THE LIFECYCLE OF WOOD FROM TROPICAL FORESTS IN COSTA RICA

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Table of Contents

Chapter 1

Introduction

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Wood is naturally multi-functional and renewable. It is strong enough to provide structural support, while allowing the flow and storage of water and chemicals along the tree (Jakes et al., 2016; Ramage et al., 2017). This natural multi-functionality has translated into multiple materials and uses, such as food, chemicals, textiles, shelter, etc. In a world that is in urgent need of transitioning onto sustainable solutions for development, bio-materials and bio-energy produced from wood seem a logical choice (Glew, Stringer, Acquaye, & McQueen-Mason, 2017; Wohlfahrt et al., 2019). Wood can be considered the cornerstone of a bio-economy which relies on traditional applications coupled with technological transformation and innovation (Scarlat, Dallemand, Monforti-Ferrario, & Nita, 2015). The substitution of non-renewable materials and energy is the main potential contribution from the bio-economy to the mitigation of man-made climate change. There are only two problems with wood. It is abundant and not highly valued (Glew et al., 2017; Oliver & Mesznik, 2006), and it requires harvesting, which is an activity that suffers largely from a negative generalized perception (Edwards, Tobias, Sheil, Meijaard, & Laurance, 2014). In this thesis I quantify the potential contribution of tropical wood production to climate mitigation and the bio-economy, with focus on Costa Rica as a case study.

Tropical forest management

Logging forests as a climate mitigation strategy may raise some controversy, as it could seem contradictory to the goals of reducing atmospheric GHG concentrations. Most forests' potential for climate mitigation is precisely avoiding emissions from deforestation or degradation, sequestrating carbon through forest growth instead, and storing carbon in the biomass and soil (Canadell & Schulze, 2014). Logging, on the other hand, causes a disturbance to the ecosystem by extracting wood and damaging the surrounding biomass in the process. In fact, damage to the ecosystem can be two or three times as high as the amount of wood that is extracted (Ellis et al., 2019; Seiji Hashimoto, 2008; Pearson, Brown, Murray, & Sidman, 2017), and this may seem inefficient. Ecosystem carbon losses arise from harvested wood, damage from felling trees, and the infrastructure required to access, store and transport trees outside forests.

Tropical forests cover a vast area, but the area estimated for production is close to 400 million ha, out of which 165 million ha are available for harvesting (Blaser, Sarre, Poore, & Johnson, 2011). Due to the scale of selectively logged forests and the high carbon density in one hectare of tropical forests, the impact of management is large. At a per hectare level, between 6.8–50.7 Mg C ha^{-1} can be lost due to harvesting (Pearson, Brown, & Casarim, 2014). Tropical degradation due to logging accounts for 1.1 Gt CO_2 emissions annually (Pearson et al., 2017). Damage caused to forests is by far the largest source of emissions, but these vary according to harvesting practices and the intensity of harvest (Martin, Newton, Pfeifer, Khoo, & Bullock, 2015; Piponiot et al., 2018). Tropical forest management generally applies a selective logging system where only a few trees per hectare are harvested and the damage is inherent to these systems. Although some damage is still inevitable, there are opportunities to avoid 30-40% of carbon losses through reduced impact logging techniques (Ellis et al., 2019; Francis E. Putz et al., 2008).

Immediately after harvesting, forests are allowed to recover for a period of 15 to 60 years depending on local standards (MINAE, 2002; Rutishauser et al., 2015; Sasaki, Chheng, & Ty, 2012). This timeframe for recovery may vary due to forest type, local conditions and the intensity of the disturbance (Baccini et al., 2012; Piponiot et al., 2018; Rutishauser et al., 2015). As a result, there is debate on whether current standards for rotation or logging cycles provide enough time for recovery. This is important from an ecological perspective, but as in any human-nature interaction, socio-economic factors influence this decision as well. A rotation period partly reflects the compromise between environmental and socio-economic benefits. For example, in Costa Rica the minimum rotation period is 15 years, considered by some to be extremely short (Arroyo-Mora, Svob, Kalacska, & Chazdon, 2014). However, allowable harvest and forest sizes are low. Different from other regions characterized by large forest concessions where harvest intensities can be high and the rotation long, land use benefits need to be maximized to provide a relatively long-term but steady source of income in small, privately owned forests.

The discussion on rotation periods is certainly valid since allowing the forest to recover marks its potential to be used sustainably. If renewable forest resources are exploited beyond their ability to recover, this will result in a degraded forest (Muralikrishna & Manickam, 2017). Therefore, logging can potentially lead to degradation as it strives to find the proper balance between harvest, damage and recovery time, yet this scenario is very different from deforestation or other forms of long-term degradation (Poker & MacDicken, 2016; Sessions, 2007). Managed tropical forests tend to remain as forests and will recover at least part of the carbon lost following a disturbance (Arroyo-Mora et al., 2014; Piponiot et al., 2018; Francis E. Putz et al., 2012; West, Vidal, & Putz, 2014). In a sustainably managed forest, degradation is a temporal state, but most accounts of forest degradation fail to make this differentiation clear. The dominant view is that forest management is not environmentally important, hampering its consideration as part of conservation strategies (Edwards et al., 2014; F. Mohren, 2019; Runting et al., 2019).

Given that deforestation accounts for an average annual loss of five million hectares (an area the size of Costa Rica) and 12% of global CO_2 emissions (Harris et al., 2012; Keenan et al., 2015), many tropical countries are in the early process of implementing or developing national plans and policies for the conservation of forests. These efforts have been triggered by the creation of a global mechanism to Reduce Emissions from Deforestation and Degradation (REDD+) under the United Nations Framework Convention on Climate Change. REDD+ is broadly intended to guide these plans and create ways by which these can be financed (Culas, 2012). Despite REDD+ including forest management among potential measures for emissions reduction, current climate mitigation policies mainly favour a protection approach to forest conservation (Merry, Soares-Filho, Nepstad, Amacher, & Rodrigues, 2009; Sasaki et al., 2016, 2012).

The 400 million hectares of potentially productive natural tropical forests (Blaser et al., 2011) can be an important global asset for REDD+ as they could contribute both to conservation and socio-economic benefits (Merry et al., 2009; Poker & MacDicken, 2016; Sessions, 2007). Additionally, these forests can supply a growing demand for products that would otherwise compromise strategies aimed exclusively at protecting forests (Parker, Merger, Streck, Tennigkeit, & Wilkes, 2014; Sasaki et al., 2016, 2012). The imbalance caused by protection policies has limited opportunities to make a better use of forest resources and forest-based climate mitigation (Ellison, Petersson, Lundblad, & Wikberg, 2013).

Sustainable forest management for climate mitigation

The ecological basis for sustainable forest management and its potential contribution to climate mitigation is that due to saturation, carbon uptake in biological systems tends towards a state of dynamic equilibrium (Intergovernmental Panel on Climate Change, 2014; Schlamadinger et al., 1997). Therefore, by extracting some trees from a mature forest, competition is reduced and the availability of resources (e.g. light and water) triggers a more productive state (Finegan, 2012). If mature forests were to continue growing perpetually, protecting them would probably be the best mitigation option (Bellassen & Luyssaert, 2014). Yet, despite some controversy, the general understanding is that tropical forests follow the dynamic equilibrium hypothesis, or might even be reducing growth due to environmental changes (Finegan, 2012; Roitman, Vanclay, Hay, & Felfili, 2016; Zuidema et al., 2013). Under these conditions, additional carbon sequestration due to protection is low and harvesting could provide additional benefits (Lippke et al., 2011).

If forests are harvested and this harvest decomposes immediately or is combusted (as assumed by estimates of forest degradation), there will be no carbon-related gains from forest management. In fact, this would be counterproductive as carbon once stored is released to the atmosphere, causing increased atmospheric concentrations of CO_2 that will take years before being sequestered back through forest regrowth. However, decomposition is not immediate and even if a large part slowly decomposes on-site, carbon contained in the harvested wood can remain stored for long periods. In addition, even though combustion may be immediate, biomass used as an energy source could substitute fossil fuels and avoid emissions from non-renewable sources (Nabuurs, Arets, & Schelhaas, 2017; Sikkema, Junginger, McFarlane, & Faaij, 2013). Thus, the sustainability of managing forests for wood production depends on processes occurring inside the forest, while additional climate mitigation is determined by storage and substitution effects occurring outside the forest.

Carbon stored in wood products

Due to the multi-functionality of wood, products made from this material are numerous and can include everything from fuels, chemicals, textiles and paper, to sawn-wood for a multitude of uses, and even less known applications such as activated carbon or carbon nanostructures (Jakes et al., 2016; Ramage et al., 2017). Depending on their use, wood products can last for years before being discarded and decomposing, with lifetimes ranging from some months to hundreds of years in the most extreme cases (Brunet-Navarro, Jochheim, & Muys, 2017). Lifetime is not so much dependent on the physical properties of the material but on socio-economic factors such as obsolescence or less utilitarian factors such as fashion (IPCC, 2014; Pingoud & Wagner, 2006; Suter, Steubing, & Hellweg, 2016). Products that have long-term uses such as structural wood or almost any wood used in construction are preferred for carbon storage (Intergovernmental Panel on Climate Change, 2014; Lun, Li, & Liu, 2012).

Carbon will remain stored throughout the lifetime of products, which will accumulate to form a carbon stock of "products in use" outside forests (Brandão et al., 2013; Pingoud, Skog, Martino, Tonosaki, & Xiaoquan, 2006; Pingoud & Wagner, 2006). The size of the stock and the rate at which it grows and decomposes is significant for climate mitigation (Cowie, Pingoud, & Schlamadinger, 2006). Globally, the stock of wood products in use has accumulated around 15-20 Gt CO_2, growing at a rate of 335 – 540 Mt CO_2 per year, and offsetting approximately 1% of global emissions (Iordan, Hu, Arvesen, Kauppi, & Cherubini, 2018; Johnston & Radeloff, 2019). At regional or country levels, this contribution may vary depending on the characteristics of the local forest sector and wood consumption patterns. Growth in the carbon stock is mostly due to increasing harvest levels (Cláudia Dias, Louro, Arroja, & Capela, 2009; Pilli, Fiorese, & Grassi, 2015; Pingoud, Pohjola, & Valsta, 2010) with opportunities to further increase the stock with even higher harvesting. The main limitation is that important changes in wood consumption would be required first (Suter et al., 2016; Werner, Taverna, Hofer, Thürig, & Kaufmann, 2010).

Once wood products are discarded from use, they face several end-of-life options (EoL), including re-utilization, recycling, storage, combustion or incineration. Storage refers to common local municipal solid waste management practices where disposal takes place in solid waste disposal sites (SWDS). These are broadly classified as managed anaerobic (i.e. landfills), unmanaged shallow, unmanaged deep, and uncategorized (e.g. open dumps) (Pipatti et al., 2006). Open dumps are still common in many developing countries so the transition to landfills is seen as an improvement (Ziegler-Rodriguez, Margallo, Aldaco, Vázquez-Rowe, & Kahhat, 2019). Re-utilization and recycling are still not common practices, as well as incineration of waste with or without energy recovery. In tropical countries the open burning of waste is common (Wiedinmyer, Yokelson, & Gullett, 2014; Yadav & Samadder, 2018; Ziegler-Rodriguez et al., 2019), but the information on the scale of open burning and EoL processes in general is extremely uncertain (Akagi et al., 2011; Bogner et al., 2008; Clavreul, Guyonnet, & Christensen, 2012; Pingoud & Wagner, 2006; L. Zhang, Sun, Song, & Xu, 2019).

The type of EoL of wood products partly determines their potential contribution to climate mitigation since the anaerobic conditions found in SWDS promote carbon storage. Under these conditions, lignin becomes recalcitrant as it limits bacterial decomposition and its carbon content becomes enriched during the decomposition of holocellulose (De la Cruz, Chanton, & Barlaz, 2013; F. Ximenes, Björdal, Cowie, & Barlaz, 2015). Site conditions and the physical and chemical properties of lignin determine bacterial decomposition but the process is mostly driven by species specific factors such as the concentrations of cellulose, hemicellulose and lignin, rather than climate (Barlaz, 2006; F. Ximenes et al., 2015). For these reasons, a large fraction of the wood disposed of in SWDS will accumulate indefinitely and the stability of the carbon it stores is mostly challenged by changes in management practices (e.g. higher rates of reutilization, recycling or incineration). This carbon stock is sometimes ignored when accounting for technospheric carbon storage because of the large uncertainties on waste flows towards EoL management, and because the decomposition of organic materials under anaerobic conditions may result in the release of methane, a more potent greenhouse gas (IPCC, 2014; Pingoud & Wagner, 2006). However, when carbon stored in SWDS is included, it may be significantly higher than the amount of carbon in products that are in use and given that only a small fraction of wood decomposes, methane emissions are usually not large enough to offset storage (Ingerson, 2011; Levasseur, Lesage, Margni, & Samson, 2013; Lun et al., 2012; Skog, Pingoud, & Smith, 2004; L. Zhang et al., 2019). Therefore, it is the combined effect of carbon storage in products in use and SWDS, which provide part of the additional contribution to climate mitigation from using wood from a sustainable forest management system.

Substitution

Besides storing carbon, wood can potentially substitute other products or energy sources and avoid the emissions associated to their production and use (Helin, Sokka, Soimakallio, Pingoud, & Pajula, 2013; R Miner, 2010). This is perhaps the main argument to promote forest management and use of wood products as a climate mitigation option, given that the effect from avoiding these emissions is much higher than the contribution from storing carbon (Côté et al., 2002; Ingerson, 2011; Intergovernmental Panel on Climate Change, 2014; Lun et al., 2012). The basis for this argument is that producing and manufacturing wood products consistently shows that it is less energy and input-intensive than producing a similar functional product from other materials (Intergovernmental Panel on Climate Change, 2014; Leskinen et al., 2018; Suter et al., 2016). Most opportunities to avoid emissions can be found in the construction sector, where long-term wood products can substitute more energy-intensive materials such as concrete or steel (Dodoo, Gustavsson, & Sathre, 2009; Perez-Garcia, Lippke, Comnick, & Manriquez, 2005).

A common indicator for this substitution effect is the displacement factor, which describes the units of carbon used in production of non-wood materials and that are displaced by units of carbon in wood products (Helin et al., 2013; Lippke et al., 2011; Pingoud et al., 2010; Sathre & O'Connor, 2010). The most commonly used displacement factor is an average of 2.1 Mg C per Mg C in dry wood (Knauf, Köhl, Mues, Olschofsky, & Frühwald, 2015; Sathre & O'Connor, 2010), but more recent estimates show lower results in a range of 0.8 - 1 kg C per kg C in wood (Geng, Zhang, & Yang, 2017; Keith, Lindenmayer, Macintosh, & Mackey, 2015; Leskinen et al., 2018; Lippke et al., 2011; Rüter et al., 2016). Although these values can vary largely depending on the product used for comparison, most wood products show a contribution to climate mitigation through substitution.

There are two main criticisms when using displacement factors to claim climate mitigation from wood product use: the choice of reference product and the assumption that all wood products substitute other materials (Buchholz, Hurteau, Gunn, & Saah, 2016; Cherubini et al., 2009; Dale & Kim, 2014; Gustavsson & Sathre, 2006; Helin et al., 2013; Lippke, Wilson, Meil, & Taylor, 2010; Pingoud et al., 2010; Plevin, Delucchi, & Creutzig, 2014b, 2014a; Sathre & Gustavsson, 2006; Sathre & O'Connor, 2010; Wolf, Klein, Weber-blaschke, & Richter, 2015). The combination of these two assumptions can lead to large overestimations of the benefits of wood products (Law & Harmon, 2011). When choosing the reference system, the risks are being minimized as more information that can be used as a reference for comparison becomes available from a broader set of circumstances and products (Wolf et al., 2015).

Better estimations of the magnitude of substitution effects are needed but these depend on the goal of the study and require more than just better data. When considering a single product and under unique circumstances, the comparison can be direct and simple, i.e. what is the effect from substituting product x with product y. However, when up-scaled to a sectorial or national level and considering all wood production, the effect from substitution becomes unclear, as there is not a direct relationship between an increase in the production of x and a decrease in the use of y. In these cases, the comparison is not against a reference product, but a reference scenario such as those used in project-based mitigation mechanisms, where the potential benefits are estimated using a 'what-if' scenario compared to a baseline. When observed trends in wood consumption have been used to create such baseline scenarios, these show wood has been replaced by other products, demonstrating there is potential to revert these trends (Pingoud et al., 2010; Suter et al., 2016). Substitution benefits represent the most important contribution to climate mitigation from wood products if these can be attributable to the product system. The allocation of benefits remains controversial mainly because it is case-specific (Helin et al., 2013).

Assessing the lifecycle climate impact of tropical forest management

It has been hypothesized that carbon storage and product substitution will also provide a climate mitigation contribution when using wood from natural tropical forests. This was part of the justification of including forest management under the REDD+ mechanism (Butarbutar, Köhl, & Neupane, 2016; Sasaki et al., 2016, 2012). However, a complete assessment of all processes leading to emissions, storage, sequestration and potential substitution has not yet been conducted for tropical forests managed for wood production. As mentioned, most of the existing evidence of the impacts from logging in the tropics is limited to carbon losses in the biomass of forests (i.e. biogenic carbon) and is therefore an overestimation of its climatic impact. When biogenic carbon balances for tropical forest logging have included assumptions on product use and forest regrowth, the result is a system that still leads to increased carbon emissions but a balance that is closer to neutral (Richard A. Houghton, 2013; Numazawa, Numazawa, Pacca, & John, 2017; Piponiot et al., 2016). However, these studies have only focused on biogenic carbon and estimation of storage in wood that is largely based on assumptions on product allocation (i.e. how wood is used). There is a large gap between these studies and those that have estimated the GHG emissions associated to forestry operations, manufacturing and use of specific products without including biogenic carbon in their estimation (Adu & Eshun, 2014; Eshun, Potting, & Leemans, 2010, 2011; Jankowsky, Galina, & Andrade, 2015; Ramasamy, Ratnasingam, Bakar, Halis, & Muttiah, 2015; Ratnasingam et al., 2015; Rinawati, Sari, & Prayodha, 2018).

To bridge this gap, a lifecycle assessment (LCA) framework provides guidance on how to integrate and assess the different processes that conform a product system to provide a complete account of the climate impact of tropical forest management. The main aim of the LCA framework is to standardize the collection of data, the estimation of impacts and the interpretation of results in a way that comparisons between similar product systems or functional units are possible (Helin et al., 2013; Iritani, Silva, Saavedra, Grael, & Ometto, 2015). Wood production is a special case of LCA, characterized by the challenges to integrate biospheric and technospheric processes. This gap between forest carbon balances and product carbon footprints has also been common in LCA, where biogenic carbon has usually been excluded under the assumption of carbon neutrality (Brandão et al., 2013; Knauf et al., 2015; Newell & Vos, 2012). Carbon neutrality is a broad simplification of the system that is based on the assumption that forest management itself is sustainable. This assumption is now recognized as a major weakness and methodological adjustments and recommendations are constantly being done to address the challenges of a complete LCA for bio-materials and bio-energy (Cardellini et al., 2018; Helin et al., 2013; Johnson, 2009). Yet, there are issues that remain unresolved (Breton, Blanchet, Amor, Beauregard, & Chang, 2018; Helin et al., 2013; Klein, Wolf, Schulz, & Weber-Blaschke, 2015; Knauf et al., 2015; Lippke et al., 2011). Since lack of sustainability is the main criticism in the case of tropical forest management, estimating the climate impact excluding biogenic carbon would be incomplete and misleading.

Challenges of an LCA for wood products are related to the characteristics of the material, which require the system to be bounded both spatially and temporally (Helin et al., 2013). Because wood can be renewable, the balance between emissions and sequestration needs to be accounted for, and both occur under different timeframes. As described previously, there is large variation in the recovery rate of forests, although this timeframe is artificially defined by the rotation period. Then, carbon stored in products is released back to the atmosphere at a slow pace, but this time is independent from the rotation cycle. To account for carbon emissions/storage of products, a 100-yr period is suggested based on the assumption that delaying emissions during this period can have a potential climate mitigation benefit (Reid Miner, 2006). This timeframe is consistent with those used under other mechanisms and in the definition of time horizons used to estimate global warming potentials (GWP). Although, the decision to use this timeframe is also arbitrary and is mainly aimed to serve policy making (Brandão et al., 2013; Cowie et al., 2006). To simplify the decision on the temporal boundary, one rotation period has been suggested as this is the standard boundary used in forestry and it broadly defines the period in which a new cycle of logging will take place (Klein et al., 2015). However, it does not capture the effect from the mismatch between the occurrence of emissions (usually early in the lifecycle) and sequestration on atmospheric GHG concentrations. Several approaches have been recommended to address the timing of emissions, but this probably is the main unresolved issue in the LCA of bio-products (Breton et al., 2018).

Carbon emissions associated with wood use depend largely on the allocation of wood into products. Usually, wood from forests has multiple uses, which to a large degree determine the processes and inputs required to transform and use products (Klein et al., 2015). Products can vary from fuelwood, that requires no or little transformation and induces immediate emissions, to sawn-wood, which can be transformed using a variety of energy-intensive manufacturing processes and induces emissions at a much longer timescale. Additionally, not all wood effectively ends as products due to transformation efficiencies. During sawmilling, efficiencies are close to 50% (Butarbutar et al., 2016; Ofoegbu, Ogbonnaya, & Babalola, 2014; Ramasamy et al., 2015; Sasaki et al., 2016) while further transformation residues can be around 8% (Winjum, Brown, & Schlamadinger, 1998). Wood residues have traditionally been considered waste with, a small margin for re-utilization; and although large differences between countries are expected, this has been changing globally (Mantau, 2015; Sikkema et al., 2013).

The appropriate choice of temporal and spatial system boundaries and functional unit has direct implications in the correct representation of lifecycle processes and their impacts (Newell & Vos, 2012). However, boundaries and functional units are likely to change depending on the goal and scope of the assessment and the information that is available (Reap, Roman, Duncan, & Bras, 2008). In a way, these are subjective, highlighting the complexity and relevance of standardizing procedures used in the lifecycle framework to estimate the environmental impact of products or services. Results from an LCA are aimed at supporting decision-making processes, so these will be compared, e.g. between products, and such comparisons need to be fair. Still, differences in approaches or assumptions are to some extent inevitable, but this is not problematic so long as it is communicated clearly (Helin et al., 2013). In this regard, the estimates of tropical forest degradation are a very good example. Despite limitations in their scope, these estimates highlighted a problem to be addressed, started a public debate, and played a major role in the discussion on whether wood production from forests can be a climate mitigation strategy. Currently, the potential climate impact from harvesting the 400 million hectares of productive tropical forests remains uncertain.

Tropical forest management in Costa Rica

In Costa Rica, natural tropical forests with the capacity for wood production cover approximately 17% of the territory. At this moment, less than 1% of this area is being managed for wood production (Camacho Calvo, 2015; Pedroni, Espejo, & Villegas, 2015; Werger, 2011). Managed forests are mainly located in the Northern Caribbean region of the country (86%) and correspond to tropical moist and tropical wet forests according to Holdridge's classification system. These are all privately owned forests within farms and have an average size of 80 hectares. Currently, these provide 5% of national harvest (Barrantes & Ugalde, 2018). As a consequence of these small harvest levels from forests, sawmills that traditionally depended on this wood source have been disappearing and the total number is currently estimated to be close to 40 (Serrano & Moya, 2011).

Since the 1990s, wood production in the country was transformed by a combination of environmental policies and incentives to forest plantations, which now represent 77% of national harvest (Barrantes & Ugalde, 2018). Planted forests were initially intended to substitute wood from natural forests but are now mainly used for pallets and packaging for agricultural exports (I. Jadin, Meyfroidt, & Lambin, 2016; Isaline Jadin, Meyfroidt, Zamora Pereira, & Lambin, 2016; Santamaría, 2015). Wood used in construction has declined, and for years there have been claims that it has been substituted by concrete, iron and aluminium (Santamaría, 2015; Serrano & Moya, 2011; Werger, 2011).

These changes in forest management are largely explained by Costa Rica´s efforts to recover and protect its forests, a resource that was largely depleted due to an agricultural expansion taking place between 1950-1970 (Santamaría, 2015). As a result of policies to revert deforestation and incentivize reforestation since the 1980s, almost half of the country is now under forest cover (Camacho Calvo, 2015). This cover is comprised of forests under different stages of succession. A keystone for the conservation of Costa Rica´s forests was the establishment of a nationwide payment for environmental services program (PES) which has been running since 1997, and that is now the basis for the country´s REDD+ Strategy. This program was intended to compensate forest owners for the conservation of forests and their services, and included forest management as an activity entitled to an environmental payment. However, managed forests were excluded from this program for almost 10 years (Werger,

2011) and during this time, some regions even set administrative bans and rejected all attempts to obtain logging permits from forests (Camacho, 2015; Santamaría, 2015).

A drawback from Costa Rica´s approach to forest protection is that since the establishment of the PES program, the country has never been able to fully compensate forest owners and an important part of these owners never participated due to lack of funding. Partly for this reason, the country has been a promoter of a REDD+ program under the UNFCCC, and more recently has reconsidered the role of forest management within conservation strategies. As a first step, it reintroduced managed forests in the country´s PES system and later incorporated measures to promote sustainable production and consumption of wood into its REDD+ Strategy (Pedroni et al., 2015; Santamaría, 2015).

Objective

The objective of this PhD study is to determine the potential contribution to climate mitigation of natural forest management in Costa Rica. It is essentially a case study for the Costa Rican forest sector, but the mechanisms behind processes leading to emissions and carbon storage are applicable to other countries as well. Therefore, the evidence put forward in this thesis may help clarify the role of forest management in the tropics in the climate change debate. In addition, there may also be lessons drawn that are applicable to countries in the processes of defining conservation policies under REDD+ programs.

Outline

Chapter 2

In the second chapter, we study the biogenic lifecycle carbon balance of tropical forest management in Costa Rica (Figure 1.1). It focuses on biogenic carbon only, to be comparable with current estimates of tropical forest degradation due to logging. To understand the combined effect of emissions, storage and sequestration we estimate the net balance using one hectare as the functional unit and a rotation period of 15 years as the temporal boundary. To trace wood along this boundary, we used a material flow and lifetime analysis. Spatial boundaries include all processes until the end of life of wood products. The aim is to estimate the net carbon balance of logging forests in Costa Rica, considering all processes of emissions, storage and sequestration.

Chapter 3

In the third chapter, we developed a harvested wood product (HWP) carbon inventory for Costa Rica (Figure 1.1) following a Tier 2 level according to IPCC guidelines (Pingoud et al., 2006). That is, we used country specific information on harvest and product categories. By using this method, we not only increase the accuracy of the inventory but also trace each wood product to its source, i.e. natural forests, plantations and agricultural lands. Based on observed patterns in wood sourcing and product allocation during the analysed period (1990 – 2016), we hypothesized that the stock of carbon in products should be reacting to these changes. Changes in allocation cause important changes in the stock's half-life, and understanding how the stock reacts to these changes can help clarify mechanisms leading to increased climate mitigation by increasing product lifespan.

Chapter 4

In the fourth chapter, we develop a lifecycle assessment for tropical forest management based on empiric data collected for Costa Rica (Figure 1.1). We used the same system boundaries as in Chapter 2, but now included all GHGs from the combustion or decomposition of biogenic carbon and the emissions from fossil fuels, from wood extraction to the end of life. As in Chapter 2, we trace all products from an average hectare of natural forests in Costa Rica, to provide a weighted account at a per hectare level. However, in this case we present results for all products individually treating them as functional units. We assess the net balance at a per hectare level, and test these results for a 100-year temporal boundary and a 20-year time horizon.

Chapter 5

In the final chapter, I integrate the results from all chapters and discuss their implications. I first address the trade-offs from using local empiric data on the uncertainty of the system. I discuss the results from all chapters considering the local context and how these provide a better understanding of the potential contribution to climate mitigation from managing forests. Finally, I discuss whether we should manage productive tropical forests for climate mitigation considering lessons from a lifecycle approach and the main findings from this thesis.

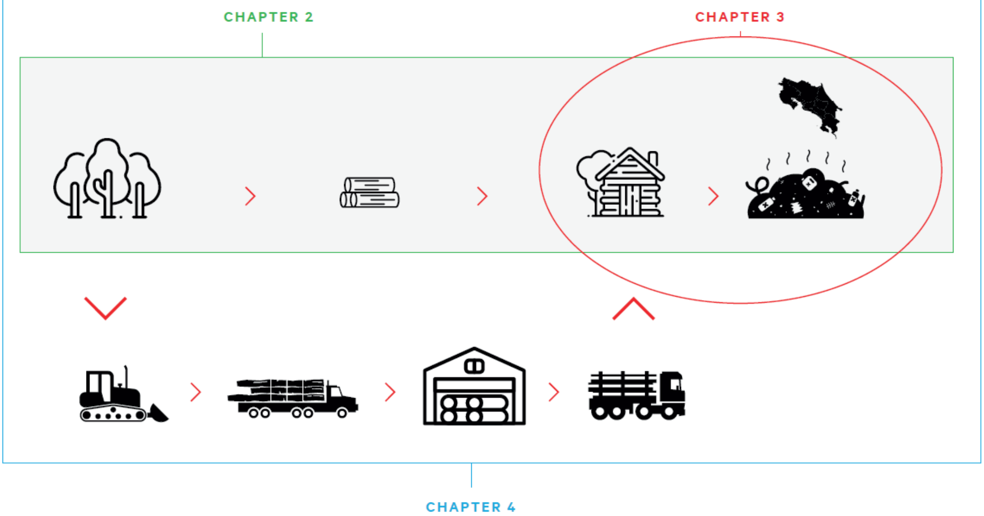

Figure 1.1. Conceptual framework: Chapter 2 (green) - biogenic lifecycle carbon balance; Chapter 3 (red) – Carbon stock of HWP in Costa Rica; Chapter 4 (blue) - lifecycle assessment for tropical forest management.

Chapter 2

The lifecycle carbon balance of selective logging in tropical forests of Costa Rica

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Abstract

The effect of logging on atmospheric carbon concentrations remains highly contested, especially in the tropics where it is associated to forest degradation. To contribute to this discussion, we estimated the carbon balance from logging natural tropical forests in Costa Rica through a lifecycle accounting approach. Our system included all major lifecycle processes at a regional level during one rotation period (15 years). We used mass flow analysis to trace biogenic carbon based on data from all logging operations in the Costa Rican NW region (107 management plants), a sample of industries transforming wood into final products (20 sawmills) and national reports. We estimated a surplus of -3.06 Mg C ha^{-1} 15 yr^{-1} stored within the system. When accounting for uncertainty and variability in a Monte Carlo analysis, the average balance shifted to -2.19 Mg C ha^{-1} 15 yr^{-1} with a 95% CI of -5.26 to 1.86. This confidence interval reveals probabilities of a net increase in atmospheric carbon due to harvesting although these are smaller than those from a system that acts as a reservoir. Our results provide evidence for the carbon neutrality of biomaterials obtained from natural forests. We found that anthropogenic reservoirs play a determinant role in delaying carbon emissions and that these may explain differences with previous carbon balance studies on tropical forest management. Therefore, the climate mitigation potential of forest-derived products is not exclusive to forest management, but measures should be considered throughout the processes of wood transformation, use and disposal.

Introduction

Sustainable forest management for wood production is a potential climate mitigation option as wood products may accumulate in anthropogenic reservoirs for relatively long periods, avoiding carbon locked in wood from reaching the atmosphere (Brandão et al., 2013; Intergovernmental Panel on Climate Change, 2014). Anthropogenic reservoirs consist of products in use, where harvested wood products (HWP) can last for 5-100 years (Brunet-Navarro et al., 2017) and solid waste disposal sites (SWDS) where some HWPs can last even longer after they are retired from service (De la Cruz et al., 2013; F. Ximenes et al., 2015). Globally, these reservoirs are known to be growing (Butarbutar et al., 2016; S Hashimoto, Nose, Obara, & Moriguchi, 2012; Seiji Hashimoto, 2008) and this storage component is an increasingly important part of the land-use related carbon balance.

The climate mitigation potential of forest management depends on storage in anthropogenic reservoirs, but also importantly on the losses associated with timber processing; from harvesting to final application. In the tropics, 1.1 Gt CO_2 emissions have been estimated due to logging and it is therefore commonly described as the main cause of forest degradation (R. A. Houghton, Byers, & Nassikas, 2015; Pearson et al., 2017). For example, to extract one cubic meter of timber, an associated 1 to 3 Mg C may be lost from the forest due to log extraction, logging damage and the infrastructure needed for forest operations (Pearson et al., 2014). From the extracted timber only a fraction will be transformed into products, out of which an even smaller fraction will remain stored in long-term anthropogenic reservoirs, and the allocation of harvested timber to different product classes determines overall residence time of carbon in wood products.

A third component determining the magnitude of the climate mitigation potential of forest management is the recovery of carbon emissions from the forest. However, these emissions are temporary (as long as no land use change occurs) and will recover through forest regrowth (R. A. Houghton, 2012; R. A. Houghton et al., 2015; Richard A. Houghton, 2013; Intergovernmental Panel on Climate Change, 2014). Recent evidence shows that growth rates after logging tend to increase (Piponiot et al., 2018), although uncertainties on the rates of carbon sequestration due to spatial variation still remain (Baccini et al., 2012). In the case of tropical logged forests, biomass losses have been shown to partly explain the recovery time (Rutishauser et al., 2015) and have been used to estimate the carbon balance.

Estimating the carbon balance is a step closer to the potential climate impact of logging given that it considers all processes leading to carbon emissions, storage and sequestration; which can take place at different spatial and temporal scales (Newell & Vos, 2012). To integrate these processes, a lifecycle carbon accounting approach has been recommended (Geng, Zhang, et al., 2017; Hauschild, Rosenbaum, & Olsen, 2018; Klein et al., 2015; Knauf, 2015; Lippke et al., 2011). This approach accounts for changes in biogenic carbon (BioC; i.e. carbon stored in biomass) from the forest and up until the end of life (EoL) of wood products. That is, it includes the decomposition of biomass in the forest due to logging damage; carbon storage in anthropogenic reservoirs over time, and; depending on the type of EoL, the moment when carbon is released back to the atmosphere via combustion or decomposition. Also, this approach allows the inclusion of carbon sequestered by the forest via regrowth, to produce a complete biogenic carbon lifecycle balance (BioC-LC).

In the tropics, efforts have been made to integrate the lifecycle of timber harvesting and wood use to determine an overall BioC-LC, but attempts so far have been restricted by a lack of empirical data (Murphy, 2004; Numazawa et al., 2017; Piponiot et al., 2016). This situation is not unique for tropical forests since only few lifecycle studies include a complete BioC-LC (Aleinikovas et al., 2018; Cardellini et al., 2018; De Rosa, Schmidt, Brandão, & Pizzol, 2017; Downie, Lau, Cowie, & Munroe, 2014; Helin et al., 2013; Liu et al., 2017; Newell & Vos, 2012). It is often assumed that the overall integration of these processes results in a carbon neutral outcome, but this assumption has raised a considerable debate (Cardellini et al., 2018; Helin et al., 2013; Johnson, 2009).

For tropical logging, it has been shown that including a detailed biomass lifecycle can result in delayed carbon emissions (Piponiot et al., 2016). Although in that study tropical forests producing timber are reported mainly as sources of carbon, zero net emissions (i.e. carbon neutrality) are included within the 95% confidence interval. Therefore, given the conservative assumptions on product use (e.g. HWP being one third of harvest with the rest assumed to be sawdust) and the exclusion of the EoL phase of wood products, it is possible that lifecycle processes leading to carbon storage may increase the chance of a carbon neutral outcome.

Here we present the carbon balance of selectively logged tropical forest in Costa Rica, in which all processes until the end of life of wood products are integrated using a lifecycle approach. We do this based on a mass flow analysis (Geng, Yang, Chen, & Hong, 2017; Jasinevičius, Lindner, Cienciala, & Tykkyläinen, 2018) using foreground data collected for natural forest harvest operations, sawmilling industry and wood product use in Costa Rica. To account for uncertainty and variation, we perform a Monte Carlo Analysis (Clavreul et al., 2012; European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010a; Heijungs & Huijbregts, 2004; Heijungs & Lenzen, 2014; Huijbregts, 1998; Lo, Ma, & Lo, 2005). We address whether, under the circumstances found in our case study, a disturbance to the natural carbon cycle in tropical forests due to human interventions lead to accumulation or loss of carbon, or whether the system can be considered neutral in terms of carbon cycling. Such questions are especially useful in tropical countries, as answers to these bear relevance about the potential of forest management in national REDD+ (Reduced Emissions from Deforestation and Degradation) or other climate mitigation strategies.

Methods

Approach and system boundaries

This study focuses on the exploitation of natural wet and moist forests in the Northern Caribbean region of Costa Rica. This region is responsible for 83% of timber harvest from natural forests in the country (MINAE, 2011, 2012, 2013). Changes in biogenic carbon pools are quantified along all stages within the product system, i.e. harvesting operations, sawmilling, transformation into end products, product and co-product use, and end of life (EoL) management. Products are defined as the intended output of the milling process (e.g. sawnwood), while co-products are by-products (i.e. slabs, bark, edges, off-cuts, sawdust and shavings) with a market value (e.g. as fuelwood or pellets).

The temporal boundary used here is one rotation period which in tropical forests can vary from 15 to 60 years (Rutishauser et al., 2015; Sasaki et al., 2012). In Costa Rica, the rotation period is determined based on information on forest recovery through a pre-harvest inventory. There is variation among production forests but it has not been quantified. For this reason, we fixed this period to the minimum allowable length of 15 years according to national legislation

(MINAE, 2002; MINAET, 2009). As a result, our estimate of recovery of forest carbon stocks are probably conservative, given that in practice cutting cycles are likely considerably longer.

Based on these boundaries, we approximate carbon emissions due to biomass decomposition and combustion when they occur, together with forest regrowth. This is done for an average hectare of natural tropical forest in Costa Rica, where timber is extracted for wood products and co-products. Therefore, the sum of all carbon gains and losses are allocated to one hectare of natural tropical forest.

Soil carbon was excluded based on probable limited changes from this stock due to small impacted area and a short duration of the impact (Pearson et al., 2014), together with large uncertainties around its estimation (Baccini et al., 2012; De Rosa et al., 2017; G. M. J. Mohren, Hasenauer, Köhl, & Nabuurs, 2012). In terms of processes, recycling was excluded given lack of data, with only 1-2% reported by the furniture industry (Solera, 2014). In both cases, evidence that a continuous cover system with a proportionally low impacted area (Pearson et al., 2014) and without residue collection can lead to soil carbon increases (Helin et al., 2013), and that the effect of recycling results in the prolongation of the life of products, seem not to challenge the conservativeness of this BioC-LC. Finally, because harvesting does not cause deforestation in Costa Rica (Arroyo-Mora et al., 2014), land use change was also excluded and the ‘no use’ scenario becomes the reference (Helin et al., 2013).

Foreground data was collected from the revision of all management plans within the study region during 2010-2016 and field questionnaires for all stages except end of life management. For EoL, background data was taken from national reports (SI-Table 2.1).

Carbon stocks in forest biomass

Plots from the National Forest Inventory within our study region (Programa REDD/CCAD-GIZ -SINAC, 2015) were used together with a site-specific allometric equation (Fonseca, Alice, Rojas, Villalobos, & Porras, 2016) (SI-Table 2.1) to determine average carbon per hectare. This equation estimates all ecosystem biomass, i.e. above and belowground tree biomass, herbaceous vegetation and necromass.

Wood harvest and logging damage

To account for wood harvest and carbon emissions from logging operations, we reviewed the management plans submitted to the regional offices of the Ministry of Environment and Energy (MINAE) in Costa Rica during the period 2010 - 2016. A total of 107 forest management plans and their corresponding audit reports were reviewed.

Reported extracted volumes over bark from felled standing trees (> 60 cm minimum harvestable diameter) and deadwood, were converted to biomass using wood densities (g cm^{-3})(Chave et al., 2009). Species were grouped according to their traditional classification as hardwoods, semi-hardwoods and softwoods, with some remaining as unclassified (Zúñiga-Méndez, 2016). This was done to accommodate species for which wood densities were not available. Average wood density was determined per group and weighted by the group's contribution to total volume. Further conversion into harvested carbon (H) was calculated using a site specific carbon fraction for tree stems (Fonseca et al., 2016).

Logging damage was estimated based on the area impacted as reported in the reviewed management plans (SI-Table 2.1) and the carbon stock in the forest biomass. We assumed that residual large trees (>40 cm DBH) are not damaged during the construction of infrastructure or during felling operations. The ecosystem carbon excluding these trees was 55.83 Mg C ha^{-1} or 55% of total ecosystem carbon and within the range of 28 – 56.2% reported in the literature (Sasaki et al., 2012). In the case of gaps from felling, we also included the additional carbon from the tree compartments from extracted logs which remain in the forest as slash (i.e. leaves, branches and roots). This was done using the biomass expansion factors and root to shoot ratios (Fonseca et al., 2016). In case of harvested deadwood, no carbon emissions due to gap formation were calculated since this extraction does not involve felling.

Decomposition was included assuming exponential decay with a 0.1 yr^{-1} (R. A. Houghton et al., 2000) decay constant. We report carbon emissions (LD_{15}) and stocks in the system after the 15-year period.

$$\boldsymbol{LD_{15}} = (Gp + LgDck + PrmRd + ScRd + SkTr\ ) \times (1 - e^{-k_1 t}) \tag{1}$$

Where:

$\boldsymbol{LD_{15}}$ = Carbon in logging damage decomposed by year 15 (Mg C ha^{-1})

Gp = Initial amount of carbon from felling gaps (Mg C ha^{-1})

$LgDck$ = Initial amount of carbon from logging decks (Mg C ha^{-1})

$PrmRd$ = Initial amount of carbon from primary roads (Mg C ha^{-1})

$ScRd$ = Initial amount of carbon from secondary roads (Mg C ha^{-1})

$SkTr$ = Initial amount of carbon from skid trails (Mg C ha^{-1})

k_1 = Decay rate of deadwood; 0.1 yr^{-1}

t = years; 15 years

Forest regrowth

As we lacked observations on forest regrowth in our study region, we used results from a meta-analysis of 10 logged Neotropical forests (Rutishauser et al., 2015) to estimate the time it takes for forest carbon to recover to pre-logging carbon stock (*RT*). RT is a function of carbon lost, i.e., the sum of logging damage (*LD*$_0$) and extracted wood (*H*). RT was used to determine the growth rate until the initial biomass was reached.

$$RT = \left(\frac{(H + LD_0) \times 100}{CSFB}\right)^{\emptyset} \tag{2}$$

Where:

RT = Recovery time (years)

H = Harvest; sum of carbon extracted, both standing trees and deadwood ($H_{St} + H_{Dw}$) (Mg C ha^{-1})

LD_0 = Carbon from logging damage $(Gp + LgDck + PrmRd + ScRd + SkTr\ )$ (Mg C ha^{-1})

$CSFB$ = Carbon stock in forest biomass (Mg C ha^{-1})

$\emptyset = 1.106 \pm 0.022$

$$\boldsymbol{FR_{15}} = min\left(H + LD_0; t \times \frac{(H + LD_0)}{RT}\right) \tag{3}$$

Where:

$\boldsymbol{FR_{15}}$ = Carbon from forest regrowth by year 15 (Piponiot et al., 2016)

t = rotation period

Sawmill biomass & carbon flow

Based on the available forest management plans we identified a total of 42 sawmills and selected those processing timber from natural forests. After an initial contact, we selected 20 sawmills to include in our survey. Most selected sawmills were located within the study region; four were located at ≈100 km distance.

We developed a questionnaire for sawmilling and gathered data on all biomass inputs and outputs. The reported types and amounts of wood products, co-products and residues were used to develop the carbon flow within the sawmill.

Products were grouped according to their end use and classified as short, mid or long-term to assign half-lives. For example, all sawnwood used as formwork was considered a "short-term" product, while the remaining sawnwood that does require further transformation at the milling stage (i.e. planing and moulding of wood flooring, boxboard, mouldings, scantlings, beams, etc.), were grouped into one single category, i.e. "construction" and classified as "long term" products.

All forms of by-products (e.g. slabs, edges, sawdust, etc.) were traced independently but grouped into co-products depending on their end use. For example, slabs, sawdust and shavings are all used for pellets, while edges and off-cuts are used in the furniture industry. Although being a co-product, this last end use was further classified as a "mid-term" product due to its half-life.

Transformation into end use products

Long-term and mid-term products (i.e. wood used in construction and furniture) require an additional transformation outside the mill before becoming products in use. For these, we used a questionnaire for the secondary transformation industry to determine the fraction of wood that becomes residues and that is sent to EoL (i.e. TL_f in equations 8 and 9 below). Other categories have no further transformation (e.g. formwork), or it makes no difference given that complete carbon loss is assumed to occur on the year of harvest (e.g. pellets).

End use phase of the lifecycle

No carbon emissions from the main wood products take place at this stage but there is a flow of carbon from products in use to EoL. To quantify this flow, we used first order decay and half-lives that varied according to products. As described above, products were classified as short, mid and long-term, and assigned half-lives of 2, 25 and 35 years (i.e. k_4, k_5 and k_6 in equations 8 and 9 below) (IPCC, 2014).

For co-products, carbon can be lost during use. In the case of fuelwood and pellets used for bioenergy, we assumed emissions take place immediately on year 0. For the decomposition of sawdust and shavings used as flooring for stables or as compost in nurseries, we assumed the same rate as forest biomass (i.e. k_1) given the conditions under which these will decompose. This may result in a slight underestimation of carbon emissions due to the smaller particle size of this co-product.

$$\boldsymbol{Plt_0} = H \times (SwdPlt_f + ShvPlt_f + SBPlt_f) \tag{4}$$

Where:
$\boldsymbol{Plt_0}$ = Initial amount of carbon in wood used for pellets (Mg C ha^{-1})
$SwdPlt_f$ = fraction from harvest that becomes sawdust and is used as pellets
$ShvPlt_f$ = fraction from harvest that becomes shavings and is used as pellets
$SBPlt_f$ = fraction from harvest that becomes slabs and bark and is used as pellets

$$\boldsymbol{Fw_0} = H \times (SwdFw_f + ShvFw_f + SBFw_f) \tag{5}$$

Where:
$\boldsymbol{Fw_0}$ = Initial amount of carbon from wood used as fuelwood (Mg C ha^{-1})
$SwdFw_f$ = fraction from harvest that that becomes sawdust and is used as fuelwood
$ShvFw_f$ = fraction from harvest that becomes shavings and is used as fuelwood
$SBFw_f$ = fraction from harvest that becomes slabs and bark and is used as fuelwood

$$\boldsymbol{SSN_{15}} = H \times \left(\left(SwdSSN_f + ShvSSN_f \right) \times (1 - e^{-k_1 t}) \right) \tag{6}$$

Where:

$\boldsymbol{SSN_{15}}$ = Carbon in wood used for stables, stalls & nurseries decomposing by year 15 (Mg C ha^{-1})

$SwdSSN_f$ = fraction from harvest that becomes sawdust and is used in stables, stalls or nurseries

$ShvSSN_f$ = fraction from harvest that becomes shavings and is used in stables, stalls or nurseries

End of life (EoL)

We approximated the EoL of products and residues by determining the amount that decompose in solid waste disposal sites (SWDS) and those that are open burned. This distribution was taken from the National GHG Inventory (Chacón, Jiménez, Montenegro, Sassa, & Blanco, 2012). Once the fraction of products reached SWDS, we used the default value of 0.5 as the decomposable degradable organic carbon fraction (DOC_f) and half-lives differentiated by wood type; i.e. 20 years for wood products, slabs and bark, and 10 years for sawdust and shavings at the mill dump (Pipatti et al., 2006).

Carbon emissions from the mill dump (Equation 7) were estimated separately for groups of wood residues due to differing half-lives. In Equation 8, we first estimate the flow of each wood product category (*STP*, *MTP* and *LTP*) into SWDS using half-lives described in the previous section (i.e. wood products retired from service). Then, once in a SWDS, we determine the outflow/emissions using exponential decay and a single half-life for all products (k_3). Transformation losses were subtracted from products and accounted separately because these flow directly to SWDS on year 0. Finally, emissions were estimated only for the fraction that is effectively lost (i.e. DOC_f).

