yr pasture	
High-grazing intensity												
C3-C												
0-10 cm	19.7	(1.8)^{a}	22.0	(1.5)^{a}	25.5	(1.5)^{a}	12.0	(0.9)^{a}	12.1	(1.2)^{a}	6.0	(0.5)^{a}
10-20 cm	15.2	(1.1)^{ab}	17.4	(0.9)^{b}	21.5	(2.3)^{ab}	7.2	(0.4)^{b}	9.1	(0.4)^{b}	6.5	(0.2)^{ab}
20-30 cm	11.8	(1.0)^{b}	13.4	(0.7)^{c}	16.2	(1.6)^{b}	5.5	(0.4)^{b}	6.7	(0.4)^{b}	5.2	(0.1)^{b}
C4-C												
0-10 cm	0.1	(0.1)	0.0	(0.0)^{a}	0.4	(0.3)	5.6	(1.1)^{a}	5.8	(0.9)^{a}	15.4	(1.2)^{a}
10-20 cm	0.3	(0.1)	0.2	(0.1)^{ab}	0.6	(0.3)	2.2	(0.3)^{b}	2.5	(0.4)^{b}	3.2	(0.2)^{b}
20-30 cm	0.4	(0.1)	0.3	(0.1)^{b}	0.6	(0.2)	1.7	(0.2)^{b}	1.5	(0.2)^{b}	2.0	(0.1)^{b}
Low-grazing intensity												
C3-C												
0-10 cm	17.6	(1.5)^{a}	20.5	(2.2)^{a}	14.3	(1.3)	20.9	(2.3)^{a}	18.2	(1.1)^{a}	25.3	(2.5)^{a}
10-20 cm	10.7	(0.9)^{b}	11.1	(0.5)^{b}	10.3	(0.5)	12.5	(1.1)^{b}	13.2	(0.7)^{b}	15.0	(1.5)^{b}
20-30 cm	8.8	(0.7)^{b}	12.7	(2.7)^{b}	10.0	(2.0)	9.2	(0.6)^{b}	11.1	(0.7)^{b}	11.7	(1.3)^{b}
C4-C												
0-10 cm	2.9	(0.3)	2.0	(0.5)	2.8	(0.6)	2.7	(0.5)^{a}	3.9	(0.8)	3.9	(0.7)
10-20 cm	2.8	(0.3)	3.8	(0.4)	2.4	(0.4)	4.4	(0.2)^{b}	3.6	(0.5)	4.6	(0.7)
20-30 cm	2.7	(0.3)	2.9	(0.8)	1.8	(0.5)	4.5	(0.3)^{b}	2.1	(0.5)	4.1	(0.5)

4.4.6. Emission factors for changes in SOCT stocks (IPCC Tier 3 approach)

IPCC (2006) recommends developing emission factors for changes in SOC pool for lands converted to grasslands, including the forest-to-pasture conversion, and the long term effects of land-use management within a Tier 3 approach. Accordingly, the following equations and parameters were fitted to the data in order to describe the change in SOC_T stocks to 30 cm depth with time due to the forest-to-pasture conversion after 20 years of deforestation in HG and LG, respectively:

$$SOC_T = 52.1024\ exp(-0.216yr) \quad (4.6)$$

and the equation:

$$SOC_T = 49.0154 + 0.8807yr \quad (4.7)$$

Table 4.5 presents the resulting emission factors for changes in SOC_T stocks for the forest-to-pasture conversion in both sub-regions.

Table 4.5 Emission factors (t CO_2e ha^{-1}) for changes in soil organic C pool after 20 years of forest-to-pasture conversion in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Values in parenthesis represent the Confidence Interval (95%).

Post-deforestation time (yr)	Soil organic C pool			
	HG		LG	
0	191.0	(15.1)	179.7	(16.0)
1	187.0	(13.7)	183.0	(14.9)
2	183.0	(12.5)	186.2	(14.0)
3	179.1	(11.4)	189.4	(14.4)
4	175.2	(11.1)	192.6	(13.4)
5	171.5	(10.7)	195.9	(12.0)
6	167.8	(10.6)	199.1	(12.3)
7	164.2	(10.8)	202.3	(12.2)
8	160.7	(11.2)	205.6	(12.1)
9	157.3	(11.8)	208.8	(13.3)
10	153.9	(12.7)	212.0	(13.2)
11	150.6	(13.4)	215.2	(13.8)
12	147.4	(14.3)	218.5	(14.6)
13	144.3	(15.1)	221.7	(15.5)
14	141.2	(15.9)	224.9	(16.4)
15	138.2	(16.9)	228.2	(17.6)
16	135.2	(17.7)	231.4	(19.4)
17	132.3	(18.5)	234.6	(20.8)
18	129.5	(19.3)	237.8	(22.7)
19	126.7	(20.0)	241.1	(23.6)
20	124.0	(20.9)	244.3	(24.7)

4.5. Discussion

The results indicate that forest-to-pasture conversion modified SOC_T in the Colombian Amazon, although the pattern of C variation after deforestation in each sub-region was very different. SOC_T increased shortly following deforestation as a result of a large input from litter, then SOC_T quickly declined to previous levels in both sites within 5 years of deforestation, after which the change in SOC_T diverged among land management regimes, decreasing in HG and increasing in LG (See Figure 4.1). A few causal mechanisms for changes in SOC as a consequence of land management are expanded upon below, and include the changes in decomposition rates following deforestation, the effects of fires on C inputs to soil, C losses from erosion and runoff and loss of C from deeper soils as a consequence of greater C in the light fraction of soil, which is easily lost.

It has been demonstrated that the slash-and-burn system widely used in the Amazon to prepare land for pasture establishment contributes to transfer large amounts of C and nutrients from forest biomass to the soil (Buschbacher, 1986; Buschbacher et al., 1988; Fearnside et al., 2001; Fearnside and Barbosa, 1998; Aragão and Shimabukuro, 2010), and that, as a result, decomposition rates increase during the first months after deforestation due to the high availability of readily decomposable organic matter at the soil surface, and increases in surface temperatures and aeration (Lavelle et al., 1993; Kauffman et al., 1995; Kauffman et al., 1998; Yakimenko, 1998; Chapin et al. 2002; Powers et al., 2009; Conant et al., 2011).

The use of fire as a management practice of pasture establishment contributed to elevate SOC_T stocks in HG and LG. In fact, depending on burning efficiency, fire contributes to eliminate on average 38% of the remnant dead wood and other non-readily-decomposable material left after deforestation in the Amazon (Kauffman et al., 1995; Kauffman et al., 1998; Fearnside et al., 2001; IPCC, 2006). As a consequence, in some areas of the Amazon region where burning efficiency is low due to short or non-existent dry seasons (Armenteras-Pascual et al., 2011), or associated with chemical or physical properties of organic matter such as wood density or moisture content (Araujo et al., 1999), farmers usually implement fire more than once in order to eliminate the greatest possible amount of woody debris (Fearnside et al., 2001) or to control the growth of secondary vegetation (Aragão and Shimabukuro, 2010). Chapter 5 of this study shows that fire does not totally consume the remnant woody debris left after cutting the forest in HG and LG, so farmers tend to burn the deforested areas in two to three fire events.

4.5.1. SOC variation with forest-to-pasture conversion in HG

The high-intensity post-deforestation grazing occurring in HG, represented by a high density of cattle population (i.e. 2.00 HFFL ha^{-1}) in pastures with low carrying capacity (i.e. 0.8-1.0 HFFL ha^{-1} according to Mahecha et al., 2002), poor maintenance of the introduced grasses and the use of machinery, has increased the soil susceptibility to lose organic C through erosion and runoff. C loss due to soil erosion has been well documented (Lal, 1998; Wairu and Lal, 2003; Fonte et al., 2014). Soil erosion contributes to deplete the organic C associated to the free and occluded light-density fractions in soils (Marin-Spiotta et al., 2009; Wagai et al., 2009), mainly composed of plant and animal material at different stages of decomposition, and gradually also affects the soil organic C in the high-density fraction consisting of organic matter attached to minerals (Schrumpf et al., 2013). Some authors have suggested that soil organic matter is related to soil clay content (Moraes et al., 1996; Moraes et al., 2002), and that susceptibility to soil erosion is higher in clayey soils where high-grazing intensity, cattle trampling, mechanization and high annual precipitation values contribute to remove small particles composing soil (Lal, 1977; Fearnside and Barbosa, 1998). While clay content was not measured directly in this study, van Engelend and Dijkshoorn (2013) suggested that the predominant soils in HG (i.e. Haplic Ferralsols and Haplic Acrisols) have high clay contents, indicating the high risk of erodibility in pastures established in this sub-region, which increases when considering that mean annual precipitation exceeds 3700 mm. Guo and Gifford (2002) suggested that land-cover change from forest to pasture in areas where annual rainfall is >3000 mm, precipitation leads to soil erosion and associated C loss. The use of machinery to eliminate the remnant dead wood after fire also contributed to reduce the SOC in HG, through the removal of part of the dead root C pool. The dead root C pool is the main source of organic C to the deeper soil layers (Ludovici et al., 2002), and the use of machinery during pasture establishment reduces the C stocks stored in roots by ~90% (See Chapter 6).

Soil compaction due to cattle trampling after 20 years of forest-to-pasture conversion in HG was also notable at the three depths assessed in this study, evidenced by an increase in soil bulk density of 36% in the top 0-10 cm layer, 27% in the 10-20 cm layer and 17% in the 20-30 cm layer compared to the original forest. As bulk density increased, a resulting decreased of soil porosity might have limited the productivity of improved pastures, consequently leading to a reduction in the SOC stocks in late stages after pasture establishment (Ellert and Bettany, 1995; Martinez and Zink, 2004). Fonte et al. (2014) found that productivity in pastures declined progressively after poor maintenance, resulting in the occurrence of patches of bare soil, invasion of non-palatable vegetation and progressive loss of the improved grass *Brachiaria spp*.

SOC_T in the original forest, and its increment during the first two years after deforestation, were strongly influenced by the high input of C3-derived C to the entire soil profile, as a result of the high organic matter input through litterfall and the implementation of slash-and-burn practices. However, the rapid loss of C3-derived C during the early years of pasture establishment and its deceleration during the later stages, could reflect the high susceptibility of the C associated to the free, unprotected light fraction of soil (Marin-Spiotta et al., 2009) and the progressive depletion of the occluded light and heavy fractions (Desjardins et al., 2004; Wagai et al., 2009). Camargo et al. (1999) found that ~90% of bulk soil at 10 cm depth corresponds to the heavy fraction at degraded and improved pastures and also at primary and secondary forests of the Brazilian Amazon, but C concentration associated to that fraction was low (2.8 ± 0.3%) compared to the high C concentration found in the light fraction (22.8 ± 3.4%). Pastures under well-managed practices have the potential to sequester more C in soils than forests and to store it even below 30 cm depth (Fisher et al., 1994; Murty et al., 2002); in HG, however, where pastures are poorly maintained, soil C4-derived C is increasing mostly in the 0-10 cm layer.

4.5.2. SOC variation with forest-to-pasture conversion in LG

Unlike HG, low intensity grazing led to a net gain of 41% in SOC_T stocks to 30 cm depth after 20 years of forest-to-pasture conversion, mainly as a consequence of the significant increase of C3- and C4-derived C input to soil at the 0-10 cm and 10-20 cm layers. The management practices implemented in LG, in which secondary vegetation is rarely eliminated from the pasture matrix, could explain the pattern of increase of C3-derived C during the entire chronosequence. In fact, due to the low cattle density (0.03 HFFL ha^{-1}) and the possibility to establish new pastures areas in the adjacent forest, it is a common practice in LG to abandon the pasture when its productivity declines allowing the reestablishment of secondary vegetation. Camargo et al. (1999) reported the contribution of C3- and C4-derived C to SOC_T in a degraded pasture in the Brazilian Amazon (i.e. the early stages of secondary succession), where C3 shrubs grew within a matrix of C4 grasses. Some other studies have also reported the transition from abandoned pastures to the different stages of secondary forest growth in the tropics (Uhl et al., 1988; Brown and Lugo, 1990; Marin-Spiotta et al., 2009).

Forest clearing in LG also resulted in a large transfer of organic matter to the soil surface, which increased the availability of readily decomposable material and allowed the input of large amounts of C3-derived C to soils. Decomposition rates increase just after deforestation in pastures and in early stages of secondary succession, because disturbance allocates large amounts of labile nutrients as C and N to the soil (Lavelle et al., 1993; Ewel, 1976). The increase in SOC_T stocks and in C3-derived C input continued until the end of the 20 years of forest-to-pasture conversion in LG, which could be related to an increase in NPP due to the rapid colonization and high growth rates of C3 vegetation supported by a high availability of light, water, and nutrients (Chapin et al., 2002; Grace, 2004). The proximity of the pastures areas to the forest matrix in LG would facilitate seed dispersal and seedling establishment of C3-plant species into the opened areas, enhancing secondary succession (Cavelier et al., 1996; Maza-Villalobos et al., 2011; Mora et al., 2015; Norden et al., 2015). In the competition for light and nutrients in areas such as abandoned pastures where early stages of succession could occur, plants have developed strategies to allocate a large fraction of C to construct leaves with high turnover rates (Selaya et al., 2008), or to increase leaf area index (Ryan et al., 1997). Brown and Lugo (1990) found larger amounts of C allocated to highly-decomposable organic matter during the first 20 years of secondary succession in humid areas of the tropics, which then was recycled through litter decomposition.

Compared to HG, the susceptibility to lose organic C through erosion and runoff is lower in LG. Soil compaction due to cattle trampling in this sub-region was only evident in the top 0-10 cm layer during 20 years of forest-to-pasture conversion, reducing the impact on soil porosity and structure (Ellert and Bettany, 1995; Martinez and Zink, 2004). The lower average values of precipitation in LG (i.e. 3351.1 ± 341.7 mm) also could contribute to reduce the risk of C loss through erosion and runoff (Guo and Gifford, 2002).

4.5.3. Nitrogen and C:N ratio

N_T stocks to 30 cm depth remained relatively similar between HG and LG during the forest-to-pasture conversion. However, notably changes in N stocks occurred in soil profiles in HG where an increase of 17% in N stocks was recorded at the end of the 20 years of pasture establishment in the top 0-10 cm layer, and a decrease of 33% and 36% was registered in the 10-20 cm and 20-30 cm layers, respectively, compared to the original forest. These results are supported by the results of the natural abundance of ^{15}N presented here that show lower values of $\delta^{15}N$ in the 0-10 cm layer compared to the deeper profiles, indicating high rates of N_2 fixation at the soil surface in the pasture sites of HG. Assessing the effects of land-cover conversion from forest to pasture on the natural abundance of ^{15}N in the western Brazilian Amazon, Piccolo et al. (1994) found that $\delta^{15}N$ values of soil surface in pastures are lower than those in forests and decrease with pasture age, which could be related to atmospheric N_2 fixation by free-living nitrogen fixing bacteria associated with pasture grasses (Piccolo et al., 1996). Grasses of the genus Brachiaria spp. obtain ~40% of their nitrogen from N_2 atmospheric fixation through the association with free-living nitrogen fixing bacteria (Boddey and Victoria, 1986; Moraes et al., 2002), explaining the high values of soil N stocks in the top 0-10 cm of HG where *Brachiaria spp*. is commonly planted. Some farmers in HG mix *Brachiaria spp*. grasses with the nitrogen fixing legumes *Arachis pintoi* or *Desmodium ovalifolium* (Alarcón and Tabares, 2007; Mosquera et al., 2012), in order to improve the N transfer to the grasses (He et al., 2009). The difference in N stocks among soil layers is less noticeable in LG, although N stocks decreased with depth in all stages of forest-to-pasture conversion. As in HG, decreasing N stocks were supported by the increase of $\delta^{15}N$ with depth in LG, but in this sub-region the reduction of $\delta^{15}N$ with pasture age was not as evident as in HG. Values of $\delta^{15}N$ in LG were greater than in HG, suggesting that atmospheric N_2 fixation is lower in LG presumably due to the absence of introduced grasses. The strong correlation between soil C and N variation during the pasture establishment in both sub-regions indicates that soil N stocks are also highly susceptible to grazing intensity. In a review of different studies, Murty et al. (2002) also found that changes in soil C and N were strongly correlated when forest is converted to agricultural land, including the forest-to-pasture conversion. SOC and N change also affected C:N ratios in each sub-region, represented by a small decrease of C:N ratios in HG and an increase in LG (see Table 4.1). Low C:N ratios are related to high rates of decomposition (Chapin et al., 2002), for example during the first year following deforestation, which is particularly evident in LG (see Figure 4.1d).

