The economic and ecological sustainability of the Amazonian timber industry

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Contents

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[1]Global Witness report available at: https://www.globalwitness.org/campaigns/environmental-activists/deadly-environment/

Abstract

Selective logging of tropical forests, particularly reduced impact logging (RIL), has long been suggested as a benign compromise between profitable land-use and biodiversity conservation. Throughout human history, slow-renewal biological resource populations have been predictably overexploited, often to extinction. This thesis examines the degree to which timber harvests beyond the first-cut can be financially profitable or demographically sustainable, both of which remain poorly understood. Data on legally planned logging of ~17.3 million m^3 of timber were obtained from 824 government-approved private and community-based concession management plans. Results indicate that neither the post-depletion timber species composition nor total value of pre-harvest forest stands recover beyond the first-cut, suggesting that commercially most valuable timber species become predictably rare or economically extinct in old logging frontiers. Additionally, smallholders appear to exert strong high-grading pressure upon high-value hardwood species, thereby accruing higher gross revenue productivity per unit area and were more likely to inconsistently report areas of unlogged forest set-asides as required by Brazilian law.

Selective logging leads to several forms of collateral damage (CD) to the residual forest stand. This pattern of structural disturbance is poorly quantified or understood despite representing a key form of forest degradation, or the second 'D' of REDD+ (United Nations Reduced Emissions from Deforestation and Degradation). A review of studies on selective logging impacts on tropical forest fauna revealed that ~90% failed to at least report or attempt to quantify CD. This thesis also examined CD associated with a certified industrial-scale RIL operation of eastern Brazilian Amazonia and finds that for every harvested tree, there is an estimated loss of ~12 damaged stems (≥10cm DBH). Over 30% of total ground sampling area of logged forest was cleared within felled-trees impacts alone. Finally, using RIL concession data from an 11-year time series where ~0.34 million trees were harvested, we estimated the total biomass and carbon stock of harvested trees, their CD, and the infrastructure damage associated with roundlog removal. If only harvested trees and their associated CD are considered, the estimated cost incurred in sparing logging-induced forest degradation through carbon financing projects such as REDD+ could compensate for the ~393 US\$ ha^{-1} yr^{-1} logging revenues accrued to concession owners.

Chapter 1: Introduction

1.1 Tropical forests and Amazonia

The Amazonian rainforest contains approximately a quarter of all terrestrial species (Dirzo & Raven 2003), and is home to 16,000 described and undescribed species of trees alone (ter Steege et al. 2013). Most of Amazonia's 6,412,000 km^2 lies within Brazil (67%); the remaining stretches of forest are distributed throughout Peru (10%), Colombia (7%), Bolivia (6%), Venezuela (6%), Guyana (3%), Suriname (2%), Ecuador (1.5%) and French Guyana (1.5%, Pereira et al. 2010). It is estimated that Amazonian forests also account for ~15% of all global terrestrial photosynthesis (Bush et al. 2004).

Over the last 25 years, the largest loss in natural (i.e. non-plantation) forest occurred in South America, at an annual average rate of 3.5 million ha per year from 1990-2000 and then slowing to 2.1 million ha per year between 2010-2015. This slow down can largely be attributed to Brazil's unprecedented decline in deforestation over the past decade. Between 2004 and 2012, deforestation was reduced by 84% making Brazil a world leader in climate change mitigation (INPE 2014). This deceleration in forest loss was driven by a mix of mutually supporting economic, institutional and technological factors (Dalla-Nora et al. 2014). These complex interdependent factors act at local or regional spatial scales (such as soil fertility, topography, distance to roads, access to rural credit, management practices) and at global scales through export commodity prices and market demands. The myriad of regional policies, however, are thought to have been disproportionately responsible for the significant decline in deforestation (Dalla-Nora et al. 2014; Nepstad et al. 2014).

Regional policies include, for instance, a 'green municipal county' programme launched in the state of Pará where agricultural credit was suspended for private properties located inside municipalities with high deforestation rates. Additionally, in 2009 many rural landholdings were georeferenced, mapped and recorded in the rural land registry or *cadastro rural* (Portuguese acronym *CAR*) thus improving land tenure and enabling property level law enforcement. Deterring public policies and risks for deforesters also included the 'soy moratorium' (2006) and a Greenpeace campaign (aided by the public prosecutor's office) to exclude livestock from areas that were deforested after 2009 (Nepstad et al. 2014). However, deforestation rates in Brazilian Amazonia are rising once again; between 2013 and 2014, 24,400 ha were cleared, representing an increase of 467% from the previous year and a 1,070% increase in forest degradation (46,800 ha, Fonseca et al. 2014).

1.2 The global tropical timber trade

In 2015 approximately 30% of all forests worldwide were designated as timber production forests (FAO 2015). The onset of the global economic crises decreased the production of tropical industrial roundlogs but by 2011 production had increased again, reaching 173.6 million m^3 (ITTO 2012). Most of the roundlog production in 2011 (59%) was harvested in the east-Asia Pacific region; in 2012 a 10% decrease in production from Malaysia alone reduced global tropical timber production to 172.5 million m^3 (ITTO 2012). There is evidence that Southeast Asian natural timber stocks have become severely depleted and as production starts to decline, pressures will likely increase on African and South American forests (Shearman et al. 2012).

Globally, plantation forests have an average annual rate of increase of 3.1 million ha per year (2010-2015) and now account for 7% of the world's forests (FAO 2015). Most of these fast-growing tree monocultures are located in temperate zones (150 million ha) whereas boreal and tropical zones account for around 57 million ha each (FAO 2015). Tropical forest plantations increased by 69% over the last 25 years (FAO 2015) but account for an average of only 5% of all tropical production forest areas (Blaser et al. 2011), thereby leaving most of the timber demand to be met by natural forests (Fig. 1.1).

Figure 1.1 Photo depicting two different tree plantations, (left) native Paricá *Schizolobium amazonicum* and (right) African mahogany *Khaya senegalensis*, taken in a logging operation of eastern Amazonia (Photo: Vanessa Richardson).

1.3 Logging in Brazil

Brazil is named after an endemic Atlantic rainforest species called Brazil-wood or Pau Brasil (*Caesalpinia echinata,* Leguminosae-Mimosoidae*),* a colonial source of red dye and hardwood that was heavily harvested from the 16th century onwards. The species was, and continues to be, particularly prized for making the bow of classic string instruments. Today, Pau Brasil is protected by law but remains on the brink of extinction. Most surviving specimens are in *ex situ* botanical gardens, with only a few small highly inbred relict populations remaining in the wild (Cardoso et al. 1998). The Atlantic rainforest used to stretch for 1.3 million km^2 along the Brazilian coast into Argentina and Paraguay. This heavily fragmented biodiversity hotspot still harbours one of the highest percentages of endemic species in the world but has been reduced to less than 13% of its original range (Ribeiro et al. 2009). By 1913 Brazilian Amazonia had also lost its main source of income, rubber, made from the latex of the endemic tree species *Hevea brasiliensis*. The rubber-boom monopoly finally came to an end as biopirated seeds by the British Royal Botanical Kew Gardens were cultivated in South-east Asia and began flooding the international market (Dean 1987). By the beginning of the 20th century, the Atlantic rainforest stock of timber was already severely depleted and attention towards untapped timber resources in the vast expanses of Amazonia, increased.

Traditionally, Amazonian logging was limited to the Amazon River estuary, the eastern edge of the state of Pará, or along floodplain forests. But the Amazon highway system (particularly in the State of Pará), including the Belém-Brasilia Highway paved in 1969-71 and the Transamazon Highway built in 1970, allowed greater access to vast tracts of *terra firme* interfluvial forest and mechanised industrial-scale logging increased exponentially (Uhl & Vieira 1989). By the mid-1990s there were over 2000 sawmills just in the state of Pará, producing 13 million m^3 of sawnwood, representing half of all timber produced in Brazil (Veríssimo et al. 2002; Uhl et al. 1997). In 2009, Pará still accounted for 47% of all roundlog production, 44% of the gross timber revenue (~US$1.1 billion), and 45% of all direct and indirect employment in the wood-related sector of the 5 million km^2 Brazilian Amazon (Pereira et al. 2010).

Brazil is estimated to hold 64% of the world's total intact forest landscapes (836 Mha: Potapov et al. 2008), 72% of which are primary and secondary forests in Amazonia (SFB 2013). Brazil also accounts for 85% of the roundlog production in Latin America/Caribbean region, and the total harvested volume was 30.8 million m^3 in 2012, even if we overlook the poorly quantified illegal trade (ITTO 2012). Approximately 95% of the Brazilian natural roundlog production is destined for the domestic market, so that only 5% is exported (ITTO 2012). International consumer market pressure to boost logging sustainability in Brazil is therefore largely ineffective.

In Brazil, there are approximately 6.7 million ha of plantation forests, 4.52 million ha of *Eucalyptus* species, 1.79 million ha of *Pinus* species, and 344,000 ha of other species, including *Acacia mearnsii, A. mangium, Araucaria angustifolia, Schizolobium amazonicum, Tectona grandis,* and *Populus* spp (Blaser et al. 2011). In 2013, most of the Brazilian timber exports (95%) by both export value and roundwood equivalent volume comprised of pulp and paper (70% and 10%, respectively) from plantation forests and were destined to the EU, China, USA and Japan (Wellesley 2014).

1.4 Illegal logging and corruption

Illegal logging activities account for 50–90% of all pantropical native forestry products worth US$30–100 billion yr^{-1} or 10–30% of the global wood trade (Nellemann 2012). The majority of illegal logging at global scale can be traced to three major producing countries, namely Indonesia (50%), Brazil (25%), and Malaysia (10%) (Hoare 2015). These figures largely reflect the size of these tropical forest countries, and their forestry sectors. Other smaller timber producing countries have higher proportions of illegally harvested roundlogs, such as Democratic Republic of Congo (DRC), where most offtakes are illegally sourced (Hoare 2015), but total illegal logging is lower.

The EU has responded to the issue of illegal logging through measures such as the EU Forest Law Enforcement, Governance and Trade (FLEGT) Action Plan (2003), bilateral Voluntary Partnership Agreements (VPAs in 2006), and the EU Timber Regulation (2013). Similarly, the USA launched the Lacey Act Amendment (2008) and Australia introduced the Illegal Logging Prohibition Act (2012). However exports to these countries have become increasingly less significant with the growth of developing markets such as China and Brazil. Since 2000, China has become the world's principal processor of timber. In only 5 years, China's import of timber-sector products increased from an estimated 45 million m^3 to 94 million m^3 (Hoare 2015). Accordingly, between 2000-2013 the average volumetric share of exports to China from the top nine producing countries (namely, Indonesia; Brazil; Papua New Guinea; Malaysia; Cameroon; Republic of Congo; Laos; Ghana; and DRC) increased from 10% to 23% (Hoare 2015).

Most of the natural forest logging in Brazil is deemed to be illegal, with estimates suggesting that around 75% of all offtakes are illegally sourced (Wellesley 2014). The NGO Greenpeace-Brasil has recently (2015) analysed concession management plans issued by the State Environmental Secretariat of Pará (SEMA) in collaboration with the state Environmental Agency and Brazil's Public Prosecutor's Office (MPF: www.mpf.mp.br). On the ground inspections revealed widespread corruption and fraudulent activities including large volumes of illegal timber laundering through false official documentation and bribery

(Greenpeace 2014), some of which has recently (August 2015) resulted in arrests and prosecutions of high-level government officers (http://www.theguardian.com/environment/2015/aug/24/dawn-raids-brazil-illegal-timber-laundering-operation).

1.5 A brief history of Reduced Impact Logging

Logging can severely damage forests. As target trees are felled, small neighbouring stems are crushed, and many trees have large portions of bark removed exposing cambial tissue which leads to pathogen invasion through the xylem, heartrot, and eventually death (Romero & Bolker 2008). In the impact zones of felled canopy and emergent trees, huge gaps averaging up to $1000m^2$ are created from the canopy to the forest floor (Jackson et al. 2002). Logging operations also damage tropical forests through logging roads and dense networks of skid trails for tractors to haul large roundlogs out of the forest (Fig. 1.2). In unplanned conventional logging (CL) as much as 26% of harvested tree volume is lost (versus 1% in planned operations) due to hollow boles, felling errors that damage the bole, or simply because loggers cannot find, access or retrieve the felled round logs (Barreto et al. 1998). Damage to the residual stand (collateral damage) is also significantly greater in CL than in planned reduced impact logging (Fig. 1.3.).

Figure 1.2 Structural damage in a Reduced Impact Logging operation of eastern Amazonia; **(a)** Primary logging road; **(b)** Secondary road; **(c)** Primary skid trail (to haul harvested logs out of the forest); **(d)** Log deck (where harvested roundlogs are stock-piled for transport) and an adjacent secondary road; **(e)** Collateral damage in a logging gap (Photos: Vanessa Richardson).