For the fraction that is open burned (Equation 9), we applied the same logic were transformation losses are subtracted from products, and the outflow from products in use is estimated based on each products' half-life. The main difference is that all carbon was assumed to be lost as soon as residues were disposed of or products were retired from service.

$$\begin{aligned}\boldsymbol{SmR_{15}} = \Big(H \times \left(SwdR_f + ShvR_f\right) \times (1 - e^{-k_2 t})\Big) \\ + \Big(H \times \Big(SBR_f \times (1 - e^{-k_3 t})\Big)\Big) \times DOC_f\end{aligned} \tag{7}$$

Where:

$\boldsymbol{SmR_{15}}$ = Carbon in sawmill residues decomposing by year 15 (Mg C ha^{-1})

$SwdR_f$ = fraction from harvest that becomes sawdust residues during milling

$ShvR_f$ = fraction from harvest that becomes shavings residues during milling

SBR_f = fraction from harvest that becomes slabs and bark residues during milling

DOC_f = fraction of degradable organic carbon that can decompose; 0.5

$k_2 = \ln(2)/10$

$k_3 = \ln(2)/20$

$$\begin{aligned}\boldsymbol{SWDS_{15}} = SWDS_f \\ \times \Bigg(\sum_{i=0}^{14} \Big(\Big(H \times \big(\big(STP_f \times e^{-k_4 i} \big) \times (1 - e^{-k_4}) \big) \\ + \big(\big(MTP_f \times (1 - TL_f) \times e^{-k_5 i} \big) \times (1 - e^{-k_5}) \big) \\ + \big(\big(LTP_f \times (1 - TL_f) \times e^{-k_6 i} \big) \times (1 - e^{-k_6}) \big) \Big) \times \big(1 - e^{-k_3 (t-i)} \big) \Big) \\ + \Big(H \times \Big(\big(MTP_f \times TL_f \big) + \big(LTP_f \times TL_f \big) \Big) \times (1 - e^{-k_3 t}) \Big) \Bigg) \times DOC_f\end{aligned} \tag{8}$$

Where:

$\boldsymbol{SWDS_{15}}$ =Carbon in wood decomposing at SWDS during the 15-year period (Mg C ha^{-1})

$SWDS_f$ =fraction of wood decomposing at SWDS

STP_f=fraction of wood harvest that becomes short-term products

MTP_f=fraction of wood harvest that becomes mid-term products

LTP_f=fraction of wood harvest that becomes long-term products

TL_f =fraction of wood that becomes residues and is sent to EoL during the final transformation of mid and long-term products

k_4 =ln (2)/2 (STP)

k_5 =ln (2)/25 (MTP)

k_6 =ln (2)/35 (LTP)

i =years 0 -14

$$\begin{aligned}\boldsymbol{OB_{15}} = & \left(1 - SWDS_f\right) \\ & \times \Bigg(\left(\left(H \times STP_f\right) \times (1 - e^{-k_4 t}) \right) \\ & + \Bigg(\left(\left(\left(H \times MTP_f\right) \times \left(1 - TL_f\right) \right) \times (1 - e^{-k_5 t}) \right) \\ & + \left(H \times MTP_f \times TL_f\right) \Bigg) \\ & + \Bigg(\left(\left(\left(H \times LTP_f\right) \times \left(1 - TL_f\right) \right) \times (1 - e^{-k_6 t}) \right) \\ & + \left(H \times LTP_f \times TL_f\right) \Bigg) \Bigg) \end{aligned} \tag{9}$$

Where:

$\boldsymbol{OB_{15}}$ =Carbon in wood products open burned during the 15-year period (Mg C ha^{-1})

System balance, uncertainty and sensitivity analyses

All previous equations were combined to obtain the system's net carbon balance (summarized in Equation 10 for illustration purposes only). Although we did not account for carbon storage in products or SWDS directly, the difference between the inflow and outflow from these reservoirs indicates storage.

$$\boldsymbol{System\ Balance} = (LD_{15} + SmR_{15} + Plt_0 + Fw_0 + SSN_{15} + EoL_{15}) - FR_{15} \tag{10}$$

To evaluate the effect of parameter variability and the uncertainty of the carbon balance, we determined the probability density functions for all 32 parameters used in the analysis (SI-Table 2.2). We randomly sampled from their distributions, calculated the carbon balance and repeated this procedure through Monte Carlo simulations 10,000 times. We then calculated the mean and confidence interval of the carbon balance.

To assess the sensitivity of carbon balance to variation in parameters, we also performed a sensitivity analysis. This differs from a Monte Carlo analysis in that it applies equal changes (+/- 10%) to each parameter separately and then evaluates the effect on the carbon balance.

Results

Harvest, logging damage and forest regrowth

A total of 7756 ha were harvested in the region during 2010-2016, with an average area per forest management plan of 80 ha (n= 97; σ=67), ranging from 3.5 to 325 ha. Average standing tree harvest was 11.08 m^3 ha^{-1} (n= 65; σ=6.23) and deadwood was 1.95 m^3 ha^{-1} (n= 54; σ=7.02), resulting in a total harvest of 13.03 m^3 ha^{-1} (over bark).

Average wood density of hardwoods, semi-hardwoods, softwoods and "unclassified" species were 0.66, 0.45, 0.33 and 0.52 g cm^{-3} respectively. The weighted average wood density (0.4965 g cm^{-3}) was used together with a carbon content of 0.447 (Fonseca et al., 2016) to determine carbon in the harvest. Total harvested carbon was 2.89 Mg C ha^{-1}, distributed in 2.46 Mg C ha^{-1} (n= 65; σ=1.38) from felled trees and 0.43 Mg C ha^{-1} (n= 54; σ=1.56) for deadwood. Carbon stock at the ecosystem level was estimated using NFI average basal area (25.8 m^2 ha^{-1}; n= 9; σ=8.79) and resulted in 101.72 Mg C ha^{-1} (n= 9; σ=32.9). Carbon in the diametric classes damaged during harvesting (i.e. DBH <40 cm) represented 55.83 Mg C ha^{-1} (n= 9; σ=15.81).

According to the 31 forest management plans that reported area impacted by logging, gaps from tree felling represented the largest amount with 3.62 ha or 5.3% of the total forest area. Secondary roads represent 1.01 ha (1.8%) and skid trails 0.8 ha (1.4%). Primary roads and logging decks inside the forest caused only a marginal impact with 0.25 (0.2%) and 0.11 ha (0.2%), respectively. Total carbon impacted during logging, excluding harvest, was 5.26 Mg C ha^{-1}. Of this amount, 4.09 Mg C ha^{-1} was lost due to decomposition during the 15-year period, while 1.17 Mg C ha^{-1} (22.3%) remains in the forest as necromass (Figure 2.1, SI-Table 2.3).

We estimated 9.99 years for the full recovery of carbon stocks in logged forests. During this period, carbon stocks increased at a rate of 0.82 Mg C ha^{-1} yr^{-1}. This estimate was obtained using total harvested volume plus all carbon from logging damage, i.e. 8% of total initial ecosystem carbon or 8.15 Mg C ha^{-1}. In order not to overestimate recovery time and because the original model (Rutishauser et al., 2015) assumes committed emissions, we used logging damage on year 0 instead of our 15-year estimate considering decomposition.

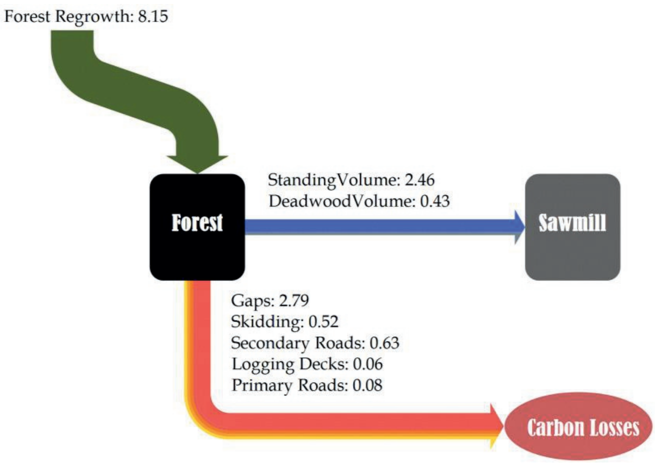

Figure 2.1. Forest carbon flows during one logging cycle of 15 years for an average hectare of exploited tropical forest in Costa Rica (Mg C ha^{-1} $15yr^{-1}$; box sizes are not indicative of the size of the stock but black boxes represent carbon storage).

Sawmill and wood transformation

The main products from milling are boards (27%) and laths (20%) used as formwork, which is a low quality and short-term product used to mould concrete in construction. The category "construction" or long-term products (i.e. laths for framing, beams, scantlings, mouldings, floors, boardbox, etc.) represent an additional 17%, and together constitute the 64% milling efficiency (SI-Table 2.4; Figure 2.2).

The remaining 36% is distributed among slabs and bark (13%), edges and off-cuts (13%), sawdust (8%) and shavings (2%; SI-Table 2.4), of which some became co-products. For example, sawmills reported selling 100% of edges and off-cuts to the furniture industry, 6.2% of slabs and bark is used to produce pellets for bioenergy, 4.6% for fuelwood, and only 2.6% of slabs and bark end at the mill dump (SI-Table 2.5). In the case of sawdust and shavings, the most important use is flooring for stables, stalls or as organic matter used in nurseries (7% and 2% respectively). Small amounts of sawdust and shavings were also used for pellets, fuelwood or will be discarded (SI-Table 2.5).

Products and co-products flow to a next transformation stage or directly to the use phase of the lifecycle (Figure 2.2). From the 2.89 Mg C from harvest, only 0.09 Mg C end at the mill dump.

Given that slabs and bark represent over 80% of wood going to this dump and have a half-life of 20 years, only 0.02 Mg C (0.7% of harvest) are lost during the 15-year period, while 0.07 Mg C (3% of harvest) remain stored at the dump.

The transformation of products into end uses was limited to those used in construction or edges and off-cuts for the furniture industry. Out of the 0.49 Mg C of long-term and 0.37 Mg C of mid-term products that result from the milling process, 9.8% (SD=7.29) or 0.05 and 0.04 Mg C respectively, are sent to EoL management (Figure 2.2).

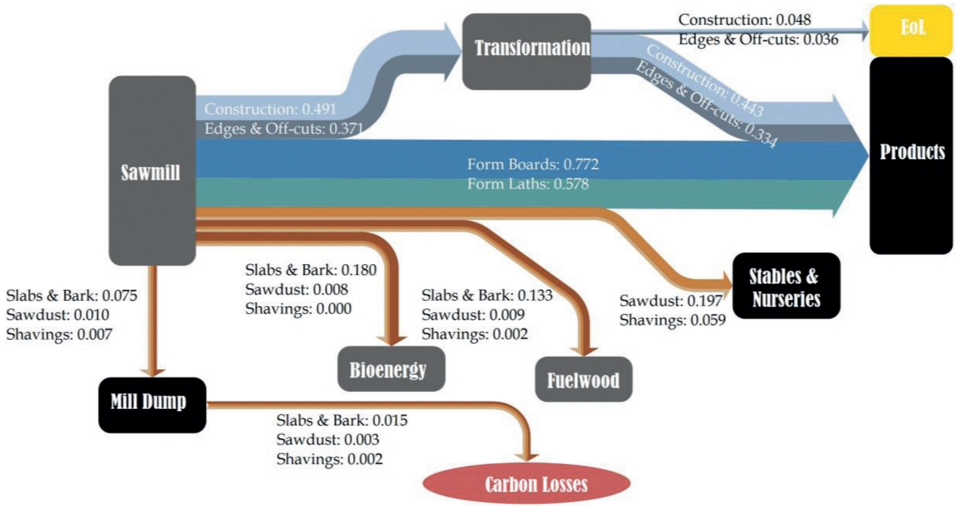

Figure 2.2. Carbon flows during sawmill processing and final product transformation of wood sourced from one hectare of natural forest in Costa Rica, during one logging cycle of 15 years (Mg C ha^{-1} $15yr^{-1}$; box sizes are not indicative of the size of the stock but black boxes represent carbon storage).

Product use & end of life

During the 15-year period and given the 2-year half-life, 99.4% of all formwork has been transferred to EoL (Figure 2.3). However, 67% of construction wood and 60% of edges and off-cuts used in the furniture industry remain stored in the product pool. Carbon emissions during this phase corresponded to decomposition of sawdust and shavings used in stables and nurseries or the combustion of firewood and pellets, both assumed to have occurred during the year of harvest.

At SWDS, carbon emissions were determined by the fraction of degradable organic carbon that is effectively lost due to biomass decomposition (DOC_f) and the 20-year half-life assumed for all types of wood. As a result, from the 1.57 Mg C transferred to SWDS from products in use and the 0.09 Mg C from the final transformation shown previously, only 0.45 Mg C were lost during the 15-year period while 1.22 Mg C remained stored (Figure 2.3).

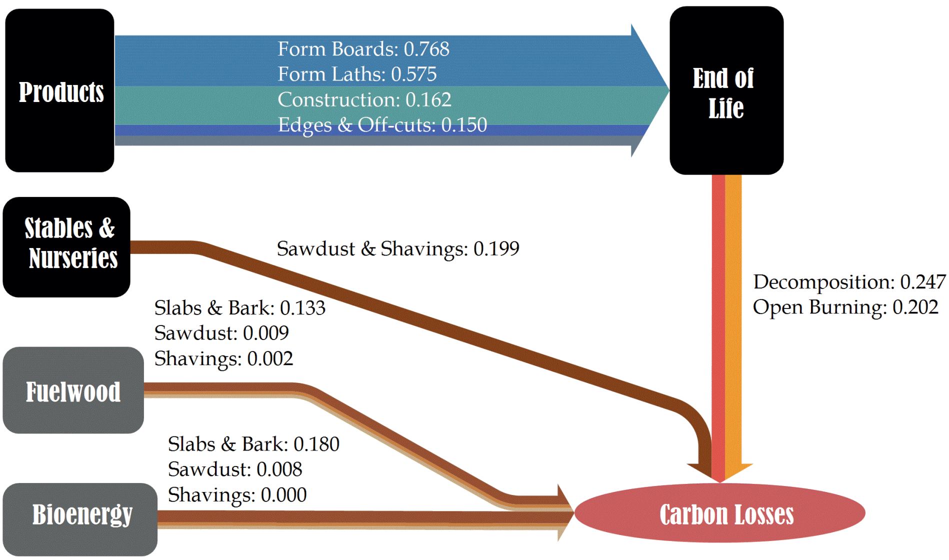

Figure 2.3. Carbon flows incurred by retirement (to EoL) and loss of wood products obtained from one hectare of exploited natural forest in Costa Rica, during one logging cycle of 15 years (Mg C ha^{-1} 15yr^{-1}; box sizes are not indicative of the size of the stock but black boxes represent carbon storage).

System balance

Lifecycle carbon emissions from the management of natural tropical forests for wood production in Costa Rica were 5.09 Mg C ha^{-1} 15 yr^{-1} and were dominated by the damage from harvesting operations (Figure 2.4). Logging damage was responsible for 80% of all carbon lost, followed by SWDS (9%), pellets (4%), stables, stalls and nurseries (4%), and fuelwood (3%). However, an important part of the ecosystem carbon (i.e. 3.08 Mg C ha^{-1}) was transferred across pools and remained stored along the system after the 15-year period.

Anthropogenic reservoirs hold 58% of carbon, especially SWDS (40%). The remaining carbon can still be found at the forest (38%), where it was transferred from living biomass to necromass. These reservoirs delay carbon emissions and together with forest regrowth determined the balance. As a result, the difference between carbon sequestration via regrowth (i.e. -8.15) and lifecycle carbon emissions was -3.06 Mg C ha^{-1} 15 yr^{-1}.

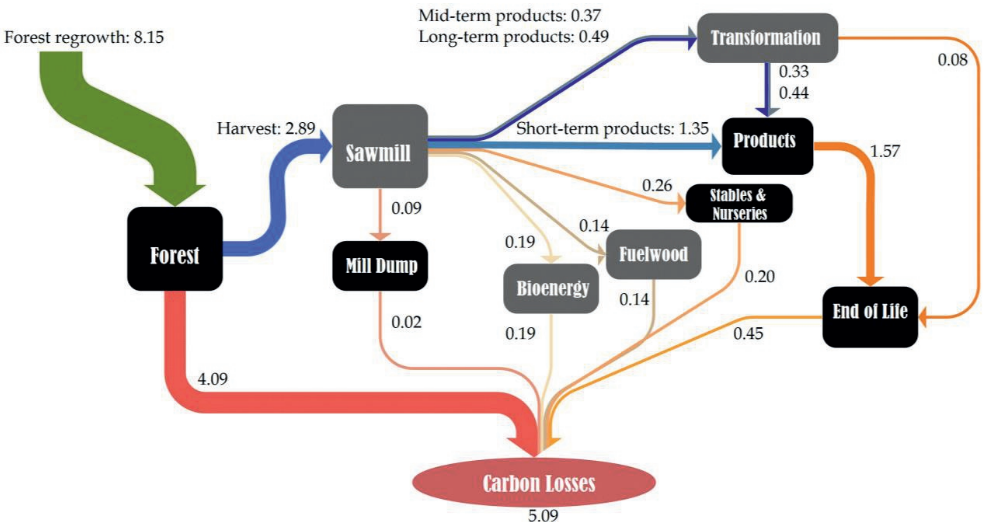

Figure 2.4. Carbon flow of wood products from one hectare of exploited natural forest in Costa Rica during a 15-year rotation period (Mg C ha^{-1} 15yr^{-1}; box sizes are not indicative of the size of the stock but black boxes represent carbon storage).

Uncertainty and sensitivity analyses

The Monte Carlo simulations shifted the average carbon balance from -3.06 to -2.19 Mg C ha^{-1} 15 yr^{-1}. This shift is due to the asymmetry of the distributions of some parameters (e.g. standing and deadwood harvest). The 95% confidence intervals of the carbon balance when taking parameter uncertainty and variation into account ranged from -5.26 to 1.86 Mg C ha^{-1} 15 yr^{-1}. This confidence interval includes the value of 0, implying that parameter variation can lead to carbon emissions from natural forest management. Yet, the probability of finding negative values is considerably larger, approximately 80%.

Sensitivity analyses showed that carbon balance was most sensitive to rotation length (SI-Figure 2.1). All other things remaining equal, a longer rotation length resulted in higher carbon emissions due to its effect on retirement and decomposition rates. The decay rate of biomass at the forest (i.e. k_1) was the second most important parameter. This effect also shows the important role of carbon in necromass in the forest for the carbon balance after one logging cycle. The third most important parameter influencing the balance is the fraction of wood ending in SWDS ($SWDS_f$).

We found clear differences in the output of the sensitivity analysis when conducted for emissions and regrowth separately. For example, parameters such as harvest (*H*) and logging damage (gaps in particular) only affected regrowth despite *H* interacting with most parameters or logging damage being associated to carbon emissions. In both cases, an increase resulted in lower carbon emissions due to larger forest and anthropogenic reservoirs. In the case of the rotation period, it mainly affected emissions and had a marginal effect on regrowth under these circumstances.

None of the parameter changes in the sensitivity analysis resulted in a positive carbon balance. Yet, the results of the MC analysis showed that if multiple parameters are varied simultaneously, positive balance values can be obtained. Combined, this suggests that in scenarios with long logging cycles, high harvest intensity, high damage, high shares of short-term products and/or low retirement to landfill, carbon balance will likely be positive. The cases with a positive carbon balance in our Monte Carlo simulations represent such (combinations) of variables.

Discussion

The ecosphere meets the technosphere

This study presents a complete lifecycle carbon balance for wood harvesting in the tropics following recommendations from the LCA framework. We find that indeed there are large probabilities for a carbon neutral outcome and confirm that it is at the forest where the largest exchanges of carbon occur (Butarbutar et al., 2016; Newell & Vos, 2012). Therefore, ignoring this phase from the lifecycle of wood and biogenic carbon in general under the carbon neutral assumption is not justified (Geng, Yang, et al., 2017; Klein et al., 2015; Knauf, 2015; Lippke et al., 2011). Especially considering the probabilities for the system to become a source of carbon emissions (Keith et al., 2015; Piponiot et al., 2016).

On the other hand, assuming committed emissions (i.e. the immediate release of carbon when harvesting takes place) is known to overestimate losses (Iordan et al., 2018). In our study, this methodological difference largely explains why results for carbon emissions (4.06 Mg C ha^{-1} or 0.31 Mg C m^{-3}; SI-Table 2.3) are well below those reported (Pearson et al., 2014) (i.e. 6.8–50.7 Mg C ha^{-1} or 0.99–2.33 Mg C m^{-3}), although site specific circumstances also play a role. For a better interpretation of these differences, we discuss three possible local practices that help to explain our results.

First, harvest intensity is known to determine forest carbon emissions (Martin et al., 2015; Francis E. Putz et al., 2008). In our study, harvest intensity (13.03 m^3 or 2.89 t C ha^{-1} per logging cycle) is in the lower end of ranges reported in the literature (i.e. 10 to above 30 m^3 ha^{-1} and 1.5 - 8.5 t C ha^{-1} (Pearson et al., 2014; Rutishauser et al., 2015; Sasaki et al., 2016), while logging damage is comparable with that from even lower reported harvest intensities (i.e. 9 m^3 ha^{-1} and 6.7 Mg C ha^{-1}) (Pearson et al., 2014). This is partly explained because in Costa Rica ~ 20% of the harvest is collected deadwood which does not require felling, and felling is the largest source of carbon emissions in a continuous cover harvesting system (i.e. 38-51% (Pearson et al., 2014); and 80% from this study). As a result, during the logging cycle (even if this is a relatively short one), carbon emissions are fully recovered in our study system.

Second, because of the small size of forest patches in Costa Rica (3.5 - 325 ha), logging hardly requires infrastructure such as primary roads and logging decks inside the forest. Furthermore, national standards for forest management (MINAE, 2002) require measures to reduce road impact consistent with those recommended in the literature (Laurance, Goosem, & Laurance, 2009). Most importantly, roads are closed once harvesting activities have taken place and are left for the forest to recover. This reduces the damage while increasing the contribution of gaps from felling on the overall damage.

Finally, different from the 50 cm threshold used in similar studies (Numazawa et al., 2017; Pearson et al., 2014; Piponiot et al., 2016), we assumed instead that trees >40 cm DBH did not experience logging damage. This is based on the outcome of questionnaires to harvesting operators and foresters, who reported even lower tree sizes depending on the type of damage (i.e. between 10 - 30 cm). Consistent with our findings from these questionnaires, skidding together with cable winch as done in Costa Rica has been reported not to cause damage to trees >10 cm (Griscom, Ellis, & Putz, 2014).

Clarifying these differences or assumptions is also important given that we estimate regrowth based on total biomass lost (Rutishauser et al., 2015). Despite the relatively low logging damage, the sum of harvest and damage represents 9% of the initial ecosystem carbon and is within the expected range (3–15%) (Pearson et al., 2014). Furthermore, as a measure of the conservativeness of the recovery rate used, the resulting mean annual increment of forest carbon stocks (i.e. 0.82 Mg C ha^{-1} yr^{-1}) is close to the lower end of those found in the Amazon, Borneo & Nicaragua (0.66-1.5 Mg C ha^{-1} yr^{-1}) (Rutishauser et al., 2015; Sasaki et al., 2016).

Finally, the amount of biomass damaged due to harvesting and left at the forest to decompose also represents an important temporal carbon stock at the end of the analysed period. Regardless of efforts to correctly estimate damaged biomass, this stock depends on decay rates that are associated to large uncertainties (Pearson et al., 2014; Piponiot et al., 2016). In tropical forests, half-lives for biomass decay can vary from 1 to 69 years (Pearson et al., 2014) depending on the type of necromass. For this reason, other studies differentiate decay rates based on this (e.g. fine and large necromass) (De Rosa et al., 2017; Lun et al., 2012; Piponiot et al., 2016), but the largest uncertainties are mostly related to large necromass. To avoid overestimating this stock we used a conservative decay rate with a half-life of ~7 years, which is close to the average from the 0.6 – 14 yr reported range (Hérault et al., 2010).

The lifecycle of biogenic carbon in the technosphere

A mass flow analysis such as the one used to trace carbon along the lifecycle of wood has been recommended in the literature to consider the multifunctionality of wood and avoid misrepresenting its contribution in the overall balance (Geng, Yang, et al., 2017; Jasinevičius et al., 2018). Essential for this analysis was the use of foreground data to determine all milling outputs (products, co-products and residues) and estimate milling efficiency. By doing so, we were able to categorize wood products far beyond commonly used classifications (e.g. sawnwood, panels, pulp & paper) and had more flexibility to assign specific half-lives until the EoL.

The relatively high milling efficiency (i.e. 63.67%) is the result of the main products, i.e. boards and laths (46% of the total harvest), being used as formwork. This is a very low quality end product that allows the maximization of sawnwood use. Most reported milling efficiencies tend to be around 50% (Butarbutar et al., 2016; Ofoegbu et al., 2014; Ramasamy et al., 2015; Sasaki et al., 2016), but our result is still within a range of 40% to 70% reported in the literature for tropical countries (Ofoegbu et al., 2014; Sasaki et al., 2016).

Despite the effect that formwork has on the milling efficiency, it was also because of the short half-life from this wood product that the stock of carbon from HWPs was relatively small, in accordance with the strong effect of retirement rates of wood products on carbon storage (R Miner, 2010). In this work the carbon stock in products was largely determined by products with half-lives larger than a rotation period, and according to our sensitivity analysis there is no effect from prolonging this half-life. Since this is contrary to what has been repeatedly found in the literature (Brunet-Navarro et al., 2017), it is important to clarify that it is not small changes in half-life what affects the balance (e.g. +/- 10% used in the sensitivity) but a radical change in wood use (e.g. from formwork to mid or long-term products).

The most important anthropogenic reservoirs delaying carbon emissions were solid waste disposal sites (SWDS). Few studies include this reservoir given the limited data (Clavreul et al., 2012) which was also the case in our study. To partly compensate for a potential overestimation of carbon allocated to landfills, we chose the 0.5 default value for all types of residues (Pipatti et al., 2006) as the fraction of carbon that will be lost (i.e. DOC_f) through anaerobic biomass decomposition. This fraction can vary from 0 to 0.65 (Barlaz, 2006; De la Cruz et al., 2013; Micales & Skog, 1997; F. Ximenes et al., 2015), and under tropical conditions, an average value of 0.18 has been reported (F. Ximenes et al., 2015). Therefore, it is very likely that our choice overestimates carbon emissions.

Uncertainties and variation due to the choice of system boundaries

In lifecycle studies the choice of system boundaries can be highly subjective and have a large effect on results (Geng, Yang, et al., 2017; Klein et al., 2015; Knauf, 2015; Lippke et al., 2011; Newell & Vos, 2012). We made an attempt to avoid these decisions by including all mayor processes and by collecting foreground or local data as far as possible. By doing so, we reduced some of the model's uncertainty but increased parameter variability. Variability being the most common measure of uncertainty in LCA (Heijungs & Huijbregts, 2004). However, decisions regarding the reference unit to which the impact is attributed i.e. the functional unit, could not be avoided. In our study, this functional unit could be described as one hectare of natural tropical moist or wet forests in Costa Rica from which wood for various uses is harvested in 15-year rotation cycles.

Using a hectare as part of the functional unit is possible because we trace biogenic carbon exclusively and it is usually measured using this unit, especially at a regional level. It basically only allows the comparison with other forests (perhaps other land uses), and is therefore not frequently used in LCA where the common metric for wood products is cubic meter (Lippke et al., 2011). Its use can be further justified based on the goal of the analysis (De Rosa et al., 2017; Lippke et al., 2011; Perez-Garcia et al., 2005), and has the additional benefit of avoiding part of the multifunctionality problem, i.e. the allocation of impacts to products and co-products based on some allocation rule, mass being the most common (Sandin, Peters, & Svanström, 2016).

The most critical aspect of this unit is the choice of the analysed period. Long-term processes associated to forestry have always conflicted lifecycle studies (Ter-Mikaelian, Colombo, & Chen, 2015) because these rely on the assumption of a static system (Clavreul et al., 2012). We followed the "whole rotation approach" (Klein et al., 2015) given that it provides a time frame that allows the inclusion of regrowth, decomposition and carbon storage until a next cycle and because it is the most common in the LCA of forestry (Ter-Mikaelian et al., 2015). However, this approach is not entirely free from criticisms.

Given that it assumes that the forest has never been logged before, it is subject to what has been termed the "start-up effect" (Reid Miner, 2006). Wood will continue to be retired from the system, combusted or will decompose in the years following this rotation, and these emissions are not accounted for, thus leading to an overestimation of carbon storage during the first rotation. To correct for this effect, methods used in HWP inventories estimate "inherited emissions" (i.e. carbon emissions from previous harvests) (Pingoud et al., 2006) while other proposed methods recommend estimating the existing stocks after 100 years (Reid Miner, 2006). The main advantage from this approach is that reliable data on previous harvests can be difficult to obtain.

To present the most conservative estimate for the carbon balance, we recalculated the balance and its uncertainty for a 100-yr period. This resulted in a lower average -1.36 Mg C ha^{-1} 100 yr^{-1} where the 95% CI [-4.43, -0.14] is always negative. Due to a longer rotation period, logging damage is almost fully decomposed but there is still some carbon stored in products and mainly in SWDS. Most importantly, this longer period reduces chances of forests not recovering the initial carbon.

The main contribution from this work has been to show the importance of biogenic carbon and the effects of expanding the system boundaries to include all major processes in the lifecycle of tropical timber, something that was lacking in the literature (Murphy, 2004; Numazawa et al., 2017; Piponiot et al., 2016). Our results provide evidence for the hypothesis that managed forests could potentially contribute more to climate change mitigation than unmanaged forests (Lundmark, Bergh, Nordin, Fahlvik, & Poudel, 2016), although this remains highly controversial and opposing evidence is also available. Converting managed forests to protected areas has been shown to lead to higher carbon accumulation (Keith et al., 2015), especially when the reference scenario involves high harvest intensities and logging damage. In any case, under circumstances were forests are being harvested, it is a combination of short logging cycles, low harvesting intensities and high mass allocation into long-term products what has the greatest probabilities of avoiding some carbon emissions (Liu et al., 2017).

Conclusions

According to our analysis, selective logging in a 15-year cycle with subsequent timber use, may delay biogenic carbon emissions due to the storage of carbon in forests and anthropogenic reservoirs; allowing the forest to recover before the next cycle of use. However, forest management may also act as a disturbance leading to an acceleration of carbon emissions, e.g. through higher harvesting intensities with high logging damage, leading to insufficient recovery time until the next logging event, or by allocation of wood to short-term uses. When considering carbon storage, low impact logging and long-term product use are crucial.

Our results imply that forest management and subsequent use of wood products may indeed contribute to total carbon storage, also when considering harvest and wood processing losses.

Decisive factors in that case are low-intensity and low-impact selective logging, efficient wood processing, and allocation of wood to product categories that have substantially long life-spans. In addition, end-of-life is important, and final allocation of used wood products to landfill may comprise an important storage component, although the final allocation to landfill may be unwanted for other reasons, and should be reconsidered for re-use or use for bioenergy in which fossil fuels are replaced. This allows sustainable forest management combined with efficient product use to contribute to carbon storage, while a continued resource use adds to valuation of forested land, and thereby supports conservation of the forest resource.

Acknowledgements

We would like to thank the regional offices of the Costa Rican Ministry of Environment and Energy (MINAE) in San Carlos, Pital, Tortuguero and Puerto Viejo; the Fundación para el Desarrollo de la Cordillera Volcánica Central (FUNDECOR) and the Comisión de Desarrollo Forestal de San Carlos (CODEFORSA) for their support during data collection.

Supporting information

Table SI-2.1. Sources of information.

	Source	Sample	
Carbon stock in forest biomass	NFI (Programa REDD/CCAD-GIZ -SINAC, 2015)	9 plots	Database; SINAC, 2014. Inventario Nacional Forestal de Costa Rica. Sistema Nacional de Áreas de Conservación, San José.
	(Fonseca et al., 2016)	NA	$CSFB = 5.02746 + 3.74799 \times \mathrm{G}$ G = Basal area (m^2 ha^{-1})
Wood harvest	Management plans	89	Data retrieved from management plans include: total forest area (ha), effective managed area (ha), extracted standing volume (m^3), extracted species, extracted deadwood (m^3), forest area impacted by felling gaps, primary and secondary road construction, skid trails and logging decks (ha or % area). Out of the 107 forest management plans studied, 58 had been closed or finalized and 31 were ongoing. Information on wood extraction was retrieved from ongoing and closed plans, while logging damage from closed plans only.
Logging damage	Management plans	31	
Forest regrowth	(Rutishauser et al., 2015)	NA	
Harvest operations	Questionnaire	20	
Sawmill carbon flow	Questionnaire	21	All sawmills have very similar or identical processes that involve headsawing, resawing, edging, moulding, planing and in some cases groove and tongue. All wood is air dried or not dried at all.
Final transformation	Questionnaire	4	
End of life management	(Chacón et al., 2012)	Na	

Table SI-2.2. Parameters used in the uncertainty and sensitivity analyses

Parameters	Distribution	Justification
Gp	$\Gamma(7.56, 2.1)$	Gamma distribution for positive values
LgDck	$\Gamma(0.45, 5.73)$	
PrmRd	$\Gamma(0.23, 2.19)$	
ScRd	$\Gamma(1.34, 1.64)$	
SkTr	$\Gamma(1.2, 1.79)$	
H_{St}	$\Gamma(3.16, 1.29)$	
H_{Dw}	$\Gamma(0.08, 0.18)$	
$SwdR_f$	*Dir* (0.17)	Proportion; multivariate generalization of the beta distribution
$ShvR_f$	*Dir* (0.12)	
SBR_f	*Dir* (0.30)	
$SwdPlt_f$	*Dir* (0.07)	
$ShvPlt_f$	*Dir* (0.05)	
$SBPlt_f$	*Dir* (0.53)	
$SwdFw_f$	*Dir* (0.12)	
$ShvFw_f$	*Dir* (0.05)	
$SBFw_f$	*Dir* (0.29)	
$SwdSSN_f$	*Dir* (1.32)	
$ShvSSN_f$	*Dir* (0.62)	
STP_f	*Dir* (2.59)	
MTP_f	*Dir* (1.18)	
LTP_f	*Dir* (0.54)	
TL_f	β (2.18, 20.18)	Proportion
DOC_f	$N(0.50, 0.01^2)$	±20% uncertainty range and assumed positive normal distribution.
$SWDS_f$	$N(0.88, 0.19^2)$	Assumed half mean as the uncertainty range and positive normal distribution.
k_1	$N(0.1, 0.0004^2)$	±15% uncertainty range and we assumed positive normal distribution.
k_2	$N(0.07, 0.12^2)$	Assumed a positive normal distribution based on reported $t_{1/2}$ of 8-12 years.
k_3	$N(0.035, 0.0064^2)$	Assumed a positive normal distribution based on reported $t_{1/2}$ of 14-23 years.
k_4	$N(0.35, 0.48^2)$	50% uncertainty range and we assumed positive normal distribution.
k_5	$N(0.03, 0.0036^2)$	Assumed half mean as the uncertainty range and positive normal distribution.
k_6	$N(0.02, 0.0016^2)$	
ϴ	$N(1.11, 0.15^2)$	Assumed a normal distribution.
CSFB	$\Gamma(13.34, 0.13^2)$	Gamma distribution for positive values
t	15	The rotation period was fixed at 15 years

Table SI-2.3. Carbon losses from logging damage. Year 0 is the amount of carbon from logging damage and year 15 represents the amount lost due to decomposition.

	Year 0				Year 15	
	Mg C m^{-3}	Mg C ha^{-1}	N	SD	Mg C m^{-3}	Mg C ha^{-1}
Gaps	0.38	3.59	31	0.15	0.21	2.79
Skid Trails	0.09	0.67	28	0.68	0.04	0.52
Secondary roads	0.08	0.81	31	0.07	0.05	0.63
Primary roads	0.01	0.11	31	0.02	0.01	0.08
Logging decks	0.01	0.08	31	0.01	0.00	0.06
Total	0.57	5.26			0.31	4.09

Table SI-2.4. Reported percentages of products and by-products from the milling process.

Products	AVG	N	SD	SE	SE%
Form boards	26.70	21	16.05	4.91	13.49
Form laths	19.99	21	16.42	3.58	18.43
Construction*	16.98	21	25.34	5.53	12.94
TOTAL	63.67				
By-Products	AVG	N	SD	SE	SE%
Edges & off-cuts**	12.81	19	10.65	2.44	18.68
Slabs & Bark	13.40	19	5.71	1.31	9.58
Sawdust	7.74	18	5.18	1.22	15.45
Shavings	2.38	20	2.43	0.54	22.37
TOTAL	36.33				

*Combination of all long-term wood products used in construction.

**Considered also a co-product given that it is used entirely in the furniture industry.

Table SI-2.5. Reported percentages for the distribution of by-products into co-products and residues.

By-product	Co-product/Residue	AVG	N	SD	SE	SE%
Edges & off-cuts	Furniture	12.81				
Slabs&Bark		13.40				
	Residue	2.60	20	4.24	0.95	39.16
	Fuelwood	4.59	20	7.70	1.72	38.70
	Pellets	6.21	19	7.70	1.76	28.82
Sawdust		7.74				
	Residue	0.34	18	0.83	0.20	57.15
	Fuelwood	0.31	18	0.90	0.21	66.80
	Pellets	0.27	18	1.02	0.24	87.35
	Stables, stalls or nurseries	6.83	18	5.71	1.35	19.31
Shavings		2.38				
	Residue	0.25	20	0.72	0.16	63.08
	Fuelwood	0.07	20	0.29	0.06	97.47
	Pellets	0.01	20	0.05	0.01	97.47
	Stables, stalls or nurseries	2.05	20	2.58	0.58	27.48

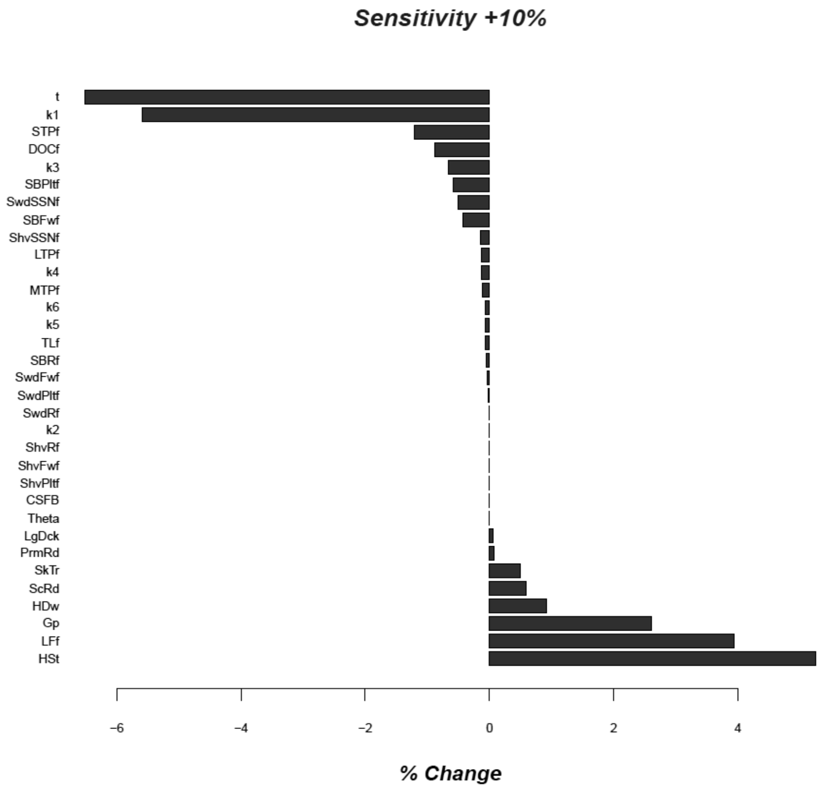

Figure SI-2.1. Effect on the carbon balance from a 10% change on each individual parameter to evaluate the sensitivity of the model (t = rotation period).

Chapter 3

The effect of changes in wood source and product allocation on the carbon stock of harvested wood products in Costa Rica

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Submitted

Abstract

Carbon storage in wood products has been growing globally and based on a predicted increase in wood production, the growth of this stock is expected to continue. However, with bioenergy and short-lived products becoming the dominant end use of wood, significant future increases in carbon storage might be compromised. Here we investigate how a major shift in wood sourcing and product use in Costa Rica during the last 26 years, has affected carbon storage in harvested wood products. We estimated changes in carbon stocks through an inventory of carbon in harvested wood products (HWP) following the "production approach" and using country specific data for domestic harvest (1990 - 2016). The material flow analysis used allowed tracing independently seven categories of wood products, differentiated by wood sources (i.e. agricultural lands, plantations and forests). Carbon storage from HWP in Costa Rica has increased since 1990 mainly due to increased wood production and is currently responsible for an annual storage equal to 30% of land use carbon emissions. In 2016, the net carbon accumulation in anthropogenic reservoirs was -412 Gg CO_2 (95% CI between -447.2 and -376.4), of which 77% is in solid waste disposal sites. We found clear differences between reservoirs. While the contribution from products in use has been rather stable, storage in disposal sites has almost doubled due to changes in the allocation of wood into products. Changes in allocation also resulted in a significant reduction in weighted half-life and carbon density of the stock of products in use but without leading to important changes in the current or future HWP contribution. Stock changes from products in use are mainly affected by harvest levels, explaining why carbon stocks from some commodities and sources reached a steady state and were even responsible for annual carbon losses due to inherited emissions. Assuming constant production, the time for the HWP carbon stock to reach steady state is very long (>500 years) because of stored carbon in solid waste disposal sites. This period is insensitive to changes in half-life or carbon density. When excluding carbon in disposal sites and considering products in use only, the time to steady state does respond to changes in half-life, but only if these are large (i.e. ±20%). This stock is more sensitive to harvest levels, with 1% changes causing carbon in products in use to increase significantly or reach equilibrium at very short timescales (around 20 years). Increasing storage by prolonging the lifetime of the stock beyond current levels is constrained by physical limits, by the inertia of carbon stock and by trends in wood production. To overcome these constraints, demand-side measures (such as increased

wood use) are inevitable if harvests or lifespan are to be increased. Such measures need to take inherent trade-offs between lifespan and harvest level into account.

Introduction

The forest sector's contribution to the stabilization of climate is large given the combined potential to avoid greenhouse gas (GHG) emissions to the atmosphere by controlling deforestation and degradation, and to increase carbon storage in forests through afforestation and reforestation (Canadell & Schulze, 2014; Grassi et al., 2018). Additionally, forest management may enhance this mitigation potential by using harvested wood products (HWP) to substitute other energy intensive materials while storing carbon outside forests (Bergman, Puettmann, Taylor, & Skog, 2014; Matsumoto et al., 2016). So far, product substitution still faces challenges of appropriately attributing and allocating carbon benefits to forestry, providing little incentives for the sector to maximize this potential. Carbon storage in HWP however, has a direct effect on climate that is entirely attributable to the forest sector and its quantification is already compulsory for Annex I countries under the Kyoto Protocol (Aleinikovas et al., 2018).

Globally, the carbon stock in HWP is known to increase by 26 - 139 Tg C yr^{-1}, i.e. around 0.2 - 1.2% of total annual global carbon emissions (Brown, Lim, & Schlamadinger, 1998; Donlan, Skog, & Byrne, 2012; Ji, Cao, Chen, & Yang, 2016; Johnston & Radeloff, 2019; E. S. Marland, Stellar, & Marland, 2010; Pilli et al., 2015; Quéré et al., 2018; Winjum et al., 1998). This annual increase may be small compared to fluxes from natural sinks (between 0.7 – 3.7% of the global terrestrial sink), but it can be significant for individual countries, where it can vary from 5 to 30% of total land use emissions (Aleinikovas et al., 2018; Cláudia Dias et al., 2009; Pilli et al., 2015; Quéré et al., 2018; Winjum et al., 1998). In the European Union for example, it is estimated that storage from HWP is approximately 10% of the land use carbon sink and could offset around 1% of EU total annual GHG emissions (Pilli et al., 2015).

Since the default assumption in national GHG inventories (GHGIs) is the instantaneous oxidation of carbon after management practices such as harvesting (Aalde et al., 2006), not accounting for HWP may result in the overestimation of emissions from forest management in national and global carbon budgets (Iordan et al., 2018; Pingoud & Wagner, 2006; Skog et al., 2004). Out of the 4 Gt CO_2 yr^{-1} difference in land use emissions between modelled global estimates and aggregated estimates from national GHGIs, approximately 3.2 Gt CO_2 yr^{-1} is likely partially due to the way managed lands are accounted (Grassi et al., 2018). Emissions from managed lands are thus important for accurate global carbon budgets, and HWP constitutes a key component of this budget in wood producing countries.

Carbon stocks in HWP are commonly estimated using exponential first-order decay (FOD) for its simplicity to model dynamic systems (Pingoud et al., 2006; Pingoud & Wagner, 2006). FOD uses two parameters to estimate stocks at the end of an analysed period (M_t); the initial mass (M_0) and a decay constant (k). In the inventory of carbon in wood products, M_0 is equal to the carbon from harvested wood at the beginning of the period (usually taken as 1 year) plus the inflow of wood from that year's harvests. k is estimated using wood product half-life and it is used to determine the mass fraction lost during each period. An increase or decrease in carbon stocks (M_t) from year to year (a.k.a. the "HWP contribution"), is an approximation of the net carbon fluxes from or to the atmosphere (Cowie et al., 2006).

As would be expected from this FOD model, estimated carbon stocks in HWP are mainly influenced by harvest levels and product half-life (Donlan et al., 2012; Pingoud et al., 2010; Skog et al., 2004). Other factors affecting the stock are the allocation of wood into product categories (Aleinikovas et al., 2018; Jasinevičius et al., 2018; Pilli et al., 2015; Pingoud et al., 2010), which essentially determines half-life; and the conversion factor of wood volume into carbon (Donlan et al., 2012; Skog et al., 2004), relevant for its direct relationship on harvest levels. Additionally, the type of end of life management, e.g. combustion, disposal or recycling, is important in those inventories where this phase of the lifecycle of wood products has been included (Donlan et al., 2012; Pingoud et al., 2010; Skog et al., 2004).

Solid waste disposal sites (SWDS) are a significant carbon stock, and in some cases their carbon stock exceeds that of products in use (Skog et al., 2004). Together, products in use and SWDS are known as anthropogenic reservoirs and are responsible for so called technospheric carbon storage. Since carbon in SWDS is also assumed to follow a FOD function, the combined effect of both reservoirs results in the extended lifetime of products (E. S. Marland et al., 2010). Additionally, due to the anaerobic conditions found in SWDS, there is an inert fraction of the stock that largely explains the size of this reservoir (Pingoud & Wagner, 2006).

Increasing carbon storage in HWP can be a potential climate mitigation option (Intergovernmental Panel on Climate Change, 2014), achieved by increasing harvest levels, increasing product half-life, or both (E. Marland & Marland, 2003). Since increasing harvest levels may be constrained in some regions by the availability of productive lands or by trade-offs with forest carbon accumulation (Pilli et al., 2015), the dominant recommendation is a more efficient wood use to prolong lifespan (Intergovernmental Panel on Climate Change, 2014; Lun et al., 2012). This higher efficiency can be understood as directly prolonging a product's half-life (e.g. using it for 100 instead of 35 years), or through a cascading effect where wood is recycled for subsequent uses until it degrades or is used for energy (Sathre & Gustavsson, 2006).

Increasing half-life has a linear and slightly larger effect on changes in carbon stocks than cascading, where a 20% increase in half-life results in a 10% increase in the stock (Brunet-Navarro et al., 2017). If the default half-lives for sawn wood, boards and panels, and pulp and paper used in HWP inventories are 35, 25 and 2 years respectively (IPCC, 2014), this 10% mitigation potential could theoretically be reached by increasing half-lives to 42, 30 and 2.4 years. It is possible to increase half-lives given that these are determined by socio-economic factors (Pingoud & Wagner, 2006), but similar to the median from a sample, relatively large changes in product lifetimes are required before observing small changes in half-life. Furthermore, due to large variability and lack of theoretically based estimates of lifetime values (E. S. Marland et al., 2010; Pingoud & Wagner, 2006), it is possible that small changes in half-life may simply fall within the uncertainty range.