4.6. Conclusions

To my knowledge, the results presented in this chapter are the first Tier 3 information and emission factors on SOC pool and its dynamics during 20 years of forest-to-pasture under different management practices in the Colombian

Amazon. My results emphasize the dual importance of land conversion and subsequent management practice for SOC dynamics in the Amazon and contribute to improve the accuracy of SOC stocks data for REDD+ initiatives in the region. They also highlight the necessity to implement low-grazing intensity practices during cattle activities, in order to preserve SOC stocks and to reduce C emissions after land-cover change from forest to pasture in the Colombian Amazon. Consequently, in order to accurately estimate the impact of land-cover change on soil C dynamics, it is essential to develop a detailed description of all the land uses and associated management practices occurring in deforested areas, and spatially-explicit maps describing their location.

Chapter 5: Conversion from forests to pastures in the Colombian Amazon leads to differences in dead wood dynamics depending on land management practices

5. Conversion from forests to pastures in the Colombian Amazon leads to differences in dead wood dynamics depending on land management practices

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5.1. Abstract

Dead wood, composed of coarse standing and fallen woody debris (CWD), is an important carbon (C) pool in tropical forests and its accounting is needed to reduce uncertainties within the strategies to mitigate climate change by reducing deforestation and forest degradation (REDD+). To date, information on CWD stocks in tropical forests is scarce and effects of land-cover conversion and land management practices on CWD dynamics remain largely unexplored. Here I present estimates on CWD stocks in primary forests in the Colombian Amazon and their dynamics along 20 years of forest-to-pasture conversion in two sub-regions with different management practices during pasture establishment: high-grazing intensity (HG) and low-grazing intensity (LG) sub-regions. Two 20-year-old chronosequences describing the forest-to-pasture conversion were identified in both sub-regions. The line-intersect and the plot-based methods were used to estimate fallen and standing CWD stocks, respectively. Total necromass in primary forests was similar between both sub-regions (35.6 ± 5.8 Mg ha^{-1} in HG

and 37.0 ± 7.4 Mg ha^{-1} in LG). An increase of ~124% in CWD stocks followed by a reduction to values close to those at the intact forests were registered after slash-and-burn practice was implemented in both sub-regions during the first two years of forest-to-pasture conversion. Implementation of machinery after using fire in HG pastures led to a reduction of 82% in CWD stocks during the second and fifth years of pasture establishment, compared to a decrease of 41% during the same period in LG where mechanization is not implemented. Finally, average necromass 20 years after forest-to-pasture conversion decreased to 3.5 ± 1.4 Mg ha^{-1} in HG and 9.3 ± 3.5 Mg ha^{-1} in LG, representing a total reduction of between 90% and 75% in each sub-region, respectively. These results highlight the importance of low-grazing intensity management practices during ranching activities in the Colombian Amazon to reduce C emissions associated with land-cover change from forest to pasture.

5.2. Introduction

Coarse dead wood, also called coarse woody debris (CWD), is one of the C pools defined by the Intergovernmental Panel Climate Change (IPCC) to report on C stocks changes and GHG emissions/removals associated with land-use changes within the Agriculture, Forestry and Other Land Use (AFOLU) sector (IPCC, 2006). CWD includes standing and fallen dead trees and all dead wood pieces with diameter ≥ 10 cm that together store on average ~30% of the total C stored in the above-ground C pools in tropical forests (Harmon et al., 1995; Clark et al., 2002; Creed et al., 2004; Rice et al., 2004; Baker et al., 2007; Palace et al., 2007). CWD is also an important component in many ecological processes in the forest as it provides a habitat for some micro- and macro-organisms (Gibss et al., 1993; Eggleton et al., 1995; Grove, 2002; Pedlar et al., 2002), participates in nutrient cycling and influences the energy flux within the ecosystem (Harmon et al., 1986; Chao et al., 2008).

Due to its importance within the tropical forests, C emissions from CWD potentially contribute a considerable amount of total CO_2 emissions associated with deforestation (Palace et al., 2008). However, information on CWD stocks and variation with land-cover and land-use changes is scarce (Baker et al., 2007; Palace et al., 2012). Recently, two studies were published on the impact of forest degradation on dead wood C pool in primary forests in eastern Brazilian Amazon (Berenguer et al., 2014) and Malaysia (Pfeifer et al., 2015) with the potential to improve GHG emissions accountability. However, there are no studies assessing the impact of the land-cover change from forest to pasture on CWD stocks and C dynamics.

Strategies to mitigate CO_2 emissions associated with deforestation such as REDD+ require reporting of emission factors, defined as the emissions/removals of all important GHG associated with land-cover conversion in all relevant C pools (i.e. total changes in C stocks), and activity data, referring to the size of a deforested or degraded area (Verchot et al., 2012). Both emission factors and activity data should be included within the forest reference emission levels (FREL) that countries willing to access result-based payments through REDD+ activities must submit to the UNFCCC, as benchmarks for assessing country's performance (FAO, 2014). Due to lack of information, dead wood C pool was not included in any of the FRELs recently submitted to the UNFCCC by Brazil, Colombia, Ecuador, Guyana, Malaysia and Mexico (Conafor, 2014; GFC, 2014; MADS, 2014; MAE, 2014; MMA, 2014; MNRE, 2014). Colombia, in particular, included the above- and below-ground biomass C pools in forests within its FREL, but neither the dead wood C pool nor their change after conversion from forest to any post-deforestation land-use category were included (MADS, 2014). Nonetheless, under the UNFCCC Stepwise Approach (UNFCCC, 2012), countries have the option to improve their initial FRELs by incorporating high-quality data, improved methodologies and additional C pools developed from country- or region-specific information and field measurements following an IPCC Tier 3 approach.

In this chapter I present new Tier 3 information and emission factors on dead wood C pool and its dynamics during 20 years of forest-to-pasture conversion under different management practices in the Colombian Amazon. In this study I addressed the following general question: to what extent land-cover change from forest to pasture and subsequent land management practices affect dead wood C pool in the Colombian Amazon? Therefore, I aimed to better quantify CWD stocks and changes with forest-to-pasture conversion in the Colombian Amazon in support of REDD+ initiatives. Specifically, my objectives were to:

1. Quantify the volume, wood density and necromass of CWD in primary forests of the Colombian Amazon.
2. Quantify the changes in CWD stocks in two sub-regions of the Colombian Amazon and describe the influence of the high- and low-grazing intensity management practices after 20 years of forest-to-pasture conversion on the dead wood dynamics.
3. Determine the emission factors of dead wood C pool in both sub-regions according to IPCC (2006), by applying region-specific equations developed in this study describing the CWD dynamics along 20 years of forest-to-pasture conversion.

5.3. Materials and methods

5.3.1. Fallen CWD volume

The line intersect method was used to estimate the volume of fallen CWD (van Wagner, 1982, Harmon et al., 1986). Ten 200 m sub-transects were established at every one of the six stages of each chronosequence in both sub-regions of the Colombian Amazon. Sub-transects were established in parallel with a separation of ~30 m from each other in order to avoid double accounting, and were south-north oriented. When the size of the measured area limited the establishment of all sub-transects, the orientation was modified towards east-west or in some cases following a transversal direction until all sub-transects were established. When sub-transects were established on steep terrain, the sub-transect distance was corrected as follows:

$$D_H = D_I * \cos\alpha \qquad (5.1)$$

where D_H (m) is the horizontal distance, D_I (m) is the inclined distance, and α (rad) is the terrain inclination angle.

Fallen CWD were defined as all dead wood pieces of stems, branches or roots exposed by a fallen tree with diameter ≥ 10 cm. Every piece of fallen CWD was only measured if the central longitudinal axis of the piece intersected the transect line, and if at least two points of the piece were in contact with the ground. The diameter of each piece of fallen CWD was measured in parallel to the central longitudinal axis at the point of intersection with the transect line. The inclination angle of each piece respect to the horizon was measure along its longitudinal axis with a clinometer and the orientation was recorded using a compass.

Total fallen CWD volume was calculated as:

$$V = \frac{\pi^2}{8L} \times \sum(d_i^2) \tag{5.2}$$

where V (m^3 ha^{-1}) is the total volume of fallen CWD per unit area at every stage of the chronosequence, L (m) is the total length of the transect and d_i (m^2) is the diameter at the intersection point of every fallen CWD piece i (Harmon et al., 1986). When the inclination angle of any fallen CWD piece with respect to the horizon was > 0°, the following modification to Eqn. 5.2 was applied:

$$V = \frac{\pi^2}{8L} \times \sum\left(\frac{d^2}{\cos\sigma_i}\right) \tag{5.3}$$

where σ_i (rad) is the inclination angle of the fallen CWD piece i (van Wagner, 1982).

As recommended by De Vries (1986) and Keller et al. (2004), the standard error for estimations of fallen CWD volume can be calculated as the variance (σ^2) of the mean volume weighted by L for n sub-transects j as:

$$\sigma^2 = \frac{\left[\sum L_j\left(V_j - \bar{V}_j\right)^2\right]}{\left[(n-1)\sum L_j\right]} \tag{5.4}$$

5.3.2. Standing CWD volume

The plot-based method was used to estimate standing CWD volume. Five 0.1 ha plots were established at every one of the six stages of each chronosequence in both sub-regions. When the plots were established on steep terrain, the distance of the plot limits was corrected using Eqn. 1. Standing CWD was defined as all standing dead trees with a diameter ≥ 10 cm measured at its base (~5 cm above the ground), located within the plot. Each standing CWD was measured for total height and diameter at its base. Diameter measurements for buttressed stumps were taken ~5 cm above the buttress.

Assuming a cylindrical-shaped, total standing CWD volume within each 0.1 ha plot was calculated as:

$$V = \sum \left(\frac{\pi}{4} d_i^2 * h_i \right) \tag{5.5}$$

where V (m^3) is the total volume of the standing CWD per 0.1 ha, d_i (m^2) is the diameter at the base of each standing CWD piece i, and h_i (m) is its total height. The results were extrapolated to m^3 ha^{-1}.

5.3.3. CWD wood density

CWD wood density (ρ) was predicted using a model that incorporates data collected in the field on depth penetration into the wood using a dynamic penetrometer. Penetrometer measurements of fallen CWD were taken at the intersection point of every piece crossing the transect line, holding the instrument in a vertical position. In the case of standing CWD, all stumps within the plot were measured on the north side, keeping the instrument at a 45° angle, which yields similar results to those collected with the instrument in a vertical position (Larjavaara and Muller-Landau, 2010).

The dynamic penetrometer includes a moving weight of 1.0 kg that was repeatedly dropped a distance of 25 cm, hitting a pin of 20 cm long located at one end of the instrument. The total length of penetration after 20 hits was measured with a ruler. However, when the pin penetrated completely in 20 or less hits, the number of hits was recorded instead of the length of penetration (Larjavaara and Muller-Landau, 2010). When total length was measured, penetration was calculated as:

$$P_i = \frac{l_i}{20} \tag{5.6}$$

where P_i (cm) is the penetration into the CWD piece i and l_i is the length of penetration into the CWD piece i. On the other hand, when the number of hits was recorded, penetration was calculated as:

$$P_i = \frac{20}{(g - 0.5)} \tag{5.7}$$

where g (hits cm^{-1}) is the number of hits required to completely penetrate the CWD piece i. P was transformed using log10+1 for the analysis. When CWD pieces with 10 ≤ d > 20 cm were penetrated completely by the pin in 10 to 20 hits (9 CWD pieces in total), P_i was calculated by dividing the diameter of the piece by 10.

From the total standing and fallen CWD measured in the field, 59 CWD samples were collected using a chainsaw and carried to the laboratory to measure their wood density (Lab ρ). The volume of each collected sample was measured using the water displacement method, after which all samples were dried using a greenhouse oven in HG (~8 days) and an electrical oven adjusted at 65°C in LG (~2 days). Dry weight of each sample was recorded and wood density was then calculated as:

$$Lab\ \rho_i = \frac{m_i}{v_i} \tag{5.8}$$

where Lab ρ_i (g cm^{-3}) is the wood density of the sample i determined at the laboratory, m_i (g) is the dry weight of the sample i and v_i (cm^3) is the volume of the sample i.

The information of Lab ρ and P of every one of the 59 samples collected in this study was combined with the information on these two similar parameters presented by Yepes et al. (2011a), in order to develop the model to predict CWD wood density from data on penetration. Thus, the model was finally developed using a total of 139 data on Lab ρ and P.

5.3.4. Void space proportion

A total of 89 samples were used to estimate void space proportion (S_V) of CWD, defined by Baker et al. (2007) as the empty region surrounded by CWD in more than 180°. A disc of approximately 3.0 cm long was cut from every chosen CWD piece using a chainsaw. Digital photographs of the disc cross section were taken and analysed for void space proportion using the software ImageJ. The void space proportion of all samples from a specific stage was averaged, so that the solid space proportion (S_S) of CWD at every stage of each chronosequence established in HG and LG was estimated as:

$$S_s = 1 - S_v \tag{5.9}$$

5.3.5. Necromass

Mean necromass N (Mg ha^{-1}) of standing and fallen CWD at each stage s_t of both chronosequences was calculated as:

$$N_{st} = V_{st} \times \rho_{st} \tag{5.10}$$

and the standard error of N was calculated following Taylor (1997), as:

$$SE_{Nst} = SE_{\rho st} V_{st} + SE_{Vst} \rho_{st} \tag{5.11}$$

where $SE_{\rho st}$ and SE_{Vst} represent the standard error of wood density and volume, respectively. Corrected necromass of standing and fallen CWD was obtained by multiplying N_{st} by S_S, and total necromass for a specific stage was obtained as the sum of the corrected standing and fallen CWD necromass in the same stage. Propagated standard error for total necromass was calculated as the square root of the sum of the squared standard error of the corrected standing and fallen CWD necromass.

5.3.6. Statistical analysis

All data were tested for normality and homogeneity of variance. The Rayleigh's test for circular uniformity was performed to verify that the orientation of fallen CWD was randomly distributed (Zar, 1999). In order to establish possible differences in mean wood density, volume and necromass of standing and fallen CWD between similar stages of the forest-to-pasture conversion between HG and LG, and among the six stages of each chronosequence, t tests of independent samples and analyses of variance (ANOVA), respectively, were performed. When significant differences were found, post hoc Tukey-HSD tests were conducted. The model to predict CWD wood density was developed using the package Lattice version 0.20-29 of the software R (R version 3.1.1). In order to obtain the pattern of variation of dead wood during the forest-to-pasture conversion under the specific management practices implemented in each sub-region, linear, exponential and logarithmic regression analyses were tested and the model which best fitted the data in each case was selected. The selected equations were used to establish the emission factors for the total change in dead wood C pool due to the forest-to-pasture conversion after 20 years of deforestation in both sub-regions of the Colombian Amazon, according to the IPCC (2006) Tier 3 approach. Thus, the results of CWD necromass expressed in Mg ha^{-1} were converted to Mg C ha^{-1} by using a factor of 0.5 assuming that C corresponds to 50% of total necromass (Feldpausch, et al., 2004; IPCC, 2006), and then converted to tonnes of CO_2 equivalent per hectare (t CO_2e ha^{-1}) by using the C-to-CO_2e conversion factor of 44/12 (UNFCCC, 2012). All the analyses were evaluated with a significance level of $P < 0.05$.

5.4. Results

5.4.1. Void space and wood density

Mean void space proportion in CWD was extremely low in both sub-regions, ranging from 2.2% and 3.9% in the forests to 0.0% and 0.3% in the 20-year-old pastures in HG and LG, respectively (Table 5.1). There were no significant differences in mean void space proportion either between similar stages of each chronosequence in HG and LG, or across different stages within the same sub-region. The best model to predict wood density of standing and fallen CWD based on penetration data using the dynamic penetrometer and calibrated with laboratory measurements of wood density ($r^2 = 0.65$; $P < 0.0001$; df = 135; AIC = -175.7; Figure 5.1), is expressed as:

$$\rho_i = 0.68574 - 0.46883(log_{10}P_i + 1) + 0.33388I + 0.36301(log_{10}P_i + 1)I \quad (5.12)$$

where ρ_i (g cm^{-3}) is the wood density of the CWD piece i and P_i (cm) is the penetration of the penetrometer into the wood of the CWD piece i (I = 0, if P ≤ 1; I = 1, if P > 1).