For over a century, studies have described how damaging logging practices can be and how much of this structural disturbance can be avoided (Bryant 1914; Nicholson 1958). However it was only in 1993 that the term Reduced Impact Logging and its acronym 'RIL' was coined. The authors define it as *intensively planned and carefully controlled timber harvesting conducted by trained workers in ways that minimize the deleterious impacts of logging* (Putz & Pinard 1993). There is no universal RIL standard because tropical forests vary greatly in structure and composition, as do the availability of marketable timber species, management objectives, and disagreements between logging experts. Instead, regional RIL practices and guidelines have developed and prevailed (Putz et al. 2008). Nonetheless most RIL practices include conducting pre-harvest forest inventories, the use of wheeled tractors over crawler tractors, cutting vines and high-climbing woody lianas prior to logging, directional felling, road and skid trail planning (Fig. 1.3), and maintaining buffers of unlogged forest cover along watersheds to protect against runoff and erosion (Fox 1968; Froehlich et al. 1981; Pinard & Putz 1996; Vidal et al. 1997; Putz et al. 2000).

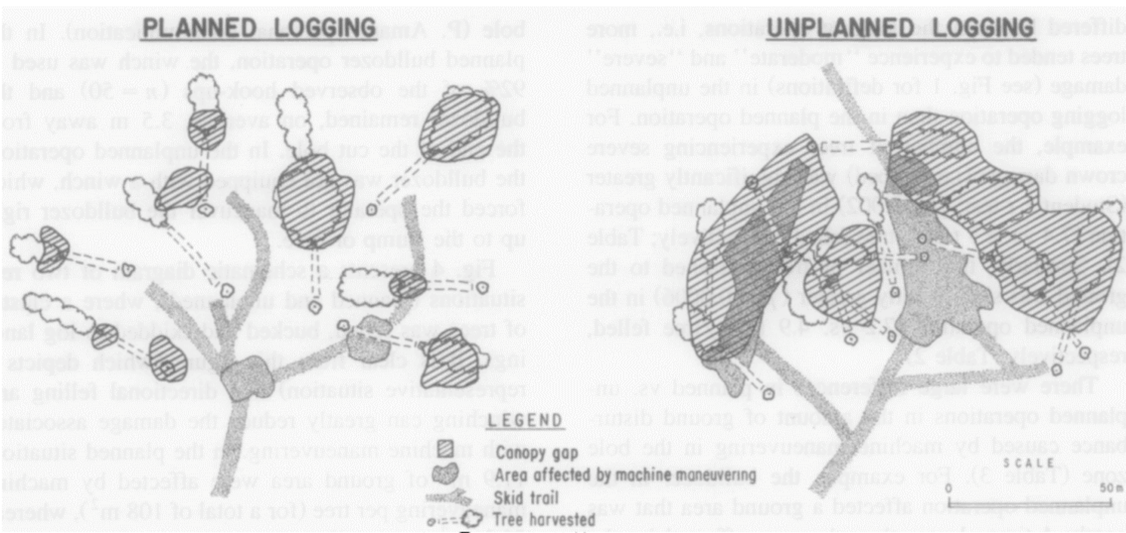

Figure 1.3 Canopy opening and ground area damage associated with planned (left panel) versus unplanned logging (right panel) in the State of Pará, eastern Brazilian Amazonia. Dotted double lines represent harvested tree boles; hashed areas represent canopy gaps; and shaded areas the cleared ground areas for roads and machine manoeuvring. Source: Johns et al. 1996.

Voluntary market-based certification schemes such as the Forest Stewardship Council (FSC) began in 1993. These may adopt RIL techniques but guidelines vary regionally. By 2015, 184 million ha of forests at a global scale had been FSC-certified, but most of these were plantation forests and only ~15% were tropical forests (FSC 2015). If all international certification programmes are considered, 438 million ha of global forests were under forest management certification by 2014. However, 90% of these areas were in boreal and temperate forests (FAO 2015).

1.5.1 Reduced Impact Logging in Brazil

In 1999, a group of NGOs including the Center for International Forestry Research (CIFOR), the Brazilian subsidiary of the Tropical Forestry Foundation (FFT) and the Amazon Institute for People and Environment (IMAZON) together with the Brazilian Corporation for Agricultural Research (EMBRAPA) established the first set of RIL guidelines in upland *terra firme* forests of Brazilian Amazonia (Sabogal et al. 2000). Guidelines were developed from the FAO Model Code of Forest Harvesting Practices (Dykstra & Heinrich 1996) and in consultation with stakeholders with experiences of logging within Brazilian Amazonia such as government officials, researchers and practitioners. Two industrial-scale timber companies (JURUA and CIKEL) tested these guidelines in an International Tropical Timber Organization (ITTO) funded project called the EMBRAPA / CIFOR 'Sustainable Management of Production Forests at the Commercial Scale in the Brazilian Amazon' (Pokorny & Adams 2003).

The Brazilian Forest Certification programme (CERFLOR) also began in the 1990s, but was only operational in 2003 (for planted forests), and in 2005 it was endorsed by a voluntary market-based certification scheme: Programme for Endorsement of Forest Certification (PEFC). By 2010, 1.25 million ha of plantation forests were certified under CERFLOR, but only one natural forest logging operation was included, and 73,000 ha within the Amazonian state of Rondônia that had been banned for noncompliance (Blaser et al. 2011).

In 1996, the FSC working group in Brazil also set out regional criteria and indicators and prevailed to become the leading international certification body for both plantation and natural forests. By 2010, 6.2 million ha of plantation and natural Brazilian forests were FSC certified (Pereira et al. 2010). By 2014, 1.2 million ha of natural forests were FSC certified, but this represented only ~4% of Brazil's entire natural forest timber production (Wellesley 2014). This may be because certification has not financially rewarded logging companies, but companies appear to have benefited from higher market access, particularly for international export (Araujo et al. 2009).

Although most natural timber offtakes in Brazil are illegal, Brazilian legislation does regulate timber harvesting. Since 1965 the Brazilian Forest Code (Law 4.771/1965, article 15) has required Amazonian landowners to set aside most of their properties (80%) as forest reserves, in addition to riparian corridors. For legal timber harvesting in private or public lands, management plans must be granted by state environment agencies (Law 11.284/2006) and these require landowners to follow some RIL practices (Normative Ruling no. 05 of 2006) including the following:

- Forest inventories
- Species-specific minimum cutting diameter or 50cm for any undetermined species
- When harvests are mechanised, a maximum volumetric harvest of $30m^3$ h^{-1} and harvest cycle of 35 years
- For small scale or non-mechanised harvests, a maximum volumetric harvest of $10m^3$ h^{-1} and harvest cycle of 10 years
- For every 100 ha of logged forest, 15% of each harvested tree species must be spared in the landscape for seed production
- A tracking or chain of custody system is required
- Moreover, species listed in the endangered flora of Brazil (IBAMA 2008, Normative ruling no. 06 of 2008) such as broadleaf mahogany (*Swietenia macrophylla*), Brazil-nut trees (*Bertholletia excelsa*) and rosewood (*Aniba rosaeodora*) cannot be legally harvested.

For a glossary of forest–related terms, other legal frameworks and legislation see Appendix 1.

1.5.2 Sustainable timber yields

There is no definitive prescription for sustainable forest management (SFM). Following Pearce et al (2003) SFM and Sustainable Timber Yields (STY) are distinguishable; STY should ensure that the same species-specific timber volume can be harvested again from a given timber tree population in the future, whereas SFM can encompass management objectives towards STY in addition to non-timber forest products and services.

When RIL was first promoted, there was considerable optimism for the future of SFM and STY (Fig. 1.4) in that RIL substantially reduces damage to the residual stand. Currently, however, there is wide consensus that RIL alone does not ensure STY (Wadsworth & Zweede 2006; Sist & Ferreira 2007; Zarin et al. 2007; Putz et al. 2008; Peña-Claros et al. 2008; Schulze et al. 2008; Macpherson et al. 2012). To prevent fire risk, promote recruitment of non-pioneer high-value timber species, and confine the overwhelming growth of pioneer vegetation some prescriptions may include restricting harvest intensities to fewer than 5 trees per ha, a minimum cutting diameter of 60cm, restricting logging gaps to 500 m^2, aggregate gap areas should not surpass 10% of the canopy area, and 85% of the stand basal area should be maintained (Sist et al. 2003; Zimmerman & Kormos 2012). Moreover, there is growing consensus that the frequency of timber harvests across tropical forests must be lowered (Sist et al. 2003; Putz et al. 2012; Zimmerman & Kormos 2012) and cutting cycles between 60-300 years have been proposed for neotropical stands under RIL management (Kammesheidt et al. 2001).

SFM that incorporates management techniques towards STY can incur additional costs, including training, capacity building and purchasing of additional equipment. Rarely is SFM the most lucrative land-use option in tropical forests, where the greatest profits are made by harvesting the most valuable hardwood timber species immediately and then either vacating the area or adopting more profitable land uses such as oil palm, soybean or tree plantations (Rice et al. 1997; Pearce et al. 2003; Wilcove et al. 2013).

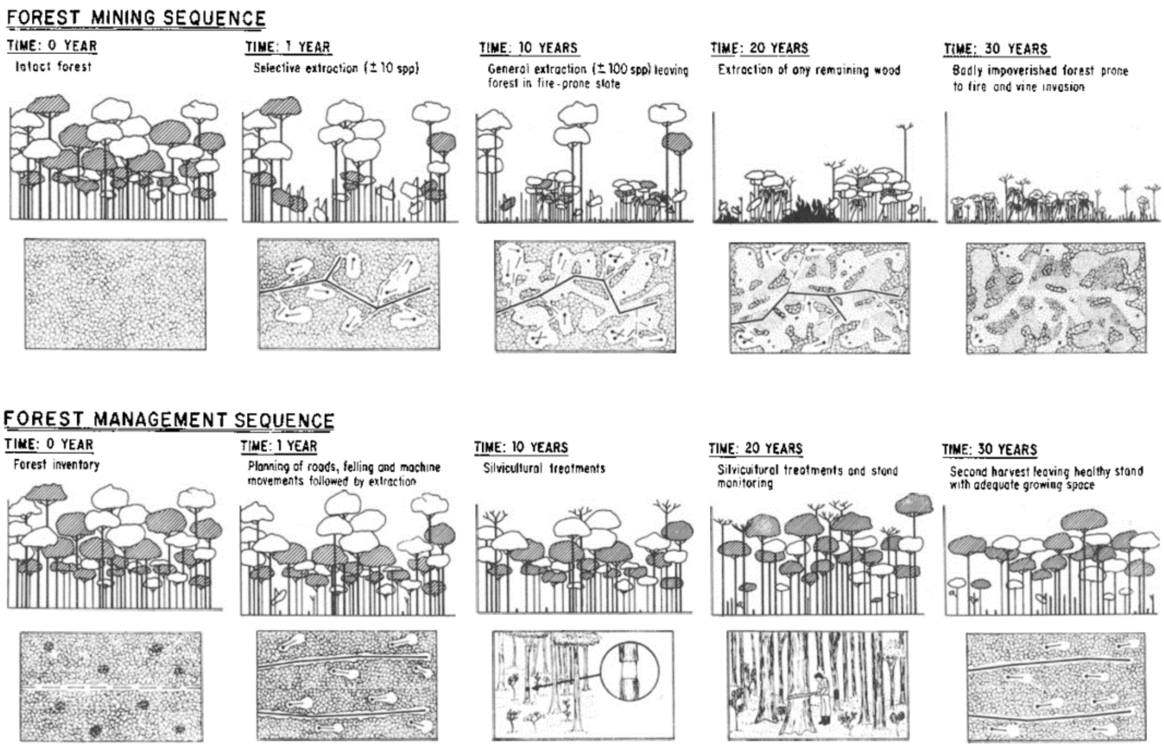

Figure 1.4 *Unsustainable forest mining sequence and sustainable forest management sequence as envisioned by Uhl et al. (1997). They describe the panels as follows*: (Top) Typical terra firme logging ('mining') practices that lead to forest degradation in eastern Amazonia. (Bottom) The alternative is forest management and includes conducting forest inventory, planning of extraction activities, and silvicultural treatments. With this approach, sustainable cutting cycles might be reduced from 70-100 years to 30-40 years. For each step in these sequences, the upper panel shows a side view of the forest, and the lower panel shows a view from above or a close-up side view. *Source: Uhl et al. (1997).*

1.6 Current understanding of the ecological impacts of selective logging

Logging intensities vary among paleotropical and neotropical forests, and are further mediated by regional timber stocks, extraction capacities, markets and topography. In the Afrotropics and Neotropics, offtakes are somewhat lower (1-5 trees per ha) but in South-East Asia, dipterocarp-rich stands have the highest offtakes, often ~10 trees or over 100m^3 per ha (Edwards et al. 2014). One pantropical meta-analysis of faunal impact studies suggests that as logging intensity increases, species richness of invertebrates, amphibians and mammals decreases and that the effect varies by continent and taxonomic group (Burivalova et al. 2014). The effects of disturbance on forest fauna tends to be more severe in the Neotropics than in the Indomalayan or African tropics

(Burivalova et al. 2014). Nevertheless there is a lack of consensus on the overall impacts of selective logging; most impact studies fail to conduct baseline sampling, quantitatively describe habitat structure or even report on logging intensities (Laufer et al. 2013), limiting inference and comparability.