Changes in lifespan can also be interpreted considering that wood product categories collectively form the stock of carbon from HWP. Then, the stock's half-life depends on its composition which is in turn determined by how wood has been traditionally allocated into product categories. By modifying product allocation, changes in the stock's half-life can be achieved so long as the existing demand for wood uses is satisfied. In this case, it is the lifetime of the stock rather than the lifetime of products what increases carbon storage. If most products are short lived, the stock will tend towards a saturation point or reach steady state much sooner (i.e. the moment when the inflow is balanced by the outflow). More storage can then be achieved by producing more long-term products. However, as long as harvest levels are constant, all stocks regardless of their lifetime will eventually settle down to a constant fraction (E. S. Marland et al., 2010; Pilli et al., 2015; Werner et al., 2010).

Opposite to how storage can be increased, the steady state is reached when annual harvest and emission rates are equal (i.e. inflow ≤ outflow). Annual losses come from products harvested in the past (i.e. inherited emissions) and are the result of previous decisions on wood production and use. Past allocations determine the half-life of the present stock, and similarly, current decisions on wood production and use will modify the half-life of the future stock. Therefore, to modify carbon storage in HWP there is a potentially considerable lag effect due to historic and existing trends in land management, wood production and use (Birdsey & Pan, 2015; Pilli et al., 2015; Poker & MacDicken, 2016).

In Costa Rica, wood production has increased considerably since 1990. Here we study how changes in wood sources (i.e. agricultural lands, forest plantations and natural forests) and product allocation (e.g. construction, furniture, packaging, exports) has affected and will affect the carbon stock of wood products. We analyse the potential mitigation benefits from increased carbon storage in HWP, and the opportunities to increase this potential. We developed a HWP carbon inventory for Costa Rica using material flow analysis and country specific data on wood production, use and disposal. This approach and data provide the opportunity to trace carbon flows of individual product categories, wood sources and for every year during the analysed period. Understanding the dynamics of wood production and carbon storage in wood products is becoming relevant as these are being considered within the REDD+ strategies of tropical countries (Butarbutar et al., 2016; Sasaki et al., 2012).

Methods

Study area

We estimated the contribution of wood products to net CO_2 emissions in Costa Rica (1990-2016) through a HWP inventory following the "production approach" and using a material flow analysis to determine annual carbon stocks and changes in anthropogenic reservoirs, i.e. products in use and products in solid waste disposal sites (SWDS). The term "approach" refers to how system boundaries for GHG accounting and reporting purposes are defined, not the methods used to estimate stocks and emissions (Cláudia Dias et al., 2009; Lim, Brown, &

Schlamadinger, 1999). In the production approach the accounting boundaries are set within the producing country only, and includes all wood harvested domestically even if it is consumed elsewhere (IPCC, 2014; E. S. Marland et al., 2010). This requires assumptions over the use and end of life of exported wood products, but tracing processes occurring outside these boundaries would be hard or even impossible (Downie et al., 2014).

In Costa Rica, domestic harvest comes from three different wood sources: 1) natural forests; 2) forestry plantations, and; 3) agricultural lands without forest cover. Managed natural forests are forest areas with a documented management plan; plantations are forests established through planting, and; agricultural lands or trees outside forests are wooded lands not classified as forests due to low density tree cover (Birdsey & Pan, 2015). Managed natural forests in Costa Rica are mostly tropical wet and moist forests (MINAE, 2011, 2012, 2013). Forestry plantations include over 10 native and non-native species but are dominated by non-native species such as *Gmelina arborea* and *Tectona grandis*. Agricultural lands without forest cover are characterized by dispersed trees within agricultural sites from > 30 species, although few dominate (e.g. *Enterolobium cyclocarpum*, *Cordia alliodora*, *Cedrela odorata and Samanea saman*). Trees can be remnants from forests historically cleared for agriculture, natural regeneration or planted.

Harvest from all sources was divided into seven categories of wood products (Barrantes & Ugalde, 2012, 2015, 2017); i.e. construction, furniture, packaging, exports of roundwood and sawnwood (coniferous and non-coniferous), and other products (Figure 3.1). Two additional categories are fuelwood/bioenergy used in the country and wood residues from transformation processes. Fuelwood is outside the scope of the inventory, but its quantification is an important part of the material flow analysis of wood production. Residues were considered as an additional category in the inventory, for which we assumed mill dumps as the end of life. Altogether, wood products, fuelwood and residues comprise total roundwood production.

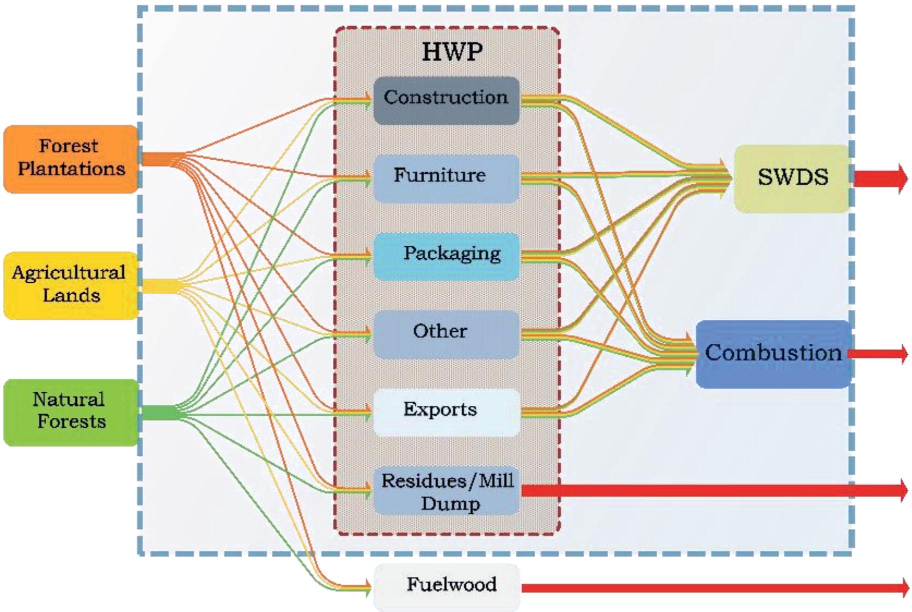

Figure 3.1. Material flow analysis of harvested wood products (HWP) in Costa Rica (2016). The square bordered by a dashed blue line contains all the carbon stocks derived from HWP considered in this study.

Domestic harvest and product use

National statistics on domestic harvest and product use has been collected since 2000 by the National Forest Office (Barrantes, Paniagua, & Salazar, 2011; Barrantes & Salazar, 2007, 2008, 2009, 2010; Barrantes & Ugalde, 2012, 2013, 2014, 2015, 2016, 2017). Since the most complete dataset for all product categories and wood source is from 2007 onwards, harvest and uses during 1990 - 2006 had to be reconstructed based on national statistics, reports, or were estimated based on their average share in product allocation during 2007-2017. Harvest data per wood source between 1990 - 2000 was taken from national reports (Arce & Barrantes, 2004; Calderón, 2000; INEC, 2015; Moya, 2004) and statistics (1990- 1998) on the number of permits and volume harvested (Canet et al., 1996; DGF, 1988, 1993, 1994). During 1994 – 1997 information was lacking for agricultural lands and forests, so we assumed agricultural lands represented 41% of harvest, while natural forests was assumed as the sum of estimates for different product categories.

To estimate the use of construction wood during 1990 – 2000 we collected national statistics on constructed area (m^2) between 1983 – 2017 (INEC, 2018) and used ONF data for 2004 – 2017 to estimate the relationship between m^3 of wood per m^2 constructed. The use of wood in furniture (1990 -2000) was mostly taken from the literature (Mckenzie, 2003) and assumed to be distributed equally between agricultural lands and natural forests. For the period 2001 – 2006, we used a linear regression based on harvest and wood use per wood source from ONF database (2007 -2017). Data on wood use for packaging (1990 -1997) was derived from agricultural exports (DGEC, 1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997) and from 1998 – 2006 from agricultural and industrial exports (PROCOMER, 2018). The share of "other wood products" per wood source was mostly assumed to be the same as in ONF database for 2007 – 2017, although for natural forests we also considered the reports from the wood panel industry that operated until the early 2000's (DGF, 1988; Serrano & Moya, 2011). During 1990 – 1997 we assumed plantations did not contribute to this category. To estimate wood exports we collected data for 1998 - 2016 (PROCOMER, 2018) and from the Dirección General de Estadística y Censo for 1990-1997 (DGEC, 1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997). During 1990 - 2000 we assumed all exports from conifer species were sourced from planted forests while non-conifers from forests. From 2001 – 2006, we used the share reported for each wood source during 2007 – 2017. Data on fuelwood/bioenergy was obtained from the ONF database and through secondary sources (García, 2013; Ramírez Hernández, Carazo Fernández, Roldán Villalobos, & Villegas Barahona, 2007; SEPSE, 2018).

The proportion of wood volume harvested that is converted into wood products was estimated using milling efficiencies differentiated by wood source. For 1990 - 2000, a 0.48 milling efficiency was used for wood sourced from forests and agricultural lands (DGF, 1988). After 2000, the annual milling efficiencies reported in the ONF database were used. Since the efficiencies reported for 1990-2000 focused on natural forests and agricultural lands, for forest plantations we used those from the ONF database throughout the analysed period. Overall, efficiencies ranged between 0.41 – 0.55 for all sources and uses.

Estimating inherited emissions (1900 – 1990)

To estimate the total amount of carbon in wood products and its dynamics, we included carbon in products from previous harvests and their decay. Such inherited emissions are carbon emissions during a given year from harvests prior to that year. Historic annual harvest (1900-

1990) was estimated based on product allocation and harvest from 1990 together with a regional rate of change in wood consumption for each source and product category taken from (Pingoud et al., 2006) (Equation SI 1). These are important to avoid the overestimation of the stock (Reid Miner, 2006), but historical inflows do not challenge the accuracy of present time changes in the stock (Pingoud & Wagner, 2006).

Carbon in wood products

To transform wood volumes into dry biomass, weighted average wood densities were estimated using tree species and volumes harvested from natural forests in Costa Rica between 2010 – 2016 (161 species) and plantations (13 species), resulting in 0.5 and 0.47 kg m^{-3} respectively. Due to lack of data on species harvested from agricultural lands, we assumed density of wood products to be equal to that of the unweighted average for the 36 most common species reported in the ONF databases (i.e. 0.49 kg m^{-3}). Wood densities were taken from the literature (Blanco, Carpio, & Muñoz, 2005; Chave et al., 2009).

The amount of carbon in the dry biomass was calculated using fractions for each wood source. For natural forests we used the average stem carbon fraction for moist and wet forests in Costa Rica of 0.45 (Fonseca et al., 2016). The carbon fraction for forest plantations (i.e. 0.45) was estimated from the average fractions reported for the 13 most common species planted in Costa Rica (Solano, 2017). Agricultural lands used the 0.43 average fraction reported for two of the most common species (Ruiz García, 2002) (Equation SI 2).

Size and annual changes of carbon stock from products in use

The carbon stock per product category and its annual change were estimated through the first order decay model commonly used in HWP inventories (Pingoud et al., 2006). Every product was assigned a half-life selected to best represent their characteristics (Table 3.1). Carbon stocks were estimated for every year (1900-2016). Each annual stock includes the inherited stock and the inflow of products from that year's harvest. The latter is corrected to account for an inflow that happens throughout the year (Equation SI 3). Changes in carbon stocks are estimated as the difference between years (Equation SI 4).

Carbon stock and annual stock changes from products in solid waste disposal sites (SWDS)

To estimate the stock of carbon in SWDS, we first determined the amount of HWP retired from service that flow to the end of life (EoL). This is basically the inverse of the estimation of stocks (Equation SI 5). To differentiate the type of EoL (SWDS or combustion), we used the fraction of waste sent to SWDS reported by the Costa Rican GHG Inventory; i.e. 0.88 (Chacón et al., 2012).

The carbon in wood products reaching SWDS was divided into two different stocks. The Decomposable Degradable Organic Carbon (DDOC), i.e. carbon that can decompose under anaerobic conditions; and the inert stock that accumulates (DOC_a). For this, a 0.5 fraction (DOC_f) was used (Pipatti et al., 2006) and the stock that decomposes can be estimated using first order decay (Equation SI 6). Different half-lives were used according to the type of SWDS. Prior to 1990, it was assumed that all wood products were disposed of in open dumps, where the half-life of wood products was 16.5 years (L. Zhang et al., 2019). After 1990, wood was sent to landfills and its half-life increased to 20 years (Pipatti et al., 2006) (Table 3.1). Wood residues not combusted as fuelwood, remained in mill dumps and were assigned a 10 year half-life (Pipatti et al., 2006).

The carbon reservoir from HWP in SWDS for any given year results from the sum of DDOC and DOC_a (Equations SI 6-8), with changes estimated as the difference between consecutive years (Equation SI 9). This same approach was used to estimate carbon storage in mill dumps, which was done independently (Equation SI 10 – 13).

Table 3.1. Selected wood product retirement rates and decay rates (k) for harvested wood products in solid waste disposal sites.

Wood Use	Half-life (yr)	k	Reference
Construction Furniture Round & sawn wood exports	35	0.020	(IPCC, 2014)
Other	25	0.028	(IPCC, 2014)
Packaging	6	0.116	(Skog & Nicholson, 1998)
End of Life			
Mill dump	10	0.069	(Pipatti et al., 2006)
Landfill	20	0.035	
Open dumps	16.5	0.042	(X. Zhang, Yang, & Chen, 2018)

Harvested wood product contribution

The "HWP contribution" is the sum of the annual changes in the main carbon pools (i.e. products in use and solid waste disposal sites) and for all wood sources and products. We expressed it as Gg of CO_2 and with a negative sign when emissions to the atmosphere were avoided via carbon storage (Equation SI 14).

Carbon released to the atmosphere

We also estimated carbon released to the atmosphere due to combustion or decomposition of wood products or residues and report other GHG depending on the type of EoL (Equations SI 15-19); i.e. N_2O and CH_4 emitted during combustion and CH_4 from biomass decomposition under anaerobic conditions in SWDS. For emissions due to combustion of fuelwood and open burning, we used a 0.95 and 0.58 oxidation factors for CO_2; an emission factor of 0.15 g N_2O/kg dry matter and an emission factor of 6,500 g/Mg wet weight for CH_4 (Guendehou et al., 2006). CH_4 emissions from SWDS were estimated using a methane correction factor differentiated by

SWDS type (i.e. 1 for managed anaerobic, 0.4 for unmanaged shallow, 0.8 for unmanaged deep, and 0.6 for uncategorized). From the 0.88 fraction of waste that ends in SWDS in Costa Rica, 0.51 are classified as managed anaerobic sites, 0.09 are unmanaged shallow, 0.09 are unmanaged deep, and 0.18 are uncategorized sites (Chacón et al., 2012) (Equation SI 20). The fraction of CH_4 generated in landfills (0.47), the fraction of CH_4 that is recovered at the EoL (i.e. 0.23) and the oxidation factor of zero were all taken from the National GHG inventory (Chacón et al., 2012).

Uncertainty analysis

We performed Monte Carlo simulations using the @Risk Software (Palisade, 2016) to determine the uncertainty of the HWP contribution to net CO_2 emissions in Costa Rica for 2016. Probability density functions were determined for all input variables (Table 3.2), including all wood volumes for every product category and source since 1990 (not shown in this table). Wood volumes were all assigned a triangular distribution (Skog et al., 2004), except for exports of coniferous sawnwood from forests and coniferous roundwood from forest and agricultural lands, which were excluded given that no exports for these categories were reported throughout the period. We used Latin Hypercube to sample from the distributions of each variable and ran 10000 simulations to calculate the mean and 95% confidence intervals.

Table 3.2. Probability density functions for the Monte Carlo Analysis

Variables		Distribution	Justification
Wood Density	Agricultural lands	*N* (0.49; 0.0195^2)	Positive normal distribution
	Forests	*N* (0.5; 0.00004^2)	
	Plantations	*N* (0.47; 0.014^2)	
Carbon fraction	Agricultural lands	*N* (0.425; 0.00005^2)	
	Forests	*N* (0.45; 0.008^2)	
	Plantations	*N* (0.45; 0.003^2)	
Decay rates	Construction	*N* (0.02; 0.000009^2)	27-50 (CI= 95 %) (Skog et al., 2004)
	Furniture	*N* (0.02; 0.000009^2)	
	Roundwood & sawn wood exports	*N* (0.02; 0.000009^2)	
	Other uses	Triangular (0.014; 0.14; 0.03)	
	Packaging	Triangular (0.09; 0.69; 0.12)	
	Mill dumps	Triangular (0.05; 0.09; 0.06)	0.058 – 0.087 (CI= 95 %) (Pipatti et al., 2006)
	Landfills	Triangular (0.028; 0.053; 0.035)	0.03 – 0.05 (CI = 95 %) (Pipatti et al., 2006)
	Open dumps	Triangular (0.026; 0.051; 0.046)	0.03 – 0.05 (CI = 95 %) (Pipatti et al., 2006)
$SWDS_f$		*N* (0.88; 0.08^2)	Assumed 1/2 mean as uncertainty range and positive normal distribution.
DOC_f		*N* (0.5; 0.01^2)	20% uncertainty range and we assumed positive normal distribution.
U		*N* (0.022; 0.00004^2)	Assumed positive normal distribution (Pipatti et al., 2006; Skog et al., 2004)

The effect of changes in wood source and product allocation in the carbon stock

To quantify the effect of changes in wood sourcing and product allocation on carbon stocks, we estimated annual weighted averages for the conversion factors of wood volume into carbon (wood density and carbon fraction) and half-life. This estimate was based on the proportion of each source and product category on each stock and year (products in use or SWDS). We chose these variables since the conversion factor of volume into carbon serves as an indicator of changes in wood source, and the weighted stock's half-life as an indicator of changes in product allocation. We did this for domestic harvest since 1990 (i.e. the inflow), and the stock of products in use. Weighted half-lives for solid waste disposal sites were also estimated. The half-life for the overall stock was determined as the sum of products in use and SWDS. We tested for significant trends in these weighted averages and estimated the HWP contribution using the initial and final weighted values (i.e. 1990 and 2016) as in a sensitivity analysis. This, with the aim of estimating not only if there has been a significant trend but its absolute effect on the stock. We report changes in the 2016 HWP contribution due to a change in each variable.

Because changes due to shifts in half-life may be slow and the effect delayed, we used the time for the stock to reach the stabilization point (time to steady state, T_{ss}) as an indicator for long-term changes in carbon stocks, i.e. the moment when the outflow equals the inflow (Pilli et al., 2015). T_{ss} was estimated analytically (Equation SI 21) and using our model to verify this estimate for all years since 1990. We tested the significance of these trends and compared changes in T_{ss} considering the initial (1990) and final (2016) conditions to estimate the absolute effect on the stock.

Finally, since carbon stock changes are also influenced by changes in harvest levels, we developed four different scenarios and determined T_{ss}. These scenarios are extrapolations of conditions on carbon density and product allocation found in 2016, and involved modifying only domestic harvest (m^3). In the first scenario harvest remained constant, in the second we used the average harvest rate for the last 10 years (2007-2016), and the last two used a +/- 1% change in harvest rate as we considered this to be more realistic than 10% changes as used in some sensitivity analysis.

Results

Domestic harvest and product use

Costa Rican wood production has significantly increased during 1990-2016 ($r=0.9$; $p<0.001$). During this period, the sourcing of wood also changed considerably. Initially natural forests and agricultural lands were the main source of wood, accounting for 98% of domestic harvest

(Figure 3.2). At the end of the period, natural forests were replaced by plantations as a wood source: their share decreased from 60% of domestic harvest in 1990 to less than 5% after 2008. Plantations became an important source of wood late in the 1990s and increased gradually until reaching 80% of domestic harvest in 2016. Wood from agricultural lands experienced a small increase towards the end of the 1990s and early 2000s, when a small peak occurred. During this period, agricultural lands substituted forests until plantations established in the early 1990's were ready for logging.

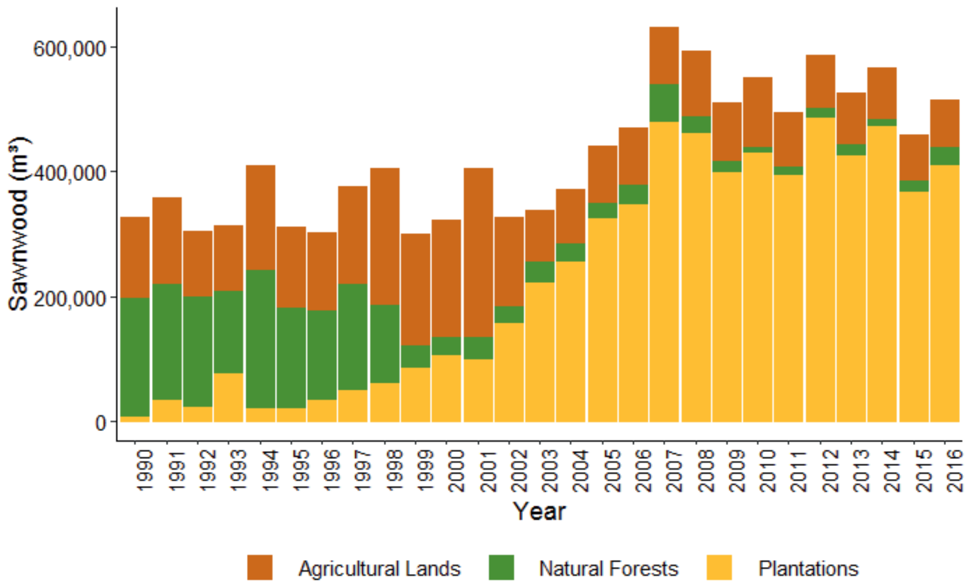

Figure 3.2. Changing source of sawn wood production in Costa Rica, 1990 -2016.

During the same period, major changes in wood use were also observed (Figure 3.3). Construction was the main wood use in the country but fell from 73% of domestic harvest in 1990 to 26% in 2016. Packaging, consisting mainly of wooden pallets grew (alongside agricultural exports) to 44% in 2016. An increase in wood exports is the other relevant change, reflecting the recent increase in round wood teak exports.

Given the normal range of wood transformation efficiencies in the forest sector (0.41 – 0.55), the amount of wood residues was large, with years (2000 - 2007) in which residues were more than the combination of all product categories. On average, 390,000 m^3 of residues were estimated to be produced per year (280,247 – 641,946 m^3 yr^{-1}). Additionally, the use of biomass for energy increased from 65,000 m^3 in 1990 to 90,535 m^3 in 2016.

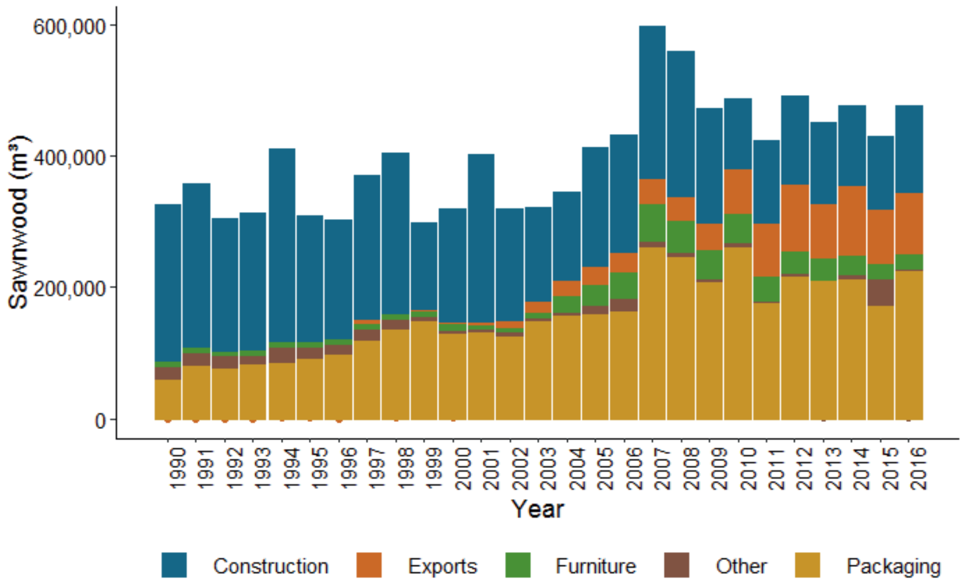

Figure 3.3. Sawn wood production per wood use in Costa Rica 1990 – 2016 (m^3-s).

Carbon stock and changes in carbon stocks from HWP

Carbon stocks in anthropogenic reservoirs have grown consistently during the study period, from 3,957 Gg C in 1990 to 7,066 Gg C in 2016. Carbon in products in use reduced from 36% in 1990 to 31% in 2016, but increased from 63 to 69% by 2016 in solid waste disposal sites (SWDS), 44% of which were wood residues in mill and open dumps. The main components to carbon storage in Costa Rica were products in use and wood residues, accounting for 75% of all carbon from harvested wood.

The annual change in the stocks of carbon from HWP (i.e. the HWP contribution) ranged between 73-209 Gg C between 1990-2016, with 112 Gg C stored in 2016. Solid waste disposal sites were responsible for 77% of the change in carbon stocks during the last year of the inventory (i.e. landfills, mill and open dumps; Figure 3.4). Although the stock of carbon from products in use is still growing, annual changes seem to be decreasing.

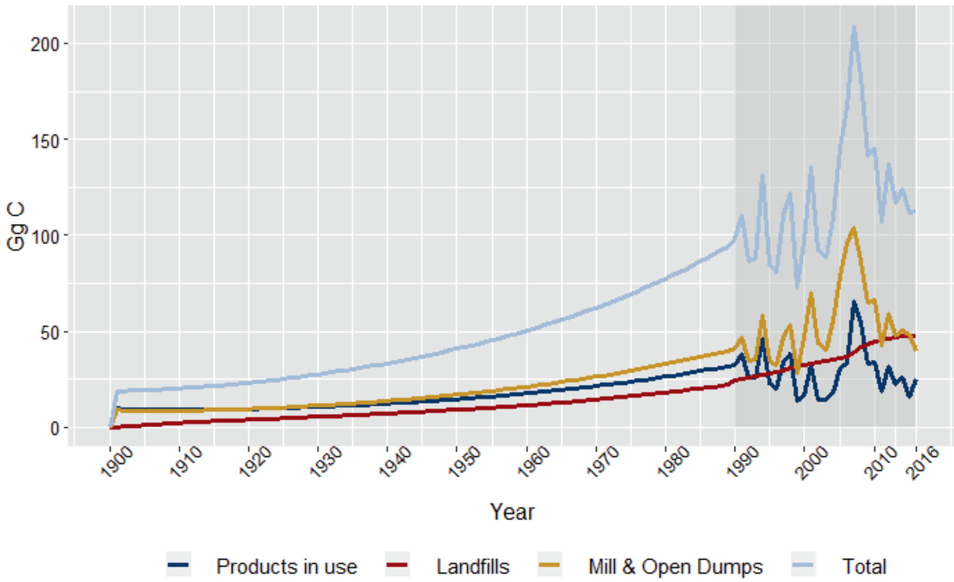

Figure 3.4. Reconstruction of the changes in carbon stocks per type of reservoir for Costa Rica (1900 – 2016). The shaded area represents the period for which annual data on wood sources and use was available.

Changes in carbon stocks varied depending on the source of wood. Due to the noticeable reduction in wood harvest after 1998 and a large inherited stock, carbon stored in products from natural forests became a source of carbon, losing 6 Gg C in 2016. Agricultural lands have remained rather stable and during 2016 products from this wood source stored 10 Gg C. HWP from planted forests account for 96% of the change in carbon stocks in 2016 and have been responsible for most of the carbon storage since the early 2000's (Figure 3.5).

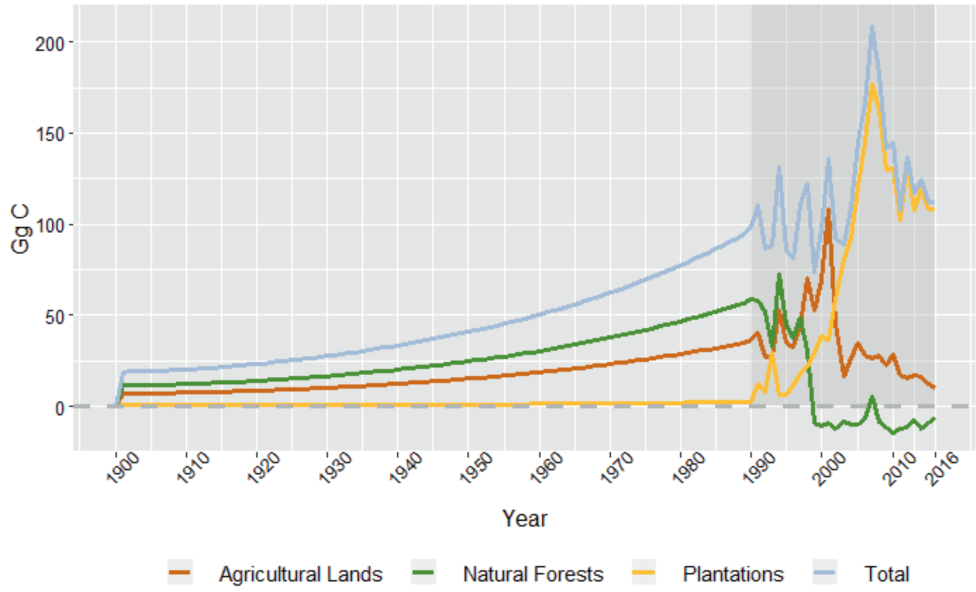

Figure 3.5. Reconstruction of the changes in carbon stocks per source of wood for Costa Rica (1900 – 2016). The shaded area represents the period for which annual data on wood sources and use was available.

When the allocation of products is taken into consideration we can determine which products contributed more to the changes in carbon stocks and how this has changed over time (Figure 3.6). Construction dominated changes in the carbon stock of HWP until the late 1990s but was replaced by packaging in 2003. By 2016, changes in the carbon stock of construction wood represented 18% of the overall change, while exports and packaging accounted for 26 and 49% respectively. Furniture increased slightly throughout the analysed period while "other" wood products remained constant except for the peak in 2015.

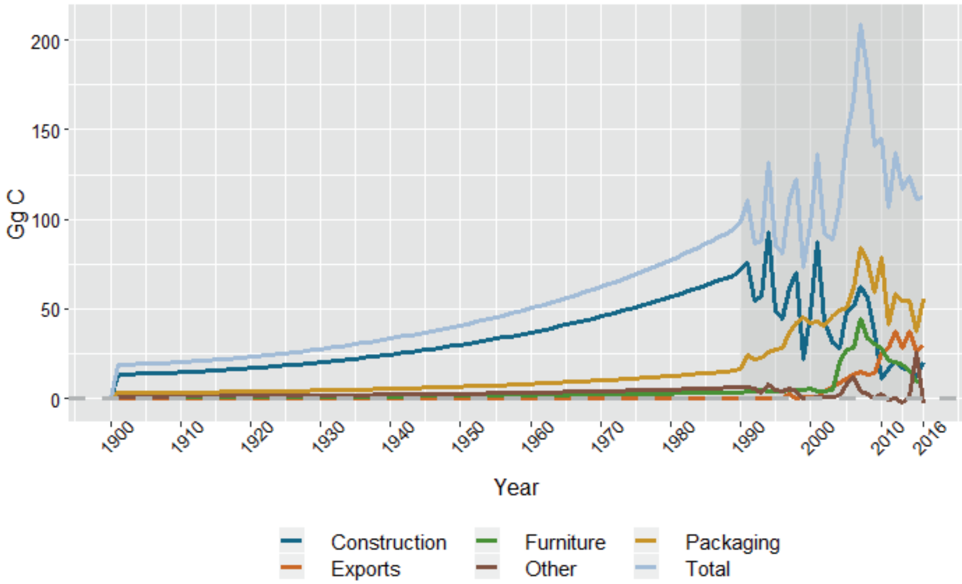

Figure 3.6. Reconstruction of the changes in carbon stocks per product type (1900 – 2016) for Costa Rica. The shaded area represents the period for which annual data on wood sources and use was available.

Changes in stocks were more evident in the "products in use" category given their susceptibility to changes in harvest levels. The stock of products in use shows that during 2016 in Costa Rica, carbon storage was only due to forest plantations while natural forests and agricultural lands were a source of carbon (Figure 3.7). Therefore, the net balance from the stock of products in use is lower than changes in the carbon stock of plantations. Lower harvest levels and inherited emissions determined changes in the stock from these wood sources.

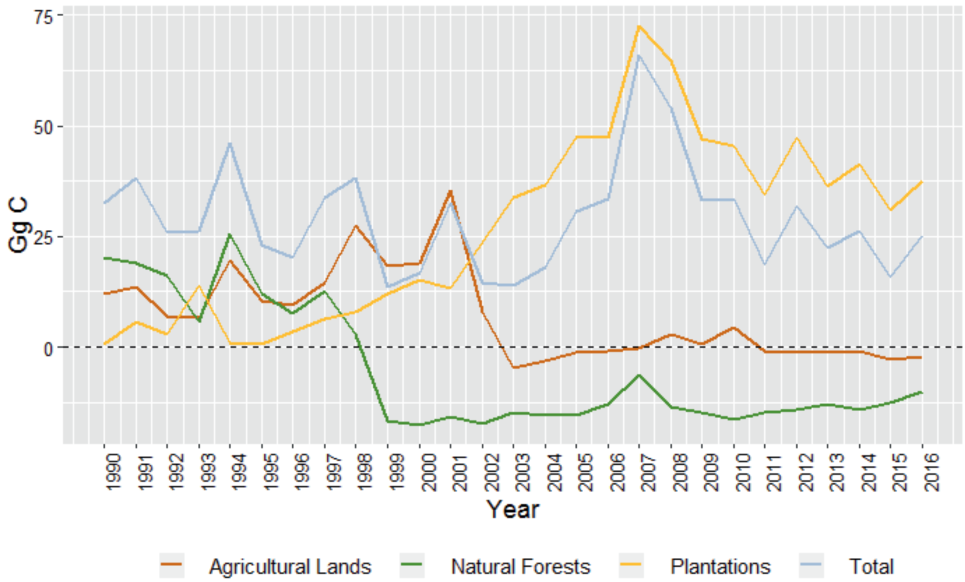

Figure 3.7. Changes in carbon stocks from products in use for Costa Rica, per wood source (1990 -2016).

Carbon stock changes per product category (Figure 3.8) were mostly due to shifts in wood exports (66% of the change). Carbon losses from construction and "other" uses are larger than the inflow, and even packaging with an increasing production rate has experienced net losses (in 2015). Higher losses show that the inherited stock determines the carbon balance of products in use, while new products are retired faster due to a shift in wood use (from construction to packaging).

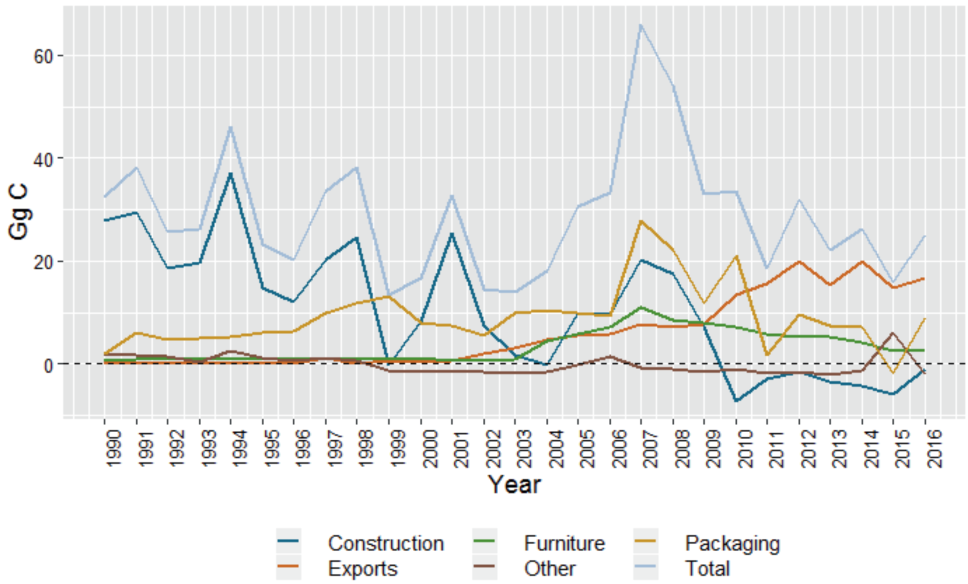

Figure 3.8. Changes in carbon stocks for products in use for Costa Rica, per product type (1990 -2016).

HWP contribution & uncertainty analysis

In 2016, the HWP contribution to net emissions in Costa Rica was -412 Gg CO_2. The uncertainty range estimated through Monte Carlo analysis resulted in a 95% confidence interval of -447.2 and -376.4 Gg CO_2, or +/- 8.6% below and above the mean. Overall, this implies that in 2016, HWP in Costa Rica were accumulating carbon.

The uncertainty range from this estimate is due to the variability in the input data on domestic harvest; more specifically, domestic harvest from forest plantations. In order of importance, planted wood used for packaging and the residues from its transformation, followed by construction wood and exports also from plantations were the most important sources of uncertainty. This is not surprising since production from forest plantations grew from 2% in 1990 to 80% in 2016 and are responsible for 96% of the change in carbon stocks in 2016 (Figure 3.5).

The effect of changes in wood source and product allocation in the carbon stock

Our results show a significant increase of wood sourcing from plantations, and a decrease in sourcing from natural forests and agricultural lands (Table 3.3). Similarly, there were increasing trends in wood use for packaging, furniture and exports and a decrease in construction wood. The only category where no significant difference was observed were "Other" uses which include boards and panels. These trends show a potential association between change in wood source and product allocation, e.g. through a positive correlation between decreasing harvest levels from natural forests and less construction wood (cor 0.72; $p < 0.001$). However, other socio-economic causes may better explain these patterns (e.g. increased agricultural exports) and we therefore focused on changes that have a direct effect on carbon stocks.

A significant declining trend ($p < 0.001$) in the weighted average carbon conversion factors from 1990 to 2016 suggests a reduction in the carbon density of harvested wood products in Costa Rica (Figure 3.9a). However, since the difference between the initial wood densities and carbon fractions for the three wood sources was already small, the resulting changes in the carbon stocks were also small. Agricultural lands and forest plantations shared an almost exact factor of 0.21 Mg C m^{-3} while forests had a slightly higher conversion factor of 0.22 Mg C m^{-3}. When combined in a weighted average, the difference for the inflow and the stock of products in use between 1990 and 2016 were 0.004 and 0.002 Mg C m^{-3} respectively. When the conversion factors in our model were substituted with the weighted averages from 1990 and 2016 (i.e. products in use and SWDS), we estimated a 1.2% difference in the HWP contribution.

The annual weighted half-lives of the inflow of HWP also showed a significant declining trend from 29 to 21.2 years (Figure 3.9b). Despite this 8-year difference, the weighted half-life of the stock of products in use decreased by just 2 years during this period (Figure 3.9c i.e. from 32.4 to 30 years). SWDS, which also included mill dumps, increased slightly from 13.4 to 13.7 years; Figure 3.9d), while the overall stock's weighted half-life decreased from 45.8 to 43.8 years (Figure 3.9d). The difference between the 1990 and 2016 weighted half-lives for SWDS and the total stock were not significant. When we substituted the half-lives in our model to test the effect of changes in the weighted half-life from 1990 to 2016 in the overall contribution, we estimated only a 1% difference between both results. This shows that although changes in weighted half-life were significant, their effect on the changes in the stock are marginal.

Calculations for the time to steady state (T_{ss}) using analytical methods revealed significant trends between 1990 to 2016 (Table 3.3). Given the change in half-lives during this period, T_{ss} is reduced but this reduction was smaller in magnitude than that of certain pools which reached a steady state abruptly (e.g. packaging in 2015) or during a time frame comparable to the analysed period. When we used our model to estimate T_{ss}, we found that SWDS, mill dumps and consequently the overall stock would take extremely long periods of time to reach the steady state (>500 years). However, the stock of products in use reached steady state earlier than estimated analytically but with no differences between initial and final conditions. Using weighted carbon conversion factors for 1990 and 2016 we estimated 150 years as the T_{ss} for products in use, while changing the weighted half-lives for this same stock resulted in a 4-year difference (i.e. from 145 to 141 years).

If the inflow is constant, our harvest scenarios show that T_{ss} of mill dumps, landfills and consequently the overall stock will be reached in extremely long periods of time (>500 years); with a 1% increase in the production rate prolonging T_{ss} even further. The last 10 years' average production rate varied according to wood source, where forests and agricultural lands showed decreasing rates (i.e. -0.02 and -0.06 respectively), while production grew by 0.005 in plantations. This scenario resulted in a less pronounced reduction in T_{ss} than a constant -0.01 rate applied to all ecosystems. For the total stock T_{ss} is 262 years under the 10-year average scenario, and 155 years under a -0.01 scenario. These shifts in T*ss* are mainly driven by the dynamics in SWDS and mill dumps.

In contrast to the stocks in SWDS and mill dumps, the carbon stock in products in use responds strongly to changes in harvest levels. For instance, when reducing the inflow by 1% annually, T_{ss} reduces from 147 years to 21 years. This demonstrates that abrupt changes in the *steady state* of products in use only occur as a result of changes in harvest levels.

Table 3.3. Trends in variables related to harvested wood products in Costa Rica during 1990-2016**.**

		1990	**2016**	**r^2**	**p-value**
Sawnwood (Gg C)	Harvest	151	206	0.8973	<0.001
Carbon Stock Changes (Gg C)	Total	97.98	112.26	0.1843	0.02
	Products in use	32.63	25.25	0.0028	0.8
	SWDS	24.48	47.56	0.9789	<0.001
	SWDS (Mill dumps)	40.87	39.44	0.1156	0.08
Weighted average carbon fraction	Harvest (Mg C m^{-3})	0.2163	0.2123	0.4936	<0.001
	Products in use (Mg C m^{-3})	0.2167	0.2142	0.978	<0.001
Weighted average half-life (*yr*)	Harvest (inflow)	29	21	0.6581	<0.001
	Products in use	32.44	30.09	0.9801	<0.001
	SWDS	13.37	13.68	0.8031	<0.001
	Total	45.81	43.77	0.964	<0.001
Time to Steady State (*yr*)	Products in use	378.36	347.64	0.9801	<0.001
	SWDS	138.90	142.54	0.8031	<0.001
	Total	557.17	529.41	0.9637	<0.001

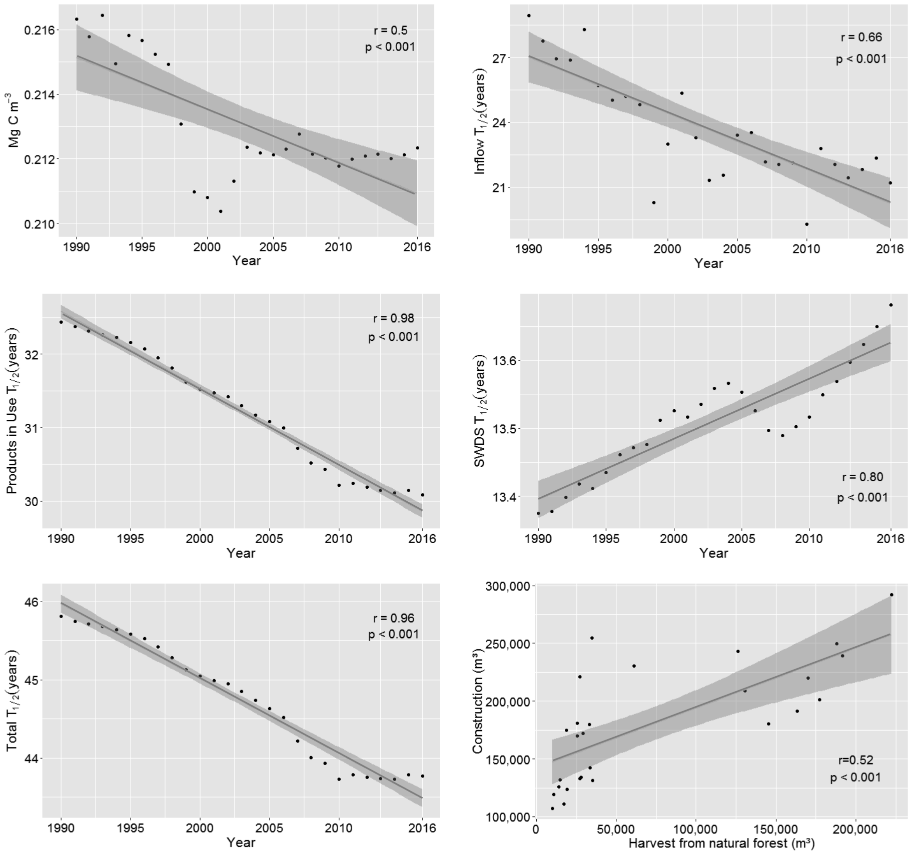

Figure 3.9. Trends in variables related to harvested wood products in Costa Rica during 1990-2016. a) Changes in the weighted average conversion factor of wood into carbon; **b)** Changes in the weighted average half-life for the inflow of wood products; c**)** Changes in the weighted average half-life for the stock of products in use; **d)** Changes in the weighted average half-life for the stock of products in SWDS; **e)** Changes in the weighted average half-life for the overall stock of wood products in Costa Rica; f) Changes in wood use for construction as a result of changes in the harvest levels of natural forests.

Discussion

Harvested wood products in Costa Rica store an increasing amount of carbon, with a net accumulation in 2016 of 112 Gg C or -412 Gg CO_2. This corresponds to 3% of the country's total emissions (Chacón et al., 2012). The contribution to total country emissions for Costa Rica is just above reports for Finland, Europe and globally (0.7 - 1%; see Johnston & Radeloff, 2019; Pilli et al., 2015; Pingoud et al., 2010). This difference is partly explained as Costa Rica's national GHG inventory does not account for emissions from managed forest lands (Chacón et al., 2012), even though these can be significant (Grassi et al., 2018).

Based on annual harvest per wood source we estimate the country's managed lands' GHG emissions for 2016 can be roughly 1,500 Gg CO_2 and the HWP contribution equal to 27% of these emissions (i.e. including N_2O and CH_4 from combustion and decomposition). This percentage is similar to those reported for other countries (Winjum et al., 1998) and confirms that excluding HWP may strongly overestimate land use related carbon emissions (Iordan et al., 2018; Pingoud et al., 2006; Skog et al., 2004). Furthermore, although this also shows there are large material losses along the production chain to produce a small fraction of HWP, these carbon losses can be recovered through forest regrowth (R. A. Houghton et al., 2015; Smith et al., 2013). Given the increasing reforestation rates and having almost completely stopped the country's deforestation, it is possible that this has been the case in Costa Rica (I. Jadin et al., 2016). If so, carbon stored in products becomes additional and managed lands may be a net carbon sink (Winjum et al., 1998).

Local activity data and a material flow analysis

Activity data on domestic harvest and product allocation collected for this inventory is the best available for Costa Rica, allowing an accurate estimate of the HWP contribution. By using local data, we reduced uncertainties attributed to inconsistencies on country reporting to international databases which may be as high as +/-50% for non-OECD countries (Grassi et al., 2018; Jasinevičius et al., 2018; Pilli et al., 2015). Also, because of the differentiation between wood sources (natural forests, plantations or agricultural lands) carbon conversion factors could be assigned per wood source instead of product categories.