Table 5.1 Mean void space proportion (± SE) and solid space proportion of standing and fallen CWD. HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

Forest-to-pasture conversion	HG			LG		
	Void space		n	Void space		n
Primary forest	0.022	(0.012)	10	0.039	(0.019)	10
1-yr pastures	0.002	(0.001)	10	0.006	(0.003)	10
2-yr pasture	0.004	(0.003)	10	0.007	(0.004)	10
5-yr pasture	0.004	(0.003)	5	0.010	(0.005)	5
12-yr pasture	0.005	(0.005)	5	0.009	(0.006)	5
20-yr pasture	0.000	(0.000)	4	0.003	(0.003)	5

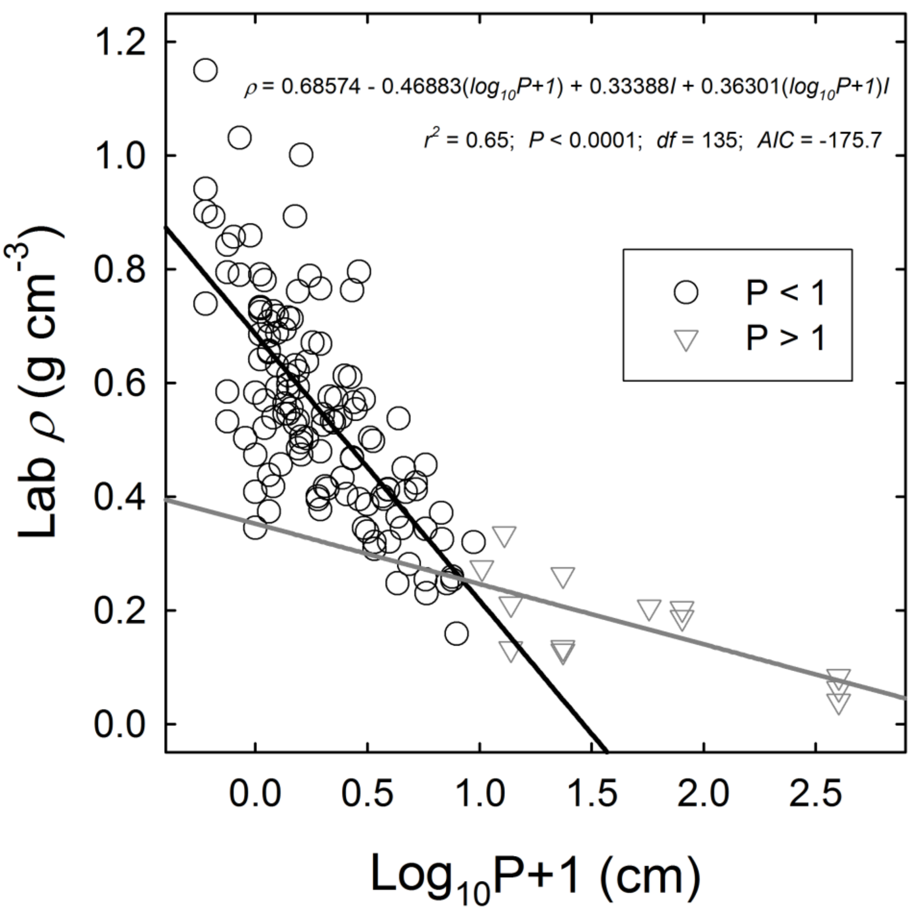

Figure 5.1 Relationship between penetration (P) into the wood of the CWD using a dynamic penetrometer and wood density measured at the laboratory (Lab ρ). $r^2 = 0.65$; $P < 0.0001$; df = 135; AIC = -175.7.

Mean wood density was significantly higher in LG in the forest and 2-year-old pasture in the case of standing CWD, and in the 1-, 2-, and 20-year-old pastures in the case of fallen CWD. Even though the rest of the comparisons between sub-regions were not significant, wood density of standing and fallen CWD were generally lower in HG compared to LG (Figure 5.2; Table 5.2). Wood density of fallen CWD was significantly lower in the forests of both sub-regions, and increased with pasture age with no significant differences among them. On the other hand, although wood density of standing CWD was lower in forests in HG and LG, there were no significant differences in CWD wood density across most of the pasture stages in each sub-region (Figure 5.2; Table 5.2).

5.4.2. CWD volume and necromass

There were no significant differences in the volume of standing and fallen CWD between similar stages of each chronosequence in HG and LG. Volume of standing CWD varied significantly among forest-to-pasture stages in HG, ranging from 21.1 ± 5.4 and 24.7 ± 4.6 m^3 ha^{-1} in the forest and 1-year-old pasture stages to 2.3 ± 1.0 m^3 ha^{-1} in the 20-year-old pasture. No significant variation in the volume of standing CWD among stages was found in LG, although the highest values were registered during the first two stages of the chronosequence (Figure 5.2; Table 5.3).

Significant differences were found in the volume of fallen CWD among stages of both chronosequences. Volume of fallen CWD in forests in HG was 56.0 ± 3.6 m^3 ha^{-1}. However, during the first stage of pasture establishment it increased 112% from its original value, after which there was a progressive reduction until a value of 3.8 ± 0.1 m^3 ha^{-1} recorded in the 20-year-old pasture. A similar pattern occurred in LG, where the volume of fallen CWD at forests was 66.8 ± 5.6 m^3 ha^{-}

^{1} and increased 71% after deforestation. Unlike HG, the decrease in volume between the 2- and 5-year-old pastures was less pronounced in LG (i.e. from 52.9 ± 3.4 to 9.1 ± 0.1 m^3 ha^{-1} in HG vs. 40.7 ± 1.6 to 25.7 ± 0.7 m^3 ha^{-1} in LG), and the value registered in the 20-year-old pasture was 8.4 ± 0.1 m^3 ha^{-1} (Figure 5.2; Table 5.3).

Mean standing and fallen CWD necromass values slightly lower when corrected for void space proportion. Total (standing + fallen) CWD corrected necromass was similar between HG and LG during the first three stages of pasture establishment, where values increased 124% in HG and 123% in LG just after deforestation, and then decreased during the 2-year-old pastures to values slightly lower than those recorded at forests. However, while there was a sharp reduction of 82% in total CWD between the 2- and 5-year-old pastures and a small amount recorded in the 20-year-old pasture (3.5 ± 1.4 Mg ha^{-1}) in HG, total CWD only was reduced by 41% between the 2- and the 5-year-old pastures and decreased to 9.3 ± 3.5 Mg ha^{-1} at the last stage of the chronosequence in LG (Table 5.4).

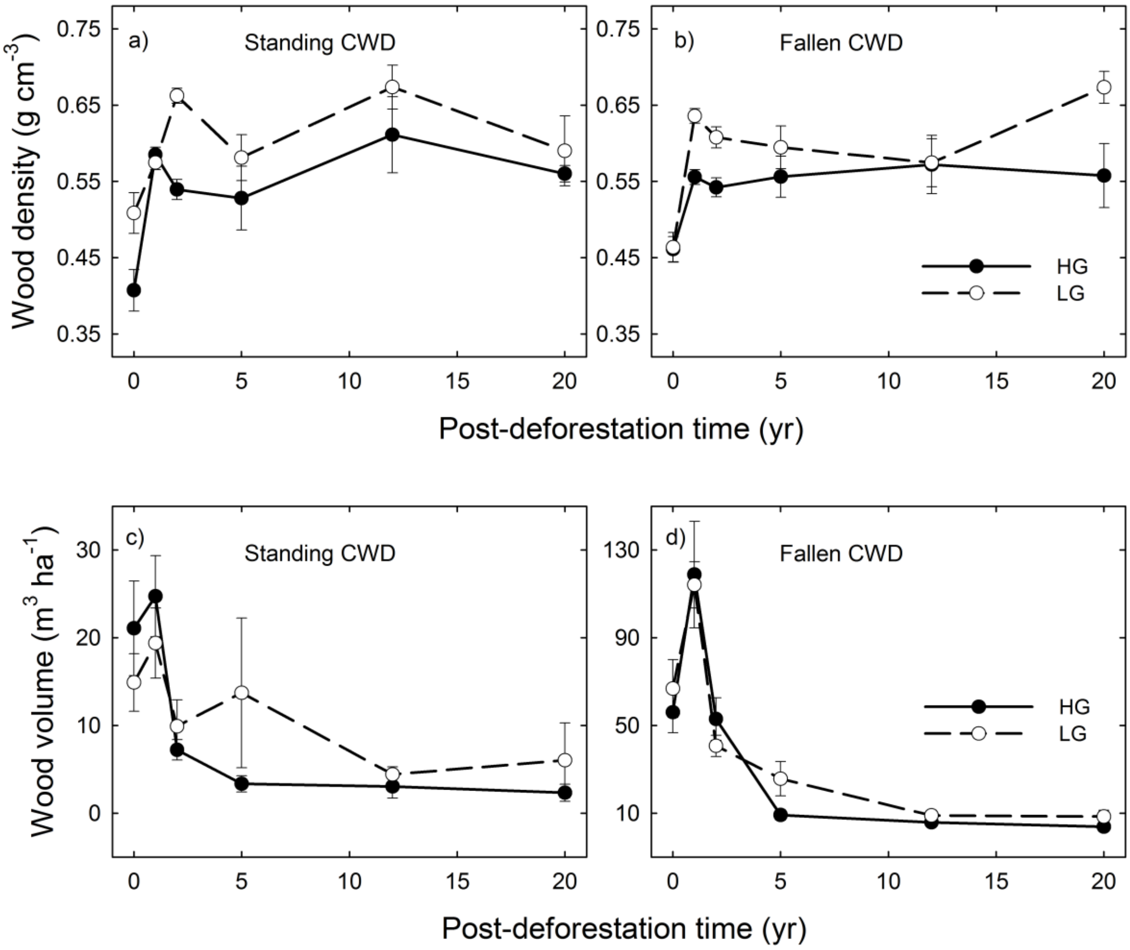

Figure 5.2 Mean wood density (a and b; g cm^{-3}) and volume (c and d; m^3 ha^{-1}) of standing and fallen CWD in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Bars represent the standard error of the mean (n for wood density is presented in Table 5.2; n for volume is presented in Table 5.3).

Table 5.2 Mean wood density (± SE, g cm^{-3}) of standing and fallen CWD in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Letters indicate significant differences among stages of the same sub-region (P < 0.0001); the plus and asterisk symbols represents significant differences in wood density of both standing and fallen CWD, respectively, between similar stages from each sub-region (P < 0.0001).

Forest-to-pasture conversion	HG						LG					
	Standing		n	Fallen		n	Standing		n	Fallen		n
Primary forest	0.41	$(0.03)^{a+}$	39	0.46	$(0.02)^{a}$	91	0.51	$(0.03)^{a+}$	26	0.46	$(0.02)^{a}$	98
1-yr pasture	0.58	$(0.01)^{b}$	170	0.56	$(0.01)^{b*}$	182	0.57	$(0.01)^{ab}$	158	0.64	$(0.01)^{b*}$	175
2-yr pasture	0.54	$(0.01)^{ab+}$	93	0.54	$(0.01)^{b*}$	118	0.66	$(0.01)^{bc+}$	73	0.61	$(0.01)^{b*}$	80
5-yr pasture	0.53	$(0.04)^{ab}$	16	0.56	$(0.03)^{b}$	18	0.58	$(0.03)^{abc}$	19	0.59	$(0.03)^{b}$	28
12-yr pasture	0.61	$(0.05)^{b}$	9	0.57	$(0.04)^{b}$	10	0.67	$(0.03)^{c}$	14	0.57	$(0.03)^{b}$	18
20-yr pasture	0.56	$(0.01)^{ab}$	4	0.56	$(0.04)^{b*}$	4	0.59	$(0.05)^{abc}$	12	0.67	$(0.02)^{b*}$	15

Table 5.3 Mean volume (± SE, m^3 ha^{-1}) of standing and fallen CWD in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Letters indicate significant differences among stages of the same sub-region ($P < 0.0001$).

Forest-to-pasture conversion	HG						LG					
	Standing		n	Fallen		n	Standing		n	Fallen		n
Primary forest	21.1	(5.4)^{a}	5	56.0	(3.6)^{b}	10	14.9	(3.3)	5	66.8	(5.6)^{b}	10
1-yr pasture	24.7	(4.6)^{a}	5	118.8	(19.6)^{a}	10	19.4	(4.0)	5	114.1	(13.9)^{a}	10
2-yr pasture	7.2	(1.2)^{b}	5	52.9	(3.4)^{b}	10	9.9	(3.0)	5	40.7	(1.6)^{bc}	10
5-yr pasture	3.3	(0.9)^{b}	5	9.1	(0.1)^{cd}	10	13.7	(8.5)	5	25.7	(0.7)^{c}	10
12-yr pasture	3.0	(1.3)^{b}	5	5.8	(0.1)^{cd}	10	4.4	(0.9)	5	8.9	(0.1)^{c}	10
20-yr pasture	2.3	(1.0)^{b}	5	3.8	(0.1)^{d}	10	6.0	(4.3)	5	8.4	(0.1)^{c}	10

Table 5.4 Mean necromass (± propagated SE, Mg ha^{-1}) corrected by void space proportion of standing and fallen CWD, and total mean necromass (standing + fallen) in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Lowercase letters indicate significant differences among stages of the same sub-region (P < 0.0001), and uppercase letters represent significant differences in uncorrected necromass of fallen CWD between similar stages from each sub-region (P < 0.0001).

Forest-to-pasture conversion in HG	Necromass						Forest-to-pasture conversion in LG	Necromass					
	Standing		Fallen		Total			Standing		Fallen		Total	
Primary forest	9.6	(2.7)^{ab}	26.1	(5.2)^{b}	35.6	(5.8)	Primary forest	7.2	(2.0)	29.8	(7.1)^{b}	37.0	(7.4)
1-yr pastures	14.4	(2.9)^{a}	65.1	(14.6)^{a}	79.6	(14.9)	1-yr pastures	11.5	(2.5)	71.1	(7.8)^{a}	82.5	(8.2)
2-yr pasture	3.9	(0.7)^{bc}	28.7	(5.9)^{b}	32.7	(5.9)	2-yr pasture	6.3	(2.1)	25.3	(3.5)^{b}	31.6	(4.1)
5-yr pasture	1.4	(0.6)^{c}	4.9	(1.6)^{Ac}	6.3	(1.7)	5-yr pasture	7.4	(5.3)	14.5	(5.3)^{Abc}	22.0	(7.5)
12-yr pasture	1.9	(1.0)^{c}	3.4	(1.3)^{c}	5.3	(1.6)	12-yr pasture	2.8	(0.7)	5.3	(1.6)^{c}	8.0	(1.8)
20-yr pasture	1.3	(0.6)^{c}	2.2	(1.2)^{c}	3.5	(1.4)	20-yr pasture	3.9	(2.8)	5.4	(2.1)^{c}	9.3	(3.5)

5.4.3. CWD necromass turnover and emission factors (IPCC Tier 3 approach)

Total CWD significantly decreases along the 20 years of forest-to-pasture conversion in both sub-regions of the Colombian Amazon ($r^2 = 0.25$, $P < 0.0001$ in HG and $r^2 = 0.26$, $P < 0.0001$ in LG; Figure 5.3a), as a consequence of the significant reduction of fallen CWD ($r^2 = 0.38$, $P < 0.0001$ in HG and $r^2 = 0.41$, $P < 0.0001$ in LG; Figure 5.3b) and standing CWD ($r^2 = 0.50$, $P < 0.0001$ in HG and $r^2 = 0.13$, $P < 0.0001$ in LG; Figure 5.3c) during the same period. IPCC (2006) recommends developing emission factors of dead wood C pool changes for lands converted to grasslands, including the forest-to-pasture conversion, and the long term legacy effects of land-use management within a Tier 3 approach. Accordingly, the following equations and parameters were fitted to the data in order to describe the change in total CWD stocks with time since conversion in HG and LG, respectively:

$$CWD_T = 33.7748\ exp(-0.2052yr) \tag{5.13}$$

and the equation

$$CWD_T = 44.8387\ exp(-0.1994yr) \tag{5.14}$$

Table 5.5 presents the resulting emission factors for changes in dead wood C pool for the forest-to-pasture conversion in HG and LG in tonnes of CO_2 equivalent per hectare (t CO_2e ha^{-1}).