Moreover, species richness does not fully explain the effects of logging on fauna. Birds demonstrate an increase in species richness following logging but this has been largely attributed to a net increase in habitat generalists moving into degraded areas despite declines in understorey specialists or species restricted to terrestrial microhabitats (Johns 1991). Species with particular ecological requirements and life history and functional traits are more vulnerable to the impacts of logging and their population declines can cause unknown cascading effects on community-wide food-web structure and function (Reiss et al. 2009). Functions and processes that may be affected include nutrient decomposition, pollination, predation and seed dispersal (Dobson et al. 2006). These vulnerable species include large-bodied frugivores (Johns & Skorupa 1987) which also play vital roles as seed-dispersers of tree species. Their demise has thus been shown to reduce seedling recruitment in Amazonian, African and Bornean forests (Terborgh et al. 2008; Levi & Peres 2013; Harrison et al. 2013; Poulsen et al. 2013). Logging is most often the first door or gateway to forest degradation and conversion (Lewis et al. 2015). Loggers begin the cycle of degradation through the construction of roads (Kirby et al. 2006; Southworth et al. 2011), which open previously inaccessible primary forest to increased forest fragmentation and anthropogenic impacts such as hunting (Peres 2001; Espinosa et al. 2014; Kleinschroth et al. 2015).

Plant life history traits may render particular tree species more vulnerable to the impacts of logging (Martini et al. 1994, Fig 1.5). Subsequent deleterious impacts of logging thus also include compositional shifts as densities of high-value overexploited timber species plummet to very low levels (Zimmerman & Kormos 2012), consolidating the dominance of light-wooded fast-growing pioneer species in the residual stand (Phillips et al. 2004; Macpherson et al. 2012). There are also increased risks of fire and feedback systems that can severely compromise regional to global scale climatic stability and carbon storage (Nepstad et al. 1999; Cochrane 1999; Siegert et al. 2001; Nepstad et al. 2004; Houghton et al. 2000; Espírito-Santo et al. 2014).

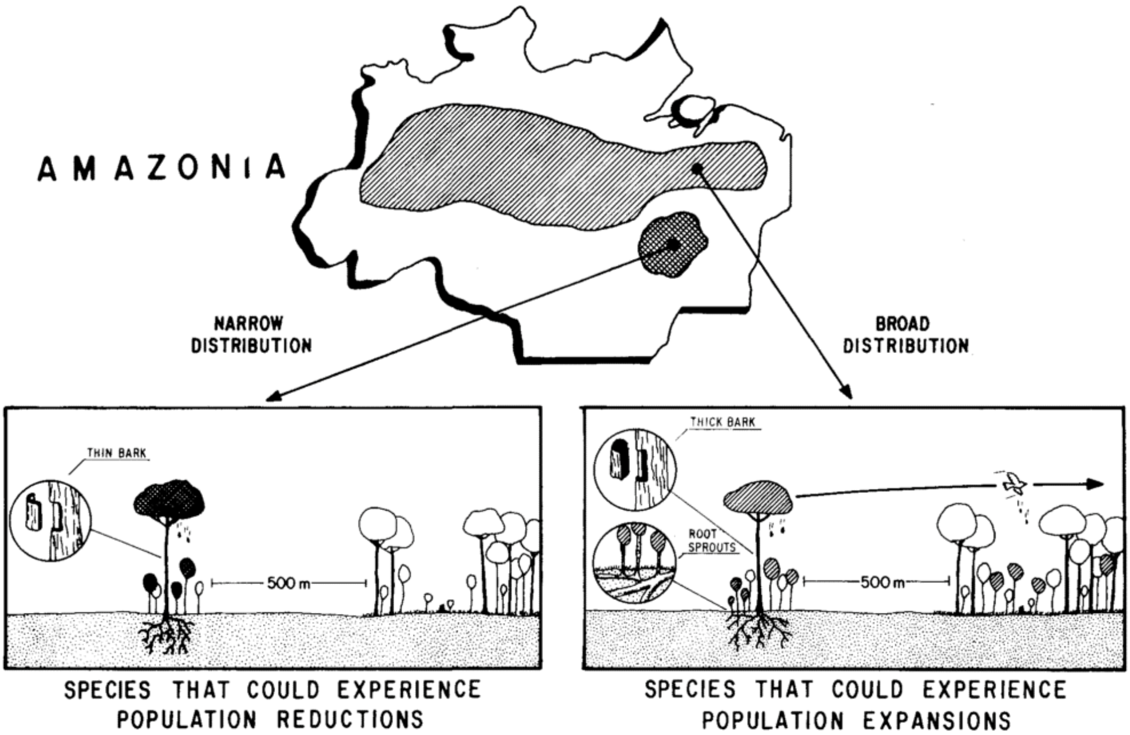

Figure 1.5 The ecological characteristics of timber species could make them vulnerable to logging impacts. For example, the hypothetical species presented on the left has its range restricted to the eastern Amazon (where logging activities are concentrated), lacks a system of long-distance dispersal, has few saplings in the understorey, is unable to resprout after cutting or crushing, and is killed by ground fires (owing to its thin bark). In contrast, a second hypothetical species (right) may actually be favoured by logging because it is distributed throughout Amazonia, has effective long-distance seed dispersal, is abundant in the regeneration, resprouts readily, is fire resistant, and regenerates vigorously in canopy openings. Source (Martini et al. 1994).

1.7 Research context and objectives

This study was conducted in collaboration with the Brazilian Belém based NGO IMAZON. For over 25 years, they have been conducting applied conservation and public policy research across the Amazonian region. Among their various programmes, IMAZON has been scrutinizing government-approved logging concession management plans in order to assess the effectiveness and quality of forest plans. To date, this has been done spatially through the processing of satellite images and landholding boundaries (e.g Monteiro et al. 2012). The NGO Greenpeace-Brasil, in collaboration with SEMA (State Environmental Secretariat of Pará) and Brazil's Public Prosecutor's Office, carried out a systematic review of management plans in Pará between 2006 and 2013 to assess the extent to which ipê (*Tabebuia serratifolia, T. impetiginosa*) timber laundering occurred (Greenpeace 2014). At the time of writing and to the best of our knowledge, an assessment of the species-specific composition of timber offtakes declared within these forest management plans had not yet been examined.

After reviewing the current state of knowledge, Chapter 2 aims to:

- Examine the historical and environmental determinants of the structure of timber offtake by volume and tree species along variable-aged logging frontiers in eastern Amazonia.
- On the basis of both geographic and historical variables associated with each logging concession, explain broad patterns of available timber stocks, estimated revenues derived from those stocks, and the timber species composition of residual stands.
- Highlight important implications for current legal frameworks governing logging concessions and explore the degree to which the current model may be economically viable and demographically sustainable.

In Chapter 3 the assessment of forest management plans was extended to examine economies or diseconomies of scale using; the economics and general indicators of sustainability of legally sanctioned private and community-based logging concessions representing 824 small, medium, and large landholdings. Specifically, the aim was to understand:

- What is the role of smallholders in the context of the Amazonian logging industry or how does their timber offtake decisions or productivity compare against that of large landowners?
- The degree to which smallholders are able to manage tropical forest timber stocks sustainably.
- To what degree landowners have complied with legal frameworks that require permanent riparian forest set-asides and at least 80% of landholdings to be maintained as native forest reserves.
- The implications of landholding size in the context of rural development policies governing logging concessions and the global timber industry.

Because most of the timber offtakes in Amazonia continue to be illegally harvested it can be difficult for best-practice RIL concessions to remain financially competitive (Rice et al. 1997). I was fortunate to be offered an opportunity to discuss these issues with the owners and sustainability representatives of a certified industrial-scale RIL operation of eastern Brazilian Amazonia (CIKEL) and conduct fieldwork in their landholding. Following the dip in timber productivity after the financial crises, the company attempted to evaluate the extent of collateral damage resulting from their logging operation. Logging induced degradation is poorly quantified or understood even though this represents the second 'D' of REDD (Reduced Emissions from Deforestation and Degradation) as recognised by both the UN Framework Convention on Climate Change and the Intergovernmental Panel on Climate Change.

In Chapter 4, the aim was to:

- Understand why and how collateral damage is currently defined.
- Assess the extent to which studies on selective logging impacts in tropical forests mention or attempt to quantify collateral damage to the residual stand.
- Estimate using different approaches the collateral damage associated with a certified industrial-scale RIL operation in the eastern Brazilian Amazonia.

In Chapter 5, data from Chapter 4 were combined with large-scale forest inventory data from the CIKEL RIL operation to address the following objectives:

- Examine the extent of collateral damage as biomass resulting from both felled-tree impacts and infrastructure damage of a reduced impact logging concession in eastern Amazonia
- Estimate the gross monetary value of harvested trees and logging profits.
- Estimate the monetary value of biomass loss through the harvested trees and associated collateral damage.
- Estimate the cost of sparing logging-induced degradation through carbon financing schemes such as REDD+.

1.8 Thesis structure

All data chapters (2-5) were written in the format of peer-reviewed papers. My supervisor Carlos Peres made important contributions regarding the conception of the study and in the ecological interpretations of the results. This contribution is recognised through co-authorship of Chapters 2-5. At the time of writing, Chapter 2 has been submitted to PLoS ONE and Chapters 3, 4 and 5 were written with specific journals in mind, and are intended to be submitted before the end of 2015.

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1.9 Appendix 1

1.9.1 Glossary*

APU (Annual Production Unit)
Term used to annually label or define the different parts of a landowner's property that was logged in different years. APU's are temporally and spatially distinct.

Bole
Main part of the stem of a tree before it separates into branches.

Breast height
1.3 metres (43 inches) above the ground on the highest side. Point at which diameter or girth is measured on a standing tree.

Broad-leaves
Term used to describe trees other than conifers. Broadleaves may be deciduous e.g. oak, or evergreen e.g. holly.

Canopy
The foliage and small branches of tall trees in a wood when these have interlaced to form continuous cover.

Crown
Branches and upper part of the stem of tree.

Diameter at breast height (DBH)
The standard way to measure standing trees using a girth tape and measuring at 1.3 metres above ground level.

Enrichment
Addition of a few young trees by planting to an area already established but not fully stocked.

Hardwood
The wood of broadleaved trees, a term sometimes used for the broadleaved trees themselves.

Increment
The amount of new wood put on by a tree or a stand in a year or in the period between thinning measured either in cubic metres or in cubic metres per hectare.

Monoculture
Growing one species as a crop.

Native species
Species that have arrived and inhabited an area naturally, without deliberate assistance by man.

Natural regeneration

Young seedlings that have arisen from seed falling from trees nearby or dispersed by wind or native forest species.

Regeneration
The production of a new crop by artificial or natural regeneration.

Silviculture
Cultivation of trees as crop with the primary objective of producing wood products.

Skidder
Tractor used to pull pole length timber along the ground.

Stumpage
The price a private firm pays for the right to harvest timber from a given land base. Expressed as per m^3 standing.

Succession
Changes that occur in vegetation as bare ground is progressively colonised by different species.

Understorey
Trees and/or shrubs below the canopy.

***Definitions are my own and from the UK Forestry Commission available at http://www.forestry.gov.uk/fr/INFD-5UWJWZ**

1.9.2 List of Brazilian legal frameworks and forestry related legislation

- Law 4771 (1965), Forest Code and amended by Law 12.651, March 25th 2012.
- Law 5197 (1967), Protection of Fauna.
- Law 6938 (1981), National Environmental Policy
- Law 9433 (1997), Water Resources Policy
- Law 9605 (1998), Environmental Crimes.
- Decree 3179 (1999), determines penalties for forest crimes.
- Decree 3420 (2000), forming the National Forest Programme.
- Decree 4340 (2002), regulates articles of the Forest Code and other laws. Additionally, it also provides guidelines for the utilization and clear-cutting of forests, forest restoration, and licences to transport forest by-products.