The categories of semi-finished products recommended for HWP inventories include: 1) sawn wood; 2) boards and panels, and; 3) pulp and paper (Pingoud et al., 2006). However, since in Costa Rica these last two categories are not an important part of wood production, the country's domestic harvest would have been classified almost entirely as sawn wood. As a result, an inventory based on a single category of products would only require a single carbon conversion factor and half-life and would largely overestimate the stock. Tracing different pools of wood products with different half-lives is the most accurate way to estimate the HWP contribution (Aleinikovas et al., 2018; Jasinevičius et al., 2018; E. Marland & Marland, 2003; Pingoud & Wagner, 2006). Due to the use of local data, this can be interpreted as a Tier two method for reporting the HWP contribution according to the IPCC Guidelines for National Greenhouse Gas Inventories (Pingoud et al., 2006), and provides the opportunity to analyse local trends in wood sourcing and product allocation.

The effect of changes in wood source and product allocation on the carbon stock of harvested wood products

Wood sourcing experienced important changes in the country during the 26-year period studied, with an increase in domestic harvest from planted forests from 2% in 1990 to 80% in 2016. This was the result of policies to increase forest cover which intensified in the early 1990's and the highest reforestation rates occurred (Arce & Barrantes, 2004). Eventually, as this wood source became more prominent in national wood production, it removed some of the pressure on natural forests confirming an important co-benefit from planted forests (Blaser et al., 2011; Bodegom, Berg, & Meer, 2008).

Globally, planted forests are becoming the most important wood source today (Birdsey & Pan, 2015; Blaser et al., 2011; ITTO, 2015; Sessions, 2007) because of their high productivity and potential to supply the world's timber with just 10% of the forest area (Oliver & Mesznik, 2006). In some tropical regions, plantations replaced the harvesting of timber from natural forests which peaked in the 1990s and declined since then (Blaser et al., 2011; Oliver & Mesznik, 2006; Shearman, Bryan, & Laurance, 2012; Tomaselli, 2007). In Costa Rica, wood production from tropical forests dropped from 45% of domestic harvest to 9% in only four years (1997-2000).

This dramatic change coincides with the 1997 Forestry Law 7575, which among other things rendered deforestation illegal, regulated forest management, and established the Program of Payments for Environmental Services (Arroyo-Mora et al., 2014; Pagiola, 2008). Immediately after this law came into force, wood shortages due to an unofficial ban on natural forest management were experienced (Carrillo, 2001) and effectively this Law transformed forest valuation in Costa Rica, from production to service provision (Villalobos & Navarro, 2017). Besides the consequences of the sudden halt on harvesting natural forests for the country's forest industry, indirect effects on land use carbon emissions are shown in our results. Initially, planted forests could not make up for wood shortages because they were not ready for harvesting, and timber from agricultural lands had to compensate for the protection of forests (1998 -2001 in Figure 3.2). Yet, a high percentage of the harvest from agricultural lands during this period was in fact from forests where the understory had been cleared for pastures (Arce & Barrantes, 2004), a clear example of how degradation leads to deforestation and of carbon leakage.

This lasted until plantations completely took over domestic harvest in 2002 to become the most important source of wood. Among the most important species planted in Costa Rica is teak (*Tectona grandis*), which plays a determinant role in the results from this HWP inventory. It is included mostly under the category of wood exports, which is reported in the country's carbon stock when using the production approach (IPCC, 2014; Pingoud et al., 2006). Teak exports have gone from 0 to approximately 18% of total harvest in the last 6 years, and this growth has partly compensated for the potentially negative effects on carbon storage that are associated to plantations.

Wood densities reported for teak (i.e. 0.5 to 0.7 kg m^{-3}; Chave et al., 2009) are comparable with many from old growth forests, causing just a small difference between the carbon conversion factors from plantations and natural forests in our study (i.e. 0.21 and 0.22 Mg C m^{-3} respectively). Therefore, although we report a significant decrease in the weighted average carbon conversion factor of the stock, changes in the total carbon stock are hardly affected by this change. We were expecting an important effect since carbon conversion factors are known to be determinant in the estimation of stocks (Donlan et al., 2012; Skog et al., 2004) and because wood from fast growing plantations is usually less carbon dense.

This allegedly lower quality has led to the believe that wood from plantations would never be functionally equal and able to completely replace wood from old-growth forests, and that planted tropical forests are partly responsible for the growth of industries such as pulp, paper, boards and panels (Angelsen & Wertz-Kanounnikoff, 2008; Oliver & Mesznik, 2006; F E Putz & Romero, 2015; Tomaselli, 2007; Werger, 2011). However, teak is among the few planted tropical species able to provide solid wood (F E Putz & Romero, 2015), and was thus classified as a long-term product. Since half-life groupings are more important for carbon storage than the functional use of wood, instead of the reported 73 to 26% decrease in construction wood, long-term products only decreased from 75% of domestic harvest in 1990 to 49% by 2016. Thus, the increase in roundwood exports from teak plantations and furniture compensated for the decrease in construction wood, making the change in the weighted half-life of the stock less pronounced.

This decreasing trend in long-term products is significant and may not be unique to Costa Rica since product allocation between developed and developing countries tends to be similar (Winjum et al., 1998). Sawn wood production has increased globally but wood based panels showed an exponential growth and is now the most important wood product (Johnston & Radeloff, 2019). In China, the use of wood based panels is currently 20 times higher than in the 1990s and the carbon stock from sawn wood became a carbon source during 2000–2003 (L. Zhang et al., 2019). Evidence of less wood used in construction is common in many parts of the world (Oliver & Mesznik, 2006) and the expected global growth in wood production appears to be driven by a growing demand for bioenergy (Akagi et al., 2011; Birdsey & Pan, 2015; Oliver & Mesznik, 2006; Pilli et al., 2015; Poker & MacDicken, 2016). These trends in wood production and wood use may have strong implications for future carbon storage.

Changes in product allocation have led to a significant reduction in the weighted half-life of the inflow but the stock was barely affected, despite the importance generally attributed to half-lives for the estimation of carbon storage (Aleinikovas et al., 2018; Brunet-Navarro et al., 2017; Donlan et al., 2012; Jasinevičius et al., 2018; Pilli et al., 2015; Pingoud et al., 2010; Skog et al., 2004). Significant decreasing trends were observed for the stock of products in use and for the total stock, but the difference in years is minimal (2 yr). This is explained by the fact that in these stocks, it is the inherited stock rather than the inflow that determines the half-life.

In 1990, the ratio of long and short-lived products from domestic harvest (inflow) was 75/19 and the change to an almost 50/50 by 2016 resulted in a 26% decrease in the weighted half-life. Due to a fast outflow of short-term products, and with long-term products accumulating over the years in the overall amount of products in use, these ratios changed from 87/7 in 1990 to 81/16 in 2016. Therefore, there was only a 7% reduction in the weighted half-life of products in use. This shows that the inertia of the stock of products in use requires substantial and continued changes in the allocation of the inflow before changes in the half-life are enough to affect the stock.

Both the timing and the scale of the potential for climate mitigation through HWP storage are partly shown by our results from modelling the time to steady state. This analysis was included to observe the long-term effect of changes in conversion factors, half-life and harvest (i.e. current trends and potential to increase storage) and how stocks respond individually and combined. We have shown that despite significant changes in half-life or carbon conversion factor, these changes did not lead to an important shift in the time for the stock to reach a steady state. The total stock is mostly unaffected by any reduction in half-life due to the combined effect of products in use and SWDS (E. S. Marland et al., 2010; Skog et al., 2004) and only large changes in overall half-life affect the stock of products in use.

This is relevant for climate mitigation as it shows that prolonging lifespan suffers from mechanisms such as the lock-in effect that prevents the transition to cleaner technologies in other sectors (Klitkou, Bolwig, Hansen, & Wessberg, 2015). That is, previous decisions on wood production and wood use determine the stock and its half-life, and it takes time to revert the effect from previous allocation and product use (Birdsey & Pan, 2015; Pilli et al., 2015; Poker & MacDicken, 2016). Even so, there are physical limits to the increase in potential storage in HWP via increased lifespan. Based on the modelling exercise as reported here, we confirm that a ~10% increase in the stock may be achieved by a ~20% increase in half-life (Brunet-Navarro et al., 2017). However, since the half-life of the stock of products in use for Costa Rica in 2016 is 30 years, an unrealistic 100% allocation of harvest to long-term products only results in a 17% increase in half-life.

Carbon density and half-life might determine the overall size of the stock, but annual change is driven by harvest levels. Only by modifying this variable we were able to reproduce observations of pools reaching a steady state during the analysed period. Based on the rather stable trend of wood production in Costa Rica for the last 10 years, we estimate the stock of products in use will reach a steady state before 2050. Similar results have been shown as a reference scenario based on historic data for Europe and could be expected elsewhere since a relatively stable production has been the global trend during the past 30 years (Akagi et al., 2011; Birdsey & Pan, 2015; Oliver & Mesznik, 2006; Pilli et al., 2015; Poker & MacDicken, 2016).

The role of end of life carbon in wood products

In Costa Rica, wood is retired fast from the stock of products in use making SWDS responsible for most of the HWP contribution (77%), but their exclusion from national inventories (IPCC, 2014) will largely underestimate the country's stored carbon. This result is consistent with those found in similar studies (Ingerson, 2011; Skog et al., 2004; L. Zhang et al., 2019). The size of the SWDS stock is partly due to end-of-life practices where incineration and open burning of wood residues are not common, although data on open burning is generally extremely uncertain (Akagi et al., 2011). Many developing countries are still in the transitioning process to landfills as the preferred end of life management system (Ziegler-Rodriguez et al., 2019). Technologies that challenge SWDS carbon storage through incineration or combustion with energy recovery are not yet common. In addition, as the outflow of carbon from products in use continues to grow, we may expect SWDS to become an even larger carbon pool in many regions.

The increasing trend in the production of short-term commodities resulted in a doubling of the HWP contribution from SWDS between 1990 and 2016 (Figure 3.4) and in a reduction in the contribution from products in use (although not significant). Increased carbon storage in SWDS is another indirect effect from changes in product allocation, reported first in China where open burning without energy recovery reverted this effect (L. Zhang et al., 2019). Biomass combustion as an end of life management option is the main factor challenging the long-term storage of carbon in SWDS (Pilli et al., 2015). For example, in our inventory we find residues in mill dumps among the most important components to carbon storage (35%). Residues have traditionally been disposed of in mill dumps, but this management practice is changing rapidly due to environmental regulations and the increased demand for wood as an energy source. We already report an increase in bioenergy that mostly rely on wood waste and as a result, mill dumps are likely to disappear.

Other than this shift towards bioenergy, the carbon reservoir in SWDS is rather stable because 1) it is rather insensitive to changes in harvest levels; 2) all products are assumed to decompose at the same rate, and; 3) there is an inert part of the stock that accumulates (Pingoud & Wagner, 2006). While all of these play a role, the fraction of decomposable degradable organic carbon (DOC_f) has the largest effect, and there seems to be an agreement on the mechanisms leading to carbon storage under these conditions (Barlaz, 2006; De la Cruz et al., 2013; Micales & Skog, 1997; O'Dwyer, Walshe, & Byrne, 2018; F. A. Ximenes, Kathuria, Barlaz, & Cowie, 2018; F. Ximenes et al., 2015). Based on this, using the IPCC 0.5 default fraction as in this inventory, appears to overestimate losses from wood products (O'Dwyer et al., 2018).

A determinant factor in the estimation of carbon stocks in SWDS is the mass flowing from products in use into SWDS. HWP inventories assume this is the result of half-lives, but in a mass flow approach appropriately determining the actual end of life (e.g. the fraction combusted or landfilled) is much more important (Aleinikovas et al., 2018; Pilli et al., 2015; Pingoud et al., 2010). The quality of this data is known to be rather poor (Akagi et al., 2011; Bogner et al., 2008; Clavreul et al., 2012; Pingoud & Wagner, 2006; L. Zhang et al., 2019) and this is worrisome as this data gap transcends the inventory of carbon in HWP. In addition to carbon storage, SWDS are responsible for 18% of global methane emissions (Bogner et al., 2008). Improving data on allocation to SWDS is important to reduce the uncertainty and to assure that this stock continues to be accounted for in the carbon balance of wood products.

Harvested wood products within climate mitigation strategies

Despite this discussion, carbon stocks have been growing in Costa Rica and globally and will continue to be relevant for the carbon budget of many countries. Increasing this storage requires the continued growth of harvest levels and measures should be considered on the supply and demand side of the forest product chain (Suter et al., 2016). On the supply side, if incentives for forest management and wood products are included within REDD+ strategies of tropical countries, wood production systems can be strengthen to assure the sustainability of the resource (Sasaki et al., 2016, 2012). By doing so, changes on the demand side can be expected given that historical bad practices are largely responsible for a negative perception of wood use (Blaser et al., 2011). Unfortunately, lag times due to the need to modify culture and institutions will also be experienced.

Besides carbon storage, the substitution of non-renewable materials through wood products is perhaps the main contribution to reduce GHG emissions and here, physical limits are not a constraint (Werner et al., 2010). Attribution of these benefits to the forest sector can be improved through monitoring, reporting and verification systems as long as forest management and wood use are included within climate mitigation strategies (Butarbutar et al., 2016; Ellison et al., 2013; Khun & Sasaki, 2014; Werner et al., 2010). So far, climate mitigation opportunities have been missed by not making this connection clear. For example, the observed trends in wood production in Costa Rica such as disincentives on forest management and changes in product allocation had an important cumulative effect on national emissions. If we consider an average displacement factor of 2.1 Mg C saved for every Mg C in wood products (Sathre & O'Connor, 2010), a persistent allocation of construction wood over this period (i.e. assuming the same 70% from 1990) could have avoided approximately 5080 Gg CO_2 of national emissions. This is, if most of the reduction in construction wood has taken place at the expense of other substitute materials such as concrete, metal or plastic. The links between wood product use and substitution effects need to be measured and highlighted clearly if wood use is expected to be part of a future bio-economy.

Conclusions

The contribution to climate mitigation through carbon storage in HWP in Costa Rica is significant. If a complete land use carbon balance is performed that includes harvesting losses and forest regrowth, the stored carbon in 2016 may still be additional and could potentially offset 3% of the countries GHG emissions. Opportunities to increase this storage mainly rely in the country´s possibilities to increase harvest rates. This could be feasible given that because of policies to protect natural forests, the country now harvests less than 1% of productive natural forests annually. Increasing this harvest could potentially revert natural forests from being a source of carbon.

Forest plantations represent 1.5% of the country´s total area but play a major role supplying most of the current domestic harvest and in 2016 were responsible for all the HWP contribution. Plantation wood can deliver commodities that serve both long and short-term uses, but the latter have predominated. As a result, important changes in the way wood has been allocated took place during the 26-year analysed period and several direct and indirect effects from these changes on the carbon stock have been observed.

Among the direct effects from changes in wood sourcing and product allocation are significant changes in the carbon content and half-life of carbon stocks in HWP. However, changes experienced by the stock have not been significant. At least in the stock of products in use, this is mainly because of an increasing harvest rate that dominated changes in the stock and because of the long-term characteristics of the inherited stock. The other direct effect from changes in wood sourcing and product allocation has been a steady increase in the stock of carbon in solid waste disposal sites. Changes in this stock have nearly doubled during the period analysed and this is entirely attributable to an increasing part of domestic harvest allocated to short-term products.

In Costa Rica, if the stock of carbon in SWDS is excluded from harvested wood product inventories, the contribution will largely be underestimated. This stock may continue to grow due to the short lifespan of the current product allocation and because, like many other developing countries, landfilling is becoming the preferred end of life management system. The main process challenging the long-term storage of carbon in this stock are decisions over the type of end of life. That is, incineration or the use of biomass as an energy source. Other than this, there is also a general agreement that the data used to approximate the end of life of products is largely uncertain; but since poor accounting of the flow of products (biogenic or not) has impacts beyond the scope of HWP inventories, collective efforts should be taken to correct this lack of data.

Physical limits characterize climate mitigation through increased carbon storage in wood products. There may be limitations to increase harvest in many regions, and; significantly prolonging the lifetime of products or the overall stock is constrained by the inertia of the system and hence the effect may take a significant amount of time to show. Even then, there are physical limits to the amount of carbon that can be contained in products in use or SWDS. The size of the stock from products in use largely depends on the amount and type of wood products that are used, and this inevitably requires demand side measures to incentivize wood consumption.

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Supplementary information

Inherited stocks (1900 – 1990)

$$\mathrm{V}_{pjt} = \mathrm{V}_{1990} \times \mathrm{e}^{[\mathrm{U}*(\mathrm{t}-1990)]} \tag{1}$$

Where:
$\mathrm{V_t}$ = annual production of product *p* from source *j* for year t; Mg C yr^{-1}.
t = year
V_{1990} = annual production of product *p* from source *j* for 1990; Mg C yr^{-1}.
U = rate of change in industrial roundwood consumption for the region that includes the reporting county between 1900 and 1961 (0,022), yr^{-1}.(Pingoud et al., 2006)

Carbon in wood products

$$C_{pjt} = V_{pjt} \times WD_j \times CF_j \tag{2}$$

Where:
C_{pjt} = carbon content in product *p* from source *j* for year *t*; Mg C yr^{-1}.
V_{pjt} = annual production of product *p* from source *j* for year *t*; m^{-3}.
WD_j = wood density for source *j*, g cm^{-3}.
CF_j = carbon fraction for source *j*, fraction.

Carbon stock and annual stock changes from products in use

$$IU_{pj}(i+1) = IU_{pj}(i) \times e^{-k_p} + Inflow_{pj}(i) \times \left[\frac{\left(1-e^{-k_p}\right)}{k_p}\right] \tag{3}$$

Where:
i = year
$IU_{pj}(i+1)$ = carbon stock from product *p* and source *j* in use at the end of year *i*; Mg C.
$IU_{pj}(i)$ = carbon stock from product *p* and source *j* in use at the beginning of year *i*; Mg C.
k_p = first order decay constant for product *p*, ($k_p = {\ln 2}/{HL_p}$) where HL is half-life of product *p*, yr^{-1}.
$Inflow_{pj}(i)$ = inflow of product *p* from source *j* during year *i*; Mg C yr^{-1}.

$$\Delta IU_{pj}(i) = IU_{pj}(i+1) - IU_{pj}(i) \quad (4)$$

$\Delta IU_{pj}(i)$ = changes in the carbon stock of product p and source j during year i; Mg C yr^{-1}.

Carbon stock and annual stock changes from products in solid waste disposal sites (SWDS)

$$IU_{pj\,EoL}(i+1) = IU_{pj}(i) \times \left(1 - e^{-k_p}\right) + \left[1 - \left(\frac{\left(1 - e^{-k_p}\right)}{k_p}\right)\right] \times Inflow_{pj}(i) \quad (5)$$

Where:
$IU_{pj\,EoL}(i+1)$ = carbon from product in use p and source j that is retired from service to the EoL during year $i+1$; Mg C yr^{-1}.
$IU_{pj}(i)$ = carbon stock from product p and source j in use at the beginning of year i; Mg C.
k_p = first order decay constant for product p, ($k_p = {}^{\ln 2}/_{HL_p}$) where HL is half-life of product p, yr^{-1}.
$Inflow_{pj}(i)$ = inflow of product p from source j during year i; Mg C yr^{-1}.

$$\begin{aligned}&\mathrm{DDOC}_{pj}(\mathrm{i}+1) \\ &= \left(\mathrm{DDOC}_{pj}\,(\mathrm{i}) \times \mathrm{e}^{-\mathrm{k_p}}\right) \\ &+ \left[\left(\mathrm{IU}_{pj\,EoL\,Inflow}\,(\mathrm{i}) \times \mathrm{SWDS}_f\right) \times \mathrm{DOC}_f \times \mathrm{e}^{-\mathrm{k_p}}\right]\end{aligned} \quad (6)$$

Where:
$\mathrm{DDOC}_{pj}\,(\mathrm{i}+1)$ = Decomposable Degradable Organic Carbon from product p and source j in SWDS during year $i+1$; Mg C.
$\mathrm{DDOC}_{pj}\,(i)$ = Decomposable Degradable Organic Carbon from product p and source j in SWDS at the beginning of year i; Mg C.
$\mathrm{IU}_{pj\,EoL\,Inflow}\,(\mathrm{i})$ = flow of product p from source j of products in use to end of life during year i; Mg C yr^{-1}.
SWDS_f = fraction of product p disposed of in SWDS, fraction.
DOC_f = degradable organic carbon that decomposes in SWDS, fraction.
k_p = first order decay constant for product p in SWDS, yr^{-1}.

$$\mathrm{DOCa}_{pj}(i+1) = \mathrm{DOCa}_{pj}\,(i) + \left(\mathrm{IU}_{pj\,EoL\,Inflow}\,(i) \times SWDS_f \times (1 - DOC_f)\right) \quad (7)$$

Where:

$\mathrm{DOCa}_{pj}\,(i+1) =$ Degradable Organic Carbon from product p and source j that accumulates in SWDS during year *i+1*; Mg C.
$\mathrm{DOCa}_{pj}\,(i) =$ Degradable Organic Carbon from product p and source j that accumulates in SWDS at the beginning of year *i*; Mg C.
$\mathrm{IU}_{pj\,EoL\,Inflow}\,(\mathrm{i}) =$ flow of product p and source j from products in use to end of life during year *i*; Mg C yr^{-1}.
$SWDS_f =$ fraction of product p disposed of in SWDS, fraction.
$DOC_f =$ degradable organic carbon that decomposes in SWDS, fraction.

$$SWDS_{pj}(i+1) = \left(\mathrm{DDOC}_{pj}\,(i+1) + \mathrm{DOCa}_{pj}\,(i+1)\right) \quad (8)$$

Where:
$SWDS_{pj}\,(i+1) =$ carbon stock from product p and source j in SWDS during year *i+1*; Mg C.
$\mathrm{DDOC}_{pj}\,(i+1) =$ Decomposable Degradable Organic Carbon from product p and source j in SWDS during year *i+1*; Mg C.
$\mathrm{DOCa}_{pj}\,(i+1) =$ Degradable Organic Carbon from product p and source j that accumulates in SWDS during year *i+1*; Mg C.

$$\Delta SWDS_{pj}(\mathrm{i}) = SWDS_{pj}(i+1) - SWDS_{pj}(i). \quad (9)$$

Where:
$\Delta SWDS_{pj} =$ changes in the carbon stock of product p and source j in SWDS during year *i*; Mg C yr^{-1}.
$SWDS_{pj}(i+1) =$ carbon stock from product p and source j in SWDS during year *i+1*; Mg C.
$SWDS_{pj}(i) =$ carbon stock from product p and source j in SWDS during year *i*; Mg C.

Carbon stock and annual stock changes from wood milling and transformation residues in solid waste disposal sites (SWDS)

$$\mathrm{DDOCr}_{pj}(\mathrm{i}+1) = \left(\mathrm{DDOCr}_{pj}\,(\mathrm{i}) \times \mathrm{e}^{-\mathrm{k_p}}\right) + \left(\mathrm{R}_{pj\,Inflow}\,(\mathrm{i}) \times \mathrm{DOC_f} \times \mathrm{e}^{-\mathrm{k_p}}\right) \quad (10)$$

Where:
$\mathrm{DDOCr}_{pj}(i+1) =$ Decomposable Degradable Organic Carbon from wood milling and transformation residues of product p and source j in SWDS during year *i+1*; Mg C.
$\mathrm{DDOCr}_{pj}\,(i) =$ Decomposable Degradable Organic Carbon from wood milling and transformation residues of product p and source j in SWDS at the beginning of year *i*; Mg C.

$\mathrm{R}_{pj\,Inflow}\,(i)$ = flow of wood milling and transformation residues from product p and source j into SWDS during year i; Mg C.
DOC_f = degradable organic carbon that decomposes in SWDS, fraction.
k_p = first order decay constant for product p in SWDS, yr^{-1}.

$$DOCar_{pj}(i+1) = DOCar_{pj}\,(i) + \left(\mathrm{R}_{pj\,Inflow}\,(i) \times (1 - DOC_f)\right) \quad (11)$$

Where:
$DOCar_{pj}\,(i+1)$ = Degradable Organic Carbon from wood milling and transformation residues of product p and source j that accumulates in SWDS during year $i+1$; Mg C.
$DOCar_{pj}\,(i)$ = Degradable Organic Carbon wood milling and transformation residues of product p and source j that accumulates in SWDS at the beginning of year i; Mg C.
$\mathrm{R}_{pj\,Inflow}\,(i)$ = flow of wood milling and transformation residues from product p and source j into SWDS during year i; Mg C.
DOC_f = degradable organic carbon that decomposes in SWDS, fraction.

$$SWDSr_{pj}(i+1) = \left(\mathrm{DDOCr}_{pj}(i+1) + DOCar_{pj}\,(i+1)\right) \quad (12)$$

Where:
$SWDS\mathrm{r}_{pj}(i+1)$ = carbon stock from wood milling and transformation residues of product p and source j in SWDS during year $i+1$; Mg C.
$\mathrm{DDOCr}_{pj}(i+1)$ = carbon stock from Decomposable Degradable Organic Carbon from wood milling and transformation residues of product p and source j in SWDS during year $i+1$; Mg C.
$DOCar_{pj}\,(i+1)$ = carbon stock from Degradable Organic Carbon of wood milling and transformation residues of product p and source j that accumulates in SWDS during year $i+1$; Mg C.

$$\Delta SWDSr_{pj}(i) = SWDSr_{pj}(i+1) - SWDSr_{pj}(i) \quad (13)$$

Where:
$\Delta SWDSr_{pj}(i)$ = changes in the carbon stock of wood milling and transformation residues of product p and source j in SWDS during year i; Mg C yr^{-1}.
$SWDSr_{pj}(i+1)$ = carbon stock from wood milling and transformation residues of product p and source j in SWDS during year i, Mg C.
$SWDSr_{pj}(i)$ = carbon stock from wood milling and transformation residues of product p and source j in SWDS at the beginning of year i; Mg C.

Harvested wood product contribution

$$HWP\ Contribution = -{}^{44}/_{12} \times \left(\Delta \mathrm{C}_{HWP\,IU\,pj} + \Delta \mathrm{C}_{HWP\,SWDS\,pj}\right) \times 10^{-6} \quad (14)$$

Where:
$HWP\ Contribution$ = harvested wood product contribution to net CO_2 emissions from AFOLU under the production approach during year i; Gg CO_2 yr^{-1}.
$\Delta C_{HWP\ IU\ pj}$ = changes in the carbon stock of product in use p and source j during the year i; Gg C yr^{-1}.
$\Delta C_{HWP\ SWDS\ pj}$ = changes in the carbon stock of product p and source j in solid waste disposal sites during the year i; Gg C yr^{-1}.

Carbon released to the atmosphere

$$EoL\ Combustion_{pj} = \mathrm{IU}_{pj\ EoL\ Inflow}\,(i) \times \left(1 - SWDS_f\right) \tag{15}$$

Where:

$EoL\ Combustion_{pj}$ = carbon in product p from source j where the end of life during year i is combustion; Mg C yr^{-1}.
$\mathrm{IU}_{pj\ EoL\ Inflow}\,(i)$ = flow of product p and source j from products in use to end of life during year i; Mg C yr^{-1}.
$SWDS_f$ = fraction of product p disposed of in SWDS, fraction.

$$EoL\ Decomposition_{pj} = \left(SWDS_{pj}(i) \times \left(1 - e^{-k_p}\right)\right) + \left(\mathrm{IU}_{pj\ EoL\ Inflow}\,(i) \times SWDS_f \times DOC_f\right) \times \left(1 - e^{-k_p}\right) \tag{16}$$

Where:
$EoL\ Decomposition_{pj}$ = carbon releases due to biomass decomposition in SWDS during year $i+1$; Mg C yr^{-1}.
$SWDS_{pj}(i)$ = carbon stock from product p and source j in SWDS at the beginning of year i; Mg C yr^{-1}.
$\mathrm{IU}_{pj\ EoL\ Inflow}\,(i)$ = flow of product p and source j from products in use to end of life during year i; Mg C yr^{-1}.
$SWDS_f$ = fraction of product p disposed of in SWDS, fraction.
DOC_f = degradable organic carbon that decomposes in SWDS, fraction.
k_p = first order decay constant for product p in SWDS, yr^{-1}.

$$\begin{aligned} EoLr\ Decomposition_{pj} &= \left(SWDSr_{pj}(i) \times \left(1 - e^{-k_p}\right)\right) \\ &+ \left(\mathrm{R}_{pj\ Inflow}\,(\mathrm{i}) \times DOC_f \times \left(1 - e^{-k_p}\right)\right) \end{aligned} \tag{17}$$

Where:

$EoLr\ Decomposition_{pj}$ = carbon releases due to biomass decomposition from wood milling and transformation residues of product p and source j during year $i+1$; Mg C yr^{-1}.
$SWDSr_{pj}(i)$ = carbon stock from wood milling and transformation residues of product p and source j in SWDS at the beginning of year i; Mg C yr^{-1}.
$R_{pj\ Inflow}$ (i) = flow of wood milling and transformation residues from product p and source j into SWDS during year i; Mg C yr^{-1}.
DOC_f = degradable organic carbon that decomposes in SWDS, fraction.
k_p = first order decay constant for product p in SWDS, yr^{-1}.

Carbon released from combustion of wood residues during transformation processes.

$$EoLr\ Combustion_{pj}\ (i) = CComb_{pj} \tag{18}$$

Where:
$EoLr\ combustion_{pj}$ (i) = carbon released through combustion of wood milling and transformation residues of product p and source j; Mg C yr^{-1}.
$CComb_{pj}$ = Carbono total destinado a la combustión para un producto p de fuente j.

$$\uparrow C_{HWP\ pj}(i) = EoL\ Combustion_{pj}(i) + EoL\ Decomposition_{pj}\ (i) + EoLr\ Decomposition_{pj}(i) + EoLr\ Combustion_{pj}\ (i) \tag{19}$$

Where:
$\uparrow C_{HWP\ pj}(i)$ = carbon released from product p and source j during year i; Mg C yr^{-1}.
$EoL\ Combustion_{pj}$ = carbon in product p and source j where the end of life is combustion; Mg C yr^{-1}.
$EoL\ Decomposition_{pj}$ = carbon released due to biomass decomposition in SWDS during year $i+1$; Mg C yr^{-1}.
$EoLr\ Decomposition_{pj}$ = carbon released due to biomass decomposition from wood milling and transformation residues; Mg C yr^{-1}.
$EoLr\ Combustion_{pj}$ (i) = carbon released through combustion of wood milling and transformation residues; Mg C yr^{-1}.

$$CH_4Emissions = \left[\sum_X \left(DDOC_{pj} \times SWDS_{fx} \times MCF_x \times F \times {}^{16}/_{12}\right) - R_T\right] \times (1 - OX_T) \tag{20}$$

Where;
$DDOC_{pj}$ = Decomposable Degradable Organic Carbon from product p and source j in SWDS during year i; Mg C yr^{-1}.

$SWDS_{fx} =$ fraction of SWDS sent to managed anaerobic sites, unmanaged shallow unmanaged deep, and uncategorized sites in Costa Rica; fraction.
MCF_x = CH_4 correction factor for aerobic decomposition in the year of deposition for the different types of SWDS (Pipatti et al., 2006); fraction.
F = fraction of CH_4, by volume, in generated landfill gas (0.47; (Chacón et al., 2012)); fraction.
${}^{16}/_{12}$ = molecular weight ratio CH_4/C; ratio.
R_T = recovered CH_4 in year T (i.e. 0.23 (Chacón et al., 2012)), fraction.
OX_T = oxidation factor in year T, (i.e. 0 (Chacón et al., 2012)), fraction.

Time to steady state

$$T_{SS} = \frac{1}{k}\log\left(\frac{J}{J - kS_1}\right) \tag{21}$$

Where;
T_{SS}= Time to steady state; yr
k =decay rate constant for each stock (i.e. products in use, SWDS, and overall); fraction
S = Stock (i.e. products in use, SWDS, and overall); Gg C
J = Harvest; Gg C

Chapter 4

The lifecycle climate impact of wood from natural tropical forests in Costa Rica

Federico E. Alice, Frits Mohren, Pieter Zuidema

Abstract

In the tropics, natural forests have traditionally been the main source of materials and energy, but this is changing due to concerns over the sustainability of harvesting practices. Logging is the main cause of tropical forest degradation and protecting these forests may seem as the best strategy for the sustained provision of ecosystem services. However, unintended environmental consequences may arise due to the continued demand for forest products or from the substitution of these products with other materials. Furthermore, financially compensating forest owners and forest dwelling communities for the protection of forests has not yet been properly addressed. In such cases, forest management can be an option to reconcile environmental and socioeconomic needs so long as it is done in a sustainable way. In this paper, we study the management of natural tropical forests for wood production in Costa Rica using a lifecycle approach to evaluate its potential climate impact. We include all possible sources of emissions, biogenic and fossil, along the most important phases of the lifecycle of wood. Biogenic carbon was included through a dynamic approach based on lifetime analysis, for which we defined a temporal boundary that is equal to a rotation period (i.e. 15 years in Costa Rica). We also evaluate the effect of extending this temporal boundary to 100 years. Activity data for most processes was collected through surveys, and the review of forest management plans and national reports. We use one hectare as the main functional unit but since this unit is the result of multiple products and co-products harvested from an average hectare of tropical forest, these were all included as independent functional units (m^3). Fossil sources were responsible for only 6% of total emissions, with harvesting operations and transportation contributing the most. The damage to the forest during harvesting was the main source of emissions. Carbon storage has an important effect on the balance, as well as end of life emissions from short-term products (i.e. formwork) and the combustion or decomposition of co-products. We found large probabilities for net negative emissions (i.e. net sequestration) for most functional units due to forest regrowth. Based on Monte Carlo simulations, we estimated that at a per hectare level this balance is -4.41 Mg CO_2-eq ha^{-1} over a 15-year period, with a 95% CI from -13.12 to 10.96. Once the temporal system boundaries are extended to 100 years, the balance for all functional units other than mid and long-term products results in net emissions. Although this boundary is suitable for products and co-products, at a per hectare level one rotation period is very likely a fair representation of the system as a longer timeframe ignores future harvesting cycles. From a climate perspective, it appears that harvesting tropical

forests in Costa Rica can contribute to climate mitigation with uncertainties mostly due to biogenic carbon from logging damage. Harvesting wood in tropical regions can have a potentially carbon neutral balance if reduced impact logging techniques and wood allocations that favour long-term products are prioritized.

Introduction

Timber from natural tropical forests has traditionally been an important source of wood materials and fuel, accounting for approximately 40% of the annual global harvest (Oliver & Mesznik, 2006; Poker & MacDicken, 2016). Wood harvest from natural forests peaked in the 1990s and has since declined to be substituted by other wood sources and materials (Blaser et al., 2011; Oliver & Mesznik, 2006; Shearman et al., 2012; Tomaselli, 2007). This decline can be partly explained by the historic overexploitation of forests (Shearman et al., 2012), the subsequent implementation of restrictions to protect forests and by changes in consumption patterns (Blaser et al., 2011; Murphy, 2004). These causes are related as protection and consumption patterns are partly driven by the global perception that logging in the tropics is unsustainable, illegal or both.

Tropical forests are a key component of the carbon cycle but are clearly threatened by deforestation and degradation, which contribute 12% to annual global CO_2 emissions (Harris et al., 2012). Although degradation due to logging and deforestation are usually seen as part of the same problem there is a weak link between them (Poker & MacDicken, 2016). However, the relationship between logging and forest degradation can be strong in many tropical regions, with carbon losses of similar magnitude as those from deforestation (Ellis et al., 2019; Francis E. Putz et al., 2008). It is therefore recommended that logging should be avoided in these forests as their protection will have a higher climate mitigation benefit (Vogtländer, Velden, & Lugt, 2013).

Perhaps this is a good example of the precautionary principle, but there are several drawbacks from this strategy. First, the need for wood products might compromise any strategy aimed at protecting standing forests exclusively (Parker et al., 2014). Then, there are forest owners that rely on forest resources as part of their livelihoods who should be compensated in case access is denied (Köhl et al., 2015). Finally, as has been occurring, reducing the use of forest products can be compensated by non-wooden products or fossil energy sources (Werner et al., 2010). Because the environmental and socioeconomic impacts from this strategy can be high, the possibility to improve how forest resources are used through sustainable forest management should not be dismissed (Ellison et al., 2013).

Reducing carbon emissions from the production of goods will require that materials from non-renewable resources are replaced by the sustainable use of biological resources (Wohlfahrt et al., 2019). In contrast to mineral or fossil materials and energy sources, the cycle of biogenic carbon emissions and sequestration associated to wood extraction may occur within human timescales, making this material potentially renewable (Breton et al., 2018). Wood is a low carbon material because the energy, chemical and other inputs required for the production and use of wood products is lower than those from other materials. When compared, an average of 2.1 Mg of C emissions can be avoided for every Mg C in wood products in use (Sathre & O'Connor, 2010). Therefore, increased use of wood can contribute to climate mitigation because wood is renewable, has low lifecycle greenhouse gas emissions and can potentially substitute other more carbon intensive materials (Intergovernmental Panel on Climate Change, 2014).

A commonly used technique to evaluate and compare the environmental performance of goods and services is the lifecycle assessment (LCA), now common in the evaluation of the climate impact of forestry and forest products (Cole, 1999; Helin et al., 2013; Klein et al., 2015). LCA standardizes the quantification of inputs and outputs through a lifecycle inventory conducted within defined spatial and temporal boundaries and is used to assess environmental impacts (Sandin et al., 2016). For climate change, the midpoint indicator for the impact is the global warming potential (GWP) and the most commonly used time horizon is 100 years (GWP100)(Brandão et al., 2013).

LCA of wood products from tropical forests are scarcely available and the environmental impact of tropical forestry is largely unquantified (Lippke et al., 2011; Murphy, 2004; Numazawa et al., 2017; Piponiot et al., 2016). This lack of information has contributed to the common association of tropical timber production with deforestation, degradation and illegality, and is possibly responsible for the reduced share of tropical wood in global timber markets (Blaser et al., 2011; Murphy, 2004). Additionally, the few LCA of tropical timbers that are available have excluded biogenic carbon from their analysis based on the carbon neutrality assumption (Adu & Eshun, 2014; Eshun et al., 2010, 2011; Jankowsky et al., 2015; Ramasamy et al., 2015; Ratnasingam et al., 2015; Rinawati et al., 2018). Assuming logging and product use have a zero net balance of carbon emitted, sequestered and stored is common in LCA as it simplifies the system and its analysis (De la Cruz et al., 2013; Liu et al., 2017). However, by doing so, the real climate impact from tropical forest management remains unknown.

The simplification of the system by the carbon neutrality assumption has been criticized given that the exclusion of this source of emissions or storage may underestimate the climate impact of forestry (Cardellini et al., 2018; Helin et al., 2013; Johnson, 2009). For instance: there are large biomass losses during harvesting, biogenic carbon will not be released immediately during use as it will remain stored in products in use or will accumulate in solid waste disposal sites (SWDS). Delaying these emissions through carbon storage can be considered equal to avoiding an emission depending on storage time (Breton et al., 2018). Furthermore, carbon will be sequestered through forest regrowth and it is the net balance from all these processes that determines the climate impact. The need to estimate this balance using a combination of forest, wood and lifecycle models, is one of the main reasons why biogenic carbon has been excluded (Newell & Vos, 2012). The other reasons have mostly been related to a lack of methodological agreement.

Even if neutrality was the result from this balance, this assumption has been criticized due to the different timing of when these processes occur. Since sequestration follows emissions and it will take time before these are fully recovered through forest regrowth, atmospheric concentrations will initially increase, and a warming effect can still be attributed to the production system (Levasseur et al., 2013). This discussion has centred mostly around bioenergy systems for which this lag time is important, although it affects biogenic carbon in general. Different metrics (e.g. discounting future emissions) or dynamic characterization methods can be used to address the dynamics of emissions and sequestration on atmospheric greenhouse gas (GHG) concentrations but there is no commonly accepted method (Brandão et al., 2013; Breton et al., 2018; Fouquet et al., 2015; Levasseur, Lesage, Margni, Deschěnes, & Samson, 2010; Røyne, Peñaloza, Sandin, Berlin, & Svanström, 2016). However, an important difference between wood products and bioenergy is that emissions from wood products occur gradually as products are retired and decompose. In fact, the time when emissions occur may be overestimated by the common assumption that wood products decay exponentially, increasing the probabilities for simultaneous regrowth and emissions (Helin et al., 2013; E. S. Marland et al., 2010). In such cases, a dynamic characterization method can even lead to a 10% reduction in global warming (Breton et al., 2018).

Wood products are a special case of lifecycle assessment due to the differentiated treatment of biogenic and fossil carbon which requires the consideration of time (Bergman et al., 2014). Among the basic principles of an LCA is the definition of boundaries, which importantly determine results but are based on a subjective decision (Newell & Vos, 2012; Reap et al., 2008). For forest management and forest product systems, a single moment (i.e. a static) and a whole rotation approach have been recommended as potential temporal boundaries, but the whole rotation is favoured as the flows of carbon along the lifecycle of wood can be measured dynamically (Klein et al., 2015). One drawback is that there will usually be differences between the chosen temporal boundary for the analysis and the actual lifecycle of products (Brandão et al., 2013). Due to the slow decay of wood, emissions will continue beyond this time frame and excluding them underestimates the climate impact of products. However, there seems to be some agreement on 100 years being a compromise between science and policy when accounting for carbon storage in wood products (Breton et al., 2018; Reid Miner, 2006).

In this study, we conduct an LCA for wood production in natural forests in Costa Rica to quantify the climate impact from logging tropical forests and using wood. We include biogenic carbon emissions together with all processes leading to GHG emissions due to harvesting, manufacturing, transport, use, and disposal of timber (Figure 4.1). Emissions and storage of biogenic carbon were quantified dynamically through a material flow and lifetime analysis that allows tracing products, co-products and wood residues independently. This analysis together with data on logging damage at the forest level is meant to partly address the question of the sustainability of harvesting practices in the tropics. We chose one rotation period as the temporal boundary but as this rotation is short in Costa Rica (15 years), we modelled the effect from extending this boundary to 100 years. Uncertainty was investigated through Monte Carlo simulations.

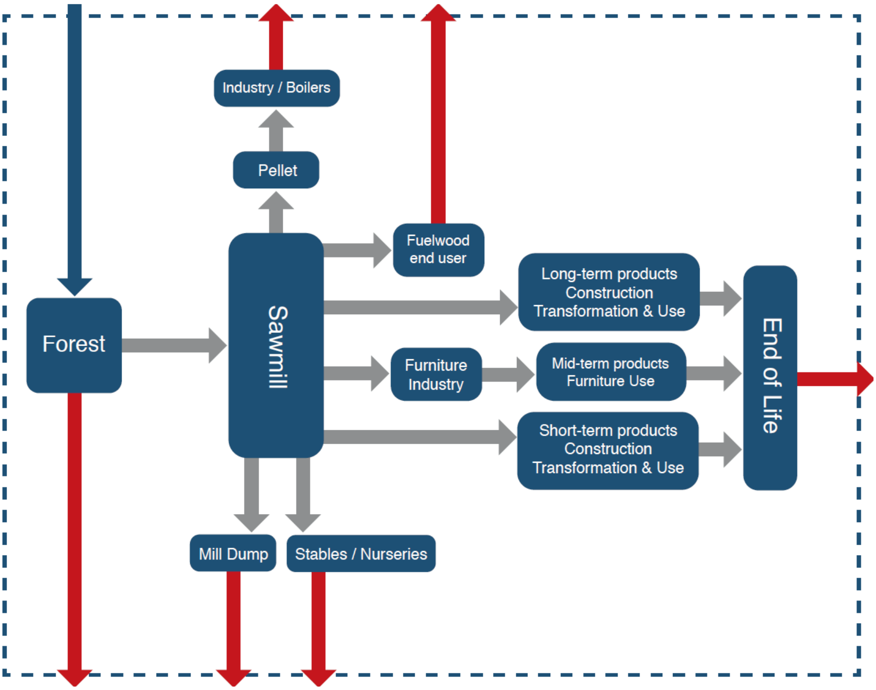

Figure 4.1. Spatial boundaries for the lifecycle of timber harvested from natural tropical forests in Costa Rica. Grey arrows indicate flows within the system, CO_2-eq emissions and uptake are represented by the red and blue arrows respectively.

Methods

Goal and scope

We developed a lifecycle assessment (LCA) to determine the potential climate impact from harvesting one hectare of tropical wet and moist forests in the Northern Caribbean region of Costa Rica. This region is responsible for 83% of timber harvest from natural forests in the country (MINAE, 2011, 2012, 2013). This LCA includes all biogenic and fossil sources of GHGs in a cradle to grave analysis. To do so, we used a lifecycle inventory (LCI) for inputs and outputs, and a material flow analysis with first order decay functions to trace biogenic carbon until the end of life (EoL).

Apart from the regional limits, the physical boundaries include all processes of harvesting, sawmilling, secondary transformation, use, and end of life (Figure 4.1). Temporal boundaries to estimate the climate impact (i.e. emissions and sequestration) were defined as one rotation period (Klein et al., 2015). In Costa Rica, the legal minimum rotation period is 15 years (MINAE, 2002; MINAET, 2009). In our analysis, climate impact is understood as the greenhouse gas (GHG) balance from all processes and is expressed as CO_2-eq. These were estimated using global warming potentials (GWP) for a 100-year time horizon as this is the most common in LCA (Brandão et al., 2013; Cherubini, Peters, Berntsen, Strømman, & Hertwich, 2011; Huijbregts et al., 2017). We also show general results for a 20-year time horizon to test the effect of short-lived GHGs in the overall GHG balance. Although biogenic and fossil carbon emissions are reported separately (Helin et al., 2013), we provide a combined final result per functional unit.

Based on our methods and the goal of this study, we provide results for several functional units, i.e.: a) one hectare of natural tropical forests used for wood production in Costa Rica; b) one (multifunctional) cubic meter of wood from natural tropical forests in Costa Rica, and; c) one cubic meter of wood from tropical forests used for each of the following products and co-products: formwork, construction, furniture, fuelwood and pellets. We performed analyses per product since the potential climate impact from the total harvest of one hectare of forests is determined by the combined impact from different products and co-products. Our analysis for a multifunctional cubic meter of wood allows comparisons of our results with those from other studies.

We excluded changes in the carbon stock in forest soils due to logging operations, as these are known to be small (Pearson et al., 2014). We excluded the cascade use of wood via recycling as this is not important in the country (Solera, 2014). Due to the complexity of data gathering and their low contribution to total climate impact (Medeiros, Tavares, Rapôso, & Kiperstok, 2017; Suter et al., 2016), emissions from inputs other than wood, and downstream emissions from use and maintenance were approximated based on assumptions. Other biophysical effects such as changes in albedo or evapotranspiration were also excluded, but in tropical forests these can be low because selective logging removes only a small number of stems per hectare (R. A. Houghton et al., 2015).

We assumed a static reference scenario (i.e. no additional regrowth in the "no harvest" scenario) and did not include emissions from deforestation. The first assumption is consistent with the cut-off period for regrowth used, and the second was chosen because in Costa Rica harvesting does not lead to deforestation (Arroyo-Mora et al., 2014). There is evidence in the country that harvesting may even lead to positive carbon leakage, as limiting harvest from forests has caused deforestation and the uncontrolled exploitation of trees outside forests (Chapter 3). Furthermore, it is likely that a dynamic reference will have a small effect on results (Buchholz et al., 2016).

Lifecycle inventory

Data collection from the wood products industry to develop the lifecycle inventory (LCI) was primarily conducted through the review of forest management plans and use of questionnaires to a sample of harvesting operators, sawmills, transportation, construction and furniture companies. Data for end of life processes was taken from the National GHG Inventory (Chacón, Jiménez, Montenegro, Sassa, & Kendal Blanco, 2014). Results from each section reflect common practices in wood production in the country.

Management plans

All forest management plans in the Northern Caribbean region of Costa Rica between 2010 – 2016 were reviewed to retrieve basic information on wood production and forest damage due to logging. A total of 107 forest management plans and their corresponding audit reports were available from the regional offices of the Ministry of Environment and Energy (MINAE).