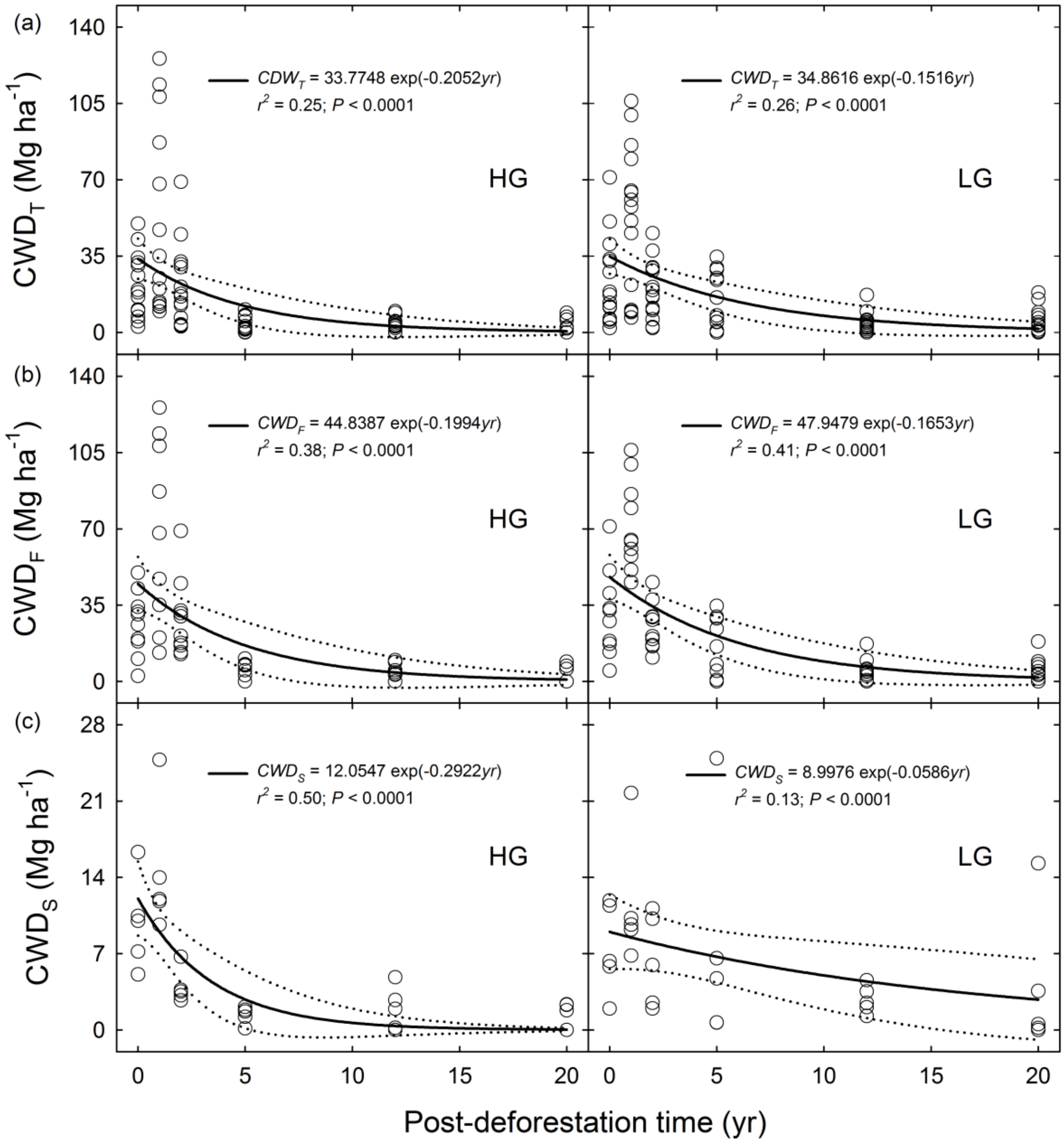

Figure 5.3 Dead wood variation of (a) total (CWD_T), (b) fallen (CWD_F), and (c) standing (CWD_S) coarse woody debris during the forest-to-pasture conversion in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. The dotted lines represent the 95% confidence interval.

Table 5.5 Emission factors (t CO_2e ha^{-1}) for changes in total (standing + fallen) dead wood C pool after 20 years of forest-to-pasture conversion in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Values in parenthesis represent the Confidence Interval (95%).

Post-deforestation	Total CWD C pool (t CO_2e ha^{-1})			
time (yr)	HG		LG	
0	61.9	(16.8)	63.9	(14.2)
1	50.4	(12.1)	54.9	(10.3)
2	41.1	(12.3)	47.2	(9.6)
3	33.5	(13.7)	40.6	(10.4)
4	27.2	(14.2)	34.9	(11.5)
5	22.2	(15.2)	29.9	(13.1)
6	18.1	(15.1)	25.7	(13.5)
7	14.7	(15.2)	22.1	(13.3)
8	12.0	(13.5)	19.0	(13.3)
9	9.8	(12.8)	16.3	(13.4)
10	8.0	(11.7)	14.0	(12.4)
11	6.5	(10.4)	12.1	(12.0)
12	5.3	(9.2)	10.4	(11.6)
13	4.3	(8.5)	8.9	(10.9)
14	3.5	(7.3)	7.7	(9.6)
15	2.9	(6.5)	6.6	(9.0)
16	2.3	(5.9)	5.7	(8.5)
17	1.9	(5.1)	4.9	(7.6)
18	1.5	(4.0)	4.2	(6.6)
19	1.3	(3.7)	3.6	(6.1)
20	1.0	(2.8)	3.1	(5.5)

5.5. Discussion

5.5.1. CWD stocks in the forests of Colombian Amazon

Total mean necromass (standing + fallen CWD) in the primary forests in both sub-regions of the Colombian Amazon presented here are in the middle of the range of those for lowland tropical forests presented by Baker et al. (2007). Estimates are slightly higher than the average values from the Peruvian Amazon (24.5 ± 2.6 Mg ha^{-1}) and the whole Amazonian terra firme forests (31.6 ± 2.7 Mg ha^{-1}) presented by Chao et al. (2009a). Reports on CWD stocks show that necromass tends to be higher in eastern and north-eastern Amazonia, reaching values of up to 86.6 ± 13.4 Mg ha^{-1} at terra firme forests in Pará, Brazil (Rice et al., 2004), and decreases towards the west of the Amazon basin (Chao et al., 2009a). Rates of tree mortality and decomposition of dead wood have been proposed as the main factors controlling CWD stocks in Amazonian forests (Baker et al., 2007). It has been demonstrated that tree mortality in primary forests also increases from east to west across the Amazon basin (Phillips et al., 2004; Chao et al., 2009b), and that decomposition of CWD in tropical forests strongly depends on wood density and diameter (Harmon et al., 1995; Chambers et al., 2000). Baker et al. (2004) found that wood density tends to be ~16% lower in northwest Amazonia compared to the central and eastern sectors, and Baker et al. (2007) showed that decomposition rates near Manaus, in the central Amazon, are lower compared to those in forests in southern Peru.

Most studies on CWD stocks, however, have been concentrated in the central and east (north and south) sectors of the Amazon (e.g. Chambers et al., 2000; Keller et al., 2004; Rice et al., 2004; Feldpausch et al., 2005; Palace et al., 2007) and just few have focused on the western region, mainly in the Peruvian Amazon

(Baker et al., 2007; Chao et al., 2008; Araujo-Murakami et al., 2011; Banin et al., 2014). Only one study on necromass of standing dead trees was carried out in a terra firme forests in the Colombian Amazon, showing CWD stocks of 26.3 ± 9.1 Mg ha^{-1} (Salarriaga et al., 1988).Therefore, reports on necromass in the HG and LG sub-regions presented here will help to reduce the uncertainty in CWD estimations in northwest Amazonia, particularly in the Colombian Amazon, and contribute with new information to strengthen regional patterns across the Amazon basin. Similarities in necromass of standing and fallen CWD between primary forests from both sub-regions of the Colombian Amazon indicate that CWD input rates and decomposition rates should also be similar between them.

One of the factors contributing to a high variability in CWD decomposition rates and necromass calculations is the error associated with wood density estimations (Chambers et al., 2000). The method of classifying dead wood into different decomposition classes has been widely used to estimate wood density (see Keller et al., 2004; Baker et al., 2007; Chao et al., 2008). Nevertheless, the inherent subjectivity of the decay classification method increases the uncertainty associated with wood density estimates and makes it difficult to replicate the method and to compare among sites where classification has been done by different people (Larjavaara and Muller-Landau, 2010). In contrast, the total variability of the model I developed to predict CWD wood density based on objective penetration data using the dynamic penetrometer is mainly explained by this parameter ($r^2 = 0.65$; $P < 0.0001$) and can be replicated in sites where wood density is measured in the field by different people.

5.5.2. Dead wood dynamics post deforestation under different management practices

The results presented here show that land-cover change from forest to pasture strongly reduced the CWD stocks in both sub-regions of the Colombian Amazon. After 20 years of pasture establishment there was a net reduction of the total CWD necromass of 90% in HG and 75% in LG compared to the original values recorded in the primary forests in both sub-regions. An expected increase of ~123% in total CWD necromass after cutting the forests followed by a reduction produced by fire events that reached similar values to those found in the forests were registered during the first two years after deforestation in both sub-regions. According to the farmers and land owners interviewed for this study, it is a common practice in HG and LG to burn the deforested areas in two to three fire events after cutting the forest to eliminate the greatest amount of dead wood produced after deforestation, as it has also been reported elsewhere in the Amazon. (Fearnside et al., 2001; Aragão and Shimabukuro, 2010). High mean annual precipitation and the very short dry season occurring in both sub-regions could explain the relatively low burning efficiency in HG and LG (Malhi et al., 2006; Armenteras-Pascual et al., 2011), in addition to chemical and physical properties of wood density or moisture content (Araujo et al., 1999). Assuming that farmers in both sub-regions perform two fire events after cutting the forest, burning efficiency in HG and LG was 29% and 31%, respectively. These results are similar to others reported for the Amazon, averaging 38% of burning efficiency (Fearnside et al., 1993; Kauffman et al., 1995; Kauffman et al., 1998; Fearnside et al., 1999; Fearnside et al., 2001; IPCC, 2006).

Despite the similarities in necromass in the first two years of forest-to-pasture conversion, significant differences in CWD stocks between HG and LG arose during the second and fifth years of pasture establishment, when total necromass reduced 82% and 41%, respectively, due to differences in management practices implemented in each sub-region. Whilst in HG the use of machinery to remove most of the residual CWD and remnant plant material not-consumed completely by fire is frequent, in LG farmers almost exclusively implement fire to remove the remnant wood produced after cutting the forest due to limitations to move heavy equipment to remote areas within the forest. In summary, the difference in management practices within a similar land-cover change category (i.e. forest to pasture) generates a pronounced difference in C emissions from dead wood C pool in the Colombian Amazon.

From the fifth year until the end of the 20-year period of forest-to-pasture conversion, in the absence of any other management practice to eliminate the dead wood remaining from cutting the forest, the reduction of the CWD stocks should be explained by the decomposition processes occurring on the pasture surface in both sub-regions. In HG total necromass decreased 2.8 Mg ha^{-1} during the last 15 years of the pasture establishment, whereas in LG the reduction was 12.6 Mg ha^{-1} during the same period (see Table 5.4). These results would be plausible when considering that the non-use of machinery and the low grazing intensity in LG would favour the growth of secondary vegetation within the pasture matrix, which would accelerate decomposition rates of organic matter by increasing resources available for decomposer organisms during the first stages of secondary succession (Brown and Lugo, 1990). In fact, as it was presented in Chapter 4, C3- and C4-derived C increases along the same chronosequence of 20 years of forest-to-pasture conversion due to the growth of secondary vegetation within the pasture matrix in LG.

Mean wood density values presented in this study tended to increase from the forest stage to the end of the 20-year chronosequence. Although it is expected that dead wood density declines as decomposition progresses with time (Harmon et al., 2000), the highest values of dead wood density registered at the old pastures in HG and LG reflect the fact that living trees with high wood density have more probability to resist decomposition and, therefore, to persist at the oldest stages of the chronosequence (Melin et al., 2009).

Studies evaluating the variation of CWD stocks with land-cover change are scarce. For example, Eaton and Lawrence (2006) reported an increase of 38% in the CWD stocks when the forest was converted to agricultural land in a dry tropical region in the south of Mexico after slash-and-burn practice was used. Saldarriaga et al. (1988) evaluated dead biomass along a chronosequence of forest succession from young forests with a CWD stocks of 1.1 Mg ha^{-1} to mature forests (see above). Some other studies on CWD necromass have compared undisturbed with degraded or secondary forests in the tropics. For instance, Berenguer et al., (2014) found that the dead wood C pool in the eastern Brazilian Amazon forests appears to be resistant to the effects of logging and fire, and shows similar CWD stocks between undisturbed and degraded forests. Some other studies have compared undisturbed forests with forests under conventional logging and the so-called ‘reduced impact logging’ where harvesting practices have been improved in order to reduce the impact on the residual forest stand in the Brazilian Amazon. Keller et al. (2004), for example, reported that average fallen necromass at Cauaxi was 55.2 ± 4.7, 74.7 ± 0.6 and 107.8 ± 10.5 Mg ha^{-1} for undisturbed, reduced impact logging and conventional logging sites, respectively, and was 50.7 ± 1.1 and 76.2 ± 10.2 Mg ha^{-1} for undisturbed and reduced impact logging at Tapajós. Palace et al. (2007) reported values of fallen necromass of 44.9 ± 0.2 and 67.0 ± 0.2 Mg ha-1 and standing necromass of 5.3 ± 1.0 and 8.8 ± 2.3 Mg ha^{-1}, for undisturbed and reduced impact logging sites in Pará, respectively.

5.5.3. Improving dead wood carbon pool emission factors (Tier 3)

Dead wood is one of the C pools recommended by IPCC to be assessed as part of the emission factors conforming the Forest Reference Emission Level (FREL) that countries implementing REDD+ activities should submit to the UNFCCC (FAO, 2014), and also to be included in the national greenhouse gas inventories within the AFOLU sector (IPCC, 2006). However, under an IPCC Tier 1 approach it should be considered that 100% of C contained in the dead wood C pool is emitted immediately after deforestation, conservatively assuming that necromass has a value of zero in a post-deforestation land category such as pastures (IPCC, 2003; IPCC, 2006). In fact, all countries that submitted their FREL to the UNFCCC by 2014 conservatively excluded the dead wood C pool from their final report (Conafor, 2014; GYC, 2014; MADS, 2014; MAE, 2014; MMA, 2014; MNRE, 2014). Countries participating in REDD+ also have the option to establish Tier 3 field-based inventories to capture the long-term legacy effects of land-cover change on dead wood C pool, in order to improve the accuracy of emission factors on CWD stocks and variation associated with land-use change activities (IPCC, 2006). However, country- or region-specific information including the dead wood C pool is limited, resulting in highly uncertain emission factors within the REDD+ national programmes (Angelsen et al., 2012).

The results presented in this study on C stocks in dead wood and their decrease after 20 years of the forest-to-pasture conversion in two sub-regions with different management practices in terms of grazing intensity in the

Colombian Amazon, are the first data set and equations developed under the IPCC Tier 3 approach using region-specific information (see Figure 5.3 and Table 5.5). Considering that countries have the option to submit their initial FRELs including a limited number of C pools, and subsequently improve them by incorporating high-quality data, improved methodologies and additional pools (UNFCCC, 2012), the information presented here has the potential to improve the emission factors to be included in the next Colombian FREL by adding Tier 3 information on dead wood C pool and its net change due to the conversion from forest to pasture in the Colombian Amazon.

5.6. Conclusions

My study suggests that conversion from forest to pasture during 20 years in the Colombian Amazon leads to a pronounced reduction in the dead wood C pool, although management practices implemented in the high- and low-grazing intensity sub-regions also greatly influences CWD stocks in the following different ways:

- Volume, wood density and necromass of standing and fallen CWD in primary forests of the Colombian Amazon are similar between HG and LG, and are within the range of CWD stocks in other Amazon forests reported elsewhere.
- During the first two years of forest-to-pasture conversion, management practices resulted in an increase of 124% in HG and 123% in LG in CWD stocks when the forest was cut, and a subsequent decrease to similar levels to those found in the original forest when fire was applied in two events with low levels of burning efficiency.

- Implementation of machinery to remove dead wood still remaining after the use of fire (i.e. between the second and fifth year of pasture establishment) generated a notably decrease of 82% of CWD stocks in HG, compared to a reduction of 41% in LG.
- During the last 15 years of pasture establishment the rate of CWD loss was greater in LG, presumably as a consequence of an increase in decomposition rates related to the secondary succession. Management practices in HG include the elimination of secondary vegetation growing in the pasture matrix, retarding the re-establishment of decomposer community.
- The implementation of low-grazing management practices after pasture establishment, including a reduction in the use of machinery, the implementation of a silvopastoral system, or the reduction of the cattle density per hectare to values equal or below to the pasture carrying capacity, could contribute to reduce CO_2 emissions following deforestation by better preserving the C stored in the dead wood after 20 years of forest-to-pasture conversion compared to pastures under high-grazing practices.

Chapter 6: Below-ground necromass stocks changes with forest-to-pasture conversion in the Colombian Amazon

6. Below-ground necromass stocks changes with forest-to-pasture conversion in the Colombian Amazon

6.1. Abstract

Below-ground necromass (BGN) is an important carbon (C) reservoir in deforested areas in the tropics, and its accounting within strategies such as REDD+ contributes to reduce uncertainties in estimates of CO_2 emissions associated with LULC change. BGN changes with land-cover conversion in the tropics have been practically unexplored. Here I present a new methodological approach and estimates of BGN stocks in primary forests in the Colombian Amazon and their dynamics along 20 years of forest-to-pasture conversion. Two chronosequences describing the forest-to-pasture conversion were identified in two sub-regions of the Colombian Amazon with high- and low-grazing intensity (HG and LG, respectively). Dead coarse roots were excavated from randomly selected stumps within pre-established plots to estimate BGN. There were no significant differences in BGN stocks between similar stages of both chronosequences, although land-cover change and management practices implemented after pasture establishment led to significant differences within each sub-region. BGN was ~16.2 Mg ha^{-1} in forests of both sub-regions and increased fourfold just after deforestation. After the second year of pasture establishment, BGN decreased from 88.2 ± 1.4 in HG and 88.3 ± 1.4 Mg ha^{-1} to 2.9 ± 0.2 and 7.5 ± 0.4 Mg ha^{-1}, respectively, after 20 years of forest-to-pasture conversion. These large changes in BGN after pasture establishment emphasize the importance of both land cover change and land management on the tropical land carbon cycle. The new methodological approach and results presented here will

contribute to improve the accuracy of the information required for REDD+ initiatives in the Colombian Amazon.