- Law 11 284 (2006) (the Public Forest Management Law), which guides public forest management for sustainable production, within the structure of the Brazilian Ministry of the Environment it establishes the Brazilian Forest Service, and creates the National Forest Development Fund (Fundo Nacional de Desenvolvimento Florestal, FNDF).
- Resolution 378 (2006), notably subjects forest exploitation to authorisations issued by the Brazilian Institute of Environment and Renewable Resources (Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis – IBAMA).
- Resolution 379 (2006), establishes and controls the database on forest management at the National Environmental System (Sistema Nacional do Meio Ambiente).
- Decree 6063 (2007), controls at the federal level, requirements of the Public Forest Management Law.
- Resolution 406 (2009), creates technical standards to be adopted in the preparation and implementation of sustainable forest management plans (Planos de Manejo Florestal Sustentável, PMFS) for logging purposes in natural forests in the Amazonian biome.
- Resolution 411 (2009), regulates inspections of operations that process charcoal and residual wood products from natural forests to ensure they do not come from illegally deforested areas.
- Law 12.187 (2009), establishes the National Climate Change Policy (NCCP) of Brazil.
- Decree 6.874 (2009) PMCF (*Programa Federal de Manejo Florestal Comunitário e Familiar*), establishes the federal community forestry program.
- Resolution 01 of 2015 updates Law 11 284 (2006); all concession areas to maintain at least 15% of the large trees or four large trees per 100 ha (per harvested species) within their Annual Production Units, although this is often difficult to ascertain.

Chapter 2: Reaping the last harvest: temporal decay in timber species composition and value in Amazonian logging concessions

2.1 Abstract

Throughout human history, slow-renewal biological resource populations have been predictably overexploited, often to the point of economic extinction. We assess whether and how this has occurred with timber resources in eastern Brazilian Amazonia. The asynchronous advance of industrial-scale logging frontiers has left a regional forest landscape with varying histories of logging. Initial harvests in unlogged forests can be highly selective, targeting slow-growing, high-grade, shade-tolerant hardwood species, while later harvests focus on fast-growing, light-wooded, long-lived pioneer trees. Brazil accounts for 85% of native neotropical forest roundlog production, and the State of Pará for almost half of all timber production in Brazilian Amazonia, the largest natural tropical timber reserve controlled by any country. Yet the degree to which timber harvests beyond the first-cut can be financially profitable or demographically sustainable remains poorly understood. Here, we use data on legally planned logging of ~17.3 million cubic meters of timber across 314 species extracted from 824 authorized harvest areas in private and community-owned forests, 446 of which reported volumetric composition data by timber species. We document patterns of timber removals by volume, species composition, and monetary value along aging eastern Amazonian logging frontiers, which are then explained on the basis of historical and environmental variables. Generalized linear models indicate that relatively recent logging operations farthest from heavy-traffic roads are the most selective, concentrating gross revenues on few high-value species. We find no evidence that the post-logging timber species composition and total value of forest stands recovers beyond the first-cut, suggesting that the commercially most valuable timber species become predictably rare or economically extinct in old logging

frontiers. In avoiding even more destructive land-use patterns, managing yields of selectively-logged forests is crucial for the long-term integrity of forest biodiversity and financial viability of local industries. The logging history of eastern Amazonian old-growth forests likely mirrors unsustainable patterns of timber depletion over time in Brazil and other tropical countries.

2.2 Introduction

Biological populations with slow life-histories and yielding commercially valuable natural resources have been predictably overexploited over the course of human history, often through a ratchet effect, stemming from heavily subsidised industries to the point of economic extinction or demographic collapse (Smith 1968; Ludwig et al. 1993). This effect may also apply to extractive industries fuelled by non-renewable resources as illustrated by peak oil (Sorrell et al. 2010) and peak phosphorous (Cordell et al. 2009). Yet overexploitation is not necessarily inevitable but rather a common consequence of low-governance and open-access systems or 'the tragedy of the commons' (Hardin 1968; Ostrom 1999). Emblematic historical collapses of myriad wild resource populations include the Antarctic blue whale, cod stocks off New England and eastern Canada, and Mediterranean bluefin tuna (Myers et al. 1997; Clark 1973; MacKenzie et al. 2009). Over 230 overharvested fish populations have shown median reductions of 83% from known historical levels (Hutchings & Reynolds 2004), often within a mere 15 years of exploitation (Myers & Worm 2003).

Economic extinction risk in terrestrial resource populations is analogous to those in marine ecosystems (Dulvy et al. 2003). For centuries, old-growth timber stocks have been rapidly mined at the expense of future cohorts (Repetto & Gillis 1988). The once abundant endemic stands of Brazil-wood or Pau Brasil (Caesalpinia echinata, Leguminosae-Mimosoidae), a colonial source of red dye and hardwood after which Brazil was named, were heavily harvested four centuries ago to the brink of extinction, except for a few small, highly inbred relict populations (Cardoso et al. 1998). Overexploitation of tropical forest tree species remains ubiquitous today, targeting prime timber and nontimber resources such as big-leaf mahogany (Swietenia macrophylla (Verissimo et al.

1995)) and Brazil-nuts (Bertholletia excelsa (Peres et al. 2003)). Amazonian rosewood populations deriving highly prized linalol essential oils (Dalbergia nigra and Aniba rosaeodora) have also been severely depleted by the high-end perfume industry to the point of widespread extinctions (May & Barata 2004).

Whilst logging operations in tropical forests are highly variable in the degree to which they can be defined as sustainable, international consensus still deems logging to be one of the best compromises between land-use revenue and forest conservation (Edwards et al. 2014), and timber to be an inherently renewable resource (FAO 2012). However, clear signs of 'peak timber' are already evident in Asian markets, as the region fast approaches a typical symmetric 'Hubbert Curve' logistic distribution seen in many overexploited non-renewable resources (Shearman et al. 2012). Counterintuitively, the demise of a high-value resource population may be the most attractive economic logic governing the behaviour of individual harvesters (Clark 1973). Multi-temporal studies indicate that high individual discount rates can encourage the liquidation of commercially valuable timber stocks even when land-tenure is secure, wherever these can provide additional short-term revenues. These can be banked or reinvested into more profitable land-use options, rather than allowing longer time-horizons of slow timber regrowth in regenerating stands (Howard et al. 1996; Rice et al. 1997; Winterhalder & Smith 2000). This often results in a highly degraded natural resource capital, with few options for alternative extractive industries. For instance, selectively logged Amazonian forests are much more likely to be deforested than unlogged forests (Asner et al. 2006), and local livelihoods generally follows a typical boom-and-bust trajectory at selectively-logged and subsequently deforested development frontiers (Rodrigues et al. 2009) (but see Weinhold et al. 2015).

Sequential harvests in old-growth tropical forests typically progress from high-value, shade-tolerant, long-lived and large-girthed tree species (with generally high wood density, and often described as hard or heavy-wooded) toward a greater reliance on short-lived, low-value pioneer species (low wood density, or light-wooded) (Macpherson et al. 2012), with over 300 tree species considered to be commercially valuable in eastern Amazonia (Martini et al. 1994). Valuable Amazonian timber tree species vulnerable to moderate and high extinction-risk include a range of canopy and emergent species such as big-leaf mahogany, Brazil-nut tree, ipê (Tabebuia serratifolia, T. impetiginosa), jatobá (Hymenaea courbaril), freijó cinza (Cordia goeldiana) and pau amarelo (Euxylophora paraensis) (Martini et al. 1994; Schulze et al. 2008). This may result in a predictable compositional shift from heavy-wooded to light-wooded species following each cutting cycle (Phillips et al. 2004). Market dynamics change to reflect this trend. In the 1990s, new logging frontiers in Peruvian Amazonia were highly selective targeting mahogany and up to 80-90 species (Rice et al. 1997), but 350 timber species were already being harvested in eastern Amazonia in the same decade (Martini et al. 1994). Industrial scale logging is highly selective in Central-West Africa, with 95% of the offtake from the Congo basin comprising only 55 species (Ruizperez et al. 2005). Notwithstanding ambiguities with species identification and nomenclature, a few hundred timber species are still currently available in Amazonian markets (DOEPA 2010) from the approximately 16,000 described and undescribed species in the Amazonian tree flora (ter Steege et al. 2013). Market studies provide only a snapshot in time, so we propose that compositional profiles of species selectivity may be crude but reliable surrogates of the degradation status of local timber stocks along the accessibility gradient of tropical logging frontiers.

The notion that industrial scale timber extraction follows a frontier gradient is not new (Souza Jr. et al. 1997; Stone 1998). In the early 1970s, large swathes of primary eastern Amazonian forests remained inaccessible, stumpage values were negative, and fiscal incentives and subsidies were handed out to convert forest to pasture. Early tropical logging frontiers are often characterized by highly selective, mobile, low-efficiency extraction targeting the most valuable species, with high transport costs and unstable property rights. As logging frontiers age, investments into transport, extraction and processing infrastructure are consolidated and high-value timber resources rapidly dwindle. Increasing volumes of timber offtake (defined here as the removal volume of harvested trees) are then required to maintain profits at much smaller margins.

Yet our understanding of the structure and composition of logged forests beyond the first old-growth harvest remains very limited (Zarin et al. 2007). Each cutting cycle continuously selects offtakes of high-value, slow-growing hardwood species, which may lead to greater functional homogenization of the remaining woody flora which can have complex and unpredictable ecological consequences on the biodiversity of exploited ecosystems (Tabarelli et al. 2012). Significant implementation of sustainable forest management is further compounded by pervasive illegal logging activities, which account for 50–90% of all pantropical native forestry products worth US$30 – 100 billion yr–1 or 10–30% of the global wood trade (Nellemann 2012), and competes with lower-impact legal logging. For instance, spectral mixing analysis (using the Normalized Differencing Fraction Index (Souza et al. 2005)) of logging-induced forest disturbance over 3 years (August 2009 – July 2012) across 358,843 ha of eastern Amazonia indicates that 69.7% of this area was logged illegally by unauthorized ('predatory') logging operations (Monteiro et al. 2012).

Brazil accounts for 85% of the roundlog production in Latin America/Caribbean region, and the total harvested volume was 30.8 million m3 in 2012, overlooking the poorly quantified illegal trade (ITTO 2012). Brazil is estimated to hold 64% of the world's total intact forest landscapes, 836 Mha (Potapov et al. 2008), of which 72% of natural (i.e. non-plantation) forests are in Amazonia (SFB 2013a). Since 2006, the Brazilian Forest Service (SFB) has granted logging concessions in National and State Forests to either companies or local communities through a national bidding process (Law 11.284 of 2006). In 2013, 5.3 Mha were available to new concessions with an additional 4 Mha becoming available in 2014 (SFB 2013b). The 125 Mha State of Pará (the second largest in Brazil) has experienced the oldest history of logging across Amazonia spanning three centuries, but still retains vast untapped timber stocks in remote unlogged forests. Native timber has become the mainstay of the Pará economy since the first road linking the state to southern Brazil was built, with over 2000 sawmills producing 13 million m3 of sawnwood in the early 1990s (Uhl et al. 1997). In 2009, Pará accounted for 47% of the roundlog production, 44% of the gross timber revenue (~US$1.1 billion), and 45% of all direct and indirect jobs in the wood-related sector in Brazilian Amazonia (Pereira et al. 2010).

Although much of the literature condemns conventional over reduced-impact logging, there are few attempts to understand the overall impact of selective logging on biodiversity (Laufer et al. 2013; Burivalova et al. 2014). For example, to what extent timber harvests beyond the first cut can be financially profitable and/or ecologically sustainable? This question remains poorly known even in low-damage logging operations. Yet the long-term economic viability of selectively-logged tropical forests is crucial if they are to persist, thereby avoiding more destructive alternative land-use options.

Here, we examine the historical and environmental determinants of the structure of timber offtake by volume and tree species along variable-aged logging frontiers in eastern Amazonia. We summarize data on legal logging operations across their entire size range, which between 2006 and 2012 were authorized to extract some 17.3 million m3 of logwood from an aggregate forest area of 638,679 ha distributed across 824 private and community-based logging concessions. Rather than being restricted to public forests, these authorized concessions to harvest old-growth timber included primarily private landholdings. On the basis of both geographic and historical variables associated with each logging concession, we then attempt to (1) explain broad patterns of available timber stocks, (2) estimate revenues derived from those stocks, and (3) estimate the timber species composition of residual stands beyond the first cut. Finally, (4) we highlight important implications for current legal frameworks governing logging concessions and explore the degree to which the current model may be economically viable and demographically sustainable.