Harvesting operations

Activity data for harvesting operations was collected on-site from a sample of 20 loggers. In Costa Rica, harvesting is performed by a logger owning basic equipment (i.e. bulldozer, skidder, tractor, truck, etc.) who subcontracts teams of 4 – 6 people (Arroyo-Mora et al., 2014). Although reported separately, activity data on transportation to the sawmill was also gathered through this questionnaire. Inputs (i.e. fossil fuels, motor oil and hydraulic fluids) were obtained per management plan and converted per hectare using the estimated regional average area (ha). Main activities were; building logging infrastructure (primary and secondary roads, and logging decks), tree felling, on-site log transport (e.g. skidding), bucking and loading. We also included the transportation of inputs and people during harvest (i.e. logistics) and pre-harvest forest inventories, permit requests and site visits by foresters (i.e. planning).

Sawmill

We sampled 20 sawmills within the study region that process timber from natural forests (although four were located at ≈100 km distance; Chapter 2). Based on the questionnaire, we analysed the process flow including all activities, machinery, inputs and outputs. Main activities were: 1) those taking place in the log yard such as loading, unloading, bucking and storing roundwood; 2) head sawing; 3) resawing: 4) edging; 5) moulding and planing, and; 6) sharpening of band saw, saws, and blades. Inputs and outputs were gathered for each of these processes to develop the mass flow and carbon footprint for milling activities.

Machinery used and inputs (i.e. fuels and electricity) were gathered at each major step in the process to avoid allocation procedures (i.e. assigning responsibility for the environmental impact of a multi-functional process amongst its functions or products) (Ford-Robertson, 2003; Reap et al., 2008). For electricity, we used running time and the energy demand of each machine (Medeiros et al., 2017) together with monthly electricity consumption. Fuels, oil and hydraulic fluids were mostly used by wheel loaders, agricultural tractors, forklifts and chainsaws and were reported monthly.

The mass flow was developed based on monthly roundwood consumption and the amount and type of products and co-products produced (Table SI-4.1). Residues were estimated using the difference between roundwood inputs and sawn wood outputs (i.e. products and co-products). Products are the intended output of the process, while co-products are by-products with a market value. Wood residues are those for which no use was reported and are disposed of in mill dumps.

Products were grouped into different categories based on the end use and the processes involved in their transformation (Table SI-4.2). Long-term products (*LTP*) are those used in construction, short-term products (*STP*) are boards and laths used as formwork in the construction sector, and mid-term product (*MTP*) are edges and off-cuts that all sawmills reported selling to the furniture industry. This is essentially a co-product but was classified as a product given the mid-term characteristics of its service life.

First order decay functions and half-lives based on product type (i.e. *STP, MTP* or *LTP*) were used to trace carbon until the end of life of products. As described above, the temporal limit was a 15-year rotation and carbon emissions and sequestration were estimated for this period. For co-products such as pellets (*Plt*), fuelwood (*Fw*), or sawdust and shavings used in stables, stalls or nurseries (*SSN*), it was assumed they combust or decompose on year zero.

Secondary processes

The transformation of timber into final products was limited to three categories: construction (*LTP*), furniture (*MTP*) and wood pellets (*Plt*). Data from 10 samples (5 for construction and 5 for furniture) were collected, aggregated and the average assumed for both, i.e. mid and long-term products. Besides data on electricity consumption, fossil fuels were only reported by the few which also deliver products to the end user. Biomass losses during this transformation of wood in construction or into furniture (10.7%) were used in the mass flow analysis for biogenic carbon. Because it is a small amount, we assumed residues flow directly into the EoL where these will be combusted or landfilled.

The relatively small sample is due to the small scale and heterogeneous nature of wood use from tropical forests in Costa Rica. The average annual harvest during 2010 – 2015 was 7300 m^3, which is less than 1% of national production (Barrantes & Ugalde, 2017). Industrial processes dependent on this wood source are non-existent. There is one pellet plant using mill residues, a relatively small sized furniture industry (Aragón-Garita, Moya, Bond, Valaert, & Tomazello Filho, 2016; Serrano & Moya, 2011; Solera, 2014) and the construction of single family houses that occasionally use this timber (Camacho, 2015; Santamaría, 2015; Serrano & Moya, 2011; Werger, 2011).

Transportation

Inputs, distances and volume transported from the sawmill to the next stages of the lifecycle were collected through a total of 27 questionnaires; 6 for the distribution of edges and off-cuts used in furniture; 5 for sawdust and shavings used in stables and nurseries; 5 for slabs and bark for pelleting and fuelwood; and 11 for sawn wood (i.e. short and long-term products).

Transportation to end users can be very diverse, especially considering intermediation in the wood market. From our questionnaires to transformation industries only four out of 10 reported transportation. This resulted in an average distance of 35 km (n= 4), which is almost half the distance reported from the sawmill to the next phases (x = 74; n= 27). Therefore, we used half of this fuel consumption to estimate transport from transformation to intermediation, and half of this (i.e. 17 km) to estimate transport from intermediation to the end user and from end use to EoL. Short-term products require no transformation, so we only included the transportation to an intermediary and from there to the end use. For pellets we used half of fuel consumption due to the distance between the plant and industries and an additional 0.1 fraction from this amount to account for the transportation of ash to EoL. The transportation included in all functional units is described in Table SI-4.3.

Lifecycle impact assessment

In the lifecycle impact assessment (LCIA), all results were expressed as CO_2 equivalent using global warming potential (GWP) for a 100 year time horizon as the midpoint characterization factors (i.e. 34 and 36 for CH_4 from biogenic and fossil sources respectively, and 298 for N_2O; kg CO_2-eq/ kg GHG) (Huijbregts et al., 2017). The net balance was estimated using a GWP for a 20-year time horizon (84 and 85 for CH_4 from biogenic and fossil sources respectively, and 264 for N_2O) to observe the effect of increasing the relative importance of short-lived gases in our results. The climate impact from biogenic and fossil carbon were estimated separately but combined in a single result to present the net GHG balance. All results are reported per hectare and per 15-year period (Mg CO_2-eq ha^{-1} 15 yr^{-1}); for a multifunctional m^3 or for a m^3 of any of the specific products and co-products (Mg CO_2-eq m^{-3} 15 yr^{-1}). To test the effect of time on the potential climate impact, we also include results for a scenario where the temporal system boundaries are prolonged to 100 years.

Biogenic carbon

The temporal boundary for this analysis was chosen because emissions from all biogenic carbon (i.e. logging residues, wood products, co-products and residues during the transformation stages) were estimated using a mass flow and lifetime analysis. That is, we estimate annual emissions and indirectly account for storage in the forest, products, mill dump and SWDS during this timeframe. In the forest, biomass decomposition from logging damage was estimated using a decay rate of 0.1 yr^{-1} (Houghton et al., 2000) and all carbon was assumed to oxidize as CO_2. Forest regrowth was estimated as a function of logging damage and total harvest (Rutishauser et al., 2015).

Using the collected data, we estimated the fraction of roundwood that becomes short, medium and long-term products and modelled the use phase until the EoL using 2, 25 and 35-year half-lives. To estimate emissions from fuelwood and pellets during their use phase, we assumed oxidation factors of 0.85 and 0.95 respectively and emission factors chosen based on the specific conditions of their combustion (Akagi et al., 2011; Delmas, Lacaux, & Brocard, 1995; FAO, 2013). SSN was assumed to oxidize as CO_2.

Residues from slabs and bark (SB), sawdust and shavings (SS) were assumed to decompose at the mill dump. To estimate emissions, we used the first order decay model with half-lives that vary according to the type of residue (i.e. 20 years for slabs and bark and 10 years for sawdust and shavings). We assumed 0.5 as the decomposable degradable organic carbon fraction (DOC_f) in all wood products and treated mill dumps as a shallow unmanaged waste disposal site (Pipatti et al., 2006). We used 0.46 as the fraction of CH_4 generated in landfill gas (Chacón et al., 2014) and 0.4 as the methane correction factor. The remaining fraction of the oxidized carbon is emitted as CO_2.

Similar procedures were followed for products decomposing in SWDS, with adjustments to the half-life, methane correction factors (MCF) and the amount of methane recovered. Waste fractions per type of SWDS in Costa Rica are divided as follows: 0.51 goes to managed anaerobic (i.e. landfills) with a MCF=1 and the only site where a 0.23 fraction of methane is recovered; 0.09 to unmanaged shallow sites, MCF = 0.4; 0.09 to unmanaged deep, MCF = 0.8, and; 0.18 to uncategorized, MCF=0.6 (Chacón et al., 2012). A single 20 year half-life was used for all sites (Pipatti et al., 2006). The remaining fraction of waste (0.12) is combusted, and CH_4, N_2O and CO_2 emissions were estimated using a default 0.58 oxidation factor, a CH_4 emission factor of 6500 g / t MSW wet weight and 0.15 g N_2O / kg dry matter (Guendehou et al., 2006).

Fossil carbon

GHG emissions from fossil sources and electricity were estimated using activity data and emission factors for N_2O, CH_4 and CO_2 that are specific for Costa Rica (IMN, 2011, 2014, 2015, 2016, 2017, 2018). We were not able to obtain data from the pellet industry, so we assumed these emissions represent 20% of harvesting and milling emissions (R Miner, 2010). A similar assumption was used for emissions from inputs such as glues, varnish, staples and nails which were reported in very small amounts during the transformation of mid and long-term products. Since we collected data on electricity use during this phase, instead of the 20% used previously we only assumed emissions were 10% of production emissions. We included an additional 10% to account for the use of mid and long-term products and avoid underestimating lifecycle emissions as we did not collect data for this phase. Downstream emissions from short-term products (i.e. formwork) were considered negligible. Emissions from using pellets, fuelwood and sawdust in stables, stalls or nurseries were limited to those explained under biogenic carbon.

Allocation

Our approach to data collection was designed to avoid allocation procedures as far as possible but it was inevitable for some data sources and functional units. Emissions from wood decomposition in mill dumps had to be allocated to all products and co-products, although assigning the waste impacts from the manufacturing of products is not necessarily allocation (Wiloso, Heijungs, Huppes, & Fang, 2016). Based on the material flow analysis we determined timber in final products and co-products and used this fraction to estimate the impact per ha and per multifunctional m^3. This was also possible when estimating the impact from a m^3 of specific products and co-products for processes that are common to all, i.e.; logging damage, forest regrowth, harvesting operations, transport from the forest to the sawmill, basic equipment during milling (i.e. Scope 1 emissions) and the emissions from wood decomposition in mill dumps. Thus, our analysis assumes that these processes are the same for a m^3 of products or co-products and the multifunctional m^3.

Emissions from the combustion or decomposition of biomass during the use or end of life phases of products and co-products (*Plt, Fw*, *SSN*, *STP*, *MTP* and *LTP*) were assigned to each specific product when they occur. Fossil emissions from electricity use during sawmilling were assigned per product depending on the processes involved in their transformation (Table SI – 4.2). For example, only long-term products require moulding and planing and were responsible for all the impact from this sub-process. For simplification, the accompanying equipment (e.g. a sawdust extractor) used in sub-processes (i.e. head saw, re-saw and moulder/planer) were grouped and their emissions were summed.

Activity data for transportation of products along the lifecycle was collected per m^3 of product, so no allocation was needed. While different product types are mixed during transport to end users or to end of life, we assumed such transports to contain only one product type. The effect of this assumption of transport per product type was assumed to be small since less volume transported per trip would require a larger number of trips.

Uncertainty

We studied the uncertainty of the system using Monte Carlo simulations. Activity data mainly followed exponential, gamma or normal distributions, which were ultimately determined based on the Akaike information criterion (Table SI-4.4). Fractions were assigned either the beta distribution or a Dirichlet distribution that is the multivariate generalization of the beta distribution (Igos, 2018). Most emission factors were assumed constant. We performed 10 000 iterations to determine the mean and the 95% confidence interval for all results. That is, results for biogenic emissions, biogenic GHG balance, fossil emissions, fossil and biogenic emissions and the complete lifecycle GHG balance for every functional unit.

Results

Lifecycle biogenic sources of emissions

GHG emissions from biogenic sources were mainly from damage to the forest during harvest operations, which are inherent to a selective logging management system. At a per hectare level, these emissions were 14.8 Mg CO_2-eq 15 yr^{-1} or 70% of all biogenic sources (Figure 4.2; Table 4.1). Logging emissions for 1 m^3 were 1.42 Mg CO_2-eq 15 yr^{-1} and were the same for all types of products, i.e. this is the impact from harvesting 1 m^3 of wood in a tropical forest in Costa Rica regardless of the end use.

End of life (EoL) emissions were important in short-term products, with CH_4 from decomposition in SWDS making up 30% of total biogenic emissions from these products. Open burning during the end of life had only a small contribution to emissions of products and were followed by mill dump emissions. Since co-products are combusted or decompose soon after harvesting, the use phase accounted for 40% of their biogenic emissions.

During the analysed period, carbon uptake from forest regrowth was enough to offset all biogenic emissions and led to a negative GHG balance for all functional units. Mid and long-term products show lower emissions due to the combination of regrowth and carbon storage. Results per hectare and the multifunctional m^3 tend to be closer to short-term products since these were 46% of total harvest (Table SI-4.1).

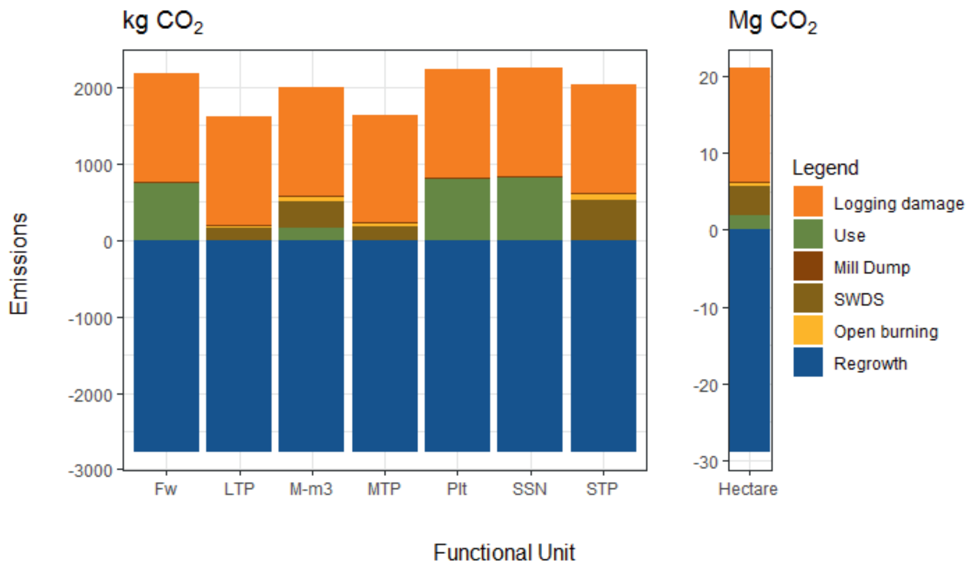

Figure 4.2. Distribution of lifecycle biogenic emissions and sequestration per functional unit. Hectare (ha), multifunctional m^3 (M-m^3), short-term products (STP), mid-term products (MTP), long-term products (LTP), pellets (Plt), fuelwood (Fw) and stables, stalls and nurseries (SSN).

Lifecycle fossil sources of emissions

Total fossil emissions per hectare and multi-functional m^3 were 1.45 Mg CO_2-eq 15 yr^{-1} and 160 kg CO_2-eq 15 yr^{-1} respectively, mainly as CO_2 from fuels (Figure 4.3; Table 4.1). The largest share of these emissions (60-80%) were due to harvesting operations and the transportation of roundwood from the forest to the sawmill. In contrast to results from the analysis of biogenic emissions, products had higher emissions than co-products due to additional transformation and transportation along the lifecycle. Sawmilling played a minor role on lifecycle emissions (6% of fossil emissions), partly because of the low emission factor from Costa Rica's electricity grid (IMN, 2018). Cradle to gate emissions were approximately 80% of fossil emissions. All transportation combined accounted for 50% of fossil emission sources.

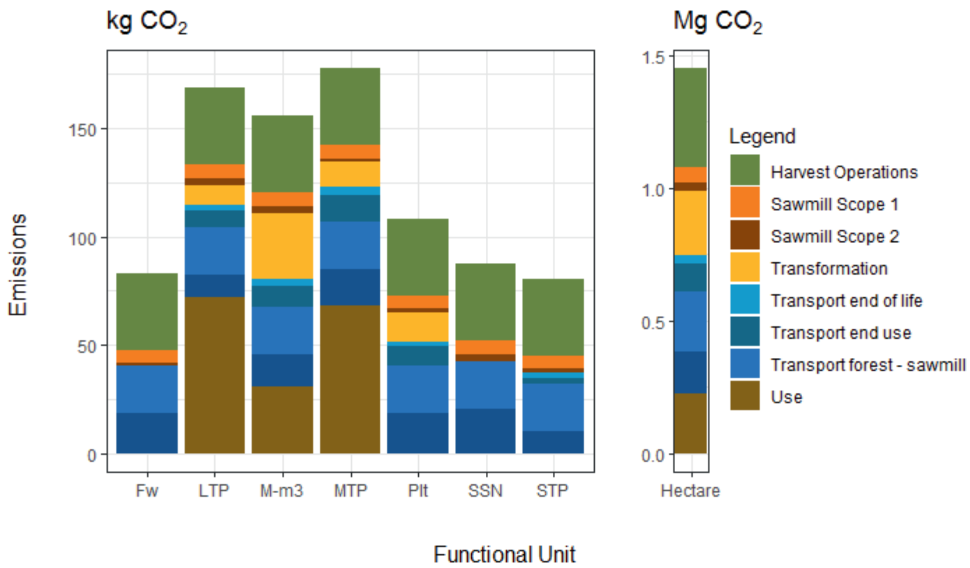

Figure 4.3. Distribution of lifecycle GHG emissions from fossil sources per functional unit. Hectare (ha), multifunctional m^3 (M-m^3), short-term products (STP), mid-term products (MTP), long-term products (LTP), pellets (Plt), fuelwood (Fw) and stables, stalls and nurseries (SSN).

Lifecycle emissions and net greenhouse gas balance

The net balance of all lifecycle GHG per hectare and multi-functional m^3 were -6.37 Mg CO_2-eq 15 yr^{-1} and -621.45 kg CO_2-eq 15 yr^{-1} respectively, mainly as CO_2 from fuels (Figure 4.3; Table 4.1). All GHG emissions from the lifecycle of wood products combined were largely dominated by biogenic sources, especially logging damage (58 - 79% depending on the functional unit; Table 4.1). In products, emissions from biomass decomposition in solid waste disposal sites were 26, 11 and 9% for short, mid and long-term products respectively. Open burning during EoL was important in short-term products but the transportation from the forest to the sawmill was more important in mid and long-term products. In co-products, the main sources of emissions were logging damage (58 -61%), combustion or decomposition during product use (34-36%) and harvesting operations and transportation which combined represented 2-6% of total emissions. Overall, GHG emissions from fossil sources accounted for only 6 to 7% of total emissions, with slightly higher percentages for mid and long-term products due to carbon storage.

Emissions from biogenic and fossil sources during the analysed period were entirely offset by forest regrowth. Since we traced all biogenic emissions temporally and assumed all fossil emissions to occur on year zero, cumulative emissions and cumulative carbon sequestration became neutral at around 3 - 4 years (i.e. with a constant 2.99 Mg CO_2 ha^{-1} yr^{-1} rate of forest regrowth). At this point, forest regrowth offsets all previous emissions, which were 40 – 50% of total emissions. From then on, annual forest regrowth offsets annual emissions and sequestered a surplus of carbon. According to the model used to estimate forest regrowth (Rutishauser et al., 2015) it will take 11 years for forest carbon to recover to initial values. Therefore, even under a different regrowth pattern the carbon payback period can be found between 4 and 11 years.

Table 4.1. Lifecycle GHG emissions (Mg CO_2-eq 15 yr^{-1}) for timber from natural forests in Costa Rica. Functional units: Hectare, multi-functional m^3 (M-m^3), short-term products (STP), mid-term products (MTP), long-term products (LTP), pellets (Plt), fuelwood (Fw) and stables, stalls and nurseries (SSN).

	Hectare	M-m^3	STP m^3	MTP m^3	LTP m^3	Plt m^3	Fw m^3	SSN m^3
Harvest Operations (F)	0.3701				0.0355			
Logging damage (B)	14.8020				1.4189			
Transport Forest – Sawmill (F)	0.2282				0.0219			
Sawmill Scope 1 (F)	0.0579				0.0061			
Sawmill Scope 2 (F)	0.0319	0.0034	0.0016	0.0016	0.0034	0.0016	0.0012	0.0034
Mill Dump (B)	0.1887				0.0181			
Transport Sawmill – Transformation (F)	0.1567	0.0150	0.0103	0.0163	0.0103	0.0186	0.0186	0.0204
Transformation (F)	0.2456	0.0305	NA	0.0113	0.0093	0.0135	NA	NA
Transport End Use (F)	0.1038	0.0099	0.0026	0.0122	0.0077	0.0093	NA	NA
Use (F)	0.2255	0.0305	NA	0.0685	0.0719	NA	NA	NA
Use (B)	1.9292	0.1586	0.0000	0.0000	0.0000	0.8035	0.7448	0.8138
Transport End of Life (F)	0.0288	0.0028	0.0026	0.0041	0.0026	0.0019	NA	NA
SWDS (B)	3.6980	0.3545	0.5312	0.1805	0.1511	NA	NA	NA
Open burning (B)	0.4550	0.0436	0.0631	0.0262	0.0215	NA	NA	NA
Total Biogenic emissions (B)	21.07	1.99	2.03	1.64	1.61	2.24	2.18	2.25
Total Fossil emissions (F)	1.45	0.16	0.08	0.18	0.17	0.11	0.08	0.09
Total Emissions	23.10	2.05	2.01	1.71	1.67	2.24	2.16	2.23
Regrowth	-28.89				-2.77			
Total Net Balance	**-6.37**	**-0.62**	**-0.66**	**-0.95**	**-0.96**	**-0.42**	**-0.50**	**-0.43**

Uncertainty and variability

The mean result and its confidence level for all sources of emissions and for each functional unit are presented in Table 4.2. Estimated mean emissions based on the average values of variables and those from simulations showed marked differences. At a per hectare level, average simulated biogenic emissions showed almost no change with respect to the sampled average, but biogenic emissions for the multifunctional m^3 increased by 75%. This increase in biogenic emissions from this functional unit determines its net balance. Since average emissions from all other functional units (i.e. 1 m^3 of products or co-products) was not as pronounced, the multifunctional m^3 seems to be highly influenced by error propagation due to the conversion of impacts into this unit. This was confirmed by the very low difference at a per hectare level. The discrepancy in average emissions between mean run (Table 4.1) and the Monte Carlo simulations (Table 4.2) is likely caused by (a combination of) parameter variability, the large number of variables included (107) and draws of improbable combinations of variables.

Fossil emissions also experienced changes in the average, but confidence intervals are still within an expected range. For example, fossil emissions from the production of a multifunctional m^3 were found between 80 and 720 kg CO_2-eq m^{-3} while biogenic emissions range from 0.9 to 9 Mg CO_2-eq m^{-3}. Fossil emissions per hectare from the simulations did not deviate much from the estimated average, while products and co-products experienced a 50% increase with respect to the mean value of the standard run. Long-term products requiring additional processes of manufacturing and transportation showed the highest difference (100%).

The uncertainty range of biogenic emissions was large compared to fossil emissions because of their magnitude (Mg instead of kg) and data variability. The net balance of all functional units included zero within the confidence interval, and in most cases this interval was biased towards negative emissions, suggesting a higher probability for net sequestration after 15 years. The simulated net balance at a per hectare level resulted in a net balance of -4.41 Mg CO_2-eq 15 yr^{-1} while the multi-functional m^3 was -80 kg CO_2-eq m^{-3} 15 yr^{-1}. The conversion of (mainly) biogenic emissions into these units tends to show the largest uncertainties due to the allocation of impact from different products and co-products.

Timing of emissions

The GHG emissions for all functional units increased when extending the system boundary to 100 years; although zero remained within the confidence intervals (Table 4.2). Due to carbon storage, the only functional units for which the balance remained negative were mid and long-term products. The small difference between these two functional units shows the 10-year difference in half-life has almost no effect on carbon storage (Figure 4.4). Slightly higher emissions from long-term products were also observed when changing the time horizon to 20 years, so a likely explanation is their higher fossil emissions. At a per hectare level, Figure 4.4 shows the mode is closer to negative emissions, but larger probabilities of extreme positive emissions determine this result. Short-term products show much higher emissions due to less storage leading to increased EoL emissions. In this case, the reduction in half-life from 35 to 5 years (from long to short-term products) has a large effect. The only additional source of long-term emissions explaining the results of all functional units are those from the decomposition of forest biomass (necromass). This is more evident in the changes observed in co-products. The 100-year net balance per hectare and multi-functional m^3 reflect the changes experienced by the individual functional units, although the larger fraction of short-term products (48%) largely influences these results.

Shortening the time horizon of the global warming potential from 100 to 20 years had a large effect on the net balance of all functional units, especially short-term products. Although methane emissions from combustion, aerobic and anaerobic decomposition are low under the reference 15-year system boundary, these were enough to increase emissions in all cases (Table 4.2; Figure 4.4). The largest changes were observed per hectare and multifunctional m^3. These are mostly driven by the change in short-term products for which methane emissions were already high. As with the 100-year boundary, the results per hectare and multifunctional m^3 are determined by small probabilities of extreme positive results (Figure 4.4).

Table 4.2. Average lifecycle GHG emissions (x) and confidence intervals for all functional units for a 15-yr system boundary and GWP100 (Mg CO_2-eq functional unit^{-1}). The net balance is shown for the reference system boundary and time horizon (15-yr and GWP100), a 100-yr system boundary and GWP100 and a 15-yr system boundary with GWP20.

		Hectare		M-m^3		STP-m^3		MTP-m^3		LTP-m^3		Plt-m^3		Fw-m^3		SSN-m^3	
Biogenic emissions	x	**20.50**		**3.49**		**2.57**		**2.42**		**2.36**		**3.20**		**3.13**		**3.18**	
	95% CI	10.7	34.1	0.9	9.3	0.6	8.1	0.6	7.4	0.6	7.5	0.8	10.1	0.8	9.5	0.8	9.8
Biogenic Balance	x	**-6.27**		**-0.29**		**-1.18**		**-1.32**		**-1.37**		**-0.54**		**-0.62**		**-0.52**	
	95% CI	-15.00	8.79	-3.66	3.34	-4.19	0.69	-4.51	0.50	-4.68	0.53	-2.40	1.50	-2.51	1.33	-2.33	1.54
Fossil emissions	x	**1.85**		**0.24**		**0.12**		**0.27**		**0.33**		**0.16**		**0.13**		**0.14**	
	95% CI	0.47	4.95	0.08	0.72	0.04	0.34	0.11	0.76	0.10	1.02	0.07	0.43	0.05	0.34	0.05	0.37
Fossil & Biogenic emissions	x	**22.24**		**3.66**		**2.74**		**2.67**		**2.69**		**3.39**		**3.25**		**3.37**	
	95% CI	12.27	36.00	1.09	9.63	0.68	8.57	0.75	7.97	0.75	7.86	0.91	9.95	0.87	9.76	0.93	10.29
Net Balance (15-yr / GWP100)	x	**-4.41**		**-0.08**		**-1.05**		**-1.06**		**-1.06**		**-0.36**		**-0.49**		**-0.38**	
	95% CI	-13.12	10.96	-3.16	3.32	-4.23	0.86	-4.15	0.79	-4.24	0.73	-2.06	1.72	-2.30	1.52	-2.09	1.71
Net Balance (100-yr / GWP100)	x	**1.90**		**0.34**		**0.86**		**-0.36**		**-0.31**		**0.10**		**0.04**		**0.10**	
	95% CI	-10.55	18.28	-1.84	3.17	-1.78	8.28	-2.42	0.58	-2.15	0.74	-0.23	0.60	-0.57	0.81	-0.52	0.92
Net Balance (15-yr / GWP20)	x	**0.28**		**0.19**		**0.06**		**-0.46**		**-0.30**		**-0.26**		**-0.32**		**-0.30**	
	95% CI	-14.03	27.31	-3.02	3.66	-4.25	7.27	-3.86	3.27	-1.98	1.95	-1.79	1.88	-1.91	1.75	-1.98	1.95

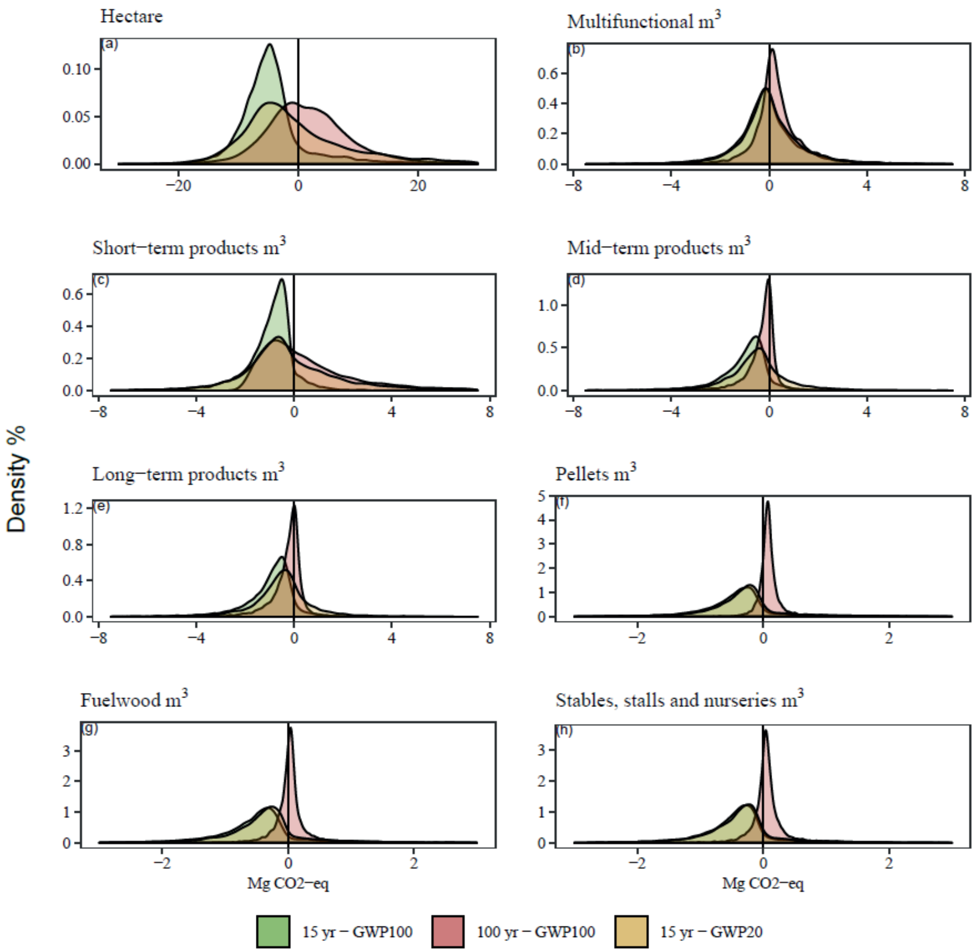

Figure 4.4. Density estimation for the Monte Carlo simulations of the net balance of all functional units under the reference system boundary and time horizon (15-yr and GWP100), a 100-year system boundary and a GWP20.

Discussion

We present the lifecycle assessment of wood harvested from natural forests in Costa Rica which included the main processes and all major sources of emissions along the lifecycle of all products from an average hectare of forests in the country. We relied mostly on empiric data that reflect wood production from forests, explaining between 77 - 100% of the estimated potential climate impact (i.e. 77% for construction and 100% for fuelwood and SSN). To avoid underestimations of product use emissions, we used assumptions taken from the literature to complete processes for which we were not able to collect activity data (Cole, 1999; Klein et al., 2015; R Miner, 2010; Wolf et al., 2015).

This is the first lifecycle assessment for tropical timbers to include a dynamic lifecycle inventory to account for biogenic carbon emissions and storage. This was partly done because of the criticisms to the carbon neutrality assumption (Cherubini, Guest, & Strømman, 2013; Helin et al., 2013; Liu et al., 2017; Suter et al., 2016), but most importantly because the loss of carbon stocks due to degradation is at the core of the discussion of the climate impact of tropical forestry. We report results for biogenic and fossil emissions separately to allow comparisons with other studies (Helin et al., 2013) but aggregate results and provide an uncertainty analysis for all functional units. We observe a large potential for the system to result in carbon neutral products and co-products. However, due to parameter variability and uncertainty, the confidence intervals were often large, leading to similar probabilities of positive (net emissions) and negative (net sequestration) balances.

Fossil sources of emissions in the lifecycle of wood from tropical forests

Similar to results from other regions where fossil and biogenic carbon have been examined (Lippke et al., 2011; Pingoud et al., 2010), emissions from fossil sources were found to be small (only 6% of total emissions). While manufacturing tends to dominate in other regions and other wood sources (Bergman et al., 2014; R Miner, 2010; Parigiani, Desai, Mariki, & Miner, 2011; Puettmann & Bergman, 2010), in this study harvesting operations were responsible for most fossil emissions due to a combination of topography and climate. In Costa Rica, low technology transformation processes (e.g. air drying) and an electricity grid that uses renewable energy (Chacón et al., 2014; Serrano & Moya, 2011) reduce the potential climate impact from manufacturing.

Transportation is consistently reported among the most important sources of emissions (Lippke et al., 2011; Pingoud et al., 2010), especially those from the forest to the sawmill. This was also the case in Costa Rica, despite the short average distance (74 km on average) compared to the commonly assumed 100 km (Merry et al., 2009; R Miner, 2010) and extremes such as 500 km reported for timber transport in Ghana (Eshun et al., 2010). Once all transport emissions were considered, these were similar to those reported in other studies (i.e. 49.6 kg CO_2-eq m^{-3} in this study vs 49 kg CO_2-eq m^{-3} in Brazil)(Medeiros et al., 2017).

When comparing fossil emissions with those found in the literature, our results tend to be higher due to data variability. For example, we report average emissions of 270 kg CO_2-eq m^{-3} (95% CI 110 – 760) for mid-term products used in furniture, while 122 kg CO_2-eq m^{-3} were reported for furniture in Brazil (Medeiros et al., 2017). For comparable products (i.e. dried air lumber from tropical forests in Ghana), emissions equal 110 kg CO_2-eq m^{-3} (Eshun et al., 2010, 2011) and are considerably lower than our mean result for a multifunctional m^3 (240 kg CO_2-eq m^{-3}; 95% CI 80 – 720). A likely explanation of this difference is that the study in Ghana used a cradle to gate system boundary. If we would apply such boundaries, our result would be approximately the same.

There are difficulties to compare results for similar functional units or under tropical conditions due to differences in system boundaries and functional units (Klein et al., 2015; Newell & Vos, 2012; Wolf et al., 2015). Comparisons are further challenged by the limited availability of such studies in the tropics (Murphy, 2004; Numazawa et al., 2017; Piponiot et al., 2016). To put our results in context, we reviewed literature and found several studies for tropical timber (Table 4.3). Studies that lacked transparency or were not exclusively for timber extracted from natural tropical forests were excluded. In our comparison, we only included those studies for which results for a specific functional unit were clearly stated or could be derived, and for which boundaries were clearly defined. For two studies included in Table 4.3, we made assumptions on the time horizon of the lifecycle impact assessment as it was not stated in the report (Gan & Massijaya, 2014; Rinawati et al., 2018).

For comparison, all results included in Table 4.3 are expressed per 1 m^3 of a certain product. Only one of the studies included biogenic emissions, although these were limited to deforestation instead of lifecycle carbon emissions. The rest follow the assumption of carbon neutrality and define system boundaries as cradle to gate or gate to gate. In these cases, results can be comparable with those from the cradle to gate GHG emissions from fossil sources presented here (Table 4.1) and our temporal system boundary becomes irrelevant. Overall, reported emissions per m^3 are higher than our cradle to gate results which range between 120 kg CO_2-eq m^{-3} for short-term products (formwork) to 330 kg CO_2-eq m^{-3} for long-term products used in construction (Table 4.2). Our multi-functional m^3 is also comparable when using a 100-yr boundary for biogenic emissions or a 20-year horizon.

Table 4.3. Lifecycle results for timber from natural tropical forests found in the literature.

Country	Functional unit	System boundary	Biogenic carbon	Time Horizon	kg CO_2-eq m^{-3}	95% CI	Reference
Costa Rica	1 multi-functional m^3	Cradle-to-grave	15-yr	20	190	[-3020 to 3660]	This study
				100	-80	[-3160 to 3320]	
			100-yr		340	[-1840 to 3170]	
Malaysia / Indonesia	1 m^3 of plywood	Cradle-to-gate	Neutral	100	446	NA	Gan and Massijaya, 2014
Malaysia	1 m^3 of rough green sawn timber	Gate-to-gate	Neutral	100	499 / 696	NA	Ratnasingam, Ramasamy, Toong, Senin, 2015; Ramasamy et al., 2015
Indonesia[1]	Unfinished chair	Gate-to-gate	Neutral	100	180	NA	Rinawati, Sari, and Prayodha, 2018
Ghana[2]	1 multi-functional m^3	Cradle-to-gate	Land use change	100	577	NA	Eshun, Potting, and Leemans 2010, 2011
Ghana	1 m^3 sawn wood	Cradle-to-gate	Neutral	100	253	NA	Adu and Eshun 2014
Brazil	1 m^3 decking boards	Cradle-to-gate	Neutral	100	73.2 – 77.3	NA	Jankowsky, Galina, and Andrade 2015

[1] Estimated based on the reported 9.01 kg CO_2-eq per 0.05 m^3

[2] Results are an extrapolation of annual harvest at a national level so emissions per m^3 are estimates based on the reported 745k tons CO_2 yr^{-1} and an annual 1.29 million m^3 produced. Products correspond to air & kiln-dried lumber, plywood, veneer & furniture.

Biogenic emissions in the lifecycle of wood from tropical forests

Our LCA results show that biogenic carbon dominates GHG emissions, mostly due to logging damage. This contribution is large, even though logging damage in our studied system is low compared to that found in other countries, i.e. 14 vs a minimum of 24 Mg CO_2 ha^{-1} reported by (Pearson et al., 2014). Tree felling is consistently reported as the main component of this damage (Ellis et al., 2019; Pearson et al., 2014; Piponiot et al., 2016). After the 15-year period, there is approximately 22% of carbon in necromass stored in the forest, so not all damage has been accounted for as an emission. Most importantly, almost 60% of products are stored in mill dumps (2.5%), products in use (19%), and solid waste disposal sites (42%). These are well-known technospheric reservoirs, and their exclusion from forest carbon balances based on the assumption of committed emissions largely overestimates emissions (Barlaz, 2006; De la Cruz et al., 2013; Iordan et al., 2018; Wang, Padgett, Powell, & Barlaz, 2013; F. A. Ximenes et al., 2018; F. Ximenes et al., 2015).

In terms of emissions, decomposition in SWDS in our study represented 9 – 26% of total emissions (17% in the average functional units; ha and M-m^3). From these emissions, approximately 80% are methane which has a larger impact on the balance and is able to offset large part of the contribution from storage (Lippke et al., 2010). These results vary depending on the type of products (i.e. their lifespan), the allocation into EoL processes (open burning or SWDS) and assumptions over their anaerobic decomposition (e.g. the fraction that decomposes). The results for EoL emissions from biogenic carbon vary depending on each functional unit but are evident in short-term products. This effect in short-term products was confirmed by using a 20-year time horizon, where these products go from storing 1 Mg of CO_2 per m^3 harvested to emit 60 kg CO_2-eq m^{-3}.

Most importantly, since this functional unit represents a large fraction of harvest (48%), it has an important effect on average units, i.e. hectare or multi-functional m^3. These units closely follow changes experienced by short-term products when modelling a longer system boundary or time horizon. This trend partly explains the results per hectare or multi-functional m^3 under these scenarios. Results for short-term products show large uncertainties, and although negative emissions were the mode (Figure 4.4), the averages for a 100-yr boundary or 20-year time horizon show positive emissions. These emissions are likely due to extreme values in the fraction of wood that becomes short-term products. Fractions were included in the model using a Dirichlet distribution to control for their correlations, but the variability of the data can still lead to extreme values and improbable combinations. Therefore, although the ranges are indicative of probable conditions, these should not be over interpreted (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010a).

Similarly, the variability from logging damage and harvest levels importantly contributed to the wide range of confidence intervals. Specially results per hectare or per multi-functional m^3 are affected by a combination of parameter and model uncertainty (Huijbregts, 1998; Lo et al., 2005). These units are estimated based on the results from all individual processes from all individual functional units, and each of these carry their own uncertainties. Mathematically, this means that certain variables are used more than once, and their uncertainties propagate during simulations. As we also use the fraction of each product and co-product that compose total harvest, this represents an additional source of uncertainty and an example of problems due to allocation. Furthermore, forest regrowth as included in this study is estimated based on harvest and logging damage (Rutishauser et al., 2015), making this effect even larger. Because these parameters are known to be positively correlated (Martin et al., 2015), reducing this uncertainty is possible but would require a different modelling approach. Although these uncertainties are important, they do not challenge the current result, but rather need to be considered during interpretation.

Land use emissions and carbon balances are generally characterized by large uncertainties as they reflect a wide range of forest types, regrowth patterns, harvesting practices, etc. (Baccini et al., 2012; G. M. J. Mohren et al., 2012). For example, the reported range for logging damage in the tropics (6.8–50.7 Mg C ha^{-1} or 24 – 185 Mg CO_2 ha^{-1})(Pearson et al., 2014) clearly demonstrates this potential for probable extreme values and provides some context to interpret the uncertainties from biogenic carbon emissions in this LCA. The range reported in the literature corresponds to extreme examples of forest management practices across the tropics but shows that our estimated uncertainty range for biogenic emissions (10.7 to 34.1 Mg CO_2 ha^{-1}) does represent the low impact logging of timber in Costa Rica.

The scale of the impact from biogenic emissions and its uncertainty show the relevance of its inclusion in the LCA of products and in the understanding of the lifecycle climate impact from logging tropical forests (Côté et al., 2002). If we simply compare the result for the 100-year system boundary against the other results presented in Table 4.3, it may seem that there is no added value from including biogenic emissions. Our results are merely close to the average from all other results that exclude this source of emissions. However, the uncertainty range broadens this interpretation providing insights into the potential limits associated to the product system (Clavreul et al., 2012). From a carbon management perspective, the opportunities and risks of lower or higher emissions become apparent and can be taken into consideration in the decision-making process.

System boundaries

We include the scenario where the system boundary is extended to 100 years because there seems to be some agreement on 100 years being a compromise between science and policy when accounting for carbon storage in wood products (Breton et al., 2018; Reid Miner, 2006). Additionally, it is probable that a whole rotation approach was suggested having temperate forests in mind, where rotations are closer to this timeframe (Klein et al., 2015). Once we model our system for this period, we found positive net emissions for all functional units except long and mid-term products, confirming the relevance of carbon storage in wood products. A 6 and 14% of carbon contained in mid and log-term products is still in use after a century.

Although carbon storage reduces considerably for products in use, 51% of wood that has been sent to SWDS remains stored for a much longer timeframe (Barlaz, 2006; De la Cruz et al., 2013; Wang et al., 2013; F. A. Ximenes et al., 2018; F. Ximenes et al., 2015). Forest carbon (necromass) has been lost entirely, along with carbon in short-term products. In these products, methane from anaerobic decomposition has a large impact and due to its higher GWP, it is able to offset large part of the contribution from storage (Lippke et al., 2010). Despite having the lowest fossil lifecycle emissions (120 kg CO_2-eq m^{-3}), short-term products have the largest lifecycle climate impact under the 100-year temporal boundary, i.e. 860 kg CO_2-eq m^{-3} (95% CI between -1780 to 8280).

Results for a 100-year system boundary confirms that a very short rotation like those practiced in Costa Rica will underestimate emissions, as wood products will decay beyond this boundary (Reid Miner, 2006). However, at a per hectare level this timeframe ignores subsequent rotations that will trigger new cycles of damage, technospheric storage, and sequestration. The 100-year average for this functional unit therefore overestimates emissions and the net balance is probably closer to the average from one rotation. For this functional unit, a long-term average from all possible rotations occurring in a 100-yr period is probably a better representation of the climate impact. In the case of products and co-products, the attribution of future sequestration or storage is not possible (Plevin et al., 2014b, 2014a) and these functional units are better represented by this longer timeframe.

Conclusions

Results for the lifecycle climate impact of forest management in Costa Rica show large probabilities of a system that is close to GHG neutral. At a per hectare level and under the conditions found in the country, our result differs largely from carbon emissions estimated for forest degradation. The inclusion of carbon sequestration and carbon storage in wood products in the lifecycle GHG balance of forest management resulted in negative emissions (i.e., net sequestration), with approximately 4 Mg of CO_2-eq ha^{-1} stored in the system during the 15-year period post-harvest. We tested the effect of extending the system boundary to 100 years, and although it then shows positive emissions, zero was still contained within the confidence interval. We interpret the wide confidence intervals as being caused by improbable combinations of product allocations. Additionally, we argue that emissions are overestimated under a 100-year period since this boundary ignores future logging cycles occurring during this timeframe.

At the product level, there is some agreement that a 100-year system boundary can be used to estimate the benefits from carbon storage, in which case the benefits become small. Only mid and long-term products show signs of any contribution from carbon storage, confirming that: 1) wood allocations that favour long-term products should be preferred and, 2) for product half-lives to influence carbon storage, changes larger than the 10-year difference between these two products are required. Further supporting this recommendation, end-of-life methane emissions from short-term products can cause the system to become a source of CO_2 emissions.

Products and co-products show positive emissions under the extended boundary, but the GHG balance for these units is within the average results reported for other tropical countries. Since other tropical results exclude biogenic carbon and use different system boundaries, the added value of this assessment relies not in the result itself but the interpretation of its variability. The fluxes of biogenic carbon are determinant in the lifecycle GHG balance of logging, and as expected in any land-use GHG inventory, these are the main source of uncertainty. Although opportunities for carbon management exist along the processes of manufacturing and use, reduced impact logging remains as the main climate mitigation opportunity for forest management in the tropics.

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Supplementary information

Table SI – 4.1. Fractions of wood products, co-products and residues harvested from natural tropical forests in Costa Rica.

	Fraction	**Substitute product**
Short-term products (Formwork)	**0.46**	Aluminium
Mid-term products (Furniture)	**0.12**	Aluminium, Polyvinyl chloride
Transformation Residues	0.01	
Long-term products (Construction)	**0.17**	
Door and window frames (Laths & scantlings)	0.07	Aluminium
Flooring	0.01	Ceramic tiles
Ceilings	0.02	Polyvinyl chloride
Mouldings	0.03	Polyvinyl chloride
Structural (e.g. beams)	0.01	Galvanized iron
Other	0.01	
Transformation Residues	0.02	
Pellets	**0.06**	Bunker fuels or diesel
Slabs & bark	0.06	
Sawdust	0.003	
Shavings	0.001	
Fuelwood	**0.06**	
Slabs & bark	0.05	
Sawdust	0.003	
Shavings	0.007	
Stables, stalls and nurseries	**0.09**	
Sawdust	0.07	
Shavings	0.02	
Residues	**0.04**	
Slabs & bark	0.03	
Sawdust	0.003	
Shavings	0.003	

Table SI-4.2. Processes and machinery involved in the milling of products and co-products process.

		PRODUCTS		CO-PRODUCTS		
Process	Machinery	STP	LTP[1]	MTP[2]	Plt & Fw[3]	SSN[4]
Log Yard	• Wheel loader • Chain saw • Forklift • Agricultural tractor	✓	✓	✓	✓	✓
Head sawing	• Head saw • Saw Carriage • Electric chain hoist • Sawdust extractor	✓	✓	✓	✓	✓
Re-sawing	• Re-saw • Sawdust extractor	✓	✓	✓		✓
Edging	• Edger	✓	✓	✓		✓
Planing & Moulding	• Planer/ Moulder • Sawdust extractor		✓			✓
Sharpening	• Sharpener	✓	✓	✓	✓	✓

[1]e.g. flooring, board box, beams, mouldings, scantlings, etc., a total of 8 secondary products were identified, plus an additional category that groups "other" products.