6.2. Introduction

After deforestation a portion of the above-ground biomass (AGB) is removed from the cleared sites and used for wood products (Winjum et al., 1998), with the remainder generally burned in situ usually more than once due to abiotic factors or biological characteristics of tree species (Araujo et al., 1999; Fernside et al., 2001; DeFries et al., 2008; Armenteras-Pascual et al., 2011). In most cases, the root system of forest trees will remain on the converted areas as the below-ground necromass (BGN) C pool, and a progressive loss of C will occur through decomposition processes (Manlay et al., 2004). Besides its function in soil nutrient and water uptake, roots are also the main source of organic matter to the deeper soil layers and their decomposition is a key process for forest productivity and fertility (Ludovici et al., 2002). Nevertheless, information on C and nutrients stored in BGN and decomposition rates of coarse roots after deforestation have been poorly investigated in tropical ecosystems (Soethe et al., 2007), and protocols and methodologies aimed to quantify C stocks in forests ecosystems rarely include the procedure to directly measure coarse roots in the field (Pearson et al., 2005; Ravindranath and Ostwald, 2008; Marthews et al., 2012) due to its high time and resources requirements (Cairns et al., 1997; Clark et al., 2001; Niiyama et al., 2010).

Although few in number, there are still more studies on root decomposition in temperate and boreal forests. For example, Chen et al. (2001) developed root decay models for Sitka spruce (*Picea sitchensis*) and Ponderosa pine species (*Pinus ponderosa*) in Oregon, U.S.A., Melin et al. (2009) developed decomposition functions of stumps and roots for Norway spruce (*Picea abies*), and Olajuyigbe et al. (2012) established the initial decomposition rate for roots of *P. ponderosa* in Ireland. Some of the few studies on below-ground biomass (BGB) in the tropics have focused on estimating its C stock rather than establishing decomposition models. For instance, Sierra et al. (2007) determined the total C stock in AGB and BGB at primary and secondary Andean forests in Colombia; Jimenez et al. (2009) estimated C stocks in fine roots at different Amazonian forests also in Colombia; and Soethe et al. (2007) established the C and nutrient stocks in coarse roots at different altitudes in Andean forests of Ecuador.

As part of the strategies aimed to mitigate climate change and reduce CO_2 emissions associated with deforestation (e.g. REDD+), countries willing to access result-based payments are required to submit their forest reference emission levels (FREL) to the UNFCCC (FAO, 2014). According to IPCC (2006), FRELs are obtained by estimating the area within which an activity, say deforestation, occurred and the C content and its total change with land-conversion of a specific C pool (i.e. emission factors). From the five C pools defined by IPCC, just AGB and BGB in forests were reported within the FRELs recently submitted by Brazil, Colombia, Ecuador, Guyana, Malaysia and Mexico to the UNFCCC (see Conafor, 2014; GFC, 2014; MADS, 2014; MAE, 2014; MMA, 2014; MNRE, 2014). None of these countries, however, included an estimate of the BGN resulting from deforestation, nor the pattern of C loss from dead coarse roots with land-cover conversion (e.g. forest-to-pasture conversion). As an alternative, the UNFCCC (2012) proposed the Stepwise Approach under which countries have the option to improve their initial FRELs by including high-quality data, improved methodologies and additional C pools developed from country- or region-specific information and field measurements following an IPCC Tier 3 approach.

In this chapter I present new Tier 3 information and emission factors on BGN C pool and its dynamics during 20 years of forest-to-pasture conversion under different management practices in the Colombian Amazon, and a new methodological approach to estimate C stocks in coarse dead roots during the conversion from forest to pasture. In my study I addressed the following general question: to what extent land-cover change from forest to pasture and subsequent land management practices affect BGN C pool dynamics in the Colombian Amazon? The aim of this study was to better quantify the BGN C stocks and changes with land-cover conversion from forest to pasture in support of REDD+ initiatives. Specifically my objectives were to:

1. Quantify and describe changes in BGN in two sub-regions of the Colombian Amazon with high- and low-grazing intensity after 20 years of forest-to-pasture conversion.
2. Determine the emission factors of BGN C pool in both sub-regions according to IPCC (2006), by applying region-specific equations developed in this study describing the BGN dynamics along 20 years of forest-to-pasture conversion.

6.3. Materials and methods

6.3.1. Root excavation and wood density estimation

From the total standing coarse woody debris (CWD) measured in the field and reported in Chapter 5, five stumps were randomly selected for root excavation at every stage of the forests-to-pasture conversion in each sub-region. In the case of the 20-year-old pasture in HG it was only possible to excavate four stumps, which corresponds to the maximum number of stumps sampled at that stage. Stumps with any signal of resprouting on their surface were discarded and replaced by a different stump. Soil surrounding each one of the selected stumps within a limit of 1 m was removed, reaching a depth of approximately 0.5 m. Once the root system was exposed, five coarse roots were randomly chosen for sampling collection. The cut section of each root was established at the points where roots had a diameter of approximately 5 cm, and samples were cut by using a chainsaw and following a 10 cm long plastic guide. Thus, from each selected root one ~10 cm sample was extracted, and three measurements of diameter and length were registered in different points of the sample. Mean volume (V; cm^3) of every sample was estimated as:

$$V = \frac{\pi}{4} d^2 * l \quad (6.1)$$

where d (cm^2) is the mean diameter and l (cm) is the mean length of each sample. Table 6.1 shows the mean (± SE) diameter, length and volume of the root samples at every stage of the chronosequence. All samples were dried until constant weight using a greenhouse oven in HG (~8 days) and an electrical oven adjusted at 65°C in LG (~2 days), and the value was recorded.

Root wood density ρ (g cm^{-3}) of each sample was calculated as follows:

$$\rho = \frac{m}{V} \quad (6.2)$$

where m (g) is the dry weight of each sample and V its volume. Thus, below-ground wood density at every stage of the forests-to-pasture conversion in each sub-region was estimated as the average of all root samples collected in the field.

Table 6.1 Mean (± SE) diameter (cm^2), length (cm) and volume (cm^3) of the root samples in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon.

Forest-to-pasture conversion	HG							LG							
	Diameter		Length		Volume		n	Diameter		Length		Volume		n
Primary forest	5.4	(0.2)	9.8	(0.1)	225.9	(1.7)	25	5.7	(0.2)	10.0	(0.2)	267.0	(2.0)	25
1-yr pasture	5.5	(0.3)	10.1	(0.2)	259.6	(2.7)	25	5.2	(0.2)	10.9	(0.2)	241.4	(1.9)	25
2-yr pasture	4.9	(0.2)	10.4	(0.1)	204.6	(1.9)	25	5.4	(0.2)	10.3	(0.2)	237.3	(1.8)	25
5-yr pasture	5.0	(0.2)	10.1	(0.2)	207.0	(1.8)	25	5.5	(0.2)	9.9	(0.2)	244.3	(2.2)	25
12-yr pasture	5.1	(0.2)	10.0	(0.2)	210.5	(1.9)	25	5.8	(0.2)	10.1	(0.2)	273.9	(1.7)	25
20-yr pasture	6.4	(0.2)	10.6	(0.2)	347.9	(2.0)	20	6.1	(0.2)	9.8	(0.2)	292.4	(1.7)	25

6.3.2. Below-ground necromass estimates

BGN was estimated following a stepwise procedure. First, it was assumed that the root system of every stump was comprised of only one idealized dead root R (Figure 6.1). Thus, mean BGN of R per unit area (BGN_{Rt}; kg ha^{-1}) in every stage t of the forest-to-pasture conversion was estimated as:

$$BGN_{Rt} = \frac{\left(\Sigma\left(\frac{dw_i * V_i * S_t}{V_t}\right)\right)}{S_t} \tag{6.3}$$

where dw_i (kg) is the dry weight of each root sample i, V_i (cm^3) its volume, V_t (cm^3) is the mean volume of all samples from the same stage t of forest-to-pasture conversion, and S_t is the total number of stumps per hectare in every stage t of the forest-to-pasture conversion reported in Chapter 5. Once BGN_R was calculated for every stage of the chronosequence, the proportion of increase or decrease of BGN_R along the forest-to-pasture conversion (P_{Rt}) with respect to the mean BGN of R in the forest (BGN_{R0}) was obtained as:

$$P_{Rt} = \frac{BGN_{Rt}}{BGN_{R0}} \tag{6.4}$$

For the second step in BGN estimates, total below-ground biomass in forests (BGB_0; Mg ha^{-1}) was estimated by using Cairns et al. (1997) equation:

$$BGB_0 = exp\left(-1.0850 + 0.9256\left(ln(AGB_0)\right)\right) \tag{6.5}$$

where AGB_0 (Mg ha^{-1}) is the above-ground biomass in forests. According to Phillips et al. (2014), AGB in the tropical moist forest of the Colombian Amazon is 328.2 Mg ha^{-1}, so the result of BGB_0 in the same type of forest by using Eqn. 6.5 is 72.1 Mg ha^{-1}.

Then, considering that BGB_0 remains equal immediately after trees have been cut (i.e. at the 1-year-old pasture stage), total BGN per unit area (BGN_t; Mg ha^{-1}) in every stage t of the forest-to-pasture conversion, except for the 1-year-old pasture, was estimated as:

$$BGN_t = P_{Rt} * \left(\frac{BGB_0}{P_{R1}}\right) \tag{6.6}$$

where P_{R1} is the proportion of increase or decrease of BGN_R in the 1-year-old pasture with respect to BGN_{R0}.

Finally, considering that BGN in the undisturbed forest (BGN_0) might persist in the 1-year-old pasture, total BGN in the 1-year-old pasture (BGN_1) was estimated as follows:

$$BGN_1 = BGB_0 + BGN_0 \tag{6.7}$$

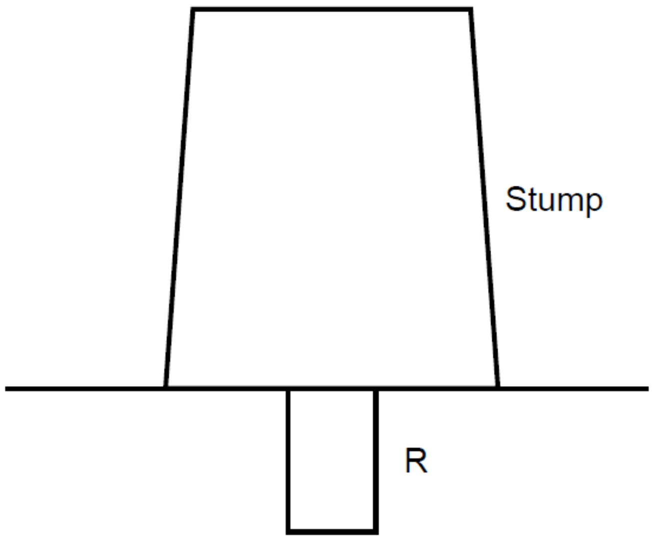

Figure 6.1 Idealized root system (R) of each stump used during the first step of below-ground necromass estimates.

6.3.3. Statistical analysis

All data were tested for normality and homogeneity of variance. Differences in root wood density and BGN_R between similar stages of the forest-to-pasture conversion between HG and LG were assessed by performing t-tests. Analyses of variance (ANOVA) were also performed to establish differences of these variables among the six stages of each chronosequence, and post hoc Tukey-HDS tests were performed when significant differences were found. Regression analyses were used to establish the pattern of variation of BGN during the forest-to-pasture conversion in both sub-regions, and the model which best fitted the data in every case was selected. Following the IPCC (2006) Tier 3 approach, the equations selected to describe the pattern of variation of BGN were used to establish the emission factors for the total change in BGN C pool due to the forest-to-pasture conversion after 20 years of deforestation in both sub-regions of the Colombian Amazon. Thus, the results expressed in Mg ha^{-1} were converted to Mg C ha^{-1} by using a factor of 0.5 assuming that C corresponds to 50% of total necromass (IPCC, 2006), and then converted to tonnes of CO_2 equivalent per hectare (t CO_2e ha^{-1}) by using the C-to-CO_2e conversion factor of 44/12 (UNFCCC, 2012). All the analyses were evaluated with a significance level of $P < 0.05$.

6.4. Results

6.4.1. Roots wood density

There were no significance differences in mean root wood density (ρ) between similar stages of each chronosequence in HG and LG, except in the 2-year-old pasture where root ρ was significantly higher in HG (Table 6.2). When assessing differences within each sub-region, mean root ρ was similar during the forest and the 1-, 2- and 12-year-old pasture stages in HG, and it was significantly lower in the 5- and 20-year-old pastures. On the other hand, significantly lower values of root ρ were found in the 2- and 5-year-old pastures in LG, compared to the other stages of pasture establishment (Table 6.2).

6.4.2. Below-ground necromass

There were no significant differences in BGN_R between similar stages of each chronosequence in HG and LG. However, BGN_R varied significantly among forest-to-pasture stages in HG, ranging from 5.5 ± 0.4 kg ha^{-1} at the forest stage to 1.0 ± 0.1 kg ha^{-1} at the 20-year-old pasture with a pronounced increase during the first year of pasture establishment that reached 24.4 ± 0.8 kg ha^{-1} (Table 6.3). A similar significant variation occurred in LG, where BGN_R varied from 5.9 ± 0.4 kg ha^{-1} at the forest stage to 2.7 ± 0.2 kg ha^{-1} at the 20-year-old pasture with a peak during the 1-year-old pasture of 27.2 ± 0.9 kg ha^{-1} (Table 6.3).

The proportion of increase/decrease of BGN_R along the forest-to-pasture conversion (P_{Rt}) with respect to the mean BGN of R in the forest increased around fourfold during the first year of pasture establishment at forests in both sub-regions, after which P_{Rt} decreased to 0.2 and 0.5 at the end of the 20 years of forests-to-pasture conversion in HG and LG, respectively (Table 6.3). Finally, total BGN in every stage t of the forest-to-pasture conversion (BGN_t) increased from ~16 Mg ha^{-1} to ~88 Mg ha^{-1} in both sub-regions during the transition from forests to 1-year-old pastures, and then decrease until 2.9 Mg ha^{-1} in HG and 7.5 Mg ha^{-1} in LG by the end of the 20 years of pasture establishment (Table 6.3).

Table 6.2 Mean (± SE) root wood density (ρ_s; g cm^{-3}) in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Letters indicate significant differences among stages of the same sub-region ($P < 0.0001$), and the asterisk symbol represents significant differences in wood density between similar stages from each sub-region ($P < 0.0001$).

Forest-to-pasture conversion	HG			LG		
	Root wood density ρ_s		n	Root wood density ρ_s		n
Primary forest	0.61	$(0.07)^{ab}$	25	0.67	$(0.09)^{a}$	25
1-yr pasture	0.69	$(0.07)^{a}$	25	0.59	$(0.08)^{ab}$	25
2-yr pasture	0.70	$(0.05)^{a*}$	25	0.52	$(0.07)^{bc*}$	25
5-yr pasture	0.39	$(0.08)^{c}$	25	0.40	$(0.08)^{c}$	25
12-yr pasture	0.60	$(0.11)^{ab}$	25	0.70	$(0.10)^{a}$	25
20-yr pasture	0.49	$(0.07)^{bc}$	20	0.62	$(0.08)^{ab}$	25

Table 6.3 Mean (± SE) below-ground necromass of R per unit area (BGN_{Rt}; kg ha^{-1}), proportion of BGN_R increase/decrease with respect to the original forest (P_{Rt}), and mean total below-ground necromass in every stage t of the forest-to-pasture conversion (BGN_t; Mg ha^{-1}) in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Letters indicate significant differences among stages of the same sub-region (P < 0.0001).

Forest-to-pasture conversion (t)	HG						LG					
	BGN_{Rt}		P_{Rt}	BGN_t		n	BGN_{Rt}		P_{Rt}	BGN_t		n
Primary forest	5.5	(0.4)^{ab}	1.0	16.1	(0.7)	25	5.9	(0.4)^{ab}	1.0	16.2	(0.7)	25
1-yr pasture	24.4	(0.8)^{c}	4.5	88.2	(1.4)	25	27.2	(0.9)^{c}	4.5	88.3	(1.5)	25
2-yr pasture	13.1	(0.6)^{b}	2.4	38.8	(1.1)	25	10.3	(0.6)^{b}	1.7	28.2	(0.9)	25
5-yr pasture	1.2	(0.2)^{a}	0.2	3.4	(0.3)	25	1.9	(0.3)^{a}	0.3	5.3	(0.4)	25
12-yr pasture	1.2	(0.2)^{a}	0.2	3.6	(0.4)	25	3.0	(0.3)^{ab}	0.5	8.2	(0.4)	25
20-yr pasture	1.0	(0.1)^{a}	0.2	2.9	(0.2)	20	2.7	(0.2)^{ab}	0.5	7.5	(0.4)	25

6.4.3. Total below-ground necromass turnover and emission factors (IPCC Tier 3 approach)

BGN tended to significantly decrease along 20 years of forest-to-pasture conversion in both sub-regions of the Colombian Amazon, following a single exponential decay function in both cases (Figure 6.2). IPCC (2006) recommends developing emission factors of BGN C pool for lands converted to grasslands, including the forest-to-pasture conversion, and the long term legacy effects of land-use management within a Tier 3 approach. Accordingly, the following equations and parameters were fitted to the data in order to describe the change in BGN with time since conversion.