2.3 Methods

2.3.1 Study areas and AUTEF management plans

Mandatory legal approval of forest management plans within the eastern Amazonian state of Pará must be issued to all timber extraction enterprises, including those in communal lands, small to medium private properties, and largeholdings controlled by logging companies, in the form of a 'Forest Exploitation Permit' (Autorização de Exploração Florestal; hereafter, AUTEF). The State Environmental Secretariat of Pará (SEMA) issues AUTEF plans, a legal requirement under both SIMLAM (Brazilian Integrated Environmental Licencing and Monitoring System) and SISFLORA (Forest Product Trade and Transport

System) for planned timber harvests of any forest site at any spatial scale. We extracted and digitized data from a total of 824 AUTEF plans across Pará sanctioned between 2006 and 2012. These included the name of the rural entrepreneur, community, landholder, or company carrying out each logging operation, the municipal county, the total landholding size, the net size of areas authorized for logging (excluding legally protected riparian forest set-asides where logging is not permitted), and the geographic coordinates of each concession. Of these, a more detailed subset of 678 AUTEF plans also included the additional set-aside areas within the landholding defined as 'Legal Reserves' according to the Brazilian Forest Bill (No. 12.727, http://www.planalto.gov.br/ccivil_03/_Ato2011-2014/2012/Lei/L12727.htm). A further detailed subset of 446 AUTEF plans (issued between 2009-2012) also included the forest type (planted or natural), the total standing volume authorized for extraction, and the total authorized volume (m^3) of inventoried timber per tree species per concession to be extracted (minimum cutting diameter at breast height of 50 cm, Normative Ruling no. 05 of 2006). We therefore use either one of these data sets (N=824, or N=446), depending on the nature of the analysis. Because timber species were identified *in situ* within concession areas by experienced tree parataxonomists hired to support management plans, we converted vernacular names into their corresponding Latin nomenclature and then removed species-level synonymia whenever necessary based on a comprehensive checklist of timber species of central and eastern Amazonia compiled from multiple sources (Silva et al. 1977; Parrotta et al. 1995; Ribeiro 1999; Lorenzi & Flora 1998; Lorenzi 2002; Lorenzi 2008).

AUTEF plans granted for exotic tree monocultures, which included eucalyptus (*Eucalyptus*), teak (*Tectona*), and pine (*Pinus*) plantations, were excluded from the analyses. Although Paricá (S*hizolobium amazonicum*) plantations were reported in landholdings (20%), this species is native to Amazonia and was therefore retained in those AUTEF plans defined as 'natural' forests. All AUTEF applications to SEMA referred to a unique forest stand of known size based on GPS fixes of property boundaries, although a few exceptionally large landholdings controlled by a logging company may have included more than one AUTEF for different logging compartments exploited in different years (UPAs, Annual Production Units).

2.3.2 Timber price data

Given that many species accrue significant value along different supply chains and export market prices are affected by complex international demands, we used regional scale logwood prices per timber species in Brazilian Reais (R$ per m^3 of lumber) available from an official source for the state of Pará that serves as a benchmark for timber merchants (DOEPA 2010). This reflects the dominant domestic market, which consumed 95% of all timber produced in Brazil in 2011 (ITTO 2012), and best reflects realistic transaction prices of unprocessed timber expected by loggers at sawmills or other points of timber sales. Timber prices (R$/$m^3$) are grouped by DOEPA (2010) into four categories, with gradually fewer timber species commercialized under increasingly higher price brackets: Class A (11 species, 6 genera): > R$75.0/$m^3$; Class B (18 species, 12 genera): R$45.0/$m^3$ - R$74.0/$m^3$; Class C (40 species, 31 genera): R$25.0/$m^3$ - R$44.0/$m^3$; and Class D (all other 245 species within 157 genera): R$1.0/$m^3$ - R$24.0/$m^3$. The logwood price data we used are deliberately conservative compared to other sources, which may take into account valuation along supply chains (Stone 1998; Bacha & Rodriguez 2007).

Data on local timber extraction costs were unavailable so our analyses are based on estimates of gross expected revenues. However, extraction costs should scale to the total volumes of timber removed (Barreto et al. 1998) and extent of logging areas exploited, which are taken into account here. Alternative sources of income that may be available to different landholders may also affect local economies of scale and timber species selectivity but are beyond the scope of this study. These may include sales of non-timber forest products and residual dead wood derived from collateral damage at logging clearings (e.g. to meet the high charcoal demand for smelting iron ore in eastern Amazonia), and value-added through timber processing capacities.

2.3.3 Geographic data

Because exact landholding boundaries of logging concessions were unavailable from AUTEF management plans as spatially explicit polygons, circular buffers of sizes corresponding to each known landholding area (range = 26 – 844,021 ha), which had been reported in all 824 AUTEF plans, were projected around their geographic coordinates using ESRI ArcMap 10.2.2. Each of these buffers was then assigned an additional 10-km radius external buffer to represent the approximate landscape structure of the forest/non-forest matrix surrounding each AUTEF landholding.

Baseline data on forest structure and composition prior to any large-scale timber extraction were unavailable for logging concession sites. However, we use data from the comprehensive RADAMBRASIL forest inventories (Brasil 1978), which were conducted by the Brazilian government from the late 1960s to the early 1970s to map timber resources across Brazilian Amazonia, to estimate the plot-scale aggregate basal area (BA, m^2/ha) and wood specific gravity (wood density, g cm^{-3}) under pre-logging conditions for each AUTEF site. We considered all tree species inventoried within each timber price bracket. RADAMBRASIL is the most extensive network of forest plots ever undertaken across the entire Brazilian Amazon, and included at least 2,345 one-hectare plots surveyed across the region (/<ftp://geoftp.ibge.gov.br/>), within which a total of 128,433 canopy trees ≥ 31.8 cm in diameter at breast height (DBH) [or ≥ 100 cm in circumference at breast height (CBH)] were sampled. This was done using an ordinary krigging interpolation of total forest BA within each timber price bracket from all 1-ha plots. We also used RADAMBRASIL data to test for proportional differences in total basal area (m^2) and total volumes (m^3) of high vs low value timber (classes A - B and C – D, respectively) available across the four major logging frontiers of varying histories containing the logging concessions examined here. These frontiers are, from the oldest to the most recently exploited: (1) East Pará, primarily along the Belém-Brasília Highway (BR-010) and the main State Highway of Pará (PA-150); (2) Terra do Meio region, along the Transamazon Highway (BR-230); (3) the Calha Norte region of northwestern Pará; and (4) along the Cuiabá-Santarém Highway (BR-163) of southwestern Pará.

Approximate dates of logging frontiers follows (Pereira et al. 2010), but these were further validated and refined by accounting for the official onset of any INCRA agrarian settlement within a 75-km buffer of each AUTEF geographic centroid (Table S2.1). These government-sanctioned agrarian settlements typically mark the arrival of first settlers into new forest frontiers as they rapidly take advantage of new roads into previously inaccessible areas (Peres & Schneider 2012). In addition, earlier cycles of logging in eastern Amazonia typically occurred within 25 km of major roads (Uhl et al. 1991), so dating of logging frontiers corresponding to each AUTEF site was further verified by accounting for the completion year of all major paved and unpaved roads (or road segments) built in previously remote forest regions based on a comprehensive compilation of historical road-building records (see Table S2.1 for a full list of explanatory variables).

The proportion of forest and deforested areas (in 2011), natural savannahs (*cerrado*), and water bodies were calculated for both internal landholding projections and external buffers using 30-m resolution data from the Brazilian Space Agency PRODES project (Table S2.1). Deforestation areas under cloud pixels (for which the deforestation year was unknown) were excluded from any landholding projection, but these amounted to only <3% of all pixels. Road traffic data for 2010, including heavy cargo and passenger vehicles, were obtained for all segments of existing paved and unpaved roads within the state of Pará from Brazil's Ministry of Transport (Table S2.1). Heavily used roads are defined as those used by a daily average of ≥1,000 heavy vehicles (buses and heavy cargo vehicles, including roundlog trucks). Sub-municipal scale human population density (HPD, persons/km^2) IBGE data, including both rural and urban populations, were extracted for all 8,919 census districts across the 144 municipal counties of the state of Pará (in 2011, (IBGE 2011) Table S2.1).

Additional analyses encompassing the entire state of Pará were conducted using a grid of 50km x 50km (2,500 km^2) cells. Peripheral cells straddling state boundaries were segmented so that only portions contained within Pará were considered in our analysis. Using ArcMap 10.2.2, we quantified for each cell (N=564) the cumulative amount of deforestation by 2012, the proportional representation of forest and non-forest land-cover types, the overall density of paved and unpaved roads (km/km^2), and the mean HPD from the 2007-2010 IBGE census, in addition to the number of AUTEF sites with centroids within a given cell. A comprehensive list of site, landscape, geographic and socioeconomic variables examined and their sources are available in Supplementary Information (Table S2.1).

2.3.4 Data analysis

Patterns of timber tree species volumetric abundance and dominance within AUTEF management plans were examined using data from *in situ* forest inventories reported for each logging concession area authorized by SEMA. We examined the correlation structure between species-specific timber market prices (R$/m^3) and total timber stock sizes (m^3) quantified within any authorized concession area.

Timber tree species within each concession were rank-ordered in terms of their overall stock value (R$), defined as their total volumetric stock (m^3) multiplied by the species-specific reported timber price per m^3 to examine the offtake distribution of species-specific timber values (Σ R$ • m^3). Using the *vegan* package in R, we then constructed Rank Abundance Distribution (RAD) curves (McGill et al. 2007) in timber stock values (on a log-scale) to derive the evenness J' (Pielou 1975) in timber revenues across all co-occurring species exploited at a concession. This provides additional insights into the degree to which loggers could maximize harvesting selectivity by focusing on high-value timber to most efficiently meet their maximum legal quota of 30 m^3/ha, as required in approved AUTEF plans. Pielou's J' evenness could thus be defined as a measure of high-grading of an assemblage of coexisting tree species within a concession area. We selected this evenness measure because it is the most widely used in ecology, and is an excellent species-abundance predictor of species richness in tropical forests (He & Legendre 2014). J' values range from 0.0 to 1.0, with larger values representing more even species distributions in reported volumetric offtakes in relation to market values of timber tree species, or a wider offtake portfolio of timber species by value suggesting lower species selectivity. Conversely, steeper RAD slopes represented by lower J' values indicate high-grading, or timber revenues disproportionately concentrated on only a few highly profitable timber species.

To examine multivariate patterns of species composition we used Nonmetric Multidimensional Scaling (NMDS) ordination based on the total volumetric abundance of different timber species declared within each AUTEF management plan. Vernacular identification of Amazonian trees is often ambiguous at the species level for several tree morphospecies but sufficiently robust at the genus level (Higgins & Ruokolainen 2004). However, identification ambiguity is a lesser problem for the much smaller subset of commercially important timber species identified by experienced parataxonomists in the field. NMDS ordinations thus used an abundance-based (Bray-Curtis) similarity matrix including 153 tree genera surveyed across the 446 concessions for which a timber tree inventory was available, once the raw data had been standardized and sqrt-transformed. Stress values at two-dimensional scaling was 0.23 or lower and ordination scores along the first axis ($NMDS_1$) are defined as an additional descriptor of the concession-scale composition of timber species in terms of their declared volumetric abundance.

Timber tree assemblage patterns at the scale of individual stands were further related to geographic and historical variables describing each concession using the BIOENV procedure in Primer version 6.0, using the Spearman rank correlation coefficient (ρ). The BEST analysis within BIOENV allows the exploration of environmental variables that best correlate with the dissimilarity patterns observed in biotic assemblages by calculating a rank correlation between the Bray-Curtis dissimilarity matrix [weighed in terms of species-specific timber volumes (m^3) per logging concession] and the Euclidean distances of the abiotic data (Clarke & Ainsworth 1993). BIOENV results were then tested using a non-parametric Mantel test (RELATE) procedure, which compares the global ρ to the distribution of ρ under the null hypothesis generated by 999 random permutations. To assess differences in genus-level composition of timber volumes among variable-aged logging frontiers we used analysis of similarities (ANOSIM), which compares within- and between-group variances using the R statistic, which ranges from −1.0 to +1.0 with a value of 0 indicating no difference among groups (Clarke & Ainsworth 1993). The significance is determined by comparing the observed R value to the distribution of R under the null hypothesis of no difference between groups ($\alpha = 0.05$).

GLMs were constructed to model three response variables associated with the total worth of timber resources per concession: (1) the mean estimated density of gross timber value (R$/ha) per concession from trees available at the time of forest inventories; (2) the timber species J' evenness in total value by volume; and (3) the first ordination axis ($NMDS_1$) describing the broad multivariate patterns of genus-level timber composition. We used a negative binomial error distribution to model the first response variable, and a Gaussian distribution to model the second and third variable. To examine these responses we included covariates that described the size, history (frontier age), geographic features, and the landscape structure of each concession (see Table S2.1).