[2] All edges & off-cuts were reported as sold to the furniture industry.

[3] Pellets and fuelwood are mostly slabs & bark generated during sawing and re-sawing.

[4] Sawdust and shavings are mostly used in stalls, stables and nurseries and are produced at all stages of milling.

Table SI-4.3. Transportation involved in the lifecycle of specific products.

	Sawmill	Transformation	Intermediation	End use	End of Life
Formwork	✓		✓	✓	✓
Construction	✓	✓	✓	✓	✓
Furniture	✓	✓	✓	✓	✓
Pellet	✓	✓		✓	✓
Fuelwood	✓			✓	
SSN	✓			✓	

Table SI-4.4. Variables and distributions used in the Monte Carlo simulations.

	Variable	Unit	Distribution	
1	Transformation biomass losses	fraction	*B* (3.18, 26.58)	Beta distribution
	Managed			
2	Anaerobic	fraction	*Dir* (0.51)	
3	Unmanaged Shallow	fraction	*Dir* (0.09)	
4	Unmanaged Deep	fraction	*Dir* (0.09)	
5	Uncategorized	fraction	*Dir* (0.19)	
6	Open Burning	fraction	*Dir* (0.12)	
7	Sawdust residues	fraction	*Dir* (0.003)	
8	Shavings residues	fraction	*Dir* (0.003)	
9	Slabs and bark residues	fraction	*Dir* (0.02)	
10	Sawdust for pellets	fraction	*Dir* (0.002)	Dirichlet multivariate generalization of the beta distribution
11	Shavings for pellets	fraction	*Dir* (0.0001)	
12	Slabs and bark for pellets	fraction	*Dir* (0.0611)	
13	Sawdust for fuelwood	fraction	*Dir* (0.003)	
14	Shavings for fuelwood	fraction	*Dir* (0.0007)	
15	Slabs and bark for fuelwood	fraction	*Dir* (0.05)	
16	Sawdust for stables, stalls and nurseries	fraction	*Dir* (0.06)	
17	Shavings for stables, stalls and nurseries	fraction	*Dir* (0.02)	
18	Short-term products	fraction	*Dir* (0.48)	
19	Mid-term products	fraction	*Dir* (0.15)	
20	Long-term products	fraction	*Dir* (0.17)	
21	Bulldozer diesel	liters	exp (0.002)	
22	Bulldozer oil	liters	exp (0.03)	
23	Bulldozer hydraulic fluid	liters	exp (0.03)	
24	Agricultural tractor diesel	liters	exp (0.003)	
25	Agricultural tractor oil	liters	exp (0.20)	
26	Agricultural tractor hydraulic fluid	liters	exp (0.35)	
27	Chainsaw gasoline	liters	exp (0.002)	Exponential distribution
28	Chainsaw oil	liters	exp (0.04)	
29	Chainsaw chain oil	liters	exp (0.004)	
30	Wheel loader diesel	liters	exp (0.002)	
31	Wheel loader oil	liters	exp (0.12)	
32	Wheel loader hydraulic fluid	liters	exp (0.09)	
33	Chainsaw sawmill gasoline	liters	exp (0.02)	

34	Chainsaw sawmill oil	liters	exp (0.58)	
35	Chainsaw sawmill chain oil	liters	exp (0.08)	
36	Sawmill agricultural tractor diesel	liters	exp (0.17)	
37	Sawmill agricultural tractor oil	liters	exp (0.11)	
38	Headsaw	kwh	exp (0.0003)	
39	Sawcarriage	kwh	exp (0.003)	
40	Electric chain hoist	kwh	exp (0.04)	
41	Headsaw dust extractor	kwh	exp (0.002)	
42	Resaw	kwh	exp (0.001)	
43	Ressaw dust extractor	kwh	exp (0.004)	
44	Edger	kwh	exp (0.007)	
45	Moulder and planer	kwh	exp (0.001)	
46	Moulder and planer dust extractor	kwh	exp (0.003)	
47	Sharpener	kwh	exp (0.014)	
48	Sawn wood transport	unit	exp (0.25)	
49	Sawdust and shavings transport	unit	exp (0.13)	
50	Harvest Deadwood	m^{-3}	exp(0.99)	
51	Harvest Standing	m^{-3}	Γ(3.15, 0.28)	
52	Slabs and bark transport	unit	Γ (6.54, 0.92)	
53	Gaps	Mg C	Γ(7.56, 2.1)	
54	Logging Decks	Mg C	Γ(0.45, 5.73)	
55	Primary Roads	Mg C	Γ(0.23, 2.19)	
56	Secondary Roads	Mg C	Γ(1.34, 1.64)	
57	Skid Trails	Mg C	Γ(1.2, 1.79)	
58	Carbon Stock in Forest Biomass	Mg C	Γ(10.72, 0.11)	
59	Logistics and Planning	liters	Γ(1.72, 0.001)	Gamma distribution
60	Logging truck diesel	liters	Γ(6.0, 0.72)	
61	Forklift diesel	liters	Γ(0.24, 0.01)	
62	Forklift oil	liters	Γ(0.19, 0.18)	
63	Forklift hydraulic fluid	liters	Γ(0.24, 0.24)	
64	Sawmill agricultural tractor hydraulic fluid	liters	Γ(0.11, 2.81)	
65	Edges and off-cuts transport	unit	Γ(19.11, 3.06)	
66	Electricity secondary processing	kwh	Γ(0.67, 0.01)	

67	Forest necromass decay rate	fraction	N(0.1, 0.0002^2)	Normal distribution
68	Sawdust and shavings mill dump decay rate	fraction	N(0.1, 0.12^2)	
69	Landfill decay rate	fraction	N(0.1, 0.006^2)	
70	Short-term products retirement rate	fraction	N(0.1, 0.48^2)	
71	Mid-term products retirement rate	fraction	N(0.1, 0.003^2)	
72	Long-term products retirement rate	fraction	N(0.1, 0.002^2)	
73	Regrowth rate	unit	N(0.1, 0.02^2)	
74	Decomposable Degradable Organic Carbon	fraction	N(0.1, 0.01^2)	
75	Wood specific density	kg m^{-3}	N(0.1, 0.0004^2)	
76	Carbon Fraction	fraction	N(0.1, 0.0001^2)	
77	Logging truck oil	liters	N(0.1, 0.0021^2)	
78	CH_4 correction factor managed anaerobic	fraction	1.00	Assumed constants
80	CH_4 correction factor unmanaged Shallow	fraction	0.40	
79	CH_4 correction factor unmanaged Deep	fraction	0.80	
81	CH_4 correction factor uncategorized	fraction	0.60	
82	CH_4 in generated landfill gas	fraction	0.47	
83	Recovered CH_4	fraction	0.23	
84	Open burning CO_2 oxidation factor	fraction	0.58	
85	Open burning CH_4 emission factor	fraction	0.007	
86	Open burning N_2O emission factor	fraction	0.0000002	
87	Pellets and fuelwood N_2O emission factor	fraction	0.00024	
88	Pellets CH_4 emission factor	fraction	0.002	
89	Pellets CO_2 oxidation factor	fraction	0.95	
90	Fuelwood CO_2 oxidation factor	fraction	0.85	

91	Fuelwood CH_4 emission factor	fraction	0.01
92	Gasoline emission factor	kg CO_2-eq l^{-1}	2.23
93	Gasoline N_2O emission factor	kg CO_2-eq l^{-1}	0.00002
94	Gasoline CH_4 emission factor	kg CO_2-eq l^{-1}	0.00035
95	Diesel emission factor	kg CO_2-eq l^{-1}	2.61
96	Diesel N_2O emission factor	kg CO_2-eq l^{-1}	0.00002
97	Diesel CH_4 emission factor	kg CO_2-eq l^{-1}	0.00038
98	Lubricants	kg CO_2-eq l^{-1}	0.51
99	Electricity emission factor	kg CO_2-eq kWh^{-1}	0.08
100	Nitrous oxide GWP100		288
101	Fossil CH_4 GWP100		36
102	Biological CH_4 GWP100		34
103	N_2O GWP20		264
104	Fossil CH_4 GWP20		85
105	Biological CH_4 GWP20		84
106	Average area per plan	hectare	41.51
107	Average sawn wood per mill	m^3	276.24

Chapter 5

General discussion

Tropical forest management for climate change mitigation

The protection of forests is one of society's main priorities. Regardless of our perceptions of what value they possess, forests remain a key component in the cycles that sustain the biosphere. We depend on their existence. The main threat to forests has always been a growing human population that demands more land and resources, which are obtained from forests or gained at their expense. As this trend is not likely to change soon and the remaining forests are still threatened by the same pressures of the past, all possible options must be considered to assure their continuity.

In this thesis, I considered the opportunities for global warming mitigation through harvesting timber from tropical forests in Costa Rica. Overall, the conditions in the country enabled a system that has contributed to the mitigation of climate change through the use of harvested wood products. In this discussion, I will review the main findings from this thesis, the uncertainties associated to the system as it occurs in Costa Rica and provide the local context for understanding results and uncertainties. I will then provide an assessment of potential climate mitigation in Costa Rica through increased forest management. Finally, I will reflect on the implications of these results, treating the existing perception of what tropical forest management is.

The lifecycle of wood from natural tropical forests in Costa Rica

In Chapter 2, I studied the biogenic lifecycle carbon balance (BioC-LC) of tropical forest management in Costa Rica. To quantify the effect of logging and compare it against forest ecosystem carbon balances, I used one hectare as the functional unit and defined the system's temporal boundary as one rotation period (i.e. 15 years). Until now, findings have supported the idea that tropical logging leads to higher carbon emissions, but there have yet been no carbon balance analyses done for these ecosystems using a lifecycle approach. By including all lifecycle processes, I show that technospheric storage, in combination with forest regrowth, result in additional storage of carbon within the system (i.e. -2.19 Mg C ha^{-1} over a 15-year period with a 95% CI of -5.26 to 1.86; or -8.00 Mg CO_2-eq ha^{-1}). Probabilities of a system that could potentially represent a source of carbon exist, as higher harvesting intensities leading to high logging damage, insufficient recovery time, or high wood allocations into short-term uses can shift this balance. However, short-term uses increase storage in solid waste disposal sites (SWDS), and it is the combined effect from technospheric reservoirs that is important for carbon storage. Using a sensitivity analysis, I found that small changes in half-lives do not have an important effect on the stock and that only large changes such as re-allocating products from short to long-term products have substantial effects on total storage.

With the purpose of exploring mechanisms of technospheric carbon storage, I developed a detailed harvested wood product carbon inventory for Costa Rica in Chapter 3. I followed IPCC Guidelines for National GHG Inventories, in correspondence with a Tier 2 accounting level, using country specific data and a material flow analysis. Harvest data collected for this study is the best currently available in Costa Rica, because it describes the evolution of wood production during the last 30 years. Carbon storage at the national level in 2016 (the last year of the inventory) was -412 Gg CO_2 (95% CI between -447.2 and -376.4). Most of this carbon was stored in SWDS (77%), partly a consequence of a high allocation of wood production into short-term products.

Given these allocation patterns were positively correlated with planted forests becoming the country's main wood source, I asked what have been (or will be) the effects of changes in wood source and product allocation on the carbon stock of harvested wood products in Costa Rica. Since plantation wood tends to be of lower quality (at least lower wood densities and carbon content), and because half-lives are consistently reported as drivers of carbon storage, I hypothesized the stock must be heading towards a steady state. Despite a significant decrease in half-life and carbon content, the stock seemed unaffected. Hence, the strong inertia from the inherited stock and its resulting resistance to change imply that its contribution to climate mitigation is likely smaller than commonly believed. Prolonging lifespan will not significantly extend the physical limits, which characterize technospheric carbon storage. It is mostly through increasing harvest levels and wood use whenever possible, that storage may be increased.

In Chapter 3, I found that wood products from natural tropical forests were a source of carbon because of emissions from the inherited stock coupled with decreasing harvest levels in these forests. There is a positive correlation between decreasing trends in wood sourced from natural forests and long-term uses. The decrease in wood production used in construction also resulted in this stock being a source of carbon. Wood products from local plantations did not make up for this change but population and the built environment have continued to grow during this period. Construction wood in Costa Rica has therefore been substituted by other more carbon intensive materials and by wood imports.

As a first step to account for the effect of this substitution on the country's GHG emissions, I assessed the lifecycle climate impact of wood from natural tropical forests in Costa Rica (Chapter 4). This work fills a gap in the understanding of the effects of logging in the tropics, where few studies have been conducted, and none of these have included the combined effect of biogenic and fossil emissions in a cradle to grave analysis for one rotation. Results for this study indicate that over a 15-year period, the system stored more carbon than it released, showing a net balance of -4.41 Mg CO_2-eq ha^{-1} (95% CI of -13.12 to 10.96). To verify these results, I compared with the effect of a shorter time horizon (i.e. 20-year global warming potential (GWP)), and extended the temporal boundaries (i.e. from 15 to 100 years). Under a 100-year system boundary, emissions increase significantly, causing the balance to shift to net emissions of 1.90 Mg CO_2-eq ha^{-1} over the entire period (95% CI of -10.55 to 18.28). I argue that the 100-year mark is not an appropriate boundary for the functional unit of 1 hectare of forest, since all possible rotations are not taken into consideration. This boundary is useful when the functional unit is a product, or as in this case, 1 m^3 of wood used for a specific product or co-product. Results for each individual wood product and co-product were also included, but for a 100-yr system boundary only mid and long-term products show a negative GHG balance due to carbon storage. Short-term products are specially affected by a change in boundary due to EoL methane emissions. Although these require almost no manufacturing, short-term products have the highest emissions per m^3, i.e. 860 Mg CO_2-eq ha^{-1} over a 100-year period. Because of the large proportion of short-term products, these have a large effect on the results per hectare or multi-functional m^3.

In Chapter 4, I was able to confirm that the inclusion of all GHG from biogenic carbon and the additional emissions from lifecycle processes did not change the conclusions from Chapter 2, but that these emissions reduced the net balance from the BioC-LC by 50%. At a per hectare level, the GHG balance of logging tropical forests offsets emissions during the analyzed period and creates a surplus of carbon stored in the system (i.e. forests, products and SWDS). Like the conclusions from Chapter 2, the probabilities and mechanisms that can shift the system to a source of carbon remain the same, confirming that measures to address on-site carbon losses are of primary importance if wood products from forests are to be included as a climate mitigation measure.

Local data and uncertainty

Publicly available lifecycle inventory (LCI) data for timber harvested from natural forests is scarce, with most LCI models relying on data from temperate forests (Newell & Vos, 2012). Due to large variability in forests, forest management systems, etc., it is difficult to extrapolate findings from temperate regions to the tropics (Lippke et al., 2011), where a lack of studies is still a major constraint (Murphy, 2004; Numazawa et al., 2017; Piponiot et al., 2016). Although scarce, some tropical lifecycle assessments have been conducted for timber products in all main tropical regions and for different functional units (Adu & Eshun, 2014; Eshun et al., 2010, 2011; Jankowsky et al., 2015; Ramasamy et al., 2015; Ratnasingam et al., 2015; Rinawati et al., 2018). These vary in terms of system boundaries and none include biogenic carbon, providing no evidence of its effect on the GHG balance of wood products. Due to concerns regarding the sustainability of logging in the tropics, biogenic carbon emissions should become a priority in future studies.

The availability of sufficiently thorough and reliable data may be a challenge in LCA, and more so when dealing with tropical timbers. To make up for this shortage, I collected data for all major processes through local surveys. By doing so, the conceptual understanding of the system was improved, even beyond its boundaries. This renewed understanding can in some cases be translated into more accurate estimates. Additionally, parameter variability is better quantified using local data and this adds specific information about the system. Overall, local data reduces all levels of uncertainty, i.e. conceptual, model and parameter uncertainty (Huijbregts, 1998). However, there are also trade-offs to this approach. Although the quantification of impacts generally benefits from a better understanding of the system, existing models are not fit for local data. They must be further developed under different circumstances. Even when parameter uncertainty or variability may be considered assets during LCA result interpretation (Clavreul et al., 2012), variability may add unnecessary noise (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010b). This is especially the case when parameters are meant to represent the whole country and contain spatial, temporal and technological variability (Clavreul et al., 2012; Huijbregts, 1998) as in this study.

Conceptual uncertainty

The conceptual understanding and consequently the quantification of processes within the studied system was mostly improved because of the material flow analysis. This method not only determines the flow of wood through its lifecycle but clarifies the processes that are involved in the manufacture of products and co-products, addressing multi-functionality (Geng, Yang, et al., 2017; Jasinevičius et al., 2018). It is data intensive (E. Marland & Marland, 2003) but reduces the need to make choices and assumptions. These are inevitable in lifecycle assessments, but are subjective and a main source of uncertainty (Huijbregts, 1998).

Among the limitations from the few biogenic carbon balances for tropical forests that include carbon stored in products is that these rely either on assumptions or small sample sizes to determine the fate of wood after milling (Numazawa, 2018; Piponiot et al., 2016). As these studies only included biogenic carbon, the implications are large but do not propagate to other processes as they would in LCA. In Costa Rica, the material flow marked the size of each pool and their direction until the end of life, in some instances showing processes that would otherwise be ignored. Co-products are perhaps the best example. They have been traditionally considered residue, so if I had simply assumed them lost after milling, I would have ignored the effects of transportation, manufacturing, use, carbon storage, end of life, and potential substitution resulting from these co-products. Quantifying this was possible since residues recently became an important part of the mill's revenue, improving the knowledge of the amount and type of co-products. Besides, these are not small amounts and altogether make up for 33% of harvest. An interesting and important implication is that this increased the milling efficiency up to 96%.

Model uncertainty

The possibility to estimate emissions from all products, co-products and residues, and the inclusion of all processes involved in their lifecycle show how a conceptual understanding indirectly improves estimation procedures. Estimations can still be wrong, but at least they will not be omitted. Differentiated lifecycle processes such as manufacturing were not the only ones included, but half-lives could be assigned to trace products independently in space and time. This tracing of individual products was especially important in Chapter 3, examining the role that changes in wood source (i.e. natural forest, plantations or agricultural lands) and product allocation play on carbon storage in harvested wood products (HWP) at the national level. Wood source was analysed using carbon content as a proxy, while half-life was used as an indicator for change in product allocation. This analysis was only possible due to the local nature of the data used, and the disaggregation into as many categories as possible (Aleinikovas et al., 2018; Jasinevičius et al., 2018; E. Marland & Marland, 2003; Pingoud & Wagner, 2006).

Thus, in Chapter 3, I argue that the results obtained were more accurate than if publicly available data were used, such as FAOSTAT together with a Tier 1 accounting level according to the IPCC Guidelines. Such data source and methods would have largely overestimated the stock and its changes. As the basis for this conclusion, consider that out of the three categories of semi-finished products recommended for HWP inventories (Pingoud et al., 2006), Costa Rica does not produce boards and panels or pulp and paper. Regardless of its actual use, all domestic harvest would have therefore been classified as "sawn wood," and assigned a single half-life that happens to be the highest (i.e. 35 years). Additionally, as domestic harvest is usually not linked to a specific wood source, and because carbon fractions are assigned according to wood category (i.e. sawn wood), all domestic harvest would again have been estimated with the highest possible carbon content.

To illustrate the effect of this estimation difference, Figure 5.1 shows the HWP contribution as calculated in Chapter 3, comparing it against an estimate based on FAO statistics. As predicted, the contribution from other categories is minimal (despite extreme results for the last 4 years), and total contribution is from sawn wood. HWP contribution (i.e. the yearly change in carbon stocks) based on FAO statistics was only estimated from products in use. Comparing changes in carbon stock, only for products in use, this wood classification and methods overestimated results, on average by 30%. This is surely within the range of uncertainty commonly reported for this source of information, i.e. +/-50% for non-OECD countries (Grassi et al., 2018; Jasinevičius et al., 2018; Pilli et al., 2015). It is worth noting that the patterns of wood production observed in both datasets tend to become similar after the year 2000, when Costa Rica improved its data collection system.

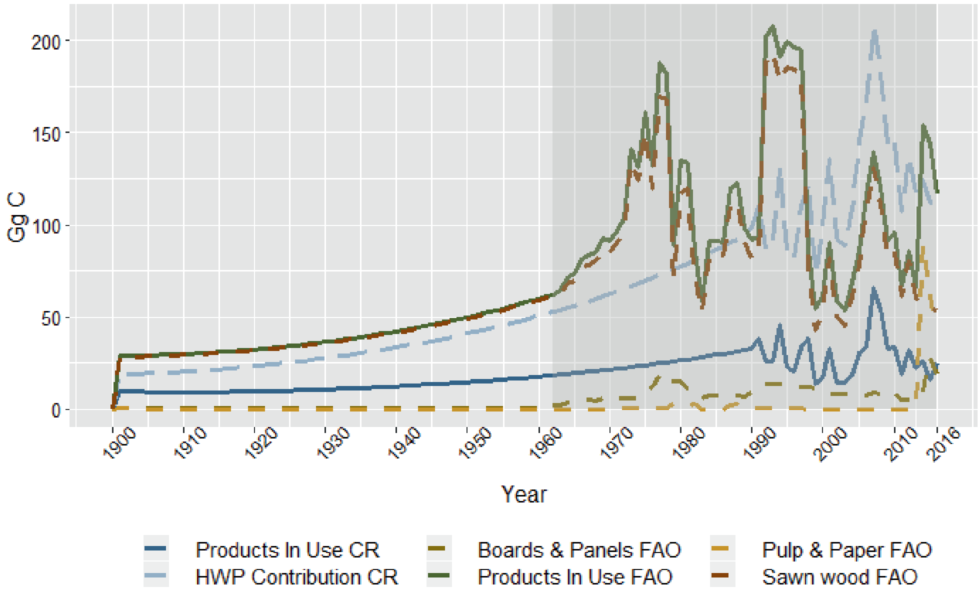

Figure 5.1. Harvested wood product (HWP) contribution for Costa Rica for products in use and considering all technospheric reservoirs based on local data (Products In Use CR and HWP Contribution CR); HWP contribution from products in use of each category of semi-finished products and all categories combined using FAO statistics (Products in Use FAO).

The local classification of wood products and their source, which is not included in most studies on carbon storage in HWP, allowed for the analysis of the effects of changes in these parameters on present and future storage. In Costa Rica, both wood source and product allocation were observed to be changing, although in opposite directions from what is needed to increase storage, i.e. “fast-wood” or more wood from fast growing plantations used in short-term products. Half-lives are consistently reported among the most important parameters affecting carbon storage, and for prolonging lifespan as a measure to increase this storage (Aleinikovas et al., 2018; Donlan et al., 2012; Jasinevičius et al., 2018; Pilli et al., 2015; Pingoud et al., 2010; Skog et al., 2004). Scenario modelling is usually the approach used to estimate the potential contribution from a change in lifespan on carbon storage (Brunet-Navarro et al., 2017). They are usually forward looking, i.e. ‘what if’ scenarios of potential change. Based on our data set, however, we used a retrospective approach. I was able to estimate this potential based on the observed changes in product allocation during the 26-year period of our analysis. By doing so, I confirm a potential 10% increase in storage can be obtained through a 20% change in half-life

(Brunet-Navarro et al., 2017). Although based on the data, increasing the half-life of the stock by 20% seems to be constraint by physical limits. This analysis serves as an example of how the use of local data provides a broader understanding of mechanisms leading to carbon storage and yields lessons that exceed the boundaries of this study.

Parameter uncertainty

Harvest and logging damage were key drivers of the GHG balance and its uncertainty. Logging damage is relevant mainly because of its scale. In a selective logging management system, losses of 6.8–50.7 Mg C ha^{-1} have been reported due to infraestructure and tree felling (Pearson et al., 2014). As shown in Chapter 2, these values can be lower depending on local practices and harvest intensities (Martin et al., 2015), but damage still remains the most important source of emissions. At a per hectare level, carbon losses due to logging in Costa Rica (assuming committed emissions for comparison) were on average 5.26 Mg C ha^{-1} (n= 31; σ=1.85).

Tracing harvested products along their lifecycle was the main goal of this work, as harvest is the base for all subsequent processes leading to emissions and storage. When accounting for carbon storage or the GHG balance of forestry, no other parameter compares with harvest levels in terms of the impact on results (Pingoud et al., 2010; Skog et al., 2004). Average standing tree harvest was 11.08 m^3 ha^{-1} (n= 65; σ=6.23), varying from 1.5 to 35 m^3 ha^{-1} for the entire rotation. Total harvest also included harvested deadwood, which on average was 1 m^3 ha^{-1} (n= 53; σ=1.55) and varied from 0.02 to an extreme case of 8.88 m^3 ha^{-1}, explaining the large standard deviation. Standing tree and deadwood harvests combined, resulted in a total average of 12.09 m^3 ha^{-1} or 2.45 Mg C ha^{-1} (n= 70; σ=1.5). In order to estimate the recovery time of forest carbon back to its initial state, both total harvest and logging damage were used in the calculation, and determine the magnitude, the rate of forest regrowth, and most of its uncertainty (Rutishauser et al., 2015).

I highlight these ranges because variability is the main source of parameter uncertainty, and the only measure for the system's uncertainty as modelled through sampling methods like the Monte Carlo simulation (Heijungs & Lenzen, 2014; Huijbregts, 1998; Lo et al., 2005). Conceptual, model, or parameter uncertainties due to sampling error, must be addressed differently, as simulations are not able to quantify their effects. I argue that although they are inherent to any system, better data could minimize the impact of these sources of uncertainty.

These ranges reflect possible levels of harvest and damage that may take place in the region, and decision making processes can usually benefit from their consideration, as long as they may be well interpreted and understood (Clavreul et al., 2012).

For example, the highest harvest levels and damage occurred in forests where wood production was not the main goal. These harvest and damage levels were associated with management plans for land use change over very small areas (3-5 ha) approved for building infrastructure (mainly electricity lines). These data should have perhaps been excluded, but as argued here, they are still within the probabilities and should not be dismissed. However, measures to reduce the effects of sampling errors on parameter uncertainty should be taken, as large variabilities can cause unwanted noise in the output of the system. This is especially important with determinant parameters such as harvest. As experienced in this study (Chapter 4), variability in harvest levels will result in very large uncertainties due to allocation at the product or co-product level. This is partly due to the decision on how to allocate (e.g. mass) but is also a result of this decision being translated into a calculation method.

The same mechanism of parameter uncertainty (or variability) is also evident propagating through calculations with correlated parameters, as in the case of harvest and logging damage (Clavreul et al., 2012; Frey, Penman, Hanle, Monni, & Ogle, 2006). It is a liability of the model developed to assume these parameters act independently, when they may in fact be positively correlated (Martin et al., 2015). This decision was taken in order not to lose information over the type of damage (i.e. roads, gaps, decks, etc.) or the source of harvest (standing trees or deadwood), but it inevitably results in higher uncertainties. Firstly, because several overlapping parameters are used in the calculation of an output that could have been estimated with one or fewer parameters. Secondly, because sampling from the distributions of these parameters may result in combinations of parameters that may be hard or impossible to find in reality. This was addressed by using the *Dirichlet* distribution (Igos, 2018) when dealing with multiple fractions of the same input.

Variability is not random. It reflects spatial and temporal differences that characterize the complexity of a lifecycle assessment of forestry (Huijbregts, 1998; Lo et al., 2005). Forests will vary spatially due to different biogeographical histories, natural disturbance regimes and the physical environment (Finegan, 2012). The diversity in these ecosystems will determine growth patterns and harvest levels, but also technospheric processes such as the conditions under which a forest is managed, wood characteristics and therefore multi-functionality. An example of spatial variability was observed in the material flow analysis, where results rely heavily on surveys from a representative group of sawmills transforming wood from forests in Costa Rica.

As described in Chapter 2, sawmills were selected based on records found in the reviewed management plans indicating where the wood was being transported. Most happened to be located within the region where wood is harvested and show clear trends in production patterns (i.e. which products and co-products were produced) and in terms of their efficiency. However, the further away these sawmills the larger the residues that were reported. This pattern shows a bias in our sampling that is entirely due to spatial differences, since residues left in mill dumps could be correlated to their distance to the pelleting plant. This plant was established recently, and soon became the main destination for a large part of these residues. As the distance from this plant to the sawmill increased, the use of fuelwood also rose, though information on fuelwood amounts is subject to higher uncertainty because fuelwood can be freely collected.

As we advance temporally and spatially through the lifecycle of wood there will be higher uncertainties due to assumptions related to end of life processes. These have been repeatedly discussed throughout this thesis, given their important role in carbon storage and the GHG balance calculations (Barlaz, 2006; De la Cruz et al., 2013; Ingerson, 2011; Skog et al., 2004; Wang et al., 2013; F. A. Ximenes et al., 2018; F. Ximenes et al., 2015). Under the assumption that wood experiences the same EoL as all the other waste in the country, no data was collected for this phase of the lifecycle. Although fuelwood is not an important source of energy in Costa Rica because the country's electricity grid covers more than 97% of its territory, by omitting EoL data I probably overestimated the flow of wood to SWDS, as a larger fraction of wood will likely be combusted for fuel. To reduce this bias it was assumed that the decomposable fraction in SWDS is the same for wood as for all other waste types, which is known to be an overestimation (O'Dwyer et al., 2018). This is evidently not the preferred way to deal with these uncertainties, and presents us with a case where lack of data tends to be a generalized problem (Akagi et al., 2011; Bogner et al., 2008; Clavreul et al., 2012; Pingoud & Wagner, 2006; L. Zhang et al., 2019).

I argue that increase in system understanding through local data partly offsets the uncertainty caused by large data variability. Additionally, the interpretation of confidence intervals varies depending on the objectives of each chapter. To understand this, is partly to understand differences between uncertainty and variability (Heijungs & Lenzen, 2014). In Chapter 3, there is an exact unknown value, i.e. the change in the carbon stock of Costa Rican products in 2016. The confidence interval, in this case, is the range within which the exact value can be found but the result is uncertain. In Chapters 2 and 4, there is no exact value but a range of possibilities that depend on circumstances, i.e. the result is variable.

Culture and institutions in the lifecycle of wood from tropical forests in Costa Rica: from local data to a local context

The results presented in this thesis show that forest management of natural tropical forests in Costa Rica has the potential for a carbon neutral outcome. So far, I have discussed the implications of local data on the results from this study, but local data is the outcome of a local context. It is therefore relevant to interpret the results from this work considering this local context. There has been a strong environmental movement during the past 30-40 years that transformed the country's relationship with forests. Forest protection became a national priority as the country managed to take advantage of a green image and profit from it, e.g. through tourism. But this was not always the case, as the country experienced high deforestation rates between 1950 and 1970 due to an agricultural expansion. Efforts to revert this trend through command and control measures, incentives and awareness raising are largely responsible for the current perception that forests are service rather than resource providers, and for the association of forestry with environmental damage (Serrano & Moya, 2011; Villalobos & Navarro, 2017; Werger, 2011). As a result, strong regulations have been put in place to control and sometimes restrict natural forest management.

One of the influential regulations in Costa Rican legislation is the 1997 Forest Law, in which the first nation-wide program of Payments for Environmental Services was published (Pagiola, 2008). This mechanism had the goal to compensate forest owners for the protection of forests. The Law recognized forest management as complementary to conservation and thus entitled managed forests to an environmental payment in order to reduce the threats from deforestation.

However, soon after the Law came into force, managed forests were excluded from this program (Werger, 2011), and some regions even set administrative bans and have rejected all attempts to obtain logging permits since then (Camacho, 2015; Santamaría, 2015). The effect these policies have on national harvest from natural forests was described in Chapter 3. In short, harvest fell from 60% in 1990 to a current 5% of national wood production. In terms of area, national forest cover is 52.4% of the country, while just 17% is considered productive (Camacho Calvo, 2015; Pedroni et al., 2015; Werger, 2011). Within this 17%, harvesting is currently taking place in 0.2 - 0.5%. Depending on the source of information (Chapter 2-4 or Chapter 3), this area corresponds to just 1432-2442 ha per year. These are all privately owned forests with an average size of 80 ha, conforming approximately 18 - 30 management plans per year.

Due to the small forest sizes and the few management plans approved per year in the country, forest management can be considered low-scale when compared to large forest concessions occurring elsewhere. One advantage is that this facilitates the enforcement of existing control mechanisms, which require: that every management plan be developed by a licensed forester, that it is approved by a regional office from the Ministry of Environment, verified through field inspections, regularly audited by the forester during its implementation, and that transportation permits are granted only against the report from this audit. Additionally, two local non-governmental organizations have been responsible for half of all management plans and play a major role in the regional forest sector. FUNDECOR (Foundation for the Development of the Central Volcanic Mountain Range) and CODEFORSA (Forestry Development Commission of San Carlos) group and assist forest owners throughout the process. Their close relationship with forest owners, the regulatory body, and the forest industry has allowed important inter-sectoral communication that reinforce existing control mechanisms.

These conditions provide further context and determine the results from this study. In Chapter 2, I discussed the reasons for the low logging damage in Costa Rica's harvest operations, which include: low harvest intensity, i.e. 12.09 $m^3\ ha^{-1}$, compared to a range of 10 - 30 $m^3\ ha^{-1}$ in other regions in Central and South America (Pearson et al., 2014; Rutishauser et al., 2015; Sasaki et al., 2016); a large share (10-20%) of deadwood harvesting that does not require felling; and small forest patches within agricultural lands that do not require extensive infrastructure inside the forest. In addition, if I consider the conditions within which these management plans take place and the strong relationship between the different actors, it is easier to assess the role that enforcement mechanisms have in the country's low impact logging.

Conditions in the wood industry also influenced the lifecycle GHG balance of Costa Rica's forest management system, which is characterized by high wood utilization efficiency. In Chapter 2, I reported a 63.67% milling efficiency, though it is common to expect efficiencies closer to 50% (Butarbutar et al., 2016; Ofoegbu et al., 2014; Ramasamy et al., 2015; Sasaki et al., 2016). I explained 46% of harvest is allocated to formwork for the construction sector (short-term products in Chapters 2 and 4), a low-quality product that maximizes wood use to mould concrete. The other 17% goes to structural and non-structural wood used in construction, classified as long-term products. The remaining fraction of co-products from the milling process was also accounted for, and overall, only 4% of wood becomes residues. Sawmill by-products also play an important role. Out of the 20 sawmills surveyed, 18 reported selling all edges and off-cuts to the furniture industry. Sawdust and shavings for stalls, stables and nurseries have been sold to local farmers for many years, and now there is even competition from the pellet plant, which additionally uses slabs and bark. I discussed the influence of distance from the sawmills to the pellet plant as a driver for the use of wood residues, and this effect may also relate to their distance from main roads or population hubs. Enforcement of environmental regulations on mill dumps plays a role here as well. Some mills report selling residues to avoid penalties from improper waste management, even if the revenue does not compensate transportation costs. Finally, a low wood supply due to low harvest levels may also partly explain the high resource utilization in the country (Brunet-Navarro et al., 2017; Suter et al., 2016).

The contribution of tropical forest management to climate change mitigation in Costa Rica

Could Costa Rica's forest management system reach a level where higher emissions are avoided or reduced? Would this provide additional mitigation benefits? A potential increase in harvest levels is worth considering, given climate mitigation relies on a human induced change against a business as usual scenario (Plevin et al., 2014a, 2014b). Thus, what interests mitigation is not the current amount of carbon stored in products or forests, but the *increase* in storage with respect to a baseline (Helin et al., 2013). Because carbon in wood products is a stock that is usually unaccounted for, this concept is sometimes confused in the literature. However, accounting an unaccounted stock does not imply true additional storage.

Increasing storage is not trivial, as it is constrained by physical limits and these are case specific (Brunet-Navarro et al., 2017; Werner et al., 2010). Potential increase in harvest levels marks the most important limit, but it also represents the largest opportunity for potential mitigation benefits. There are essentially two possibilities to obtain these benefits: increasing harvest intensity (m^3 ha^{-1}) or increasing national harvest levels. Increasing harvest intensity is an option in Costa Rica that should not be readily dismissed, as it could add by making the activity more profitable. Only one third of the volume inventoried and approved was effectively harvested in the forest management plans here considered. Forest regrowth, as estimated in this analysis (Rutishauser et al., 2015), shows that with current harvest intensity and logging damage, forests can recover their carbon stock before the end of the rotation, leaving room to increase to a maximum yield. However, increasing harvesting can also be risky, as the results from the GHG balance show that small changes in harvest intensity may result in potentially higher GHG emissions (Chapter 4). Carbon storage will increase, but probably not enough to offset the additional emissions from logging damage and EoL. Therefore, the largest opportunity in Costa Rica by far is increasing national harvest levels.

Increasing national harvest levels by increasing the annual harvest area within natural forests is also an option. In recent years (2010 - 2015), this area was 0.2 - 0.5% of the approximately 500,000 hectares considered as potentially productive forest (Camacho Calvo, 2015; Pedroni et al., 2015; Werger, 2011). If productive areas were harvested following 15-year logging cycles, annual harvest area could be increased from an average of 1,937 to 33,333 ha. This would represent an increase in harvest equal to one third of current national wood consumption (Barrantes & Ugalde, 2018). If the net GHG balance, as estimated in Chapter 4, is -4.41 Mg CO_2-eq ha^{-1} over a 15-year period, then the average climate impact from one year's harvest will be close to -8.5 Gg CO_2-eq. This projection is simply based on this study, which aims to understand the system at the forest level using hectare as a reference, and it therefore has limitations when extrapolating results at a regional level. Considering the difference between the current scenario and a potential increase reaching the national harvest's maximum sustainable yield, the resulting additional contribution to climate mitigation would be -139 Gg

CO_2-eq yr^{-1} (Table 5.1), approximately 1-2% of total national GHG emissions (Chacón et al., 2014). The main risk arises from the inability of existing institutions to cope with such a change, but the real challenge is the need to transform wood production and consumption patterns. As discussed in Chapter 3, additional mitigation benefits from the forest sector require supply and demand-side measures (Suter et al., 2016), with important lag times to be suffered as those experienced in other sectors (Klitkou et al., 2015).

Additional mitigation opportunities can be found in different combinations of product allocation and wood use, leading to higher allocations of long-term products. In the biogenic carbon balance from Chapter 2, I found that the balance remained unaffected by small changes in lifespan and concluded that in order to affect the balance, changes in a product's half-life must be large, as those caused by different product allocation. The HWP carbon inventory (Chapter 3) provided historic data on large changes in wood allocation (e.g. the significant decrease in construction wood and the increase in short-term products) to explore the effect on carbon storage. Important changes in the stock were not observed, and I questioned the feasibility of this commonly cited strategy (Intergovernmental Panel on Climate Change, 2014; Lun et al., 2012). There were methodological challenges to detect changes in the stock given the combined effect from products in use and in SWDS, and due to the large effect on the stock from annual changes in harvest levels. In fact, when estimating the time to steady state, stochastic systems are avoided, as it is simpler to analyse those where harvest is constant or where it follows some known distribution, e.g. linear or exponential (E. Marland & Marland, 2003). The small effect in the carbon stock's half-life from relatively large changes in wood product allocations with varying inflow was then verified by modelling scenarios, but a larger data set and a different approach might help clarify this mechanism. However, in a single pulse event such as one harvest, and considering all GHG from end of life processes, the result is clear (Chapter 4). A large allocation of wood into short-term products is a determining factor in the GHG balance and long-term products are consequently preferred for their additional climate benefits.

In Costa Rica, short-term products from planted forests currently represent the main use and source of wood, and despite the opportunity to increase carbon storage by increasing products with longer lifespans, this change will be slow and the effect only marginal. Wood from planted forests was expected to substitute wood from natural forests, but the strong demand for pallets and packaging for agricultural exports has transformed production and use in the country (I. Jadin et al., 2016; Isaline Jadin et al., 2016; Santamaría, 2015). Wood from natural forests used in construction decreased from an initial 73% to a 26% at present, while wood from planted forests destined for packaging represented 44% of national production in 2016. This reflects the changing policy for protection of natural forests in Costa Rica, and the efforts set on establishing plantations in the recent past. Although wood production from natural forests now mainly consists of short-term products, i.e. 46% as formwork, this may well be a result of the increased use of concrete in the construction sector (Santamaría, 2015). The production and use of these short-term products have a low climate impact, but their end of life management changes their footprint completely, making them one of the most emissions intensive products. These products play an important role in wood demand and will hardly be eliminated. The new "fast-wood" culture for producing and using wood (Cossalter & Pye-Smith, 2015) demonstrates the challenges of changing allocation patterns. Although the benefits from cascading are generally considered low (Brunet-Navarro et al., 2017; Leskinen et al., 2018), short-term products are an example where material reutilization or EoL energy recovery should be considered. Otherwise, substituting these products entirely is the alternative, but the GHG consequences from doing so can be high.

The emissions associated with producing and transforming wood products are much smaller than those for alternative products. As a result, using wood even for short-term uses can avoid GHG emissions (Leskinen et al., 2018; Suter et al., 2016). The contribution gained from effecting this substitution may be large, but there is still some debate as to whether it should be included in an attributional LCA (ALCA) such as the one presented here. The debate is partly due to limitations from varying estimation methods common in LCA (e.g. comparing systems that may have differing boundaries), but the main criticisms refer to assumptions on the counterfactual system, i.e. the choice of substitute product and the scale of this substitution (Buchholz et al., 2016; Cherubini et al., 2009; Gustavsson & Sathre, 2006; Lippke et al., 2010; Pingoud et al., 2010; Sathre & Gustavsson, 2006; Sathre & O'Connor, 2010). Results could be extremely optimistic, as one product may be compared against the worst possible option and then used to estimate the environmental benefit of a scenario where wood displaces this product completely. This is evidently incorrect and does not follow the principles that define climate mitigation, i.e. a change-based approach (Dale & Kim, 2014; Plevin et al., 2014a, 2014b).

To assess possible consequences for a given change, a reference or baseline scenario is needed. In Chapter 3, I discussed that based on the observed changes in wood production, it was possible to estimate emissions caused by substituting for other materials, and this retrospective approach indicates a reference for substitution (Suter et al., 2016). For this scenario, I estimated the cumulative emissions (1990-2016) due to substitution were 5080 Gg CO_2 of national emissions or 195 Gg CO_2 yr^{-1} that could have been avoided. There is evidence in the country supporting this substitution (Santamaría, 2015; Werger, 2011), although the scale of the effect has not yet been quantified. Aluminium has substituted door and window frames as well as some furniture and formwork; polyvinyl chloride (PVC) has substituted wood ceilings and mouldings, and structural wood (e.g. beams) has mainly been substituted by galvanized iron (Santamaría, 2015). Additionally, pellets are now being produced to substitute bunker fuels or diesel used in industrial boilers (Serrano & Moya, 2011).

Based on this evidence and on the country's potential to revert trends in the substitution of wood for other materials, I consider increased levels of wood production to supply this substitution. Country-specific displacement factors could be estimated by using results from both wood LCA and the climate impact for other materials (Hafner & Schäfer, 2018; Leskinen et al., 2018; Sathre & O'Connor, 2010). Displacement factors usually exclude carbon storage (Sathre & O'Connor, 2010), but this should only apply to cases for which a complete biogenic balance has not been conducted. Ideally, information on the climate impact from all materials should reflect local conditions, but this data may be challenging to obtain. Published displacement factors or published lifecycle GHG emissions (Cherubini et al., 2009; Gustavsson et al., 2017) and the fractions of final wood products from the material flow can provide an approximation of the scale of potential mitigation due to substitution of other materials with wood. There is a wide range of substitution factors in the literature, with the most commonly used showing an average 2.1 Mg C per Mg C of dry wood used (Sathre & O'Connor, 2010). Recently published displacement factors are much lower, and for the materials described above, they could range between 0.8 - 1 kg C per kg C in wood (Geng, Zhang, et al., 2017; Keith et al., 2015; Leskinen et al., 2018; Lippke et al., 2011; Rüter et al., 2016).

Together with the average carbon content of a harvest per hectare, these displacement factors can be used to estimate the emissions that could potentially be avoided by using more wood in Costa Rica. I have only considered the fraction of wood effectively used as products or co-products (i.e. 2.4 Mg C ha^{-1}), and the fraction of each product and co-product conforming the total harvest. Using these criteria, I estimated the weighted emissions per hectare, resulting in 1.79 Mg C ha^{-1} or 6.55 Mg CO_2-eq ha^{-1} (Table 5.1). These are negative emissions, so together with -4.41 Mg CO_2-eq ha^{-1} from the GHG balance, an average hectare of forests could potentially reduce -10.96 Mg CO_2-eq ha^{-1}. To up-scale this result to the national level, the difference between existing harvest levels and their potential increase toward the maximum national sustainable yield is the only factor considered. To account for substitution, an additional -206 Gg CO_2-eq yr^{-1} should be added to the -139 Gg CO_2-eq yr^{-1} from the GHG balance to be gained from an increased harvest. Therefore, the potential contribution of tropical forest management to climate change mitigation in Costa Rica should be close to -344 Gg CO_2-eq yr^{-1} or 3% of annual national GHG emissions.

Table 5.1. Net GHG balance and substitution from the lifecycle of wood production in Costa Rica for an average hectare, the national average annual harvest and the potential maximum sustainable yield (Gg CO_2-eq).

	Ha	m^3	GHG Balance	Substitution	Total
Average hectare	1	12.0	-0.004	-0.007	-11.0
National average annual harvest	1937	18385.1	-8.5	-12.7	-21.2
Maximum Sustainable Yield	33333	402910.9	-147.0	-218.4	-365.4
Difference			-138.5	-205.7	-344.1

Harvesting (or not) as a climate mitigation strategy in the tropics

The importance of a reference scenario

Strategies to reduce emissions from deforestation in tropical forests have mainly focused on policies aimed at the protection of forests. Broader strategies for conservation that allow for the sustainable use of resources while including protection have been dismissed (Ellison et al., 2013; Merry et al., 2009). The potential for tropical natural forests to contribute to climate mitigation is narrowed down to a perceived dichotomy between protection and management. However, these do not necessarily have to be opposing views (Finegan, 2012). The main argument against forest management is that it may lead to deforestation, but this link is weak. Even when logging has preceded deforestation, the driver is land use change for agricultural lands (Poker & MacDicken, 2016; Sessions, 2007). This perception persists despite the evidence, and it is one of the reasons why forest management has not been fully included into conservation strategies (Edwards et al., 2014).

The extent of forest degradation in the tropics has also partly been used as an argument to discourage forest management within conservation strategies (Vogtländer et al., 2013). Carbon emissions from forest degradation, mainly caused by wood production, can be as high as those from deforestation (Ellis et al., 2019; Pearson et al., 2017; Francis E. Putz et al., 2008). Although I do not question the relevance of forest degradation, I argue that the impact has been overestimated by the assumption of committed emissions (Iordan et al., 2018), generalizations over harvesting practices, the neglect of forest regrowth, and by the uncertainties on the extent of degradation caused by forest management. Carbon losses due to logging can be significant, and an average of 21 Mg C ha^{-1} or 77 Mg CO_2 ha^{-1} has been estimated from a sample of 6 countries to show the effect of logging in the tropics (Pearson et al., 2014). As shown in this thesis, logging emissions can also be significantly lower (i.e. 14 Mg CO_2 ha^{-1}; Chapter 4). Additionally, when accounting for global harvest to estimate emissions from degradation, FAO data on wood products is sometimes used (Pearson et al., 2017). As I have discussed, this may also induce overestimations. Finally, whenever forest regrowth was included, estimated gross emissions of 0.45 Pg C yr^{-1} were almost entirely compensated by forest regrowth (0.446 Pg C yr^{-1}), resulting in a net balance of 0.004 Pg C yr^{-1} (Richard A. Houghton, 2013).

The existing results from degradation are important as they highlight a problem that needs to be addressed. However, degradation is seen as the benchmark for the consequences of managing forests for wood production, and the protection of forests is inevitably the best carbon mitigation strategy for this scenario (Keith et al., 2015). Here, the dichotomy becomes evident between managing forests or not, as the options are seemingly limited to a protected standing forest on the one hand, or a forest degraded through logging on the other. The most valuable conclusion drawn from the understanding of lifecycle processes in this Costa Rican case study is that forest management does not necessarily lead to degradation, and that a protected forest should not be the default reference system.