In HG, BGN change after 20years of forest-to-pasture conversion is described by:

$$BGN = 33.6828\, exp(-0.1622yr) \tag{6.8}$$

In LG, on the other hand, BGN is described by:

$$BGN = 31.1846\, exp(-0.1219yr) \tag{6.9}$$

Table 6.4 presents the resulting emission factors for changes in BGN C pool for the forest-to-pasture conversion in HG and LG in tonnes of CO_2 equivalent per hectare (t CO_2e ha^{-1}).

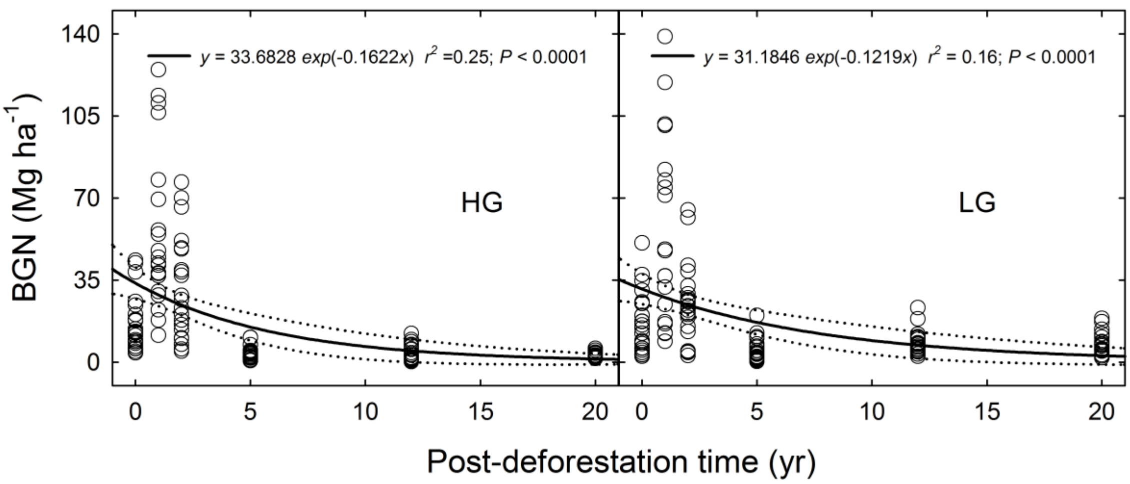

Figure 6.2 Total below-ground necromass (BGN) variation during the forest-to-pasture conversion in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. The dotted lines represent the 95% confidence interval.

Table 6.4 Emission factors (t CO_2e ha^{-1}) for changes in below-ground necromass C pool after 20 years of forest-to-pasture conversion in the high-grazing intensity (HG) and the low-grazing intensity (LG) sub-regions of the Colombian Amazon. Values in parenthesis represent the confidence interval (95%).

Post-deforestation	Below-ground necromass C pool			
time (yr)	HG		LG	
0	61.8	(12.1)	57.2	(11.9)
1	52.5	(9.4)	50.6	(9.4)
2	44.6	(8.3)	44.8	(7.8)
3	38.0	(9.0)	39.7	(7.9)
4	32.3	(9.7)	35.1	(8.8)
5	27.4	(10.3)	31.1	(9.9)
6	23.3	(11.0)	27.5	(10.7)
7	19.8	(11.1)	24.4	(10.6)
8	16.9	(10.7)	21.6	(10.8)
9	14.3	(10.3)	19.1	(10.8)
10	12.2	(9.9)	16.9	(10.9)
11	10.4	(11.5)	15.0	(10.5)
12	8.8	(8.5)	13.2	(10.7)
13	7.5	(8.2)	11.7	(9.9)
14	6.4	(7.6)	10.4	(9.7)
15	5.4	(7.0)	9.2	(9.4)
16	4.6	(6.1)	8.1	(8.6)
17	3.9	(5.9)	7.2	(8.0)
18	3.3	(5.1)	6.4	(7.4)
19	2.8	(5.0)	5.6	(7.4)
20	2.4	(3.5)	5.0	(6.9)

6.5. Discussion

6.5.1. Below-ground necromass turnover with land-use change

Results presented in this study show that BGN stocks decreased during the conversion from forest to pasture in both sub-regions of the Colombian Amazon. After 20 years of pasture establishment there was a net reduction of the BGN of 82% in HG and 54% in LG compared to the original values found in the primary forests in both sub-regions. As expected, BGN considerably increased just after deforestation in both sub-regions, because management practices implemented during the first year of pasture establishment do not include total removal of the cut trees (i.e. stumps) nor their associated root system. Thus, BGN found in the 1-year-old pastures was the result of the original BGN of the naturally-dead trees in the primary forests combined with the BGN produced through deforestation. Then, from the 1- to the 2-year-old pastures BGN decreased 56% in HG and 74% in LG, followed by a pronounced decrease of 91% and 81%, respectively, during the transition from the second to the fifth year of the forest-to-pasture conversion. Finally, during the last 15 years of pasture establishment BGN decrease to 2.9 ± 0.2 Mg ha^{-1} in HG, and to 7.5 ± 0.4 Mg ha^{-1} in LG. Management practices during pasture establishment in the Amazon include the implementation of fire to eliminate part of the woody debris produced after cutting the forest (see Kauffman et al., 1998; Fearnside and Barbosa, 1998; Fearnside et al., 2001; Aragão and Shimabukuro, 2010), and in some regions the use of machinery to remove the unburned or partially burned woody debris (Marin-Spiotta et al., 2009). As it was presented in Chapter 5, farmers in the Colombian Amazon tend to burn twice the deforested areas with a burning efficiency of ~30%, resulting in a reduction of the stump necromass of 71% in HG and 49% in LG from the first to the second year of pasture establishment. Chapter 5 also showed that the use of machinery is a common practice in HG, leading to a reduction of 82% in stump necromass between the second and fifth year of pasture establishment. By contrast, farmers in LG almost exclusively use fire to remove the remnant wood produced after cutting, which led to a reduction of 41% during the same period.

Despite its importance as a significant C reservoir in the tropical forests by storing ~20% of the total C stored in those ecosystems (Aragão et al., 2009), BGB and BGN have been poorly studied due to the considerably high effort required to measure and sample coarse roots in the field (Olajuyigbe et al., 2011). Only a few studies on BGB have been conducted in the tropics to establish root-to-shoot ratios or to develop allometric equations relating this component with AGB (see Saldarriaga et al., 1988; Cairns et al., 1997; Sierra et al., 2007). For example, from data collected in the field Saldarriaga et al. (1988) found that BGB in one mature forest in the Colombian Amazon was 42.0 Mg ha^{-1} and Sierra et al. (2007) estimated that BGB at primary forests in the Colombian Andes was 83.7 Mg ha^{-1}. Estimates of BGB in primary forests of the Colombian Amazon obtained by applying Cairns et al. (1997) allometric equation on AGB reported by Phillips et al. (2014) (i.e. ~72 Mg ha^{-1}), and used in this study to estimate BGN are close to the values reported by Sierra et al. (2007). A larger number of studies on estimating coarse woody root stocks and establishing root decomposition patterns of managed forests have been conducted in temperate and boreal regions, most of them following a chronosequence approach since the harvest event (e.g. see Harmon et al., 2000; Chen et al., 2001; Ludovici et al., 2002; Melin et al., 2009), highlighting the need to expand the research on BGN into tropical ecosystems.

Some studies have used root wood density to monitor the pattern of BGN loss after harvesting in boreal and temperate tree species, following a chronosequence approach (see Chen et al., 2001; Ludovici et al., 2002; Melin et al., 2009). Root wood density values presented in this study tended to decrease during the first five years of pasture establishment in both sub-regions of the Colombian Amazon, after which they significantly increased at the 12-year-old pastures and then decreased again by the end of the 20-year chronosequence. Although it is expected that root wood density declines as decomposition of BGN progresses with time (Harmon et al., 2000), as it occurred during the first five years of pastures establishment, the highest values of root wood density registered at the old pastures in HG and LG would reflect the fact that living trees with high wood density have more probability to resist decomposition and, therefore, to be found at the oldest stages of the chronosequence (Melin et al., 2009).

The chronosequence approach has been widely used to study changes in C pools associated to changes in LULC (see Moraes et al., 1996; Camargo et al., 1999; Harmon et al., 2000; Chen et al., 2001; Ludovici et al., 2002; Shorohova et al., 2008;), as a practical and accurate method to substitute long-term studies by monitoring sites with similar land-use and management at different stages after deforestation (Fearnside and Barbosa, 1998). Nevertheless, studies based on the chronosequence approach may find some issues that could lead to an over or underestimation of changes in the C pools from consecutive stages (Melin et al., 2009). In this study, for example, locating the roots in the field was carried out by establishing the presence of the stumps, and estimates of BGN depended on the average dry weight of the standardized root samples and the total number of stumps found in 1 hectare at each pasture stage (see Chapter 5). Although this is a practical and cost-effective method to locate roots in the field under the chronosequence approach, BGN could be underestimated in the case of the presence of roots in a place without a stump indicating their location. This issue is more likely to be found after the fifth year of pasture establishment, where management practices such as the use of fire or machinery could contribute to eliminate part of the stumps left after deforestation but not necessarily the BGN. Nonetheless, this new methodological approach to estimate BGN stocks contributes to generate new information on BGN C pool dynamics during conversion from forests to any post-deforestation land category such as pastures, and to reduce the uncertainty of estimates of total C emissions due to deforestation.

6.5.2. Improving below-ground necromass carbon pool emission factors (Tier 3)

BGB is one of the C pools to be assessed when deforestation occurs as part of the emission factors contributing to the Forest Reference Emission Level (FREL) that countries implementing REDD+ activities should submit to the UNFCCC (FAO, 2014), and also to be included in the national greenhouse gas inventories within the AFOLU sector (IPCC, 2006). BGB is rarely measured in the field and instead the IPCC allows the use of default forest-specific root-to-shoot ratios under its Tier 1 approach (IPCC, 2006). Under the IPCC Tier 1 approach it is also considered that 100% of C stored in BGB is emitted immediately after deforestation, conservatively assuming that BGN resulting from cutting the trees has a value of zero in a post-deforestation land category such as pastures (IPCC, 2006). Countries participating in REDD+ also have the option to establish Tier 3 field-based inventories to capture the long-term legacy effects of land-cover change on BGN C pool, with the aim of improving the accuracy of estimates of this C pool and its variation with land-use change activities (IPCC, 2006).

All the countries that recently submitted their FREL to the UNFCCC (i.e. Brazil, Colombia, Ecuador, Guyana, Malaysia and Mexico) included the BGB C pool into their reposts, obtained by using default root-to-shoot ratios or allometric equations (see Conafor, 2014; GYC, 2014; MADS, 2014; MAE, 2014; MMA, 2014; MNRE, 2014). However, following a Tier 1 approach, they assumed that 100% of C stored in the BGB was immediately emitted after deforestation. Therefore, the results presented here on BGN C pool and its decrease after 20 years of forest-to-pasture conversion in two sub-regions of the Colombian Amazon with different management practices in terms of grazing intensity are the first data set and equations developed under the IPCC Tier 3 approach using region-specific information. They have the potential to improve the emission factors to be included in the next Colombian FREL by adding Tier 3 information on BGN C pool and its net change due to the conversion from forest to pasture in the Amazon, considering that countries participating in REDD+ activities have the option to improve their FRELs by incorporating new high-quality data and improve methodologies (UNFCCC, 2012).

6.6. Conclusions

Results presented in this chapter suggest that land-cover conversion from forest to pasture led to a pronounced reduction of BGN C pool after 20 years of pasture establishment in the Colombian Amazon. There were no differences in BGN stocks between similar stages of both chronosequences, although deforestation and management practices implemented after pasture establishment led to significant differences in BGN along forest-to-pasture conversion within each sub-region. An expected increase in BGN stocks of around fourfold occurred just after deforestation in both sub-regions, preceded by a decrease in BGN by 50-80% at the end of the 20 years of forests-to-pasture conversion compared to the values in the primary forests.

This study provides a new methodological approach to estimate BGN and new Tier 3 region-specific emission factors on the BGN C pool and its change with forest-to-pasture conversion in the northwest Amazonia. The information provided here have the potential to improve FRELs of countries within the Amazon basin, in particular Colombia. The results presented here emphasize the importance of land conversion and subsequent management practise for BGB and BGN dynamics in the Amazon and contribute to improve the accuracy of the information required for REDD+ initiatives in the region. They also highlight the necessity to describe the land uses and associated management practices occurring in deforested areas, and elaborate spatially-explicit maps indicating their location.

Chapter 7: Improved estimates of net CO_2 emissions from forest-to-pasture conversion in the Colombian Amazon

7. Improved estimates of net CO_2 emissions from forest-to-pasture conversion in the Colombian Amazon

7.1. Abstract

Accurate net CO_2 emissions estimates from land-use change are fundamental to implement REDD+ activities, but the information required to produce reliable reports has been poorly developed. This study presents for the first time a Tier-3 assessment of the net CO_2 emissions from tropical forest-to-pasture conversion over the Colombian Amazon region. Conversion from forests to pasture emitted 555.8 million tonnes of CO_2 equivalent from 2000 to 2012, from which 85% were produced in areas with the highest rates of deforestation, and where pasture areas for cattle ranching are commonly subject to high-grazing intensity practices. Net CO_2 emissions based on Tier 3 region-specific information are 70% higher and substantially more accurate than estimates based on using IPCC Tier 1 information. These results highlight the urgency for countries implementing REDD+ to develop improved data and methodologies that meets the principles of accuracy, completeness, integrity and transparency established in the Paris Agreement of the UNFCCC.

7.2. Introduction

Strategies for avoiding deforestation such as REDD+ could help to reduce anthropogenic GHG emissions by around 2.9-4.8 billion tonnes of CO_2 equivalent (Gt CO_2e) per year (Don et al., 2011). Nevertheless, countries willing to access result-based payments through REDD+ activities requires the development of forest reference emission levels (FREL) as benchmarks for assessing country's performance (UNFCCC, 2010; UNFCCC, 2011; FAO, 2014; UNFCCC, 2014). In general, FREL are obtained from estimates of emission factors, which represent the emissions or removals of all important GHGs associated with land-cover conversion in all relevant C pools per hectare (i.e. total changes in C stocks), and the activity data, referring to the total extent of a deforested, degraded or regenerated area during an established period (Verchot et al., 2012).

By the end of 2014 Brazil, Colombia, Ecuador, Guyana, Malaysia and Mexico submitted their FREL to the UNFCCC as part of the requisites to participate in REDD+ activities. All FRELs, however, only included the above- and below-ground biomass C pools in forests, and conservatively excluded the other C pools and their changes related to CO_2 emissions or removals after conversion from forest to any post-deforestation land-use category (Conafor, 2014; GFC, 2014; MADS, 2014; MAE, 2014; MMA, 2014; MNRE, 2014). In fact, under the IPCC Tier 1 approach it could be assumed that 100% of C stored in the dead wood and below-ground biomass C pools is emitted immediately after deforestation, conservatively assuming no C in theses pools in a post-deforestation land category such as pastures (IPCC, 2006). Similarly, changes in C stocks in soils under a Tier 1 approach might be estimated assuming a linear transition to a new equilibrium in a post-deforestation land category, based on default information about climate, land-use and management practices supplied by IPCC (2006). However, even though the IPCC provides the default information on these parameters, the Tier 1 methodology also allows to conservatively exclude the soil organic C pool when the country submitting the FREL reports a lack of accurate information to estimate GHG emissions and removals from this C pool. Nevertheless, under the UNFCCC Stepwise Approach (UNFCCC, 2012), countries have the option to improve their initial FREL by incorporating high-quality data, improved methodologies and additional C pools developed from country- or region-specific information and field measurements following an IPCC Tier 3 approach.