To test for collinearity among variables, pairwise scatterplots and Pearson correlation coefficients were applied to all covariates, none of which exceeded our 0.6 threshold in pairwise scatterplots. Homoscedasticity assumptions, outliers and influential cases were investigated using standardized residuals and Cook's distance. Variance inflation factor (VIF) and tolerance statistics (1/VIF (model)) were calculated. Within our GLM models, all VIF values remained below a preselected threshold of 3 (Zuur et al. 2010) and tolerances were well above 0.2 (Menard 1995). The assumption of independent errors was tested using the Durbin-Watson test, which were within an acceptable range (1-3:[66]), and overdispersion was not found (Crawley 2002). Spatial autocorrelation can increase type I errors by introducing biases due to violation of the assumption of independent and evenly distributed residuals [71]. To examine the degree to which AUTEF plans were spatially autocorrelated, Moran's I correlation coefficients were calculated using the residuals of our three GLMs outlined above. No significant residual autocorrelation was found across our models and this was confirmed by inspection of spatial correlograms in R. For each response variable a global model containing all predictors was first developed and then all candidate models were ranked according to the AIC difference between the lowest AIC model and model *i* (ΔAIC). Where model sets 'best' model had a weight of <0.9 and ΔAIC < 2, model averaging was used to estimate coefficients (Burnham & Anderson 2002). All analyses were done in R version 2.15.1 (R Development Core Team 2014) and Primer version 6.0.

2.4 Results

In total, 9,568,249 m^3 of timber representing 314 native tree species were declared for legal offtakes in the state of Pará between 2006 and 2012 from 446 private concessions and community-based logging areas for which the taxonomic composition of timber trees was available. We examined the correlation structure between species-specific timber market prices (R$/$m^3$) and total timber stock sizes (m^3) quantified within any authorized concession area. Pearson correlation values are higher where declared harvests for standing high-value timber are largest, but lower and often negative where loggers target primarily low-value timber. This provides insights into the availability of the most valuable tree species to loggers, although low correlation values could suggest either evidence of local depletion of the most desirable species or deliberate restraint from forest high-grading practices.

Total timber volumes (but not species composition) were also available for a further 378 concessions exploited over the same period. These timber management plans are associated with landholding sizes ranging from 26 to 910,307 ha (mean ± SD = 13,810 ± 71,719 ha, N=824). The proportional area within a landholding boundary authorized for timber extraction was highly variable, ranging from 0.14% to 100% of the total property size (mean ± SD = 49.8 ± 27.3% N=824). Absolute timber offtake per management plan ranged between 77 and 298,612 m^3 of logwood (mean ± SD = 20,966 ± 27,292 m^3, N=824) depending primarily on sizes of authorized areas, although reported timber offtake rates per unit area were highly invariant (mean ± SD = 27.2 ± 4.5 m^3/ha, N=824) and below the legally required maximum quota of 30 m^3/ha.

Authorized timber extraction encompassed a taxonomic spectrum of 314 tree species representing 153 genera and 38 families. However, the total number of exploited taxa per concession site ranged from only 1 to 70 species (22.3 ± 10.6), 1 to 57 genera (20.9 ± 9.1) and 1 to 27 families (12.8 ± 4.5), suggesting high variance in availability of timber stocks and selectivity. Timber prices per cubic meter ranged from R$ 1/m^3 to R$86.5/$m^3$, although this does not include exceptionally high-value species, such broad-leaf mahogany, because these were not listed in any of the concession forest inventories.

Significant predictors in our GLMs explaining the numeric incidence of AUTEF plans within 2,500-km^2 grid cells across the entire state of Pará (Fig. S2.1) included the proportion of deforestation (β = 0.150, P < 0.001), paved and unpaved road density (β = 0.264, $P < 0.001$), human population density (β = –0.106, $P < 0.001$), and the interaction between road density and deforestation (β = –0.153, $P < 0.001$).

Considering the 446 AUTEF plans for which species composition was declared from volumetric inventories, the total expected monetary value attributed to each timber price category was 13.2% for the most valuable species (class A), 41.3% for class B, 19.2% for class C and 26.3% for the least valuable species (class D). However, the stand-scale representation of these timber price brackets was highly variable across logging frontiers and concession sites. In general, expected logging revenues per hectare decreased within older logging frontiers in eastern Pará compared to more recent frontiers in western and northwestern Pará, likely reflecting the historical chronological expansion of industrial scale logging throughout the state. This is consistent with the increasingly dominant proportions of low-value timber (class D: <R$24 per m^3) inventoried at concession sites in the oldest logging frontier (Fig. 2.1). Species composition of timber species harvested by volume clearly affected gross expected timber revenues per hectare derived from each concession, with revenues significantly increasing with proportional offtakes of timber classes A and B, but significantly decreasing with proportional offtakes of timber class D (Fig. 2.2).

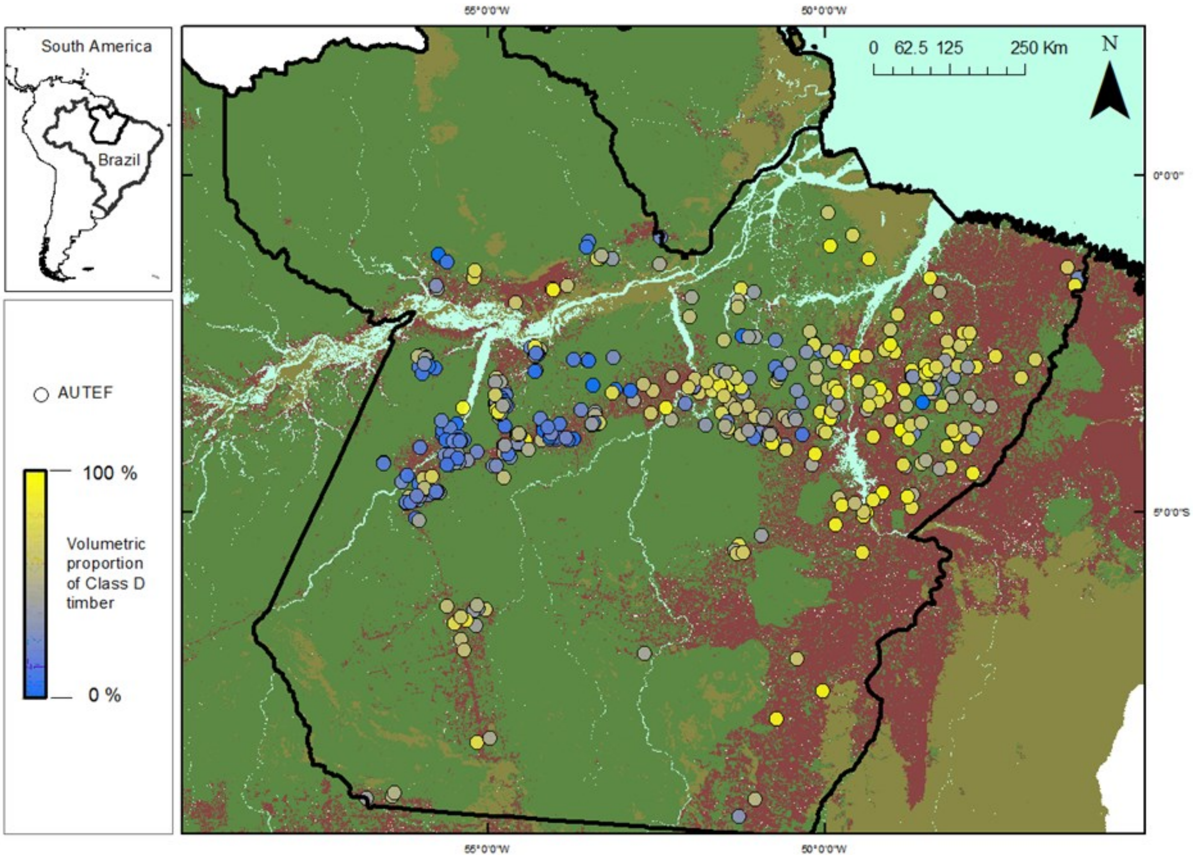

Figure 2.1 Map of the state of Pará, the second largest in Brazil, showing the geographic distribution of 446 AUTEF management plans for which the species composition of timber stocks were declared based on local forest inventories. Colour gradient shows the total fraction of low-value timber species (price class D) by volume (Σ m3) declared within each management plan. Standing volumetric stocks of low-value timber species, which comprise the cheapest lumber in local and regional markets, and represent the strongest negative predictor of concession scale logging revenues per unit area. Forest and deforested areas as of 2012 are indicated in green and dark red, respectively.

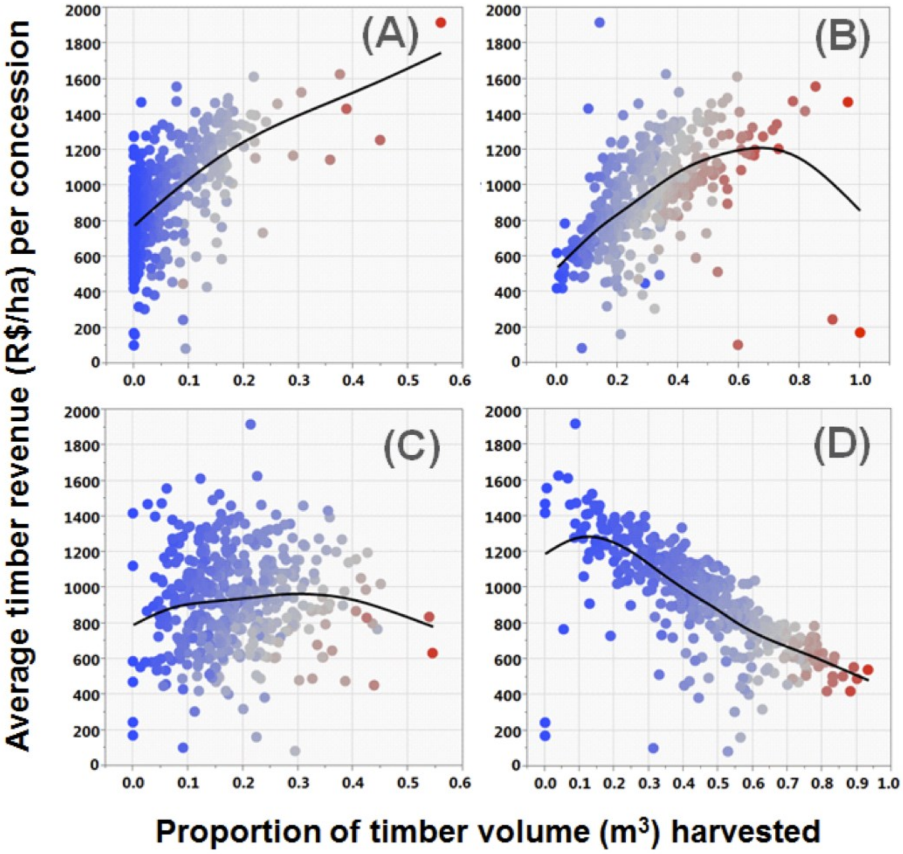

Figure 2.2 Relationship between volumetric proportions of timber species aggregated by price brackets (classes A, B, C, and D) and mean gross revenues per hectare expected from each authorized concession area. Timber value classes are ordered top to bottom from the highest (A) to the lowest (D). Symbols are colour-coded from low (blue) to high (red) values according to the respective proportion of each price class contributing to the overall timber revenue of each concession site. Solid lines represent a smoother (λ = 0.8) running through all data points.

Distance to heavy-traffic paved roads was highly variable across management plans, ranging from 2.49 to 931.7 km (mean ± SD = 464.3 ± 273.2 km). This was the most important predictor consistently appearing in the best candidate and averaged models explaining concession-scale (1) total timber revenue per hectare; (2) volumetric selectivity of tree species (J') by timber prices; and (3) the first NMDS axis describing multivariate patterns of timber genus composition (Fig. 2.3 and 2.4). The first NMDS axis was strongly positively correlated with the volumetric density of high-value species (Classes A and B; Pearson r = 0.613, $P < 0.001$, N=446) and strongly inversely correlated with the volumetric density of low-value species (Class D; r = −0.625, $P < 0.001$). Logging concession size, defined in terms of the net authorized harvesting area per landholding, was a significant predictor of both timber selectivity and genus-level composition (Table S2.2). Age of logging frontiers was also important in explaining our measure of high grading (expressed as J' evenness in timber values) and timber genus composition occurring across concession sites (Table S2.2). The proportion of forest cover throughout the landscape matrix surrounding each AUTEF landholding was also a positive predictor of timber revenues, suggesting that volumetric densities of high-value species were greater in less deforested areas. Moreover, among all possible combinations of the seven factors tested, distance to major roads and frontier age produced the highest Spearman rank correlations (BEST, $\rho > 0.468$; $P = 0.01$) with the multivariate structure of timber genera (weighed by volumetric abundance per hectare) inventoried at different concession areas.