The assumption that the reference is a forest in a state of relaxation is also common in an attributional LCA of wood products (Helin et al., 2013). This assumption is used in this study as well. This is meant to show that if the forests were not harvested, their carbon content would have remained unchanged. This is a valid assumption to some extent, but it ignores indirect effects caused by protecting forests, which commonly go unrecognised despite having important implications on the GHG balance. Based on the evidence presented in this thesis, I argue that the indirect GHG impacts occurring as a result of protecting forests should also be considered and evaluated as part of this reference scenario. These unintended consequences are among the main lessons obtained from this Costa Rican case study. It is therefore a relevant study to review.

It has been hypothesized that the need for wood products may compromise the effectiveness of strategies aimed at protecting standing forests exclusively (Parker et al., 2014). When I analysed historic wood production in the country (Chapter 3), I found evidence that harvest levels from trees in agricultural lands peaked as soon as natural forest management became restricted. These lands did not require a management plan and their harvesting is mainly attributed to deforestation (Arce & Barrantes, 2004). The forest understory was first cleared, then a logging permit was obtained, and this led to land use change. This is a clear example of how the sudden implementation of protection policies caused substantial illegal logging and deforestation in forests, (Camacho, 2015) and demonstrates the potential for carbon leakage or indirect land use change. These are unintended emissions outside the system boundary that can be attributable to the system, and that in this case result in increased emissions (Røyne et al., 2016; Taeroe, Mustapha, Stupak, & Raulund-Rasmussen, 2017).

Another unintended consequence from restricting natural forest management in Costa Rica was the transformation of wood production and wood use in the country. Increased emissions due to these changes are harder to attribute entirely to the protection of forests, but they coincide with the period in which natural forest management became restricted in Costa Rica. The approach used to supply the demand of wood originally harvested from forests was to incentivize forest plantations, but these never managed to fully substitute this wood source (Isaline Jadin et al., 2016). Today, short-term products from fast growing plantations dominate, and as shown in Chapter 4, these are an important source of end of life emissions. Simultaneously, as harvest from natural forests decreased, the inherited carbon stock of wood products from forests also became a source of carbon emissions (Chapter 3) for the country. Additionally, since reduced harvest from forests is correlated to a reduction in long-term products, a large part of emissions occurring due to the substitution of construction wood by other materials can also be attributable to a no-harvest scenario. If all these indirect effects are estimated and included in the assessment, the no-harvest reference scenario might not be the ideal state against which forest management is commonly compared.

Looking beyond...

While tropical forested countries clearly differ in wood production, industry and forest management, trends in many countries are likely similar to those in Costa Rica. Harvest from tropical forests has been declining since the 1990's (Blaser et al., 2011; Oliver & Mesznik, 2006; Shearman et al., 2012; Tomaselli, 2007) and planted forests are now responsible for 65% of tropical wood production (Birdsey & Pan, 2015; Blaser et al., 2011; ITTO, 2015; Payn et al., 2015; Sessions, 2007). Short and mid-term products are now more common than long-term uses. As a consequence, the carbon stocks from products have become sources of emissions and are more susceptible to changes in harvest levels (Johnston & Radeloff, 2019; Oliver & Mesznik, 2006; L. Zhang et al., 2019). Although harvest levels generally increase along with population, products used in construction are also in decline, and emissions from this substitution have been taking place.

Given these conditions, it can be expected that the analysis of a reference scenario against which tropical forest management must be compared is not different from that observed in Costa Rica. It is not a dichotomy, but a wide range of probable outcomes determined by local conditions. Understanding these conditions and their underlying processes is the main contribution from a lifecycle approach such as the one presented here. It is hard to derive conclusions applicable to other conditions, or to provide extrapolations for the scale of the benefits from tropical forest management based on this case study. This was not the aim of the study and attempting to do so would partly contradict our argument about the importance of local data in the understanding of systems and in the decision-making process. The main contribution from this work has been to fill in a gap in the perception towards the environmental performance of tropical forest management. Most of the findings in this thesis confirm results described for regions where LCA has been common although they have been dismissed for a tropical forest management system based on the consequences from deforestation and degradation. The possibilities of a sustainable system or of a system contributing to climate mitigation are just the same.

Increasing the potential for mitigation will rely heavily on measures outside the forest sector and this is probably the largest challenge that must be overcome. Increasing harvest levels is possible in Costa Rica and all along the tropics, but without the demand for products, this potential is constrained. To great extent, the opportunities for modifying these limits rely on making the impacts from logging explicit, but also its merits. A main argument has been that there are large opportunities for improving current estimates of forest degradation through the lifecycle approach. By doing so, a better understanding of the climate impact from degrading forests may lead to a fair representation of the system. Improving data on biogenic carbon emissions using a dynamic inventory to account for emissions, storage and regrowth when they occur can potentially improve this estimate. The processes that should be included are: logging damage; harvesting; wood allocation into products, co-products and residues; end of life processes (i.e. the fraction combusted or sent to SWDS); and forest regrowth. Forest carbon sequestration is of overriding importance in the net GHG balance of logging (Côté et al., 2002) so methods that are better at predicting growth rates should be implemented, considering the large spatial variation between forest types and regions (Baccini et al., 2012).

The conditions that led to the results of this Costa Rican case study are also not different from those commonly mentioned in the literature, and which should be at the core of any climate mitigation strategy that considers tropical forest management. Tropical forests can contribute to climate mitigation through carbon stored in products and through substitution of other materials, but it is up to forest management to identify ways to assure the sustainability of the system. Reduced impact logging remains the main mitigation opportunity, since GHG emissions from the system are largely determined by how logging is performed. This not only refers to reducing the impact, but it also means finding the correct balance between harvest intensity, damage and recovery time. The substitution of wood has been characterized by the constant improvement in other materials that make them cheaper and more effective. Although efficiency in Costa Rica is relatively high, primary and secondary transformation industries need a technological boost to maximize resource efficiency through innovation. New and different wood products will also help improve our view of wood harvesting and use, as it needs to be updated. In many cases, wood is seen as a second class material. At the policy level, wood use disincentives exist and should be identified and modified.

Additional benefits in mitigation will not be achieved without a strong forest sector. I described the importance of a strong inter-sectoral communication, which in Costa Rica was enabled by a relatively small sector. This was largely responsible for the appropriate implementation of enforcement mechanisms leading to low impact logging. Strong institutions have been responsible for the sustainability of forest management in the country. Now, it seems possible to move from a system focused on enforcement, to one that is a promotor of healthier relationships with forests. Climate smart forestry in the tropics should be driven by the urge to understand and prepare for what will come next (i.e. changes). Forest management in the tropics must be sustainably productive by improving livelihoods without disrupting natural processes, highly adaptive to changes and capable of improving the resilience of the overall system.

References

Aalde, H., Gonsalez, P., Gytarsky, M., Krug, T., Kurz, W. A., Lasco, R. D., … Verchot, L. (2006). Forest land. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use. In H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe (Eds.), *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (Vol. 4, pp. 4.1-4.83). Hayama, Japan: Intergovernmental Panel on Climate Change. https://doi.org/10.1016/j.phrs.2011.03.002

Adu, G. A., & Eshun, J. F. (2014). *Life cycle assessment for environmental product declaration of tropical African mahogany (Khaya) lumber produced in Ghana*. Yokohama, Japan.

Akagi, S. K., Yokelson, R. J., Wiedinmyer, C., Alvarado, M. J., Reid, J. S., Karl, T., … Wennberg, P. O. (2011). Emission factors for open and domestic biomass burning for use in atmospheric models. *Atmospheric Chemistry and Physics*, *11*(9), 4039–4072. https://doi.org/10.5194/acp-11-4039-2011

Aleinikovas, M., Jasinevičius, G., Škėma, M., Beniušienė, L., Šilinskas, B., & Varnagirytė-Kabašinskienė, I. (2018). Assessing the Effects of Accounting Methods for Carbon Storage in Harvested Wood Products on the National Carbon Budget of Lithuania. *Forests*, *9*(12), 737. https://doi.org/10.3390/f9120737

Angelsen, A., & Wertz-Kanounnikoff, S. (2008). *Realising REDD+: National strategy and policy options*. *Realising REDD+: National strategy and policy options*. Bogor Barat, Indonesia. https://doi.org/361

Aragón-Garita, S., Moya, R., Bond, B., Valaert, J., & Tomazello Filho, M. (2016). Production and quality analysis of pellets manufactured from five potential energy crops in the Northern Region of Costa Rica. *Biomass and Bioenergy*, *87*, 84–95. https://doi.org/10.1016/j.biombioe.2016.02.006

Arce, H., & Barrantes, A. (2004). *La madera en Costa Rica: Situación Actual y Perspectivas*. San José, Costa Rica. Retrieved from http://www.sirefor.go.cr/Documentos/Industria/2004_Arce_Barrantes_maderacr_2004.pdf

Arroyo-Mora, J. P., Svob, S., Kalacska, M., & Chazdon, R. L. (2014). Historical patterns of natural forest management in Costa Rica: The good, the bad and the Ugly. *Forests*, *5*(7), 1777–1797. https://doi.org/10.3390/f5071777

Baccini, A., Goetz, S. J., Walker, W. S., Laporte, N. T., Sun, M., Sulla-Menashe, D., … Houghton, R. A. (2012). Estimated carbon dioxide emissions from tropical deforestation improved by carbon-density maps. *Nature Climate Change*, *2*(3), 182–185. https://doi.org/10.1038/nclimate1354

Barlaz, M. A. (2006). Forest products decomposition in municipal solid waste landfills. *Waste Management*, *26*(4), 321–333. https://doi.org/10.1016/j.wasman.2005.11.002

Barrantes, A., Paniagua, V., & Salazar, G. (2011). *Usos y aportes de la madera en Costa Rica: estadísticas 2010.* San José, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica/

Barrantes, A., & Salazar, G. (2007). *Usos y aportes de la madera en Costa Rica: estadísticas 2006.* San José, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica/

Barrantes, A., & Salazar, G. (2008). *Usos y aportes de la madera en Costa Rica: estadísticas 2007.* San José, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica/

Barrantes, A., & Salazar, G. (2009). *Usos y aportes de la madera en Costa Rica: estadísticas 2008.* San José, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica

Barrantes, A., & Salazar, G. (2010). *Usos y aportes de la madera en Costa Rica: estadísticas 2009*. San José, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica

Barrantes, A., & Ugalde, S. (2012). *Usos y aportes de la madera en Costa Rica: estadísticas 2011*. Heredia, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica/

Barrantes, A., & Ugalde, S. (2013). *Usos y aportes de la madera en Costa Rica: estadísticas 2012.* San José, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica/

Barrantes, A., & Ugalde, S. (2014). *Usos y aportes de la madera en Costa Rica: estadísticas 2013*.

Heredia, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica/

Barrantes, A., & Ugalde, S. (2015). *Usos y aportes de la madera en Costa Rica: estadísticas 2014.* Heredia, Costa Rica. Retrieved from http://www.onfcr.org/media/uploads/documents/usos-y-aportes-de-la-madera-estadisticas-2014.pdf

Barrantes, A., & Ugalde, S. (2016). *Usos y aportes de la madera en Costa Rica: estadísticas 2015*. San José, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica/

Barrantes, A., & Ugalde, S. (2017). *Usos y aportes de la madera en Costa Rica: estadísticas 2016.* Heredia, Costa Rica. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica/

Barrantes, A., & Ugalde, S. (2018). *Usos y aportes de la madera en Costa Rica: estadísticas 2018.* San. Retrieved from https://www.onfcr.org/article/usos-y-aportes-de-la-madera-en-costa-rica/

Bellassen, V., & Luyssaert, S. (2014). Managing forests in uncertain times. *Nature*, *506*, 153–155. https://doi.org/10.1038/506153a

Bergman, R., Puettmann, M., Taylor, A., & Skog, K. E. (2014). The Carbon Impacts of Wood Products. *Forest Products Journal*, *64*(7–8), 220–231. https://doi.org/10.1016/j.cca.2006.05.006

Birdsey, R., & Pan, Y. (2015). Trends in management of the world's forests and impacts on carbon stocks. *Forest Ecology and Management*, *355*, 83–90. https://doi.org/10.1016/j.foreco.2015.04.031

Blanco, L., Carpio, I., & Muñoz, F. (2005). *Fichas técnicas de veinte especies maderables de importancia comercial en Costa Rica*. San José, Costa Rica: Editorial de la Universidad de Costa Rica.

Blaser, J., Sarre, A., Poore, D., & Johnson, S. (2011). Status of Tropical Forest Management 2011. *ITTO Technical Series*, *38*(June), 418.

Bodegom, A. J. van, Berg, J. van den, & Meer, P. van der. (2008). *Forest plantations for sustainable production in the tropics: key issues for decision-makers*. Wageningen, The Netherlands. Retrieved from http://edepot.wur.nl/170323

Bogner, J., Pipatti, R., Hashimoto, S., Diaz, C., Mareckova, K., Diaz, L., … Gregory, R. (2008). Mitigation of global greenhouse gas emissions from waste: Conclusions and strategies from the Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Report. Working Group III (Mitigation). *Waste Management and Research*, *26*(1), 11–32. https://doi.org/10.1177/0734242X07088433

Brandão, M., Levasseur, A., Kirschbaum, M. U. F., Weidema, B. P., Cowie, A. L., Jørgensen, S. V., … Chomkhamsri, K. (2013). Key issues and options in accounting for carbon sequestration and temporary storage in life cycle assessment and carbon footprinting. *The International Journal of Life Cycle Assessment*, *18*(1), 230–240. https://doi.org/10.1007/s11367-012-0451-6

Breton, C., Blanchet, P., Amor, B., Beauregard, R., & Chang, W. S. (2018). Assessing the climate change impacts of biogenic carbon in buildings: A critical review of two main dynamic approaches. *Sustainability (Switzerland)*, *10*(6). https://doi.org/10.3390/su10062020

Brown, S., Lim, B., & Schlamadinger, B. (1998). *Evaluating Approaches for Estimating Net Emissions of Carbon Dioxide from Forest Harvesting and Wood Products*. Dakar, Senegal.

Brunet-Navarro, P., Jochheim, H., & Muys, B. (2017). The effect of increasing lifespan and recycling rate on carbon storage in wood products from theoretical model to application for the European wood sector. *Mitigation and Adaptation Strategies for Global Change*, *22*(8), 1193–1205. https://doi.org/10.1007/s11027-016-9722-z

Buchholz, T., Hurteau, M. D., Gunn, J., & Saah, D. (2016). A global meta-analysis of forest bioenergy greenhouse gas emission accounting studies. *GCB Bioenergy*, *8*(2), 281–289. https://doi.org/10.1111/gcbb.12245

Butarbutar, T., Köhl, M., & Neupane, P. R. (2016). Harvested wood products and REDD+: looking beyond the forest border. *Carbon Balance and Management*, *11*(1), 4. https://doi.org/10.1186/s13021-016-0046-9

Calderón, J. F. (2000). *Investigación sobre la problemática del sector forestal en Costa Rica*. San José, Costa Rica. Retrieved from http://www.asamblea.go.cr/sd/Publicaciones_T_C/Investigación sobre

la Problemática del Sector Forestal en Costa Rica.pdf
Camacho, A. M. (2015). *Diagnóstico corto sobre las barreras que desalientan el manejo de bosques naturales en Costa Rica y propuestas de solución*. San José, Costa Rica.
Camacho Calvo, M. (2015). *Superficie de bosques susceptible de manejo forestal en Costa Rica y estimación de su potencial productivo*. San José, Costa Rica.
Canadell, J. G., & Schulze, E. D. (2014). Global potential of biospheric carbon management for climate mitigation. *Nature Communications*, *5*, 5282. https://doi.org/10.1038/ncomms6282
Canet, G., Chavarría, M., Gamboa, O., Garita, D., Jiménez, M., Lobo, S., … Valerio, M. (1996). *Información estadística relevante sobre el sector forestal 1972-1995*. San José, Costa Rica.
Cardellini, G., Valada, T., Cornillier, C., Vial, E., Dragoi, M., Goudiaby, V., … Muys, B. (2018). EFO-LCI: A New Life Cycle Inventory Database of Forestry Operations in Europe. *Environmental Management*, *61*(6), 1031–1047. https://doi.org/10.1007/s00267-018-1024-7
Carrillo, O. (2001). *Situación de la industria forestal costarricense*. *Proyecto Mercado e Industrialización de madera ...* (Vol. 006). San José, Costa Rica. Retrieved from http://www.sirefor.go.cr/Documentos/Industria/2001_Carillo_Sit_industria_forestal.pdf
Chacón, A. R., Jiménez, G., Montenegro, J., Sassa, J., & Blanco, K. (2012). *Inventario Nacional de Gases de Efecto Invernadero y Absorcion de Carbono*. San José, Costa Rica.
Chacón, A. R., Jiménez, G., Montenegro, J., Sassa, J., & Kendal Blanco. (2014). *Inventario nacional de emisiones de gases de efecto invernadero y de absorción de carbono 2010*. San José, Costa Rica.
Chave, J., Coomes, D., Jansen, S., Lewis, S. L., Swenson, N. G., & Amy, E. (2009). Towards a worldwide wood economics spectrum, 351–366. https://doi.org/10.1111/j.1461-0248.2009.01285.x
Cherubini, F., Bird, N. D., Cowie, A., Jungmeier, G., Schlamadinger, B., & Woess-Gallasch, S. (2009). Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. *Resources, Conservation and Recycling*, *53*(8), 434–447. https://doi.org/10.1016/j.resconrec.2009.03.013
Cherubini, F., Guest, G., & Strømman, A. H. (2013). Bioenergy from forestry and changes in atmospheric CO2: Reconciling single stand and landscape level approaches. *Journal of Environmental Management*, *129*, 292–301. https://doi.org/10.1016/j.jenvman.2013.07.021
Cherubini, F., Peters, G. P., Berntsen, T., Strømman, A. H., & Hertwich, E. (2011). CO2 emissions from biomass combustion for bioenergy: Atmospheric decay and contribution to global warming. *GCB Bioenergy*, *3*(5), 413–426. https://doi.org/10.1111/j.1757-1707.2011.01102.x
Cláudia Dias, A., Louro, M., Arroja, L., & Capela, I. (2009). Comparison of methods for estimating carbon in harvested wood products. *Biomass and Bioenergy*, *33*(2), 213–222. https://doi.org/10.1016/j.biombioe.2008.07.004
Clavreul, J., Guyonnet, D., & Christensen, T. H. (2012). Quantifying uncertainty in LCA-modelling of waste management systems. *Waste Management*, *32*(12), 2482–2495. https://doi.org/10.1016/j.wasman.2012.07.008
Cole, R. J. (1999). Construction of Alternative Structural Systems. *Building and Environment*, *34*, 335–348.
Cossalter, C., & Pye-Smith, C. (2015). *Fast-wood forestry: myths and realities [Japanese]*. *Fast-wood forestry: myths and realities [Japanese]*. https://doi.org/10.17528/cifor/001701
Côté, W. ., Young, R. ., Risse, K. ., Costanza, A. ., Tonelli, J. ., & Lenocker, C. (2002). A carbon balance method for paper and wood products. *Environmental Pollution*, *116*, S1–S6. https://doi.org/10.1016/S0269-7491(01)00240-8
Cowie, A., Pingoud, K., & Schlamadinger, B. (2006). Stock changes or fluxes? Resolving terminological confusion in the debate on land-use change and forestry. *Climate Policy*, *6*(2), 161–179. https://doi.org/10.1080/14693062.2006.9685593
Culas, R. J. (2012). REDD and forest transition: Tunneling through the environmental Kuznets curve. *Ecological Economics*, *79*, 44–51. https://doi.org/10.1016/j.ecolecon.2012.04.015
Dale, B. E., & Kim, S. (2014). Can the Predictions of Consequential Life Cycle Assessment Be Tested in the Real World? Comment on "Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation..." *Journal of Industrial Ecology*, *18*(3), 466–467. https://doi.org/10.1111/jiec.12151

De la Cruz, F. B., Chanton, J. P., & Barlaz, M. A. (2013). Measurement of carbon storage in landfills from the biogenic carbon content of excavated waste samples. *Waste Management*, *33*(10), 2001–2005. https://doi.org/10.1016/j.wasman.2012.12.012

De Rosa, M., Schmidt, J., Brandão, M., & Pizzol, M. (2017). A flexible parametric model for a balanced account of forest carbon fluxes in LCA. *The International Journal of Life Cycle Assessment*, *22*(2), 172–184. https://doi.org/10.1007/s11367-016-1148-z

Delmas, R., Lacaux, J. P., & Brocard, D. (1995). Laboratoire d'Agrologie (URA CNRS 354), Universitd Paul Sabatier, 118 Route de Narbonne 31062, Toulouse Cedex, France. *Environmental Monitoring and Assessment*, *38*(2), 181–204. https://doi.org/10.1007/BF00546762

DGEC. (1990). *Artículos exportados según peso bruto y valor FOB por sección, capítulo, partida arancelaria y país*. San José, Costa Rica.

DGEC. (1991). *Artículos exportados según peso bruto y valor FOB por sección, capítulo, partida arancelaria y país*. San José, Costa Rica.

DGEC. (1992). *Artículos exportados según peso bruto y valor FOB por sección, capítulo, partida arancelaria y país*. San José, Costa Rica.

DGEC. (1993). *Artículos exportados según peso bruto y valor FOB por sección, capítulo, partida arancelaria y país*. San José, Costa Rica.

DGEC. (1994). *Artículos exportados según peso bruto y valor FOB por sección, capítulo y partida arancelaria*. San José, Costa Rica.

DGEC. (1995). *Artículos exportados según peso bruto y valor FOB por sección, capítulo y partida arancelaria*. San José, Costa Rica.

DGEC. (1996). *Artículos exportados según peso bruto y valor FOB por sección, capítulo y partida arancelaria*. San José, Costa Rica.

DGEC. (1997). *Artículos exportados según peso bruto y valor FOB por sección, capítulo y partida arancelaria*. San José, Costa Rica.

DGF. (1988). *Censo de la industria forestal*. San José, Costa Rica.

DGF. (1993). *Anuario Estadístico de productos forestales de exportación e importación 1991 y 1992*. San José, Costa Rica.

DGF. (1994). *Boletín Estadístico Forestal N5 1990-1993*. San José, Costa Rica.

Dodoo, A., Gustavsson, L., & Sathre, R. (2009). Carbon implications of end-of-life management of building materials. *Resources, Conservation and Recycling*, *53*(5), 276–286. https://doi.org/10.1016/j.resconrec.2008.12.007

Donlan, J., Skog, K., & Byrne, K. A. (2012). Carbon storage in harvested wood products for Ireland 1961-2009. *Biomass and Bioenergy*, *46*, 731–738. https://doi.org/10.1016/j.biombioe.2012.06.018

Downie, A., Lau, D., Cowie, A., & Munroe, P. (2014). Approaches to greenhouse gas accounting methods for biomass carbon. *Biomass and Bioenergy*, *60*, 18–31. https://doi.org/10.1016/j.biombioe.2013.11.009

Edwards, D. P., Tobias, J. A., Sheil, D., Meijaard, E., & Laurance, W. F. (2014). Maintaining ecosystem function and services in logged tropical forests. *Trends in Ecology & Evolution*, *29*(9), 511–520. https://doi.org/10.1016/j.tree.2014.07.003

Ellis, P. W., Gopalakrishna, T., Goodman, R. C., Putz, F. E., Roopsind, A., Umunay, P. M., … Griscom, B. W. (2019). Reduced-impact logging for climate change mitigation (RIL-C) can halve selective logging emissions from tropical forests. *Forest Ecology and Management*. https://doi.org/10.1016/j.foreco.2019.02.004

Ellison, D., Petersson, H., Lundblad, M., & Wikberg, P. E. (2013). The incentive gap: LULUCF and the Kyoto mechanism before and after Durban. *GCB Bioenergy*, *5*(6), 599–622. https://doi.org/10.1111/gcbb.12034

Eshun, J. F., Potting, J., & Leemans, R. (2010). Inventory analysis of the timber industry in Ghana. *International Journal of Life Cycle Assessment*, *15*(7), 715–725. https://doi.org/10.1007/s11367-010-0207-0

Eshun, J. F., Potting, J., & Leemans, R. (2011). LCA of the timber sector in Ghana: Preliminary life cycle impact assessment (LCIA). *International Journal of Life Cycle Assessment*, *16*(7), 625–638. https://doi.org/10.1007/s11367-011-0307-5

European Commission - Joint Research Centre - Institute for Environment and Sustainability. (2010a).

General guide for Life Cycle Assessment - Detailed guidance. *International Reference Life Cycle Data System (ILCD) Handbook* (First edit). Luxembourg. https://doi.org/10.2788/38479

European Commission - Joint Research Centre - Institute for Environment and Sustainability. (2010b). *International Reference Life Cycle Data System (ILCD) Handbook : Review schemes for Life Cycle Assessment (LCA). EUR 24710 EN. European Commission*. https://doi.org/10.2788/39791

FAO. (2013). *Wood Fuels Handbook*. *Journal of Chemical Information and Modeling* (Vol. 53). https://doi.org/10.1017/CBO9781107415324.004

Finegan, B. (2012). Tropical Forestry. *Biotropica*, (1993), 1–28. https://doi.org/10.1111/btp.12004

Fonseca, W., Alice, F. E., Rojas, M., Villalobos, R., & Porras, J. (2016). *Construcción de Funciones Alométricas para Costa Rica en el Contexto del Proyecto de Protección Ambiental a través de la Protección de los Bosques de Centro América*. San José, Costa Rica. Retrieved from http://www.sirefor.go.cr/?wpfb_dl=11

Ford-Robertson, J. (2003). *Implications of harvested wood products accounting: Analysis of issues raised by Parties to the UNFCCC and development of a simple decay approach*. *Analysis of issues raised by parties to the …*. Retrieved from http://maxa.maf.govt.nz/forestry/publications/harvested-wood-products-accounting/harvested-wood-products-accounting-technical-paper.pdf%5Cnhttp://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:Implications+of+Harvested+Wood+Products+Accounting+Ana

Fouquet, M., Levasseur, A., Margni, M., Lebert, A., Lasvaux, S., Souyri, B., … Woloszyn, M. (2015). Methodological challenges and developments in LCA of low energy buildings: Application to biogenic carbon and global warming assessment. *Building and Environment*, *90*, 51–59. https://doi.org/10.1016/j.buildenv.2015.03.022

Frey, C., Penman, J., Hanle, L., Monni, S., & Ogle, S. (2006). Uncertainties. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Vol 1. In H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe (Eds.), *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (pp. 93–134). Hayama, Japan: Intergovernmental Panel on Climate Change.

Gan, K. S., & Massijaya, M. Y. (2014). *Life cycle assessment for environmental product declaration of tropical plywood production in Malaysia and Indonesia*. Yokohama, Japan.

García, A. (2013). Estimación del potencial de producción de biocombustibles en Costa Rica a partir de cultivos energéticos, agroenergéticos y aceites usados, al 2021, 1–196. Retrieved from https://repositoriotec.tec.ac.cr/bitstream/handle/2238/3151/estimacion_potencial_produccion_biocombustibles.pdf?sequence=1&isAllowed=y

Geng, A., Yang, H., Chen, J., & Hong, Y. (2017). Review of carbon storage function of harvested wood products and the potential of wood substitution in greenhouse gas mitigation. *Forest Policy and Economics*, *85*(159), 192–200. https://doi.org/10.1016/j.forpol.2017.08.007

Geng, A., Zhang, H., & Yang, H. (2017). Greenhouse gas reduction and cost efficiency of using wood flooring as an alternative to ceramic tile: A case study in China. *Journal of Cleaner Production*. https://doi.org/10.1016/j.jclepro.2017.08.058

Glew, D., Stringer, L. C., Acquaye, A., & McQueen-Mason, S. (2017). Evaluating the Potential for Harmonized Prediction and Comparison of Disposal-Stage Greenhouse Gas Emissions for Biomaterial Products. *Journal of Industrial Ecology*, *21*(1), 101–115. https://doi.org/10.1111/jiec.12421

Grassi, G., House, J., Kurz, W. A., Cescatti, A., Houghton, R. A., Peters, G. P., … Zaehle, S. (2018). Reconciling global-model estimates and country reporting of anthropogenic forest CO2 sinks. *Nature Climate Change*, *8*(10), 914–920. https://doi.org/10.1038/s41558-018-0283-x

Griscom, B., Ellis, P., & Putz, F. E. (2014). Carbon emissions performance of commercial logging in East Kalimantan, Indonesia. *Global Change Biology*, *20*(3), 923–937. https://doi.org/10.1111/gcb.12386

Guendehou, G. H. S., Koch, M., Hockstad, L., Pipatti, R., Yamada, M., Pipatti, R., & Yamada, M. (2006). Incineration and open burning of waste. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 5: Waste. In H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe (Eds.), *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (Vol. 5, pp. 1–26). Hayama, Japan: Intergovernmental Panel on Climate Change. Retrieved from http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/5_Volume5/V5_4_Ch4_Bio_Treat.pdf%5Cnhttp://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/5_Volume5/V5_5_Ch5_IOB.pdf

Gustavsson, L., Haus, S., Lundblad, M., Lundström, A., Ortiz, C. A., Sathre, R., … Wikberg, P. E. (2017). Climate change effects of forestry and substitution of carbon-intensive materials and fossil fuels. *Renewable and Sustainable Energy Reviews*, *67*, 612–624. https://doi.org/10.1016/j.rser.2016.09.056

Gustavsson, L., & Sathre, R. (2006). Variability in energy and carbon dioxide balances of wood and concrete building materials. *Building and Environment*, *41*(7), 940–951. https://doi.org/10.1016/j.buildenv.2005.04.008

Hafner, A., & Schäfer, S. (2018). Comparative LCA study of different timber and mineral buildings and calculation method for substitution factors on building level. *Journal of Cleaner Production*, *167*, 630–642. https://doi.org/10.1016/j.jclepro.2017.08.203

Harris, N. L., Brown, S., Hagen, S. C., Saatchi, S. S., Petrova, S., Salas, W., … Lotsch, A. (2012). Baseline Map of Carbon Emissions from Deforestation in Tropical Regions. *Science*, *336*(6088), 1573–1576. https://doi.org/10.1126/science.1217962

Hashimoto, S, Nose, M., Obara, T., & Moriguchi, Y. (2012). Wood products : potential carbon sequestration and impact on net carbon emissions of industrialized countries. *Environmental Science & Policy*, *5*(2002), 183–193. https://doi.org/10.1016/S1462-9011(01)00045-4

Hashimoto, Seiji. (2008). Different accounting approaches to harvested wood products in national greenhouse gas inventories: their incentives to achievement of major policy goals. *Environmental Science and Policy*, *11*(8), 756–771. https://doi.org/10.1016/j.envsci.2008.08.002

Hauschild, M., Rosenbaum, R. K., & Olsen, S. I. (2018). *Life Cycle Assessment: Theory and Practice*. (M. Z. Hauschild, R. K. Rosenbaum, & S. I. Olsen, Eds.). Cham, Switzerland: Springer International Publishing. https://doi.org/10.1007/978-3-319-56475-3

Heijungs, R., & Huijbregts, M. a J. (2004). A review of approaches to treat uncertainty in LCA. *IEMSs 2004 International Congress*, 8. https://doi.org/http://www.iemss.org/iemss2004/pdf/lca/heijarev.pdf

Heijungs, R., & Lenzen, M. (2014). Error propagation methods for LCA — a comparison, 1445–1461. https://doi.org/10.1007/s11367-014-0751-0

Helin, T., Sokka, L., Soimakallio, S., Pingoud, K., & Pajula, T. (2013). Approaches for inclusion of forest carbon cycle in life cycle assessment - A review. *GCB Bioenergy*, *5*(5), 475–486. https://doi.org/10.1111/gcbb.12016

Hérault, B., Beauchêne, J., Muller, F., Wagner, F., Baraloto, C., Blanc, L., & Martin, J. M. (2010). Modeling decay rates of dead wood in a neotropical forest. *Oecologia*, *164*(1), 243–251. https://doi.org/10.1007/s00442-010-1602-8

Houghton, R. A. (2012). Carbon emissions and the drivers of deforestation and forest degradation in the tropics. *Current Opinion in Environmental Sustainability*, *4*(6), 597–603. https://doi.org/10.1016/j.cosust.2012.06.006

Houghton, R. A., Byers, B., & Nassikas, A. A. (2015). A role for tropical forests in stabilizing atmospheric CO2. *Nature Climate Change*, *5*(12), 1022–1023. https://doi.org/10.1038/nclimate2869

Houghton, R. A., Skole, D. L., Nobre, C. A., Hackler, J. L., Lawrence, K. T., & Chomentowski, W. H. (2000). Annual fluxes of carbon from deforestation and regrowth in the Brazilian Amazon. *Nature*, *403*(6767), 301–304. https://doi.org/10.1038/35002062

Houghton, Richard A. (2013). The emissions of carbon from deforestation and degradation in the tropics: past trends and future potential. *Carbon Management*, *4*(5), 539–546. https://doi.org/10.4155/cmt.13.41

Huijbregts, M. A. J. (1998). Application of uncertainty and variability in LCA. *The International Journal of Life Cycle Assessment*, *3*(5), 273–280. https://doi.org/10.1007/BF02979835

Huijbregts, M. A. J., Steinmann, Z. J. N., Elshout, P. M. F., Stam, G., Verones, F., Vieira, M., … van Zelm, R. (2017). ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. *The International Journal of Life Cycle Assessment*, *22*(2), 138–147. https://doi.org/10.1007/s11367-016-1246-y

Igos, E. (2018). How to treat uncertainties in life cycle assessment studies ? *The International Journal of Life Cycle Assessment*. https://doi.org/https://doi.org/10.1007/s11367-018-1477-1
IMN. (2011). *Factores de emisión de gases de efecto invernadero - Primera Edición 2011*. San José, Costa Rica. Retrieved from http://cglobal.imn.ac.cr/index.php/cambio-climatico/factores-de-emision-de-gases-de-efecto-invernadero/
IMN. (2014). *Factores de emisión de gases de efecto invernadero - Cuarta Edición 2014*. San José, Costa Rica. Retrieved from http://cglobal.imn.ac.cr/index.php/cambio-climatico/factores-de-emision-de-gases-de-efecto-invernadero/
IMN. (2015). *Factores de emisión de gases de efecto invernadero - Quinta Edición 2015*. San José, Costa Rica. Retrieved from http://cglobal.imn.ac.cr/index.php/cambio-climatico/factores-de-emision-de-gases-de-efecto-invernadero/
IMN. (2016). *Factores de emisión de gases de efecto invernadero - Sexta Edición 2016*. San José, Costa Rica. Retrieved from http://cglobal.imn.ac.cr/index.php/cambio-climatico/factores-de-emision-de-gases-de-efecto-invernadero/
IMN. (2017). *Factores de emisión de gases de efecto invernadero - Sétima Edición 2017*. San José, Costa Rica. Retrieved from http://www.mapama.gob.es/es/cambio-climatico/temas/mitigacion-politicas-y-medidas/factores_emision_tcm7-359395.pdf
IMN. (2018). *Factores de emisión gases efecto invernadero - Octava Edición 2018*. *Instituto Meteorológico Nacional*. San José, Costa Rica. Retrieved from http://energiaabierta.cl/visualizaciones/factor-de-emision-sic-sing/
INEC. (2015). *VI Censo Nacional Agropecuario: resultados generales*. San José, Costa Rica.
INEC. (2018). Estadísticas de la construcción: Área en m2 de las construcciones y ampliaciones por años según provincia, tipo y destino de la obra 1983-2017. Retrieved August 23, 2018, from http://www.inec.go.cr/economia/estadisticas-de-la-construccion
Ingerson, A. (2011). Carbon storage potential of harvested wood: Summary and policy implications. *Mitigation and Adaptation Strategies for Global Change*, *16*(3), 307–323. https://doi.org/10.1007/s11027-010-9267-5
Intergovernmental Panel on Climate Change. (2014). Agriculture, Forestry and Other Land Use (AFOLU). In *Climate Change 2014 Mitigation of Climate Change* (pp. 811–922). Cambridge: Cambridge University Press. https://doi.org/10.1017/CBO9781107415416.017
Iordan, C.-M., Hu, X., Arvesen, A., Kauppi, P., & Cherubini, F. (2018). Contribution of forest wood products to negative emissions: historical comparative analysis from 1960 to 2015 in Norway, Sweden and Finland. *Carbon Balance and Management*, *13*(1), 12. https://doi.org/10.1186/s13021-018-0101-9
IPCC. (2014). *2013 Revised Supplementary Methods and Good Practice Guidance Arising from the Kyoto Protocol Task Force on National Greenhouse Gas Inventories*. (T. Hiraishi, T. Krug, K. Tanabe, N. Srivastava, J. Baasansuren, M. Fukuda, & T. Troxler, Eds.). Switzerland: Intergovernmental Panel on Climate Change. Retrieved from http://www.ipcc-nggip.iges.or.jp
Iritani, D. R., Silva, D. A. L., Saavedra, Y. M. B., Grael, P. F. F., & Ometto, A. R. (2015). Sustainable strategies analysis through Life Cycle Assessment: A case study in a furniture industry. *Journal of Cleaner Production*, *96*, 308–318. https://doi.org/10.1016/j.jclepro.2014.05.029
ITTO. (2015). *Voluntary guidelines for the sustainable management of natural tropical forests*. *Policy development series* (Vol. 20).
Jadin, I., Meyfroidt, P., & Lambin, E. F. (2016). International trade, and land use intensification and spatial reorganization explain Costa Rica’s forest transition. *Environmental Research Letters*, *11*(3), 35005. https://doi.org/10.1088/1748-9326/11/3/035005
Jadin, Isaline, Meyfroidt, P., Zamora Pereira, J., & Lambin, E. (2016). Unexpected Interactions between Agricultural and Forest Sectors through International Trade: Wood Pallets and Agricultural Exports in Costa Rica. *Land*, *6*(1), 1. https://doi.org/10.3390/land6010001
Jakes, J. E., Arzola, X., Bergman, R., Ciesielski, P., Hunt, C. G., Rahbar, N., … Zelinka, S. L. (2016). Not Just Lumber—Using Wood in the Sustainable Future of Materials, Chemicals, and Fuels. *Jom*, *68*(9), 2395–2404. https://doi.org/10.1007/s11837-016-2026-7
Jankowsky, I. P., Galina, I. C. M., & Andrade, A. de. (2015). Life-cycle assessment for environmental product declarations of IPE and cumaru decking strips produced in Brazil. *Life-Cycle Assessment*

for Environmental Product Declarations of IPE and Cumaru Decking Strips Produced in Brazil, (July), 46-pp. Retrieved from http://www.itto.int/sfm_detail/id=4619

Jasinevičius, G., Lindner, M., Cienciala, E., & Tykkyläinen, M. (2018). Carbon Accounting in Harvested Wood Products: Assessment Using Material Flow Analysis Resulting in Larger Pools Compared to the IPCC Default Method. *Journal of Industrial Ecology*, *22*(1), 121–131. https://doi.org/10.1111/jiec.12538

Ji, C., Cao, W., Chen, Y., & Yang, H. (2016). Carbon balance and contribution of harvested wood products in China based on the production approach of the intergovernmental panel on climate change. *International Journal of Environmental Research and Public Health*, *13*(11). https://doi.org/10.3390/ijerph13111132

Johnson, E. (2009). Goodbye to carbon neutral: Getting biomass footprints right. *Environmental Impact Assessment Review*, *29*(3), 165–168. https://doi.org/10.1016/j.eiar.2008.11.002

Johnston, C. M. T., & Radeloff, V. C. (2019). Global mitigation potential of carbon stored in harvested wood products. *Proceedings of the National Academy of Sciences*, 201904231. https://doi.org/10.1073/pnas.1904231116

Keenan, R. J., Reams, G. A., Achard, F., de Freitas, J. V., Grainger, A., & Lindquist, E. (2015). Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management*. https://doi.org/10.1016/j.foreco.2015.06.014

Keith, H., Lindenmayer, D., Macintosh, A., & Mackey, B. (2015). Under what circumstances do wood products from native forests benefit climate change mitigation? *PLoS ONE*, *10*(10), 1–23. https://doi.org/10.1371/journal.pone.0139640

Khun, V., & Sasaki, N. (2014). Cumulative carbon fluxes due to selective logging in Southeast Asia. *Low Carbon Economy*, *5*(December), 180–191. https://doi.org/10.4236/lce.2014.54018

Klein, D., Wolf, C., Schulz, C., & Weber-Blaschke, G. (2015). 20 years of life cycle assessment (LCA) in the forestry sector: state of the art and a methodical proposal for the LCA of forest production. *The International Journal of Life Cycle Assessment*, *20*(4), 556–575. https://doi.org/10.1007/s11367-015-0847-1

Klitkou, A., Bolwig, S., Hansen, T., & Wessberg, N. (2015). The role of lock-in mechanisms in transition processes: The case of energy for road transport. *Environmental Innovation and Societal Transitions*, *16*, 22–37. https://doi.org/10.1016/j.eist.2015.07.005

Knauf, M. (2015). A multi-tiered approach for assessing the forestry and wood products industries' impact on the carbon balance. *Carbon Balance and Management*, *10*(1), 4. https://doi.org/10.1186/s13021-015-0014-9

Knauf, M., Köhl, M., Mues, V., Olschofsky, K., & Frühwald, A. (2015). Modeling the CO2-effects of forest management and wood usage on a regional basis. *Carbon Balance and Management*, *10*(1), 13. https://doi.org/10.1186/s13021-015-0024-7

Köhl, M., Lasco, R., Cifuentes, M., Jonsson, Ö., Korhonen, K. T., Mundhenk, P., … Stinson, G. (2015). Changes in forest production, biomass and carbon: Results from the 2015 UN FAO Global Forest Resource Assessment. *Forest Ecology and Management*. https://doi.org/10.1016/j.foreco.2015.05.036

Laurance, W. F., Goosem, M., & Laurance, S. G. W. (2009). Impacts of roads and linear clearings on tropical forests. *Trends in Ecology and Evolution*, *24*(12), 659–669. https://doi.org/10.1016/j.tree.2009.06.009

Law, B. E., & Harmon, M. E. (2011). Forest sector carbon management, measurement and verification, and discussion of policy related to climate change. *Carbon Management*, *2*(1), 73–84. https://doi.org/10.4155/cmt.10.40

Leskinen, P., Cardellini, G., González García, S., Hurmekoski, E., Sathre, R., Seppälä, J., … Verkerk, H. (2018). *Substitution effects of wood-based products in climate change mitigation*. *From Science to Policy* (Vol. 7). Retrieved from https://www.efi.int/publications-bank/substitution-effects-wood-based-products-climate-change-mitigation

Levasseur, A., Lesage, P., Margni, M., Deschěnes, L., & Samson, R. (2010). Considering time in LCA: Dynamic LCA and its application to global warming impact assessments. *Environmental Science and Technology*, *44*(8), 3169–3174. https://doi.org/10.1021/es9030003

Levasseur, A., Lesage, P., Margni, M., & Samson, R. (2013). Biogenic Carbon and Temporary Storage

Addressed with Dynamic Life Cycle Assessment. *Journal of Industrial Ecology*, *17*(1), 117–128. https://doi.org/10.1111/j.1530-9290.2012.00503.x

Lim, B., Brown, S., & Schlamadinger, B. (1999). Carbon accounting for forest harvesting and wood products: review and evaluation of different approaches. *Environmental Science & Policy*, *2*(1999), 207–216. https://doi.org/10.1016/S1462-9011(99)00031-3

Lippke, B., Oneil, E., Harrison, R., Skog, K., Gustavsson, L., & Sathre, R. (2011). Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. *Carbon Management*, *2*(3), 303–333. https://doi.org/10.4155/cmt.11.24

Lippke, B., Wilson, J., Meil, J., & Taylor, A. (2010). Characterizing the importance of carbon stored in wood products. *Wood and Fiber Science*, *42*(Lci), 5–14.

Liu, W., Zhang, Z., Xie, X., Yu, Z., Gadow, K. von, Xu, J., … Yang, Y. (2017). Analysis of the Global Warming Potential of Biogenic CO2 Emission in Life Cycle Assessments. *Scientific Reports, Published Online: 3 January 2017; | Doi:10.1038/Srep39857*, *7*(January), 484–489. https://doi.org/10.1038/srep39857

Lo, S., Ma, H., & Lo, S. (2005). Quantifying and reducing uncertainty in life cycle assessment using the Bayesian Monte Carlo method. *Science of The Total Environment*, *340*(1–3), 23–33. https://doi.org/10.1016/j.scitotenv.2004.08.020

Lun, F., Li, W., & Liu, Y. (2012). Complete forest carbon cycle and budget in China, 1999-2008. *Forest Ecology and Management*, *264*, 81–89. https://doi.org/10.1016/j.foreco.2011.10.004

Lundmark, T., Bergh, J., Nordin, A., Fahlvik, N., & Poudel, B. C. (2016). Comparison of carbon balances between continuous-cover and clear-cut forestry in Sweden. *Ambio*, *45*(2), 203–213. https://doi.org/10.1007/s13280-015-0756-3

Mantau, U. (2015). Wood flow analysis: Quantification of resource potentials, cascades and carbon effects. *Biomass and Bioenergy*, *79*, 28–38. https://doi.org/10.1016/j.biombioe.2014.08.013

Marland, E., & Marland, G. (2003). The treatment of long-lived, carbon-containing products in inventories of carbon dioxide emissions to the atmosphere. *Environmental Science and Policy*, *6*(2), 139–152. https://doi.org/10.1016/S1462-9011(03)00003-0

Marland, E. S., Stellar, K., & Marland, G. H. (2010). A distributed approach to accounting for carbon in wood products. *Mitigation and Adaptation Strategies for Global Change*, *15*(1), 71–91. https://doi.org/10.1007/s11027-009-9205-6

Martin, P. A., Newton, A. C., Pfeifer, M., Khoo, M. S., & Bullock, J. M. (2015). Impacts of tropical selective logging on carbon storage and tree species richness: A meta-analysis. *Forest Ecology and Management*, *356*, 224–233. https://doi.org/10.1016/j.foreco.2015.07.010

Matsumoto, M., Oka, H., Mitsuda, Y., Hashimoto, S., Kayo, C., Tsunetsugu, Y., & Tonosaki, M. (2016). Potential contributions of forestry and wood use to climate change mitigation in Japan. *Journal of Forest Research*, *21*(5), 211–222. https://doi.org/10.1093/arisoc/aox016

Mckenzie, T. A. (2003). *Tendencias y perspectivas para el sector forestal nacional de Costa Rica hasta el año 2020*. *Proyecto Tendencias y Perspectivas del Sector Forestal de América Latina*. San Jose, Costa Rica.

Medeiros, D. L., Tavares, A. O. do C., Rapôso, Á. L. Q. R. e S., & Kiperstok, A. (2017). Life cycle assessment in the furniture industry: the case study of an office cabinet. *The International Journal of Life Cycle Assessment*, *2*. https://doi.org/10.1007/s11367-017-1370-3

Merry, F., Soares-Filho, B., Nepstad, D., Amacher, G., & Rodrigues, H. (2009). Balancing conservation and economic sustainability: The future of the amazon timber industry. *Environmental Management*, *44*(3), 395–407. https://doi.org/10.1007/s00267-009-9337-1

Micales, J. A., & Skog, K. E. (1997). The decomposition of forest products in landfills. *International Biodeterioration and Biodegradation*, *39*(2–3), 145–155. https://doi.org/10.1016/S0964-8305(97)83389-6

MINAE. (2002). *Principios , Criterios e Indicadores de Sostenibilidad para el Manejo de Bosques Naturales en Costa Rica*. San José, Costa Rica.