In recent years Colombia has strengthened its capacity to support and implement REDD+ projects in the country, by improving its information on deforestation rates, drivers of deforestation, and C stocks stored in the above-ground biomass in natural forests (MADS, 2014). According to the latest official available information, more than one million hectares of forest were lost in the Colombian Amazon for the period from 2000 to 2012, and conversion from forests to pastures has become the most common land-use change in the country by occupying ~83% of the converted area (Ideam, 2014a). Nevertheless, estimates of net CO_2 emissions due to deforestation, or particularly to forest-to-pasture conversion, in the Colombian Amazon are uncertain due to a lack of reliable information, especially related to the emission factors of the relevant C pools and their change with land-use conversion. Recent studies demonstrated that high-grazing intensity practices lead to a larger CO_2 emission from the below-ground, dead wood and soils C pools in the Colombian Amazon, compared to areas under low-grazing intensity practices. In fact, the soil C pool in low-grazing intensity areas tends to store more C in the first 30 cm depth after 20 years of pasture establishment, compared to the soils in the original forests (See Chapters 4, 5 and 6).

In this chapter I present the first Tier 3 net CO_2 emissions due to forest-to-pasture conversion in the Colombian Amazon for the period 2000-2012, in support of the REDD+ National Strategy of Colombia. Specifically, in my study I addressed the following two questions:

1. To what extent Tier 3 data and methodologies improve the net CO_2 emissions associated with forest-to-pasture conversion in the Colombian Amazon, compared to the IPCC Tier 1 default information?
2. To what extent management practices related to grazing intensity affect estimates of net CO_2 emissions due forest-to-pasture conversion in the Colombian Amazon under a Tier 3 approach?

7.3. Methods

Cumulative net CO_2 emissions due to forest-to-pasture conversion in both sub-regions of the Colombian Amazon during the period 2000-2012 were calculated as the product of the total deforested area converted to pastures in each sub-region during this period (i.e. the activity data), and the emission factors from the above- and below-ground biomass, soils and dead wood C pools in the forest and pasture categories. The information on the total deforestation in HG and LG during 2000-2012 (Table 7.1) was obtained from the Maps of Forest Change for the periods 2000-2002 (Ideam, 2014b), 2002-2004 (Ideam, 2014c), 2004-2006 (Ideam, 2014d), 2006-2008 (Ideam, 2014e), 2008-2010 (Ideam, 2014f) and 2010-2012 (Ideam, 2014g). Additionally, the proportion of the total deforested area specifically converted to pastures in each sub-region was obtained from the Maps of Forest Change for the periods 2000-2005 (Ideam, 2014h), 2005-2010 (Ideam, 2014i) and 2010-2012 (Ideam, 2014j). All maps have a spatial resolution of 30 m per pixel and an associated uncertainty of ~15%.

In order to compare the cumulative net CO_2 emissions due to forest-to-pasture conversion in each sub-region under the Tier 3 and Tier 1 approaches (IPCC, 2006), the following information was implemented: i) Tier 3 emission factors on: above-ground biomass in forests (Phillips et al., 2014), above- and below-ground biomass in pastures (Amézquita et al., 2008), soil organic C in forests (Chapter 4), dead wood in forests (Chapter 5), and below-ground necromass in forests (Chapter 6); ii) Tier 3 long-term stock change factors on: soil organic C during forests-to-pasture conversion (Chapter 4), dead wood during forests-to-pasture conversion (Chapter 5), and below-ground necromass during forests-to-pasture conversion (Chapter 6); iii) default Tier 1 emission factors on: above-ground biomass in forests (Table 4.7 in IPCC, 2006), above- and below-ground biomass in pastures (Tables 4.4. and 6.4 in IPCC, 2006), below-ground biomass in forests (Table 6.1 in IPCC, 2006), dead wood in forests (Table 2.2 in IPCC, 2006), and soil organic C in forests (Table 2.3 in IPCC, 2006); and iv) default Tier 1 long-term stock change factors on: below-ground biomass during forests-to-pasture conversion (Table 4.4 in IPCC, 2006), and soil organic C during forests-to-pasture conversion (Table 6.2 in IPCC, 2006). For the estimates of CO_2 emissions, below-ground biomass C pool includes the below-ground biomass and necromass in forests and pastures (see Chapter 6). A summary of the information used in this study is presented in Table 7.2 and Table 7.3. As IPCC (2006) recommends, total emissions due to forest-to-pasture conversion were marked with a negative (-) sign.

Following (GOFC-GOLD, 2014), total uncertainty of net CO_2 emissions (UT; %) during forest-to-pasture conversion in the Colombian Amazon was first calculated as:

$$U_{Tci} = \sqrt{U_A^2 + U_{Ci}^2} \qquad (7.1)$$

where, U_{Tc} (%) is the uncertainty associated with the total CO_2 emissions from each C pool (C_i) within the total extent of the converted area, and U_A (%) and U_{Ci} (%) are the uncertainties associated with both the total converted area (ha) and the emission factors of C_i (t CO_2e ha^{-1}), respectively. Finally, U_T was calculated as:

$$U_T = \frac{\sqrt{(U_{Tc1} \times x_{C1})^2 + (U_{Tc2} \times x_{C2})^2 + (U_{Tcn} \times x_{Ci})^2}}{|x_{C1} + x_{C2} + x_{Ci}|} \qquad (7.2)$$

where, x_{ci} (t CO_2e) represents the total CO_2 emissions from each C pool.

Table 7.1 Total deforested area (ha) and total area converted to pastures (ha) in the Colombian Amazon during the period 2000 and 2012 according to Ideam (2014a). HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

Period of	Total		Pasture	
deforestation	HG	LG	HG	LG
2000-2002	106,552	14,107	88,438	11,709
2002-2004	151,386	32,660	125,651	27,108
2004-2006	102,601	28,811	85,159	23,913
2006-2008	129,741	25,209	107,685	20,924
2008-2010	133,929	29,278	111,161	24,301
2010-2012	133,183	20,547	110,542	17,054
Total	757,392	150,612	628,635	125,008

Table 7.2 Tier 3 emission factors (t CO_2e ha^{-1}) for the above- and below ground biomass, dead wood, and soil organic C pools in forests and pastures, and long-term stock change factors for 20 years of forest-to-pasture conversion in the Colombian Amazon. HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

C Pool	Emission factors in forests		Emission factors in pastures		Long-term stock change factors for forest-to-pasture conversion (20 yr)
HG					
Above-ground biomass - forests (AGB_F)	601.7	(21.5)§	-		-
Above-ground biomass - pastures (AGB_P)	-		4.7	(1.5)†	-
Below-ground biomass - forests (BGB_F)	61.8	(12.1)¶	-		$BGB_F = 33.6828 * exp(-0.1622yr)$¶
Below-ground biomass - pastures (BGB_P)	-		13.4	(1.0)†	-
Dead wood (DW_T; standing + fallen)	61.9	(16.8)*	-		$DW_T = 33.7748 * exp(-0.2052yr)$*
Soil (SOC)	191.0	(15.1)¥	-		$SOC = 52.1024 * exp(-0.216yr)$¥
LG					
Above-ground biomass - forests (AGB_F)	601.7	(21.5)§	-		-
Above-ground biomass - pastures (AGB_P)	-		4.0	(1.0)†	-
Below-ground biomass - forests (BGB_F)	57.2	(11.9)¶	-		$BGB_F = 31.1846 * exp(-0.1219yr)$¶
Below-ground biomass - pastures (BGB_P)	-		9.2	(0.5)†	-
Dead wood (DW_T; standing + fallen)	63.9	(14.2)*	-		$DW_T = 44.8387 * exp(-0.1994yr)$*
Soil (SOC)	179.7	(16.0)¥	-		$SOC = 49.0154 + 0.8807yr$¥

§ Phillips et al. (2014)
¥ Chapter 2
* Chapter 3
¶ Chapter 4
† Amézquita et al. (2008)

Table 7.3 IPCC Tier 1 emission factors (t CO_2e ha^{-1}) for the above- and below ground biomass, dead wood, and soil organic C pools in forests and pastures, and long-term stock change factors for 20 years of forest-to-pasture conversion in the Colombian Amazon. HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

C Pool	Emission factor in forests	Emission factor in pastures	± Error (%/interval)	Long-term stock change factors for forest-to-pasture conversion (20 yr)
Above-ground biomass - forests (AGB_F)§	220.0	-	(120-400)	-
Above-ground biomass - pastures (AGB_P)¶	-	19.9	75%	-
Below-ground biomass - forests (BGB_F)*	81.4	-	R:S ratio 0.4	$BGB_{(yr)} = BGB_F - ((BGB_F/20)*yr)$
Below-ground biomass - pastures (BGB_P)¥	-	-	130%	-
Dead wood (DW_T; standing + fallen)†	-	-	-	-
Soil in forests (SOC_F)€	11.0	-	90%	$SOC_{(yr)} = SOC_F + (((SOC_F*F_{LU}*F_{MG}*F_I)/20)*yr)$
Land use factor (F_{LU})£	-	1.0	-	-
Management factor (F_{MG})£	-	1.3	9%	-
Input factor (F_I)£	-	1.2	7%	-

§ IPCC (2006), Table 4.7
¶ IPCC (2006), Table 6.4
* IPCC (2006), Table 4.4
¥ IPCC (2006), Table 6.1
† IPCC (2006), Table 2.2
€ IPCC (2006), Table 2.3
£ IPCC (2006), Table 6.2

7.4. Results

7.4.1. Tier 3 versus Tier 1

Cumulative net CO_2 emissions due to forest-to-pasture conversion in the Colombian Amazon using a Tier 3 approach totalling -555.8 ± 65.2 x 10^6 t CO_2e for the period 2000-2012 (Table 7.4). Nevertheless, when calculations were based on a combination of Tier 3 values for AGB in forests and IPCC Tier 1 default values and equations for the other C pools, estimates of cumulative net CO_2 emissions were ~17% lower and uncertainty increased to more than 115% compared to the Tier 3 results. Even more pronounced was the reduction of cumulative net CO_2 emissions estimates when only IPCC Tier 1 default values were used, where results reduced by more than 70%, while the uncertainty increased to 99% in comparison with the more accurate Tier 3 results (Figure 7.1).

7.4.2. High vs Low Grazing Intensity

Estimates of cumulative net CO_2 emissions due to the forest-to-pasture conversion during 2000-2012 when using a Tier 3 approach were fivefold higher in HG compared to LG, totalling -471.0 ± 64.0 x 10^6 t CO_2e and -84.8 ± 12.4 x 10^6 t CO_2e, respectively (Table 7.4). Around 85% of the total net CO_2 emissions after 12 years of pastures establishment came from the changes occurring in the AGB C pool in both sub-regions of the Colombian Amazon, whilst SOC, DW and BGB C pools contributed to the remaining ~15% of the net CO_2 emissions (Table 7.5). Nevertheless, in contrast to the other C pools, the soil organic C pool in LG acted as a C sink during the 12 years of forest-to-pasture conversion by removing 3.0 ± 0.5 x 10^6 t CO_2e (Table 7.5). The same dominant pattern of changes in AGB on net CO_2 emissions is detected when estimates are done by combining Tier 3 and Tier 1 information, or when only Tier 1 information is used. In both cases, however, total net CO_2 emissions is lower than cumulative CO_2 emissions from changes in AGB because IPCC Tier 1 default data for soil and below-ground biomass C pools conservatively assume CO_2 removals from the atmosphere associated with changes in all categories of land-use (Table 7.5). Furthermore, IPCC Tier 1 information does not include default data and equations on dead wood C pool (see Table 7.3).

Table 7.4 Cumulative net CO_2 emissions ($x10^6$ t CO_2e) from forest-to-pasture conversion in the Colombian Amazon during the period 2000-2012 under the use of Tier 3, Tier 1 and a combination of Tier 3 and Tier 1 approaches. Net CO_2 emissions were marked with a negative (-) sign according to IPCC (2006). HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

IPCC tier approach	HG		LG		Colombian Amazon	
Tier 3	-471.0	(64.0)	-84.8	(12.4)	-555.8	(65.2)
Tier 3+1*	-384.9	(522.4)	-75.2	(102.0)	-460.1	(532.3)
Tier 1	-123.9	(144.7)	-24.2	(28.3)	-148.1	(147.4)

*IPCC Tier 1 default values and models applied to all C pools, except for the above-ground biomass in forests (AGB)

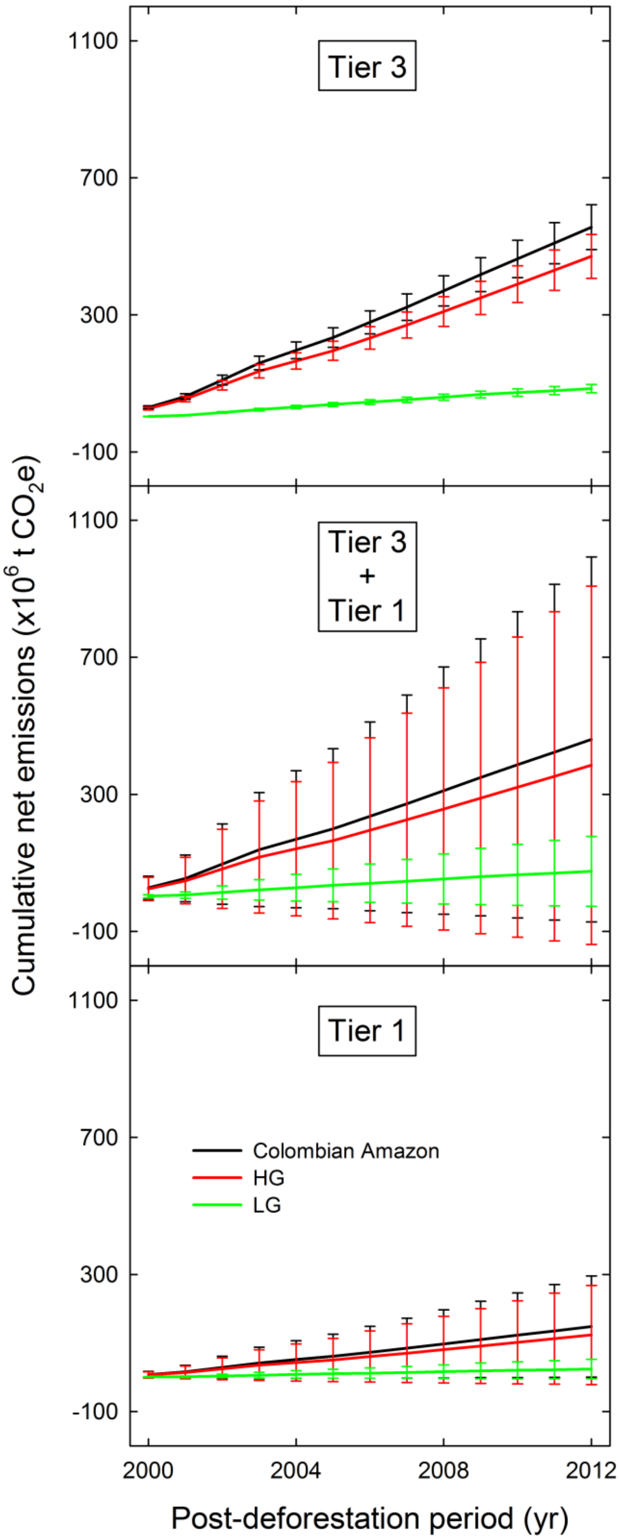

Figure 7.1 Cumulative net CO_2 emissions (x10^6 t CO_2e) from forest-to-pasture conversion in the Colombian Amazon during the period 2000-2012 under the use of Tier 3, Tier 1 and a combination of Tier 3 and Tier 1 approaches. Bars represent the propagated error. HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

Table 7.5 Tier 3 Cumulative net CO_2 emissions (x10^6 t CO_2e) from forest-to-pasture conversion in the Colombian Amazon during the period 2000-2012, from the above-ground biomass (AGB), below-ground biomass (BGB), dead wood (DW), and soil organic carbon (SOC) pools. Net CO_2 emissions were marked with a negative (-) sign according to IPCC (2006). HG: high-grazing intensity sub-region; LG: low-grazing intensity sub-region.

	HG						LG					
C Pool	Tier 3		Tier 3+1*		Tier 1		Tier 3		Tier 3+1*		Tier 1	
AGB	-408.3	(63.0)	-397.9	(520.7)	-136.9	(138.4)	-79.8	(12.3)	-77.7	(101.7)	-26.7	(27.0)
BGB	-16.5	(4.9)	9.6	(15.8)	9.6	(15.8)	-2.9	(0.9)	1.8	(3.0)	1.8	(3.0)
DW	-28.6	(8.9)	0.0	(0.0)	0.0	(0.0)	-5.1	(1.4)	0.0	(0.0)	0.0	(0.0)
SOC	-17.6	(3.0)	3.4	(3.1)	3.4	(3.1)	3.0	(0.5)	0.7	(0.6)	0.7	(0.6)

*IPCC Tier 1 default values and models applied to all C pools, except for above-ground biomass in forests (AGB)

7.5. Discussion

Cumulative net CO_2 emissions during the period 2000-2012 reached 555.8 million tonnes of CO_2 equivalent, exclusively from the conversion from forests to pastures in the Colombian Amazon. This elevated value of CO_2 emissions is associated with the fact that the highest rates of deforestation in Colombia have been historically recorded in this region (Etter et al., 2006; Ideam, 2014a; MADS, 2014), and that most of the deforested area in the whole Amazon has been occupied by cattle pastures (Fearnside and Barbosa, 1998; Asner et al., 2004; Desjardins et al., 2004). As expected, total cumulative net CO_2 emissions due to forest-to-pasture conversion in the Colombian Amazon were dominated by the emissions occurred in HG, where the total deforested area during 2000-2012 was 757,392 ha, from which 628,635 ha were converted to pastures (Ideam, 2014a). In comparison, total deforested area in LG was 150,612 ha for the same period, from which 125,008 ha corresponded to areas converted to pastures (Ideam, 2014a).