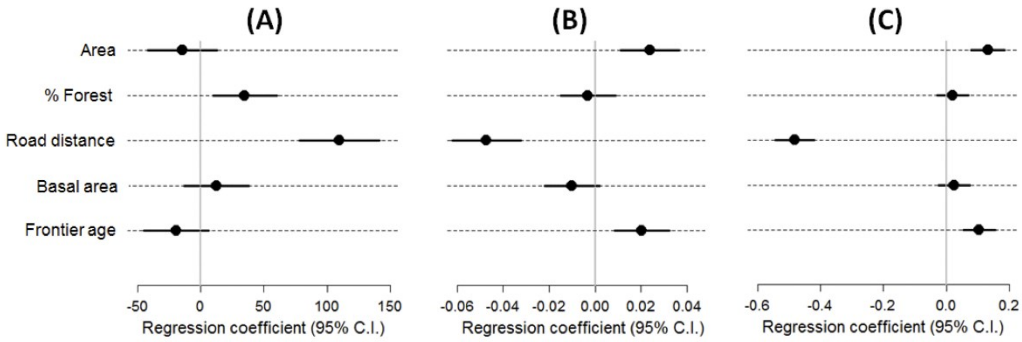

Figure 2.3 Coefficient estimates (± 95% confidence intervals) showing the magnitude and direction of effects of different forest site and landscape scale variables on timber revenue (R$/ha), species selectivity ranked by timber prices (J' evenness), and the volumetric composition of timber offtake (NMDS1). Concession-scale timber revenue, tree species selectivity, and volumetric composition of offtakes were modelled with generalized linear models using the following variables: AREA - net concession areas authorized for timber extraction; % FOREST - percentage of forest cover within a 10-km buffer outside concessions; ROAD DISTANCE - linear distance between each concession and the nearest heavy-traffic paved road; BASAL AREA - predicted pre-logging forest basal area based on an interpolation of 2,345 one-hectare plots from the RADAMBRASIL forest inventory program; and FRONTIER AGE - number of years since the onset of large-scale timber exploitation. Explanatory variables were standardized prior to analyses. For full summary of averaged models, see Table S2.2.

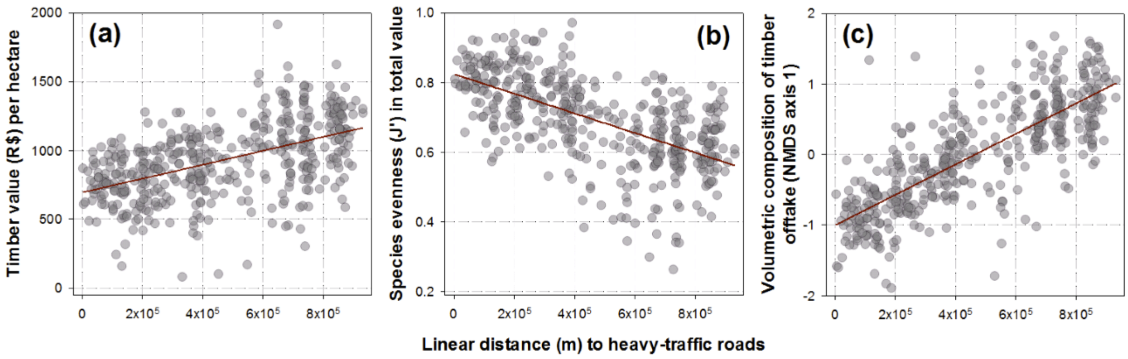

Figure 2.4 Relationships between linear distances to major access roads and timber revenue (R$/ha), species selectivity ranked by timber prices (J' evenness), and the volumetric composition of timber offtake (NMDS1). Higher revenues per ha (R$/ha) and higher levels of species selectivity (lower Pielou's J–evenness values) are observed in forest management plans farther away from heavy-use roads. Multivariate patterns of volumetric species composition (NMDS axis 1), which was inversely correlated with volumetric densities of low-value timber, was strongly positively related to distances from heavy-use roads.

These effects are unlikely to result from baseline differences in the overall abundance of valuable timber species across all concession sites prior to the emergence of mechanized operations in modern logging frontiers. For example, estimates of aggregate forest basal area within canopy tree plots sampled by the RADAMBRASIL forest inventory program yielded no significant effects on total timber revenue, species selectivity, and timber species composition across all of our models. In addition, on the basis of RADAMBRASIL tree plots coinciding with each of our logging frontiers (Fig. S2.2), we found no significant pre-depletion differences across all four variable-aged logging frontiers in the total basal area (P = 0.193) and total timber volume (P = 0.311) of high-value timber trees (classes A and B). In fact, historical patterns of timber tree abundance suggest that prior to logging in the 1960s-70s, the most depleted logging frontier (East Pará) had similar relationships between low- and high-value timber species, in terms of both total basal area and total volume, compared to less depleted frontiers (ANCOVAs, $P > 0.269$ in all cases, Fig. 2.5). In addition, there were no baseline differences across logging frontiers in the volumetric abundance of timber species according to timber price classes (Fig. S2.3). Finally, there were no differences across RADAMBRASIL plots in different logging frontiers in relation to the plot-scale variation in wood specific gravity (wood density) of their canopy trees (Fig. S2.2), suggesting that the functional composition of currently depleted frontiers included similar relative abundances of heavy-wooded trees compared to less depleted frontiers.

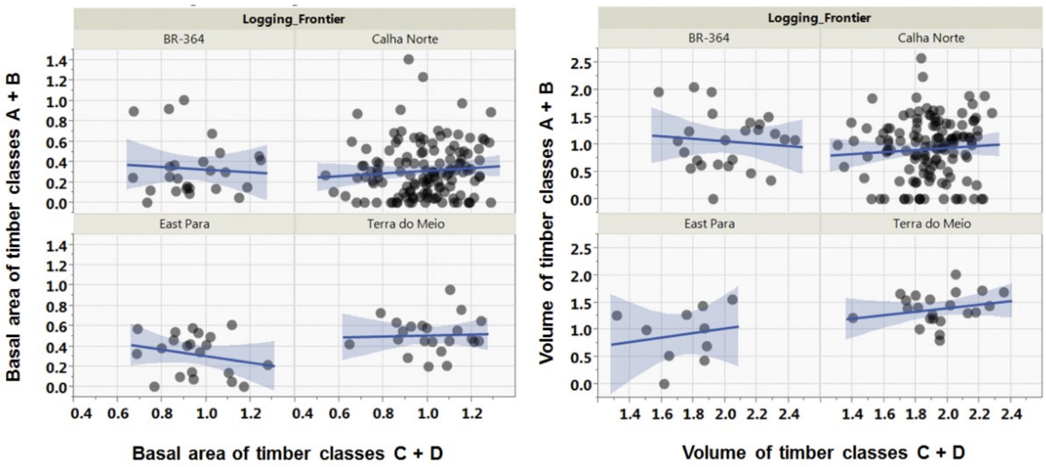

Figure 2.5. Relationships between the baseline abundance of timber species within contrasting market price classes at four major logging frontiers examined in this study, in terms of the total proportional basal area density (m^2/ha) and total volume density (m^3/ha) per 1-ha plot sampled in the early 1970s by the RADAMBRASIL forest inventory program. High-value and low-value timber species are aggregated within price classes A + B and C + D, respectively.

2.5 Discussion

We find no evidence to support the notion that old-growth tropical forest timber stocks across one of the oldest mechanized logging frontiers of lowland Amazonia have been sustainably exploited as a renewable resource capital. Both historical records and the contemporary status of a widespread set of existing forest stands suggest that the most prized and sought after timber species have been repeatedly mined to the point of subregional demographic collapse as a function of local supply/demand conditions mediated by physical access, land-tenure systems, and timber market prices.

Biological resource overexploitation often triggers a pattern of depletion of high-value emblematic species, which results in the sequential targeting of less valuable species locally or displacement of harvesting farther afield into previously undepleted areas where high-value stocks are still abundant (Peres 2009). In Brazilian Amazonia, spatiotemporal clustering of logging activity closely tracks the gradual historical expansion of the paved and unpaved road infrastructure, which opens up access into previously unlogged pristine forest areas (Verissimo et al. 1995; Souza Jr. et al. 1997). Depletion of prime tropical hardwood species thus conforms to the general pattern of other extractive industries where multiple species are exploited simultaneously, thereby ensuring that the search effort targeting more abundant species continues to subsidize the exploitation of declining species well beyond their marginal economic value. In the world's oceans, the largest and most valuable fish are exhausted first before exploitation shifts down trophic levels, sequentially targeting the next biggest fish species available (Pauly et al. 2002). This is also consistent with historical trajectories of spatial depletion of vertebrate game species by hunters in Neotropical (Jerozolimski & Peres 2003) and Afrotropical forests (Fa et al. 2005). However, comparisons with exploited fauna must be made with alarming caution, given slow-growth rates and low recruitment of hardwood trees, it is likely that the recovery of over-harvested timber species is also slower than that of most animals.

In both tropical forest and marine ecosystems, populations of fast-growing species increase via different density compensation mechanisms once slow-growing, high-value target species are removed (Myers & Worm 2003; Macpherson et al. 2012). This predicts that lower value species with fast life histories should become increasingly more common in post-depletion markets. Our data clearly supports this sequential depletion pattern for Amazonian timber species. Currently, the most valuable timber species were only available in relatively remote and more recently exploited forest hinterlands, far removed from the oldest logging frontiers, which have been served by heavy-traffic paved roads since the early 1970s, such as the BR-010 and PA-150 Highways in eastern Pará. Timber harvesting cycles under declining supply/demand conditions suggest that concession stands in the oldest frontiers may have been systematically logged twice or three times over the last 45 years. Conversely, relatively undepleted stands in the most recent frontiers, such as Calha Norte and BR-163 Highway, may still retain their full complement of slow-growing, high-value timber trees that are becoming increasingly restricted to unlogged old-growth forests. Our models further indicate that high economic returns per unit area (R$/ha) can only be realized where extensive areas of primary forest are still available in the surrounding landscape, partly because the frontier expansion history of logging and deforestation are inextricably linked (Kirby et al. 2006; Southworth et al. 2011). Indeed, high levels of timber selectivity, whereby loggers could afford to high-grade timber and prioritize allocation of volumetric offtake to high-value species, were apparently restricted to these areas.

The interplay between large areas of remaining primary forest cover and adequate road access, which results from local deforestation rates and geopolitical allocation of new road-building, largely determines where economically viable timber stocks can be legally extracted. Throughout the state of Pará, AUTEF concessions were largely located in sparsely settled areas where expanding road networks were sufficiently dense, but the extent of cumulative deforestation was still relatively low. Across the entire spectrum of forest loss in eastern Amazonia, this corresponds to areas experiencing intermediate levels of deforestation. For example, nearly one fifth of the state of Pará has been recorded as containing “no timber economic value” (Souza Jr. et al. 1997), but most of this area had either been deforested or completely exhausted of its prime timber resources long before the early 1970s. Given the inter-dependent dynamics of logging and deforestation frontiers in Amazonia, this is a transient and unstable condition because virtually all high road density areas tend to be eventually deforested outside protected areas (e.g. Laurance et al. 2002), including Indian Lands, where legally sanctioned logging concessions cannot occur. As logging frontiers grow older in the aftermath of settler occupation, selective timber extraction is gradually forced into ever more remote new logging areas that meet both logistical conditions and financial viability.

Some 90% of all timber species and 67% of the total timber volume (6, 439,474 m^3) harvested by loggers in eastern Amazonia were of low value, primarily including price classes C and D combined (cf. Verissimo et al. 1998), but this resulted in only 45.5% of all timber revenues. As expected, the volumetric proportion of the cheapest timber species (class D) in authorized offtakes was inversely related to that of high-value timber species (classes A and B; Fig. 2). Baseline estimates of forest basal area of canopy trees (>31.8 cm DBH) based on RADAMBRASIL forest inventories from the late 1960s to early 1970s consistently failed to show significant effects in any of our models, suggesting that patterns uncovered here are indeed a function of more recent depletion of high-value timber, rather than geographic differences in the preharvest price distribution of local timber stocks. This was further supported by additional RADAMBRASIL data as none of our four major regional logging frontiers showed any pre-logging differences in aggregate abundance of high-value timber tree species. In fact, forest plots in the early 1970s in the oldest and most depleted logging frontier (East Pará) on average contained the highest proportion of high-value timber species both in terms of basal area (Σ m^2/ha) and timber volume (Σ m^3/ha).

The effects of overexploitation are often exacerbated if overexploited species with slow life histories are also penalized by low abundances, poor long-distance seed dispersal, small geographic ranges, and/or infrequent pulses of seedling recruitment (Leão et al. 2014). Other traits that may render timber species vulnerable to population (harvest-sensitive species) declines induced by overexploitation includes low resprouting capacity after cutting or crushing, and high susceptibility to surface fires due to thin bark (Martini et al. 1994; Barlow et al. 2003). Largely unknown nuances restricting optimal reproductive conditions, including the disturbance-dependent episodic recruitment exhibited by mahogany (Rodan et al. 1992) and overharvesting of propagules (Peres et al. 2003), can render such detrimental consequences even more difficult to predict. Adequate timber regeneration throughout the life cycle of a full complement of tree species also depends on healthy levels of seed dispersal for even the most dispersal-limited large-seeded species, which depends on intact assemblages of large-bodied frugivores (e.g. Levi & Peres 2013). Moreover, retaining habitat quality in fully functioning forest ecosystems is crucial beyond post-logging structural damage and compositional changes, and impacts of logging disturbance on forest fauna tend to be more severe in the Neotropics than in the Indomalayan or African tropics (Burivalova et al. 2014). A long-term study at the Tapajós National Forest (Pará) that monitored residual stands prior to and after logging over 30 years indicate that changes in species composition among commercially-valuable canopy trees included average losses of 18 species per treatment area, but no overall decreases in tree species diversity (de Avila et al. 2015).