MINAE. (2011). *Reporte Estadístico Forestal 2011*. San José, Costa Rica. Retrieved from http://www.sirefor.go.cr/?wpfb_dl=4

MINAE. (2012). *Reporte Estadístico Forestal 2012*. San José, Costa Rica. Retrieved from http://www.sirefor.go.cr/?wpfb_dl=4

MINAE. (2013). *Reporte Estadístico Forestal 2013*. San José, Costa Rica. Retrieved from http://www.sirefor.go.cr/?wpfb_dl=4

MINAET. (2009). Estándares de sostenibilidad para manejo de bosques naturales: Código de prácticas. In *Resolución R-SINAC-021-2009* (pp. 24–27). San José, Costa Rica. Retrieved from http://www.sinac.go.cr/ES/normativa/Resoluciones/R-SINAC-021-2009 Estándares de Sostenibilidad para Manejo de Bosques Naturales.pdf

Miner, R. (2010). *Impact of the global forest industry on atmospheric greenhouse gases*. *Forestry*. Retrieved from http://library.wur.nl/WebQuery/clc/1941440

Miner, Reid. (2006). The 100-Year Method for Forecasting Carbon Sequestration in Forest Products in Use. *Mitigation and Adaptation Strategies for Global Change*, 20. https://doi.org/10.1007/s11027-006-4496-3

Mohren, F. (2019). Use tropical forests or lose them. *Nature Sustainability*, *2*(1), 12–13. https://doi.org/10.1038/s41893-018-0211-0

Mohren, G. M. J., Hasenauer, H., Köhl, M., & Nabuurs, G. J. (2012). Forest inventories for carbon change assessments. *Current Opinion in Environmental Sustainability*, *4*(6), 686–695. https://doi.org/10.1016/j.cosust.2012.10.002

Moya, R. (2004). Gmelina arborea en Costa Rica. *Bois et Forêts Des Tropiques*, *279*(1), 1–11.

Muralikrishna, I. V., & Manickam, V. (2017). Natural Resource Management and Biodiversity Conservation. In *Environmental Management*. https://doi.org/10.1016/b978-0-12-811989-1.00003-8

Murphy, R. J. (2004). Review of Information on Life Cycle Analysis of Tropical Timber Products, *04*(August), 66.

Nabuurs, G. J., Arets, E. J. M. M., & Schelhaas, M. J. (2017). European forests show no carbon debt, only a long parity effect. *Forest Policy and Economics*, *75*, 120–125. https://doi.org/10.1016/j.forpol.2016.10.009

Newell, J. P., & Vos, R. O. (2012). Accounting for forest carbon pool dynamics in product carbon footprints: Challenges and opportunities. *Environmental Impact Assessment Review*, *37*, 23–36. https://doi.org/10.1016/j.eiar.2012.03.005

Numazawa, C. T. D. (2018). *Material flow analysis and CO2 footprint in lumber from managed Brazilian Amazon rainforests*. Polytechnic School of the University of São Paulo. Retrieved from http://www.teses.usp.br/teses/disponiveis/3/3146/tde-31012019-073206/en.php

Numazawa, C. T. D., Numazawa, S., Pacca, S., & John, V. M. (2017). Logging residues and CO2 of Brazilian Amazon timber: Two case studies of forest harvesting. *Resources, Conservation and Recycling*, *122*, 280–285. https://doi.org/10.1016/j.resconrec.2017.02.016

O'Dwyer, J., Walshe, D., & Byrne, K. A. (2018). Wood waste decomposition in landfills: An assessment of current knowledge and implications for emissions reporting. *Waste Management*, *73*, 181–188. https://doi.org/10.1016/j.wasman.2017.12.002

Ofoegbu, C., Ogbonnaya, S., & Babalola, F. D. (2014). Sawmill Conversion Efficiency and Wood Recovery of. *Agriculture & Forestry*, *60*(1), 105–113.

Oliver, C. D., & Mesznik, R. (2006). Investing in forestry: opportunities and pitfalls of intensive plantations and other alternatives. *Journal of Sustainable Forestry*, *21*(4), 97–111. https://doi.org/10.1300/J091v21n04_07

Pagiola, S. (2008). Payments for environmental services in Costa Rica. *Ecological Economics*, *65*(4), 712–724. https://doi.org/10.1016/j.ecolecon.2007.07.033

Palisade. (2016). @Risk – Risk Analysis and Simulation Add-In for Microsoft® Excel. 798 Cascadilla St. Ithaca, NY USA 14850: Palisade Corporation.

Parigiani, J., Desai, A., Mariki, R., & Miner, R. (2011). The Carbon Footprint of an East African Forestry Enterprise. *Journal of Sustainable Development*, *4*(3), 152–162. https://doi.org/10.5539/jsd.v4n3p152

Parker, C., Merger, E., Streck, C., Tennigkeit, T., & Wilkes, A. (2014). *The land-use sector within the post-2020 climate regime*. https://doi.org/10.6027/TN2014-520

Payn, T., Carnus, J. M., Freer-Smith, P., Kimberley, M., Kollert, W., Liu, S., … Wingfield, M. J. (2015). Changes in planted forests and future global implications. *Forest Ecology and Management*. https://doi.org/10.1016/j.foreco.2015.06.021

Pearson, T. R. H., Brown, S., & Casarim, F. M. (2014). Carbon emissions from tropical forest degradation caused by logging. *Environmental Research Letters*, *9*(3), 034017. https://doi.org/10.1088/1748-9326/9/3/034017

Pearson, T. R. H., Brown, S., Murray, L., & Sidman, G. (2017). Greenhouse gas emissions from tropical forest degradation: an underestimated source. *Carbon Balance and Management*, *12*(1), 3. https://doi.org/10.1186/s13021-017-0072-2

Pedroni, L., Espejo, A., & Villegas, J. F. (2015). *Nivel de referencia de emisiones y absorciones forestales de Costa Rica (ERPD component)*. San José, Costa Rica. Retrieved from http://reddcr.go.cr/sites/default/files/centro-de-documentacion/3.4.6-_nivel_de_referencia_de_costa_rica_-_informe_final_18.09.2015.pdf

Perez-Garcia, J., Lippke, B., Comnick, J., & Manriquez, C. (2005). An assessment of carbon pools, storage, and wood products market substitution using life-cycle analysis results. *Wood and Fiber Science*, *37*(Special Issue), 140–148.

Pilli, R., Fiorese, G., & Grassi, G. (2015). EU mitigation potential of harvested wood products. *Carbon Balance and Management*, *10*(1), 6. https://doi.org/10.1186/s13021-015-0016-7

Pingoud, K., Pohjola, J., & Valsta, L. (2010). Assessing the integrated climatic impacts of forestry and wood products. *Silva Fennica*, *44*(1), 155–175. https://doi.org/10.14214/sf.166

Pingoud, K., Skog, K. E., Martino, D. L., Tonosaki, M., & Xiaoquan, Z. (2006). Harvested Wood Products. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use. In H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe (Eds.), *IPCC Guidelines for National Greenhouse Gas Inventories* (pp. 1–33). Hayama, Japan. Retrieved from https://www.ipcc-nggip.iges.or.jp/public/2006gl/

Pingoud, K., & Wagner, F. (2006). Methane emissions from landfills and carbon dynamics of harvested wood products: The first-order decay revisited. *Mitigation and Adaptation Strategies for Global Change*, *11*(5–6), 961–978. https://doi.org/10.1007/s11027-006-9029-6

Pipatti, R., Svardal, P., Joao Wagner Silva Alves, Gao, Q., López, C., Mareckova, K., … Yamada, M. (2006). *Solid waste disposal. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 5: Waste.* (H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe, Eds.), *2006 IPCC Guidelines for National Greenhouse Gas Inventories*. Hayama, Japan. Retrieved from https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/5_Volume5/V5_3_Ch3_SWDS.pdf

Piponiot, C., Cabon, A., Descroix, L., Dourdain, A., Mazzei, L., Ouliac, B., … Hérault, B. (2016). A methodological framework to assess the carbon balance of tropical managed forests. *Carbon Balance and Management*, *11*(1), 15. https://doi.org/10.1186/s13021-016-0056-7

Piponiot, C., Derroire, G., Descroix, L., Mazzei, L., Rutishauser, E., Sist, P., & Hérault, B. (2018). Assessing timber volume recovery after disturbance in tropical forests - a new modelling framework. *Ecological Modelling*, *in press*(December 2017), 353–369. https://doi.org/10.1016/j.ecolmodel.2018.05.023

Plevin, R. J., Delucchi, M. A., & Creutzig, F. (2014a). Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation Benefits Misleads Policy Makers. *Journal of Industrial Ecology*, *18*(1), 73–83. https://doi.org/10.1111/jiec.12074

Plevin, R. J., Delucchi, M. A., & Creutzig, F. (2014b). Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation Benefits Misleads Policy Makers. *Journal of Industrial Ecology*, *18*(1), 73–83. https://doi.org/10.1111/jiec.12074

Poker, J., & MacDicken, K. (2016). Tropical Forest Resources: Facts and Tables. In *Tropical Forestry Handbook* (pp. 3–45). Berlin, Heidelberg: Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-54601-3_7

PROCOMER. (2018). Exportaciones de 1998 al 2016. Retrieved August 23, 2018, from http://sistemas.procomer.go.cr/estadisticas/inicio.aspx

Programa REDD/CCAD-GIZ -SINAC. (2015). *Inventario Nacional Forestal de Costa Rica 2014-2015*. San José, Costa Rica.

Puettmann, M., & Bergman, R. (2010). Cradle-to-gate life-cycle inventory of US wood products production: CORRIM Phase I and Phase II products. *Wood and Fiber …*, *1132*(April 2009), 15–28. Retrieved from http://swst.metapress.com/index/U525K97685565245.pdf

Putz, F E, & Romero, C. (2015). *Futures of tropical production forests*. *Center for International*

Forestry Research. https://doi.org/10.17528/cifor/005766
Putz, Francis E., Zuidema, P. A., Pinard, M. A., Boot, R. G. A., Sayer, J. A., Sheil, D., … Vanclay, J. K. (2008). Improved Tropical Forest Management for Carbon Retention. *PLoS Biology*, *6*(7), e166. https://doi.org/10.1371/journal.pbio.0060166
Putz, Francis E., Zuidema, P. A., Synnott, T., Peña-Claros, M., Pinard, M. A., Sheil, D., … Zagt, R. (2012). Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. *Conservation Letters*, *5*(4), 296–303. https://doi.org/10.1111/j.1755-263X.2012.00242.x
Quéré, C., Andrew, R., Friedlingstein, P., Sitch, S., Hauck, J., Pongratz, J., … Zheng, B. (2018). Global Carbon Budget 2018. *Earth System Science Data*, *10*(4), 2141–2194. https://doi.org/10.5194/essd-10-2141-2018
Ramage, M. H., Burridge, H., Busse-Wicher, M., Fereday, G., Reynolds, T., Shah, D. U., … Scherman, O. (2017). The wood from the trees: The use of timber in construction. *Renewable and Sustainable Energy Reviews*, *68*(October 2015), 333–359. https://doi.org/10.1016/j.rser.2016.09.107
Ramasamy, G., Ratnasingam, J., Bakar, E. S., Halis, R., & Muttiah, N. (2015). Assessment of Environmental Emissions from Sawmilling Activity in Malaysia. *BioResources*, *10*(4), 6643–6662. https://doi.org/10.15376/biores.10.4.6643-6662
Ramírez Hernández, F., Carazo Fernández, E., Roldán Villalobos, C., & Villegas Barahona, G. (2007). *Encuesta de oferta y consumo energético nacional a partir de la biomasa en Costa Rica Año 2006*. San José, Costa Rica.
Ratnasingam, J., Ramasamy, G., Toong, W., Senin, A. L., Kusno, M. A., & Muttiah, N. (2015). An Assessment of the Carbon Footprint of Tropical Hardwood Sawn Timber Production, *10*, 5174–5190.
Reap, J., Roman, F., Duncan, S., & Bras, B. (2008). A survey of unresolved problems in life cycle assessment. Part 1: Goal and scope and inventory analysis. *International Journal of Life Cycle Assessment*, *13*(4), 290–300. https://doi.org/10.1007/s11367-008-0008-x
Rinawati, D. I., Sari, D. P., & Prayodha, A. C. (2018). Eco-efficiency Analysis of Furniture Product Using Life Cycle Assessment, *08005*, 1–5.
Roitman, I., Vanclay, J. K., Hay, J. D., & Felfili, J. M. (2016). Dynamic equilibrium and decelerating growth of a seasonal Neotropical gallery forest in the Brazilian savanna. *Journal of Tropical Ecology*, *32*(3), 193–200. https://doi.org/10.1017/S026646741600016X
Røyne, F., Peñaloza, D., Sandin, G., Berlin, J., & Svanström, M. (2016). Climate impact assessment in life cycle assessments of forest products: Implications of method choice for results and decision-making. *Journal of Cleaner Production*, *116*, 90–99. https://doi.org/10.1016/j.jclepro.2016.01.009
Ruiz García, A. (2002). *Fijación y almacenamiento de carbono en sistemas silvopastoriles y competitividad económica en Matiguás, Nicaragua*. Centro Agronómico Tropical de Investigación y Enseñanza.
Runting, R. K., Ruslandi, Griscom, B. W., Struebig, M. J., Satar, M., Meijaard, E., … Venter, O. (2019). Larger gains from improved management over sparing–sharing for tropical forests. *Nature Sustainability*, *2*(1), 53–61. https://doi.org/10.1038/s41893-018-0203-0
Rüter, S., Werner, F., Forsell, N., Prins, C., Vial, E., & Levet, A. L. (2016). *ClimWood2030 "climate benefits of material substitution by forest biomass and harvested wood products: perspective 2030" final report*. *Thunen Report* (Vol. 42). https://doi.org/10.3220/REP1468328990000
Rutishauser, E., Hérault, B., Baraloto, C., Blanc, L., Descroix, L., Sotta, E. D., … Sist, P. (2015). Rapid tree carbon stock recovery in managed Amazonian forests. *Current Biology*, *25*(18), R787–R788. https://doi.org/10.1016/j.cub.2015.07.034
Sandin, G., Peters, G. M., & Svanström, M. (2016). *Life Cycle Assessment of Forest Products - Challenges and Solutions*. https://doi.org/10.1007/978-3-319-44027-9
Santamaría, O. J. (2015). *Los acervos de carbono en productos de madera y derivados en Costa Rica*. San Jose, Costa Rica. Retrieved from http://reddcr.go.cr/sites/default/files/centro-de-documentacion/fonafifo_2015_-_informe_final_aumentando_los_acervos_de_carbono_en_productos_de_madera_y_derivados_en_costa_rica.pdf
Sasaki, N., Asner, G. P., Pan, Y., Knorr, W., Durst, P. B., Ma, H. O., … Putz, F. E. (2016). Sustainable

Management of Tropical Forests Can Reduce Carbon Emissions and Stabilize Timber Production. *Frontiers in Environmental Science*, *4*(August), 50. https://doi.org/10.3389/fenvs.2016.00050

Sasaki, N., Chheng, K., & Ty, S. (2012). Managing production forests for timber production and carbon emission reductions under the REDD+ scheme. *Environmental Science and Policy*, *23*(December 2011), 35–44. https://doi.org/10.1016/j.envsci.2012.06.009

Sathre, R., & Gustavsson, L. (2006). Energy and carbon balances of wood cascade chains. *Resources, Conservation and Recycling*, *47*(4), 332–355. https://doi.org/10.1016/j.resconrec.2005.12.008

Sathre, R., & O'Connor, J. (2010). Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environmental Science and Policy*, *13*(2), 104–114. https://doi.org/10.1016/j.envsci.2009.12.005

Scarlat, N., Dallemand, J. F., Monforti-Ferrario, F., & Nita, V. (2015). The role of biomass and bioenergy in a future bioeconomy: Policies and facts. *Environmental Development*, *15*, 3–34. https://doi.org/10.1016/j.envdev.2015.03.006

Schlamadinger, B., Apps, M., Bohlin, F., Gustavsson, L., Jungmeier, G., Marland, G., … Savolainen, I. (1997). Towards a standard methodology for Greenhouse Gas Balances of Bioenergy Systems in comparison with fossil energy systems. *Biomass and Bioenergy*, *13*(6), 359–375. https://doi.org/10.1016/S0961-9534(97)10032-0

SEPSE. (2018). Balances energético nacional de Costa Rica: 2012-2016. Retrieved January 14, 2019, from https://sepse.go.cr/ciena/balances-energeticos/#1544807135084-f2a106c1-7ce4

Serrano, R., & Moya, R. (2011). Procesamiento, uso y mercado de la madera en Costa Rica: aspectos históricos y análisis crítico. *Revista Forestal Mesoamericana Kurú*, *8*(21), 1–49.

Sessions, J. (2007). *Harvesting Operations in the Tropics*. New York: Springer-Verlag Berlin Heidelberg. https://doi.org/10.1007/978-3-540-46391-7_9

Shearman, P., Bryan, J., & Laurance, W. F. (2012). Are we approaching "peak timber" in the tropics? *Biological Conservation*, *151*(1), 17–21. https://doi.org/10.1016/j.biocon.2011.10.036

Sikkema, R., Junginger, M., McFarlane, P., & Faaij, A. (2013). The GHG contribution of the cascaded use of harvested wood products in comparison with the use of wood for energy-A case study on available forest resources in Canada. *Environmental Science and Policy*, *31*, 96–108. https://doi.org/10.1016/j.envsci.2013.03.007

Skog, K. E., & Nicholson, G. a. (1998). Carbon Cycling Through Wood Products: The Role of Wood and Paper Products in Carbon Sequestration. *Forest Products Journal*, *48*(7–8), 75–83.

Skog, K. E., Pingoud, K., & Smith, J. E. (2004). A method countries can use to estimate changes in carbon stored in harvested wood products and the uncertainty of such estimates. *Environmental Management*, *33*(SUPPL. 1), 65–74. https://doi.org/10.1007/s00267-003-9118-1

Smith, P., Haberl, H., Popp, A., Erb, K. H., Lauk, C., Harper, R., … Rose, S. (2013). How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals? *Global Change Biology*, *19*(8), 2285–2302. https://doi.org/10.1111/gcb.12160

Solano, J. (2017). *Impacto potencial sobre el cambio climático de las tarimas de madera elaboradas en la Región Huetar Norte de Costa Rica a través de un Análisis de Ciclo de Vida (ACV)*. Universidad Nacional de Costa Rica.

Solera, J. M. (2014). *Estudio sobre el estado de la Industria de Transformación Secundaria de la Madera en la Región del Valle Central Occidental*. San José, Costa Rica. Retrieved from http://www.fonafifo.go.cr/documentacion/biblioteca/consultorias_investigaciones/FF_informe_madera.pdf

Suter, F., Steubing, B., & Hellweg, S. (2016). Life Cycle Impacts and Benefits of Wood along the Value Chain: The Case of Switzerland. *Journal of Industrial Ecology*, *00*(0), 1–13. https://doi.org/10.1111/jiec.12486

Taeroe, A., Mustapha, W. F., Stupak, I., & Raulund-Rasmussen, K. (2017). Do forests best mitigate CO2 emissions to the atmosphere by setting them aside for maximization of carbon storage or by management for fossil fuel substitution? *Journal of Environmental Management*, *197*, 117–129. https://doi.org/10.1016/j.jenvman.2017.03.051

Ter-Mikaelian, M. T., Colombo, S. J., & Chen, J. (2015). The Burning Question: Does Forest Bioenergy Reduce Carbon Emissions? A Review of Common Misconceptions about Forest Carbon

Accounting. *Journal of Forestry*, *113*(1), 57–68. https://doi.org/10.5849/jof.14-016
Tomaselli, I. (2007). The allure of plantations. *ITTO Tropical Forest Update*, 10–13.
Villalobos, A., & Navarro, G. A. (2017). Patterns of Development, Forestry Sector and Green Growth Minimalism in Costa Rica: an historical perspective. *Revista Forestal Mesoamerica Kuru-Rfmk*, *14*(35), 2–11. https://doi.org/10.18845/rfmk.v14i35.3148
Vogtländer, J. G., Velden, N. M., & Lugt, P. (2013). Carbon sequestration in LCA, a proposal for a new approach based on the global carbon cycle; cases on wood and on bamboo. *The International Journal of Life Cycle Assessment*, *19*(1), 13–23. https://doi.org/10.1007/s11367-013-0629-6
Wang, X., Padgett, J. M., Powell, J. S., & Barlaz, M. A. (2013). Decomposition of forest products buried in landfills. *Waste Management*, *33*(11), 2267–2276. https://doi.org/10.1016/j.wasman.2013.07.009
Werger, M. J. A. (ed). (2011). *Sustainable Management of Tropical Rainforests: the CELOS Management System*. (M. J. A. Werger, Ed.), *Tropenbos Series 25*. Paramaribo, Suriname. Retrieved from www.tropenbos.org
Werner, F., Taverna, R., Hofer, P., Thürig, E., & Kaufmann, E. (2010). National and global greenhouse gas dynamics of different forest management and wood use scenarios: a model-based assessment. *Environmental Science and Policy*, *13*(1), 72–85. https://doi.org/10.1016/j.envsci.2009.10.004
West, T. A. P., Vidal, E., & Putz, F. E. (2014). Forest biomass recovery after conventional and reduced-impact logging in Amazonian Brazil. *Forest Ecology and Management*, *314*(2014), 59–63. https://doi.org/10.1016/j.foreco.2013.11.022
Wiedinmyer, C., Yokelson, R. J., & Gullett, B. K. (2014). Global emissions of trace gases, particulate matter, and hazardous air pollutants from open burning of domestic wastE. *Environmental Science and Technology*, *48*(16), 9523–9530. https://doi.org/10.1021/es502250z
Wiloso, E. I., Heijungs, R., Huppes, G., & Fang, K. (2016). Effect of biogenic carbon inventory on the life cycle assessment of bioenergy: Challenges to the neutrality assumption. *Journal of Cleaner Production*, *125*, 78–85. https://doi.org/10.1016/j.jclepro.2016.03.096
Winjum, J. K., Brown, S., & Schlamadinger, B. (1998). Forest Harvests and Wood Products: Sources and Sinks of Atmospheric Carbon Dioxide. *Forest Science*, *44*(2), 272.
Wohlfahrt, J., Frechaud, F., Gabrielle, B., Godard, C., Kurek, B., Loyce, C., & Therond, O. (2019). Characteristics of bioeconomy systems and sustainability issues at the territorial scale. A review. *Journal of Cleaner Production*, *232*, 898–909. https://doi.org/10.1016/j.jclepro.2019.05.385
Wolf, C., Klein, D., Weber-blaschke, G., & Richter, K. (2015). Systematic Review and Meta-Analysis of Life Cycle Assessments for Wood Energy Services. *Journal of Industrial Ecology*, *0*(0), n/a-n/a. https://doi.org/10.1111/jiec.12321
Ximenes, F. A., Kathuria, A., Barlaz, M. A., & Cowie, A. L. (2018). Carbon dynamics of paper , engineered wood products and bamboo in landfills : evidence from reactor studies. *Carbon Balance and Management*. https://doi.org/10.1186/s13021-018-0115-3
Ximenes, F., Björdal, C., Cowie, A., & Barlaz, M. (2015). The decay of wood in landfills in contrasting climates in Australia. *Waste Management*, *41*, 101–110. https://doi.org/10.1016/j.wasman.2015.03.032
Yadav, P., & Samadder, S. R. (2018). A critical review of the life cycle assessment studies on solid waste management in Asian countries. *Journal of Cleaner Production*, *185*, 492–515. https://doi.org/10.1016/j.jclepro.2018.02.298
Zhang, L., Sun, Y., Song, T., & Xu, J. (2019). Harvested Wood Products as a Carbon Sink in China, 1900⁻2016. *International Journal of Environmental Research and Public Health*, *16*(3), 445. https://doi.org/10.3390/ijerph16030445
Zhang, X., Yang, H., & Chen, J. (2018). Life-cycle carbon budget of China ' s harvested wood products in 1900 – 2015. *Forest Policy and Economics*, *92*(159), 181–192. https://doi.org/10.1016/j.forpol.2018.05.005
Ziegler-Rodriguez, K., Margallo, M., Aldaco, R., Vázquez-Rowe, I., & Kahhat, R. (2019). Transitioning from open dumpsters to landfilling in Peru: Environmental benefits and challenges from a life-cycle perspective. *Journal of Cleaner Production*, *229*, 989–1003. https://doi.org/10.1016/j.jclepro.2019.05.015
Zuidema, P. A., Baker, P. J., Groenendijk, P., Schippers, P., van der Sleen, P., Vlam, M., & Sterck, F.

(2013). Tropical forests and global change: filling knowledge gaps. *Trends in Plant Science*, *18*(8), 413–419. https://doi.org/10.1016/j.tplants.2013.05.006

Zúñiga-Méndez, C. (2016). *Determinación de especies comerciales y clasificación de la madera según su dureza para los bosques bajo manejo forestal sostenible en el Área de Conservación Arenal-Huetar Norte, Costa Rica*. Heredia, Costa Rica. Retrieved from https://onfcr.org/media/uploads/documents/ficha-tecnica-christian-zuniga-mendez.pdf

Summary

Chapter 1

In Chapter 1, I provide a general introduction on the implications of tropical forest management on greenhouse gas emissions, along with a brief overview of the potential contribution from sustainable forest management for climate mitigation. I describe the processes leading to increased carbon storage outside the forest (technospheric) and the potential for wood products to avoid emissions by substituting other materials. I then briefly explain the lifecycle assessment framework and the main challenges when assessing the climate impact of wood products. I conclude with an overview of the context in which forest management is performed in Costa Rica.

Chapter 2

In chapter 2, I studied the lifecycle carbon balance (BioC-LC) of tropical forest management in Costa Rica. Until now, existing findings supported the idea that tropical logging leads to higher carbon emissions but no carbon balance analysis for these ecosystems had been done using a lifecycle approach. To quantify the effect of logging and compare it against forest ecosystem carbon balances, I used one hectare as the functional unit and defined the system's temporal boundary as one rotation period. I show that by including all lifecycle processes, technospheric storage in combination with forest regrowth results in additional storage of carbon in the system (i.e. -2.19 Mg C ha^{-1} over a 15-year period with a 95% CI of -5.26 to 1.86). Just for comparison with the other results in this thesis, expressed as CO2-eq this result is equal to -8.00 Mg CO_2-eq ha^{-1} over a 15-year period. Probabilities of a system that is a source of carbon exist, as higher harvesting intensities leading to high logging damage, insufficient recovery time, or high wood allocations into short-term uses can shift this balance. However, short-term uses increase storage in solid waste disposal sites (SWDS), and it is the combined effect from technospheric reservoirs that is important for carbon storage. Using a sensitivity analysis, I found that small changes in half-lives do not have an important effect on the stock and that only large changes such as re-allocating products from short to long-term products have substantial effects on total storage. Based on these findings we highlighted the climate mitigation opportunities of forest management for timber extend beyond the forest and that measures should be considered throughout the processes of wood transformation, use and disposal.

Chapter 3

In Chapter 3, I developed a detailed harvested wood product carbon inventory for Costa Rica. I followed IPCC Guidelines for National GHG Inventories, used country specific data and a material flow analysis, corresponding to a Tier 2 accounting level according to these Guidelines. Harvest data collected for this study is the currently best available for Costa Rica describing the evolution of wood production during the last 30 years and Chapter 3 merely scratches the surface of lessons that can be extracted from this data set. Carbon storage at the national level in 2016 (the last year of the inventory) was -412 Gg CO_2 (95% CI between -447.2 and -376.4). Most of this storage was found in SWDS (77%) and was partly a consequence of a high allocation of wood production into short-term products. Given that these allocation patterns were positively correlated with planted forests becoming the country's main wood source, I asked what have been (or will be) the effects of changes in wood source and product allocation on the carbon stock of harvested wood products. Since plantation wood tends to have a lower quality (at least lower wood densities and carbon content) and half-lives are consistently reported as drivers of carbon storage, I hypothesized that the stock must be heading towards a steady state. However, despite significant decreases in half-life and carbon content, the stock seemed unaffected. Hence, the stock of wood products appears to be characterized by a strong inertia, due to the characteristics (i.e., half-life) of the material in the stock from pervious harvests. As a result of these inherited characteristics, changing stocks of wood products may take a long time. This likely implies that the contribution of this stock to climate mitigation is smaller than commonly believed. Physical limits characterize technospheric carbon storage, and prolonging lifespan may not extend these limits much further. Thus, it is mostly through increasing harvest levels and wood use that storage can be increased. In this Chapter, I highlighted that opportunities to increase storage through increased harvests or lifespan must come from the implementation of demand-side measures.

Chapter 4

In Chapter 4, I assessed the lifecycle climate impact of wood from natural tropical forests in Costa Rica (Chapter 4). This work fills a gap in the understanding of the effects of logging in the tropics, where few studies have been conducted and none of these included the combined effect of biogenic and fossil emissions in a cradle to grave analysis for one rotation. Results for the harvest of wood from a hectare of tropical forests in Costa Rica show a net balance of -4.41 Mg CO_2-eq ha^{-1} over a 15-year period (95% CI of -13.12 to 10.96), indicating that under this timeframe the system has stored more carbon than what has been released through emissions. This result was verified by studying the effect of a shorter time horizon (i.e. 20-year global warming potential (GWP)) and by extending the temporal boundaries (i.e. from 15 to 100 years). Under a 100-year system boundary, emissions increase significantly to 1.90 Mg CO_2-eq ha^{-1} over a 100-year period (95% CI of -10.55 to 18.28) but I argue that for this functional unit (i.e. ha) this timeframe is not an appropriate boundary since not all possible rotations are taken into consideration. This boundary is useful when the functional unit is a product, or as in this case a m^3 of wood used for a specific product or co-product. Results for each individual wood product and co-product were also included, but for a 100-yr system boundary only mid and long-term products show a negative GHG balance due to carbon storage. Short-term products are specially affected by a change in boundary due to EoL methane emissions. Although these require almost no manufacturing, short-term products have the highest emissions per m^3, i.e. 860 Mg CO_2-eq ha^{-1} over a 100-year period (95% CI of -1.78 to 8.28). Because of the large proportion of short-term products these have a large effect on the results per hectare or multi-functional m^3. I therefore highlighted that after the evaluation of all lifecycle processes, the largest opportunities to increase the mitigation potential of forest management in the tropics is likely through reduced impact logging techniques.

Chapter 5

In the final chapter, I integrate the results from all chapters and discuss their implications. I first address the trade-offs from using local empirical data on the uncertainty and variability of the system. I argue that the use of local data is beneficial as it leads to an overall reduction in uncertainty, a better conceptual understanding of the system, more accurate estimation methods and adds crucial information (variability) on the system. Downsides of the use of local data include that it requires adaptation of calculation methods, it increases the risk of calculation errors, it adds unnecessary noise in the calculation process, and this may hamper interpretation of results. I continue by discussing results within the context of national policies and forestry practices. This is followed by an estimation of the potential contribution of forest management in Costa Rica to climate mitigation. Based on my own results, I provide a simple scenario analysis of opportunities to increase mitigation through increased logging intensity and

increased logging area. I found that by increasing the harvest area to the maximum potential yield, a contribution of -147.0 Gg CO_2-eq yr^{-1} and the potential substitution of -218.4 Gg CO_2-eq yr^{-1}; results in a total mitigation potential of -365.4 Gg CO_2-eq yr^{-1}. Finally, I discuss the key question of whether productive tropical forests should be managed for climate mitigation. Since the main argument against forest management is that it leads to degradation, I discuss how the results from this study provide evidence that after considering all lifecycle processes this is not necessarily always true. Finally, I argue that the main contribution from this thesis and a lifecycle approach in general, is that reveals the unintended consequences of decisions to *not* manage forests (i.e. indirect land use change, changes in wood production and substitution of wood). If all of these are included in the scenario against which management is usually compared, then this would more clearly show the contribution of managing tropical forests for wood production as a mitigation strategy.

Acknowledgements

Let's see…. even kijken.

I begin by thanking those who encouraged me to pursue a career in science. These are professors from my Bachelor's in Forest Sciences at the Universidad Nacional de Costa Rica who actively helped make it real. Sonia, Wilberth and specially Dr. William Fonseca, whom I've worked with and learned from since 2005. Also, the opportunity of having Dr. Florencia Montagnini as mentor, where the combination of her guidance and my own research experience at La Selva Biological Station made for a transformative experience that turned science into a personal aspiration.

My parents, Alberto and Estrella, who influenced me personally, but whom are also largely responsible for my career path. This union between an entrepreneurial agronomist and a biologist who worked in academia all her life, shaped my views on nature conservation and sustainable production systems. I thank them for this. For all their support, I thank my parents, together with my brothers and sisters: Mariflor, María de la Paz and Ronald, Alberto and Mariela, and Daniel; my nieces and nephews: Luciana, Harry, Marianne and Fede.

The boundaries between family and friends now become a blur, a gray area where I find my cousins. They are partly responsible for keeping me grounded and for whatever sanity I might still have left. This is probably because I look up to them, and because they themselves are not too sane. Sergio, Abel, Victor, Ignacio, Esteban, Alberto and Orlando kept me updated with what was going on in Costa Rica while I was away (especially football). Mostly an expectator to their discussions, I'll have to make up with a few BBQs when I return.

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As with most things in life, I ended up in Wageningen partly by chance, though I was lucky enough to find old friends in a completely new place. Mr. Long, Marianne, Kim and San would share their house and food as in Edinburgh in 2011. This is not an easy thing to find. For me to have a home and a family outside Costa Rica has been a privilege making it all seem natural. My time here would not have been the same without your support; some Vietnamese mint tea with Dutch honey were always there at the end of the day.

To top up my Wageningen experience, I was lucky to work at the Forest Ecology and Management group. The feeling of belonging also came naturally and would only grow with time. As in the same day of my arrival, the feeling of celebration has never left me. It seems inherent to our corridor. Loud and busy but surprisingly productive. I am now convinced you build success on a good working environment.

I want to thank Ute for taking me in and broadening my small group almost since the beginning, Joke for spoiling me by making me feel special, and Marielos and Lourens for their daily effort to bridge the gap between staff and PhDs. Without you, the group dynamics would not have been the same. Thanks to all staff; Jan, Ellen, Leo, Frank, Frans, Koen, and from WENR, Gert-Jan and Mart-Jan. To all, whenever I asked for help it was my way of showing respect and admiration. I hope I was not misunderstood. I did my best to return your favor, and most times it worked. Still, I am aware, there are not enough coffee breaks and cakes I can bring to make up for the experience of being part of this group.

The learning process is the most gratifying experience from a PhD, and it was thanks to the guidance and support of my supervisors that I moved through its different phases. Their doors were always open, and I made the best of it. I could always count on Pieter, whom I owe a systems thinking approach to understanding problems. I trusted him fully when it came to solve any technical difficulty, but also to talk openly about any other matter. The same holds true for Frits, with whom I shared long talks about forestry and life in general. Despite being highly critical, he was always optimistic about my capability to do my work. I don't really know where this came from, but it was reassuring from beginning to end. I did not always win discussions with my supervisors (hardly) and in some cases the driver seat probably felt empty. Still, perhaps to their disappointment, it was an easy ride. I truly wish I could do it all over again and that this won't be the last time we work together.

Every year brought different people and experiences. These were always exciting and intense. I've described my time here as almost magical. I've seen a version of myself of which I can be proud and for this, highly esteemed colleagues, I blame you all. Wageningen, FEM and its people brought music back into my life and I've experienced it like never before.

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Even when I thought I was ready to close my Wageningen experience and return home to Costa Rica… Alan and Indira, Alejandra and her beautiful baby Camila, Daisy, Bárbara, Laura, Sophie, Carlos Moreira Miquelino Esqueleto Torres and Fabianne, Danju, Heitor, Aldicir, Maike, Rodrigo, Úrsula, Rens, Tomonari and Jazz… you make me wish I could start all over again. Thanks for your support during the toughest part of my PhD, for pointing out my severe mental problems but being quick to make them feel almost normal, for the unofficial coffee breaks, for showing me all the magical sunny corners of Lumen, and for smiles that can be larger than your faces. These have brightened my days and kept the experience exciting and fresh.

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Dr. Tiza, de los malos el mejor.

Short biography

Federico E. Alice earned his bachelor's degree in Forest Sciences from the Universidad Nacional de Costa Rica. As a student his interests were closer to forest ecology and production, yet his first job took a sharp turn towards climate change management. As the world experimented with carbon markets in the early 2000's, he joined a start-up dedicated to promoting and developing carbon projects in Latin America. He was exposed to project-based carbon accounting and climate change mitigation as a project developer, while simultaneously working on carbon accounting and finance as a research assistant for the Instituto de Investigación y Servicios Forestales. He remained active in several other research projects related to land use carbon accounting at local, regional and national scales.

In 2010 he participates in creating the Carbon Management Program at the School of Environmental Sciences (Universidad Nacional de Costa Rica). To complement this initiative, he obtains his MSc in Carbon Management from the University of Edinburgh, Scotland. He is involved, through this program, in national and municipal GHG inventories, organizational carbon footprints and lifecycle assessments, contributing, fully developing or auditing land use production systems as coffee, cacao, banana, pineapple, wooden pallets and common beans. Outside the land use sector, he has worked with transportation and waste management systems.

Outside academia he is active in carbon markets and sustainable forest management as a lead auditor of carbon and FSC projects in Latin America since 2013, and as part of Plan Vivo's Technical Advisory Panel since 2010. He has been involved in climate change negotiations under the UNFCCC, participated in the development of NAMAs at the national and global level, the submission of Costa Rica's INDCs under the Paris Agreement, and the National GHG inventory.

He is currently on a tenure track to become professor at the School of Environmental Sciences from the Universidad Nacional of Costa Rica.

List of publications

Alice, F.E., Mohren, F., Zuidema, P.A. 2019. The Lifecycle Carbon Balance of Selective Logging in Tropical Forests of Costa Rica. Journal of Industrial Ecology. DOI: 10.1111/jiec.12958

Cifuentes, M. et al. 2015. Overcoming obstacles to sharing data on tree allometric equations. Annals of Forest Science. DOI 10.1007/s13595-015-0467-8

Henry, M. et al. 2015. Recommendations for the use of tree models to estimate national forest biomass and assess their uncertainty. Annals of Forest Science. DOI 10.1007/s13595-015-0465-x

Henry, M. et al. 2015. An overview of existing and promising technologies for national forest monitoring. Annals of Forest Science. DOI 10.1007/s13595-015-0463-z

Cifuentes, M. et al. 2014. Guidelines for Documenting and Reporting Tree Allometric Equations. Annals of Forest Science. DOI 10.1007/s13595-014-0415-z

Fonseca, W., Herrera J. y Alice, F.E. 2014. Desafíos nacionales en materia de métricas de cara a la meta costarricense de alcanzar la carbono-neutralidad en el 2021. Ciencias Ambientales 47: 16-31, ISSN: 1409-2158 / 2014.

Fonseca, W., Ruiz, L., Rojas, M. y Alice, F.E. 2013. Modelos alométricos para la estimación de biomasa y carbono en Alnus acuminata. Ciencias Ambientales 46: 37-50, ISSN: 1409-2158/2013.

Fonseca, W., Navarro, G., Alice, F.E., Rey Benayas, J. M. 2012. Impacto económico de los pagos por carbono y servicios ambientales en las inversiones forestales en la región Caribe de Costa Rica. Revista Ecosistemas 21 (1-2): 21-35. ISSN 1697-2473 http://www.revistaecosistemas.net/articulo.asp?Id=711

Fonseca, W., Rey Benayas, J. M., Alice, F.E. 2011. Carbon accumulation in the biomass & soil of different aged secondary forests in the Humid Tropics of Costa Rica. Forest Ecology and Management. DOI: 10.1016/j.foreco.2011.06.036

Fonseca, W., Chávez, H., Alice, F.E., Rey Benayas, J.M. 2011. Land cover dynamics (1986-2006) and priority areas for forest restoration in the Caribbean region of Costa Rica. Recursos Naturales y Ambiente/no. 59-60: 99-107.

Fonseca, W., Alice, F.E., Rey-Benayas, J.M. 2011. Carbon Accumulation in Aboveground and Belowground Biomass and Soil of Different Age Native Forest Plantations in the Humid Tropical Lowlands of Costa Rica. New Forests. DOI: 10.1007/s11056-011-9273-9

Pioto, D. Montagnini, F., Craven, D., Alice, F.E. 2010. Silvicultural and economic aspects of pure and mixed native tree species plantations on degraded pasturelands in humid Costa Rica. New Forests 39:369–385.

Fonseca W., Alice F.E., Rey J.M. 2009. Modelos para estimar la biomasa de especies nativas en plantaciones y bosques secundarios en la zona Caribe de Costa Rica. BOSQUE 30(1): 36-47.

Fonseca W., Alice F.E., Montero J., Toruño H., Leblanc H., 2008. Acumulación de biomasa y carbono en bosques secundarios y plantaciones forestales de *Vochysia guatemalensis* e *Hieronyma alchorneoides* en el Caribe de Costa Rica. Revista Agroforestería en las Américas, No. 46, 2008. Turrialba, CATIE.

Alice F.E., Montagnini F., Montero M. 2004. Productividad en Plantaciones Puras y Mixtas de especies forestales en la Estación Biológica La Selva, Sarapiquí, Costa Rica. Agronomía Costarricense 28(2): 61-71.

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PE&RC Training and Education Statement

With the training and education activities listed below the PhD candidate has complied with the requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

Review of literature (6 ECTS)

- Understanding the carbon dioxide sink/source balance from the management of tropical forests, wood use and product substitution on atmospheric GHG concentrations

Writing of project proposal (4.5 ECTS)

- The role of tropical forest management and wood products for climate change mitigation

Post-graduate courses (7.7 ECTS)

- Introduction to R; WUR (2016)
- European forest resources and the bio-economy; WUR (2017)
- Climate change mitigation: options and policies; Radboud University (2017)
- Bayesian statistics; WUR (2018)
- Conflicting demands in European forests; WUR/SLU (2018)

Laboratory training and working visits (0.3 ECTS)

- Life cycle assessment of forest wood products; KU Leuven, Belgium (2015)

Invited review of (unpublished) journal manuscript (3 ECTS)

- Revista de Ciencias Ambientales: es efectivo el Programa País Carbono Neutralidad? (2016)

- Revista de Biología Tropical: composición florística y conservación de carbono en bosques riparios en paisajes agropecuarios de la zona seca del Tolima, Colombia (2016)
- Environmental Science & Technology: assessing the greenhouse gas mitigation potential of harvested wood products substitution in China (2018)

Competence strengthening / skills courses (2.2 ECTS)

- Effective behaviour in your professional surroundings; WUR (2015)
- PhD Workshop carousel; WUR (2015)
- Research integrity; WUR (2016)

PE&RC Annual meetings, seminars and the PE&RC weekend (2.1 ECTS)

- PE&RC First years weekend (2015)
- 2nd Wageningen PhD symposium: connecting ideas, combining forces (2015)
- PE&RC Day (2016)
- PE&RC Last years weekend (2018)

Discussion groups / local seminars / other scientific meetings (6.1 ECTS)

- REDD+ Discussion group (2015, 2016)
- FEM Journal club (2015-2018)
- Agriculture–Climate-Forests-Food PhD discussion (2017, 2018)

International symposia, workshops and conferences (6.9 ECTS)

- 60th LCA Discussion Forum Environmental Use of Wood Resources; ETH Zürich (2015)
- World Forestry Congress - IUFRO 125th Anniversary Congress (2017)
- 1er Congreso Centroamericano de Ciencias de Tierra y Mar (2017)
- Workshop on the Building of Sustainable National Greenhouse Gas Inventory Management Systems, and the Use of the 2006 IPCC Guidelines for National Greenhouse Gas Inventories for the Latin America and Caribbean Region; Rodney Bay, Saint Lucia (2017)

Lecturing / supervision of practicals / tutorials (7.5 ECTS)

- Supervision of practicals resource dynamics and sustainable utilization (2015, 2016, 2019)
- Lecture in trends in forest and nature conservation (2016, 2017, 2018)
- Supervision case study resource dynamics and sustainable utilization (2016, 2017, 2018, 2019)
- Lecture in FEM forest resources (2018)

Supervision of MSc students (18 ECTS)

- The carbon balance of the managed natural forests and their wood products in Costa Rica
- Assessing the feasibility of NDMI as indicator for forest characteristics and disturbances in Costa Rica
- Mitigación del cambio climático a través del sector forestal y uso de la tierra (FOLU) del cantón de Grecia, Alajuela, Costa Rica
- Impacto potencial sobre el cambio climático de las tarimas de madera elaboradas en la Región Huetar Norte de Costa Rica a través de un Análisis de Ciclo de Vida (ACV)
- Propuesta metodológica para un PSA campesino basada en el modelo financiero de cuantificación de los servicios agroecosistémicos
- Evaluación del carbono almacenado en la biomasa, necromasa y carbono orgánico del suelo de tres diferentes hábitats en la Península de Osa, Costa Rica

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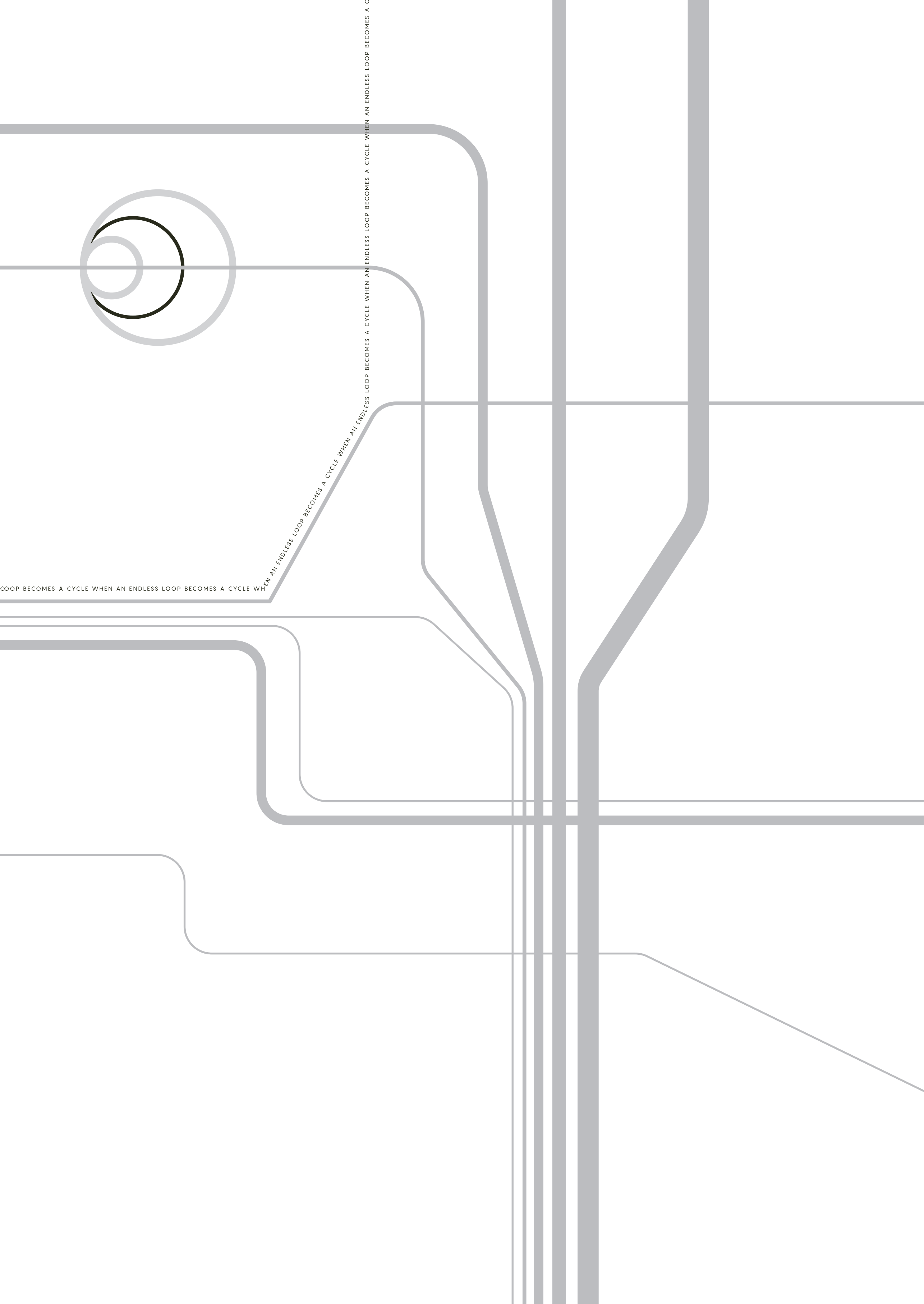
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