7.5.1. Tier 3 versus Tier 1 approaches

The results presented here also demonstrate that IPCC Tier 1 default values and models underestimate net CO_2 emissions from forest-to-pasture conversion in the Colombian Amazon by more than 70% and increase the uncertainty associated to the estimates by 99%, compared to the most accurate region-specific Tier 3 estimates. When cumulative net CO_2 emissions were estimated by combining region-specific Tier 3 emission factors on above-ground biomass in forests (Phillips et al., 2014) and IPCC Tier 1 default values emission factors and long-term stock change factors for the other C pools (IPCC, 2006), the results were underestimated by ~17% compared to the most accurate Tier 3 estimates.

Nevertheless, the uncertainty associated to these estimates also increased by 115%, due to the error propagation from the highly-uncertain IPCC Tier 1 emission factors (see Table 7.3).

On the other hand, these results also indicate that, specifically for the forest-to-pasture conversion, CO_2 emissions produced from changes in the above-ground biomass C pool (i.e. from AGBF to AGBP) comprise the larger proportion of total emissions in the Colombian Amazon. Indeed, replacement of forest AGB by pasture AGB *per se* generates a net CO_2 emission of around -597.0 ± 22.3 t CO_2e ha^{-1} yr^{-1} during the forest-to-pasture conversion in HG and LG, which represents ~85% of total CO_2 emissions when including all C pools. Nevertheless, net CO_2 emissions estimates could be different if the forest is replaced by another post-deforestation land-use (e.g. croplands), or if pastures are replaced by forests (i.e. regeneration), where AGB C pool might store higher amounts of C, resulting in lower net CO_2 emissions estimates. Unfortunately, information on C content in croplands and during secondary succession is scarce, and estimates of CO_2 emissions associated with forest-to-cropland or pasture-to-forest conversions would be highly uncertain. Although forest-to-pasture conversion is the most common land-use change in the Amazon (Fearnside and Barbosa, 1998; Asner et al., 2004; Desjardins et al., 2004), the expansion during the last decades of soybean and cocoa crops, among others, in the region (Nepstad et al., 2008; Pacheco et al., 2012) makes crucial the development of Tier 3 information and models to describe CO_2 emissions from these land-use change categories.

Regardless of the high uncertainty associated to Tier 3+1 estimates, the similar results of net CO_2 emissions due to forest-to-pasture conversion between Tier 3 and Tier 3+1 approaches is also associated with the lower net CO_2 emissions from dead wood, below-ground biomass and soil C pools in both sub-regions. Although soils store in the first 30 cm around 20% of total C stored in forests of the Colombian Amazon (see Table 7.2), and that their stocks decrease by 20% in high-grazing intensity pastures and increase by 41% in low-grazing intensity pastures 20 years after pasture establishment (see Chapter 4), their contribution to the total net CO_2 emissions from forest-to-pasture conversion, corresponds to ~6%. The similarity in estimates of CO_2 emissions between both approaches is even closer in LG (see Table 7.4), because IPCC Tier 1 long-term stock change factors conservatively assume a soil C sink after 20 years of pasture establishment, as it occurs in LG under the Tier 3 approach reported in Chapter 4. Likewise, below-ground biomass C pool conservatively increases with forest-to-pasture conversion under an IPCC Tier 1 approach, opposite to the most accurate Tier 3 estimates. Finally, due to a lack of information on C stocks stored in dead wood, IPCC does not report Tier 1 emission factors for this C pool within its default information. As well as for above-ground biomass C pool, CO_2 emissions and removals associated with soils, dead wood and below-ground biomass C pools should be assessed under different land-use changes to improve the accuracy of the estimates.

To my knowledge, this is the first report on net CO_2 emissions from forest-to-pasture conversion in the Colombian Amazon, based on recently-developed, Tier 3 emission factors and long-term stock change factors. IPCC Tier 1 information represents a starting option for reporting GHG emissions and removals associated with deforestation in countries where accessibility to accurate data and methodologies is limited (IPCC, 2006). Nonetheless, it should be progressively improved with the enhancement of capacity-building to support REDD+ projects in developing countries in terms of accuracy, transparency, completeness, consistency and avoidance of double counting, in accordance with the Paris Agreement (UNFCCC, 2015).

7.5.2. High- versus low-grazing intensity

Results of Tier 3 cumulative net CO_2 emissions highlight the contribution of each C pool to the total CO_2 emissions in both sub-regions (see Table 7.5), and demonstrate that the difference in CO_2 emissions within the Colombian Amazon is attributed to the different management practices implemented in each sub-region after pasture establishment. Indeed, low-grazing intensity management practices lead to long-term CO_2 removals from the atmosphere to the soil C pool (Chapter 4), and to a less pronounced C loss stored in dead wood (Chapter 5) and below-ground biomass (Chapter 6), compared to areas where high-grazing intensity management practices are applied. Also, as a concomitant consequence, soils under high-grazing intensity practices in the Colombian Amazon, represented by a high density of cattle population in pastures with low carrying capacity, poor maintenance of the introduced grasses and the use of machinery, also have a higher susceptibility to erode and consequently to reduce their productivity, compared to soils under low-grazing intensity practices (Chapter 4).

Overall, these results indicate that low-grazing intensity practices have the potential to help to preserve C stocks and to reduce net CO_2 emissions after land-cover change from forest to pasture in the region. Nevertheless, they also indicate that forest-to-pasture conversion emits elevated amounts of CO_2 from areas under either management practice implemented after pasture establishment in the Colombian Amazon.

7.5.3. Improving the next FREL of Colombia

According to the Article 4 of the Paris Agreement (UNFCCC, 2015), each country shall undertake rapid reductions in GHG emissions in accordance with the best available information, and communicate its own nationally determined contribution to strengthen the global response to the risk of climate change every five years. In 2014 Colombia submitted its forest reference emission level to the UNFCCC as a benchmark for assessing the country's performance in reducing emissions from deforestation, and as one of the requirements that countries implementing REDD+ activities shall meet (FAO, 2014). Nevertheless, the country only included information about the C content in the above- and below-ground biomass in natural forests of the Colombian Amazon within its FREL, but conservatively excluded the emission factors for soil, dead wood and below-ground biomass C pools, and their changes after conversion from forest to any post-deforestation land-use category, such as pastures, due to a lack of information (MADS, 2014).

The results presented here represent a step change towards the improvement of the next FREL of Colombia and, consequently, to the strengthen of the Colombian REDD+ national strategy, by providing the first accurate net CO_2 emissions from the conversion from forests to pastures in one of the most deforested region of the country: the Colombian Amazon. Further efforts are needed, however, to complement net CO_2 emission estimates due to land-use change, throughout the development of Tier 3 emission factors and long-term stock change factors in other land-use change categories (e.g. forest-to-cropland or pasture-to-forest conversions) that include all C pools. The methodology and results on net CO_2 emissions reported here could also be replicated in other regions of Colombia and in other countries around the world, where GHG emission due to deforestation, forest degradation and regeneration are contributing to increase the risk on climate change.

7.6. Conclusions

Two main conclusions emerge from this study. Firstly, the results presented in this chapter highlight the necessity for countries participating in REDD+ activities, and in this particular case Colombia, to continue building their capacities to support REDD+ projects, in relation to the development and improvement of country-specific emission factors and activity data that, in turn, will contribute to reduce the uncertainty in reported net CO_2 emissions. IPCC Tier 1 default information is important for countries with limited access to accurate data and methodologies for reporting GHG emissions associated with the agriculture, forestry and other land use (AFOLU) sector, but its highly-uncertain results makes it crucial to strengthen the efforts to produce reliable information that meets the UNFCCC principles of accuracy, completeness, consistency and transparency. The recently signed Paris Agreement opens the way for developed countries to provide financial resources to assist developing countries in building their capacities to implement strategies to mitigate climate change such as REDD+.

Secondly, it is evident that conversion from forests to pastures in the Colombian Amazon emits large amounts of CO_2, regardless of whether CO_2 is emitted from pastures under high- or low-grazing intensity practices. Nevertheless, pasture areas under low-grazing intensity practices emit less CO_2, because soils in these areas act as C sink by removing CO_2 from the atmosphere, and because dead wood and below-ground biomass C pools help to preserve the C stocks for longer. Although forest-to-pasture conversion is the most common land-use change in the Colombian Amazon, more information on CO_2 emissions from different land-use change categories and management practices is required to improve the accuracy of net CO_2 emissions in the region and the country.

Chapter 8: Conclusions and recommendations

8. Discussion, conclusions and recommendations

8.1. An optimized strategy to improve cattle ranching in Colombia

Defining a strategy to implement cattle ranching activities in a sustainable way in Colombia, depends on the trade-off between food demand/production and the maintenance of one of the most important sectors of the Colombian economy, and the preservation of the ecosystem services and biodiversity of natural and human-transformed habitats. Land sparing, a strategy in which areas for conservation are separated from high-yield farming areas, has the potential to minimize the negative impacts of food production and to optimize biodiversity conservation (Phalan et al., 2011), as well as to reduce GHG emissions from agriculture and cattle ranching (Lamb et al., 2016).

Land sparing is also useful for implementing REDD+ strategies (Phalan et al., 2011), because it helps to reduce the pressure on natural ecosystems, such as undisturbed forests, by intensifying agricultural yields in crops and cattle ranching areas. As Foley et al. (2011) pointed out, incrementing food production without deforesting and expanding the agricultural area implies improving the crops and beef production in existing agricultural lands.

Results presented here indicate that net CO_2 emissions from forest-to-pasture conversion in the Colombian Amazon during the period 2000-2012 were larger in HG than in LG, totalling 471.0 x 10^6 t CO_2e and over an area of 628 635 ha in HG and 84.8 x 10^6 t CO_2e and over an area of 125 008 ha in LG. Clearly, the high-grazing intensity system emits a larger amount of CO_2 per unit area. Nevertheless, when comparing the high- and low-grazing intensity systems in terms of emission intensity (i.e. CO_2 emissions per unit of product; Smith et al., 2014), net CO_2 emissions per head of forage-fed livestock (HFFL) increase dramatically in LG compared to HG: 22 611.9 t CO_2e HFFL^{-1} and 277.5 t CO_2e HFFL^{-1}, respectively.

An optimal cattle ranching system for the Colombian Amazon should consider a land sparing strategy, in which food production and biodiversity conservation can be achieved. Pasture lands within this optimal strategy should have a cattle density below the carrying capacity of pastures in the Colombian Amazon (i.e. 1.0 HFFL ha-1; Mahecha et al. 2002), in order to reduce CO_2 emissions and soil erosion, and to preserve the ecosystem services of the intact and transformed ecosystems.

8.2. CO_2 emission estimates are now better quantified

As it was demonstrated in Chapter 7, Tier 3 region-specific emission factors contribute to better quantify net CO_2 emissions associated with deforestation by improving the estimates and by reducing the uncertainty resulting from the use of IPCC Tier 1 default information. In turn, development of accurate estimates of CO_2 emissions will contribute to the UNFCCC objective to strengthen the global response to the threat of climate change in the context of sustainable development and poverty eradication. Our ability to hold the increase in the global temperature to below 2 °C pre-industrial levels depends, in part, on the accuracy of the information used to estimate CO_2 emissions within the AFOLU sector, and this research highlights the risk of using IPCC Tier 1 data which underestimate the actual emissions produced with land-use and land-cover changes.

Strategies to reduce CO_2 emissions from deforestation and forest degradation, the conservation and sustainable management of forests, and the enhancement of forest C stock, depend on reliable results-based payments to guarantee their permanence and effectiveness. The use of region-specific Tier 3 information to estimate CO_2 emissions associated with LULC changes has direct implications on the estimates of the opportunity costs of REDD+ projects: "*the difference in net earnings from conserving or enhancing forests versus converting them to other, typically more valuable, land-uses*" (World Bank, 2011). The opportunity costs of a REDD+ project will be substantially lower than those produced by, for example, cattle ranching activities when estimated from IPCC Tier 1 information, compared to those obtained when Tier 3 region-specific information is used. This demonstrates the urgency for countries participating in REDD+ activities to develop improved information that guarantee the transition towards a low-emissions economy.

8.3. Region-specific Tier 3 information on stocks and dynamics of soil, dead wood and below-ground biomass C pools

This research presents the first region-specific Tier 3 information and emission factors on soil, dead wood and below-ground biomass C pools and their dynamics during 20 years of forest-to-pasture conversion under different management practices in the Colombian Amazon. Results demonstrate that low-grazing intensity management practices lead to long-term CO_2 removals from the atmosphere to the soil C pool, and to a less pronounced C loss from dead wood and below-ground biomass, compared to areas where high-grazing intensity management practices are applied.

The implementation of low-grazing management practices after pasture establishment, including a reduction in the use of machinery, the implementation of a silvopastoral system, or the reduction of the cattle density per hectare to values equal or below to the pasture carrying capacity, could contribute to reduce CO_2 emissions following deforestation by better preserving the C stored in the soil, dead wood and below-ground biomass C pools after 20 years of forest-to-pasture conversion compared to pastures under high-grazing practices. However, in order to accurately estimate the impact of LULC changes on C stocks and dynamics, it is essential to develop a detailed description of all post-deforestation land-use categories and management practices, including spatially-explicit maps describing their location.

8.4. Perspectives and challenges

This research represents a step change towards the improvement of the next FREL of Colombia and, consequently, to the strengthen of the Colombian REDD+ national strategy, by providing the first accurate net CO_2 emissions from the conversion from forests to pastures in one of the most deforested region of the country: the Colombian Amazon. Capacity-building to support REDD+ projects in Colombia, however, requires grater effort to improve the information on net CO_2 emissions associated with other categories of land-use changes, for example forest-to-cropland or pasture-to-forest conversions, and to expand the research to regions in Colombia where deforestation is also high, such as the Andes and Pacific regions. In fact, the methods and results presented here could be extrapolated to the different land-use category transitions and to tropical countries implementing REDD+ activities in order to design their own methodologies and to improve their forest reference emission levels.

8.5. Analysis of the chronosequence approach

The chronosequence approach is a practical and commonly used method to monitor changes in C stocks due to land-use change from, for example, forest to pasture (e.g. Moraes et al., 1996; Camargo et al., 1999; Marin-Spiotta et al., 2009), or from abandoned agricultural lands to secondary forests (e.g. Kennard, 2000; Lebrija-Trejos et al., 2008), in the absence of long-term information from fixed locations. However, the method has received some criticism in studies of secondary succession in tropical forests, particularly because of the lack of information on land-use and management histories from the assessed sites (Feldpausch et al., 2007; Maza-Villalobos et al., 2011; Mora et al., 2015). The sites selected to represent the chronosequence of conversion from forest to pasture in this study share a similar land-use and management history in each sub-region, according to information from remote sensing, official maps of deforestation in Colombia, interviews with landowners and local people, and information on accessibility in terms of mobility and security. The assessment of the last criterion limited the availability of potential sampling sites located in remote or high-risk areas in both sub-regions, which may reduce the possibility of capturing the whole variability on C stocks and dynamics with forest-to-pasture conversion in unexplored areas in HG and LG. The lack of information on C stocks and dynamics after deforestation from long-term studies in fixed sites, makes the chronosequence approach a suitable alternative to improve the information required in support of REDD+ initiatives. Nevertheless, a complementary set of long-term dynamic studies on C dynamics after deforestation should be established in fixed sites, in order to cover the high variability in landscape and soil properties occurring within extensive regions such as the Colombian Amazon, and to reduce the uncertainty on CO_2 emissions estimates associated with land-use and management history.

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Appendices

10. Appendices

This section contains the following papers in their accepted and final version, respectively:

1. Navarrete, D., Sitch, S., Aragão, L.E.O.C., Pedron, L., 2016. **Conversion from forests to pastures in the Colombian Amazon leads to contrasting soil carbon dynamics depending on land management practices**. Global Change Biology, doi: 10.1111/gcb.13266
2. Navarrete, D., Sitch, S., Aragão, L.E.O.C., Pedron, L., Duque, A., 2016. **Conversion from forests to pastures in the Colombian Amazon leads to differences in dead wood dynamics depending on land management practices**. Journal of Environmental Management, 171, 42–51.