Our estimates of high-value timber depletion at subregional scales are likely conservative because they take no account of exceptionally valuable timber species, which may have been historically depleted in eastern Amazonia prior to the early-1970s national forest inventories. This includes exceptionally high-value tree species that illustrate chronic population declines, such as broadleaf mahogany (*Swietenia macrophylla*), rosewood *(Aniba rosaeodora*), and moderate-risk Brazil-nut trees (*Bertholletia excelsa*). None of these species were reported in AUTEF concessions because they cannot be legally harvested under Brazilian law, and are officially red-listed in the endangered flora of Brazil (IBAMA 2008, Normative ruling no. 06 of 2008). However, minor offtakes of rosewood were declared in some AUTEF plans under the vernacular name *pau-rosa* but listed as *Aniba parviflora*, and one concession authorized the offtake of *pau amarelo* (*Euxylophora spp).* To date, CITES only recognizes the need to control trade of mahogany and rosewood, both of which are listed under Appendix II. At the time AUTEF plans were authorized, Brazilian national forest policy prohibited offtakes of rare species, locally defined as < 0.03 ind. ha^{-1} [i.e. fewer than 3 large trees ≥ 50 cm DBH per 100 ha: Normative Ruling no. 05 of 2006]. Regrettably, this legislation treats tree populations of all species above this density threshold equally, threatening sustainable timber yields (STY) (Schulze et al. 2008)*.* Recent improvements through Normative Ruling no. 01 of 2015 now requires all concession areas to maintain at least 15% of the large trees or four large trees per 100 ha (per harvested species) within their Annual Production Units, although this is often difficult to ascertain.

Compared to conventional logging, reduced impact logging (RIL) treads lighter on tropical forest structure and composition, and has the potential to be more financially profitable in the long run (Barreto et al. 1998; Holmes et al. 2002). In practice, however, RIL is far from widely implemented (Pearce et al. 2003; Smith et al. 2006) and there is wide consensus that RIL alone does not ensure STY (Wadsworth & Zweede 2006; Sist & Ferreira 2007; Zarin et al. 2007; Putz et al. 2008; Peña-Claros et al. 2008; Schulze et al. 2008; Macpherson et al. 2012; Brandt et al. 2016). In East Africa, a post-logging regeneration time of 50 years is insufficient for forest structure to recover to baseline levels (Plumptre 1996), and a minimum cutting cycle of 60 years has been proposed for RIL implementation in the neotropics (Kammesheidt et al. 2001). This suggests that a second cutting cycle should not occur before ~2030 to achieve STY in many eastern Amazonian subregions, including our oldest logging frontier (East Pará), which was most likely first intensively logged in the early 1970s following the construction of the Belém-Brasília Highway. Our findings further support widespread evidence that a universal cutting cycle is both financially and ecologically unworkable and should be lengthened (Zarin et al. 2007). Moreover, sustainable forest management practices that can maintain STYs may actually violate prospects of financial profitability, whereas current practices ensure the commercial and demographic depletion of high-value timber species within three harvests in all three major tropical forest regions (Zimmerman & Kormos 2012).

AUTEF management plans merely represent a paper commitment of earmarked offtakes that must be fulfilled by law, rather than an accurate record of what timber was actually removed on the ground. The ultimate goal of a landholder is to maximize timber profits within the legal limit of 30 m^3/ha (Normative Ruling no. 05 of 2006), so that low-value timber species (classes C and D) should be logged only once classes A and B have been exhausted. As such, the management plan is still the most reliable indication of how best to capitalize on timber revenues within a given area. However, we recognize that any observed pattern of exploitation through AUTEF plans is an atypical representation of most logging practices across the Amazon. Species-specific volumetric targets could potentially be met via illegal logging of neighboring forestlands. The disrespect of concession boundaries remains pervasive in Latin America (Finer et al. 2014), and there are ample opportunities to surreptitiously boost profits with illegally extracted timber from areas outside a nominal licensed concession. The degree to which individual loggers comply with AUTEF guidelines is poorly known, but remote sensing assessments within and outside authorized concession boundaries have identified encouraging improvements. Legally sanctioned logging accounted for a mere 11% of all roundlogs extracted across Pará between 2007-2008, but this gradually improved thereafter to 14% (2008-2009), 35% (2009-2010) and 40% (2010-2011) (Monteiro et al. 2008; Monteiro & Souza 2011; Monteiro et al. 2011). This represents a significant contribution to urgently needed global efforts to reduce the scale of illegal logging (Nellemann

2012). Unfortunately the proportion of legally sanctioned logging dropped to 22% (73,535 ha) between 2011-2012, following the steady improvement of previous years (Monteiro et al. 2012). The NGO Greenpeace-Brasil, in collaboration with SEMA and Brazil's Public Prosecutor's Office (MPF: www.mpf.mp.br), carried out a systematic review of all 1,325 extant AUTEF plans in Pará between 2006 and 2013 to assess the extent to which timber laundering occurred (Greenpeace 2014). In total, 746 (56.3%) AUTEF plans listed ipê (*Tabebuia serratifolia*) in their inventories and approximately 14% overestimated volumetric offtakes (3,000 m^3 per concession or 60% above the species average of 2.4 m^3/ha). Subsequent in situ inspections revealed a plethora of fraudulent activities including illegal timber laundering through illegitimate plans where authorized areas showed no signs of logging. Electronic credit documents (which are issued with AUTEFs and deducted from loggers to credit timber buyers across the chain of custody) were crediting timber well in excess of what had been authorized by management plans before roundlogs were transferred to sawmills exporting timber worldwide.

Between 2000 and 2005 at least 20% of all tropical forests worldwide were selectively logged (Asner et al. 2009). In 2012, global scale tropical forest production of roundlogs, sawnwood and plywood combined reached 239.3 million m^3 (ITTO 2012) and timber extraction from natural forests is likely to expand (Verissimo et al. 1998; Laporte et al. 2007). It is estimates that 4.5 ± 1.35 billion m^3 of commercial timber is available in Brazilian Amazonia, 1.2 billion m^3 of which is currently profitable to harvest, resulting in an estimated total stumpage value of US$15.4 billion (Merry et al. 2009). As we exhaust Asian and African supplies of tropical hardwoods, market demands on Amazonian timber stocks will only increase. High-income and developed countries are often net wood importers and technocratic solutions aimed at producer-country inefficiencies will be insufficient to meet conservation goals (Mills Busa 2013). In meeting these goals a better understanding of the synergies between global timber demands, trade and in-country conservation capacity is crucial (Kastner et al. 2011).

The decisive goal for both biological resource managers and conservation biologists should be to conserve wild species (Mace & Hudson 1999). Effects of logging-induced forest degradation will be further exacerbated by on-going large-scale national development programs designed to meet growing infrastructure and energy demands from an ever-larger human population. A key question is then how can we bring about regional development without compromising conservation goals. This is especially true in light of the national bidding process (Law 11.284/2006) that will legally open access to timber extraction in an additional 4 million hectares of unlogged Amazonian forest from 2014 (SFB 2013b). To become competitive against illegal logging, low taxes are being lobbied to the Brazilian congress. However, this may flood timber markets, thereby slashing timber prices and forcing law-abiding RIL enterprises out of the market. Many of the current policy failures in managing harvest-sensitive timber stocks in private, communal and public natural forests will thus need to be addressed before the future onslaught of even more widespread timber exploitation.

Unlogged old-growth tropical forests are irreplaceable for biodiversity conservation (Gibson et al. 2011). However, selective logging can be described as relatively benign compared to alternative forms of tropical forest land use as long as run-away forest degradation can be curbed (Edwards & Laurance 2013; Wilcove et al. 2013; Edwards et al. 2014). This study provides further evidence that tropical timber tree populations are unwisely exploited as a renewable natural resource in terms of both the forest composition and economic potential of logged forests. Our analysis has important conservation implications, and calls for a better management of the commercial timber offtake portfolio across the entire range of timber species to ensure that high-value species can maintain demographically viable populations. Business-as-usual timber extraction that maximizes short-term profits can lead to loss of natural forest capital for livelihoods along the entire timber supply chain, resulting in severe socioeconomic repercussions in the long run. Even if we overlook the immense but poorly quantified scale of illegal logging, future Amazonian timber supplies are severely threatened by the systemic historical failure in institutional mismanagement, from both federal and state levels, which continues to perform poorly in effective planning, enforcement, and monitoring of sustainable timber yields. Our analyses suggest a rapid rate in population declines in high-value, extinction-prone slow-growing timber species over vast areas, which is unlikely to be easily reversed. We therefore urge national policy makers to curb the largely unchecked tide of widespread depletion of the most harvest-sensitive timber species.

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2.7 Supporting Information

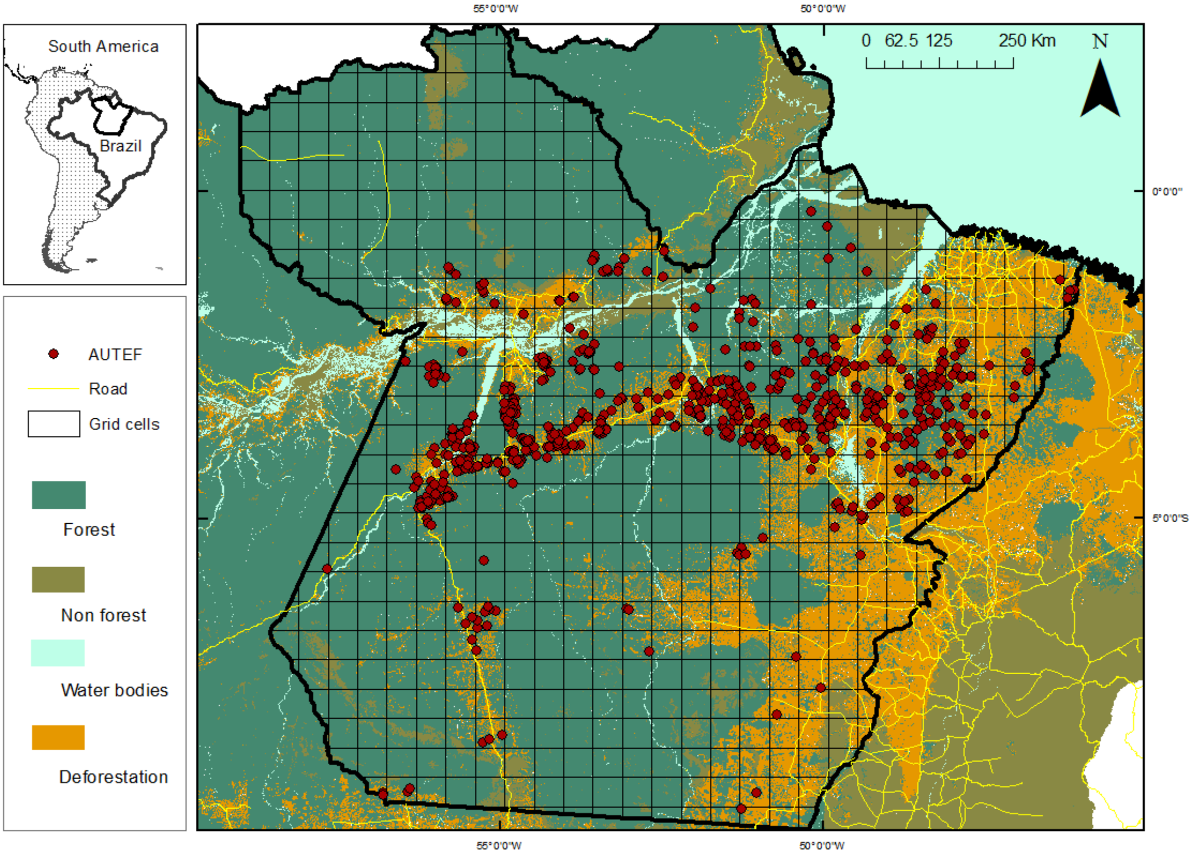

Figure S2.1. Major classes of land cover within the Brazilian state of Pará showing the spatial distribution of 824 private and community-based AUTEF forest management plans approved by SEMA between 2006 and 2012 across a grid of 564 cells of 50km x 50km. The main paved and unpaved roads are indicated in yellow; deforested areas as of 2012 are indicated in orange; non-forest areas refer to natural vegetation types, including Amazonian *cerrados*, outside the closed-canopy forest domain.

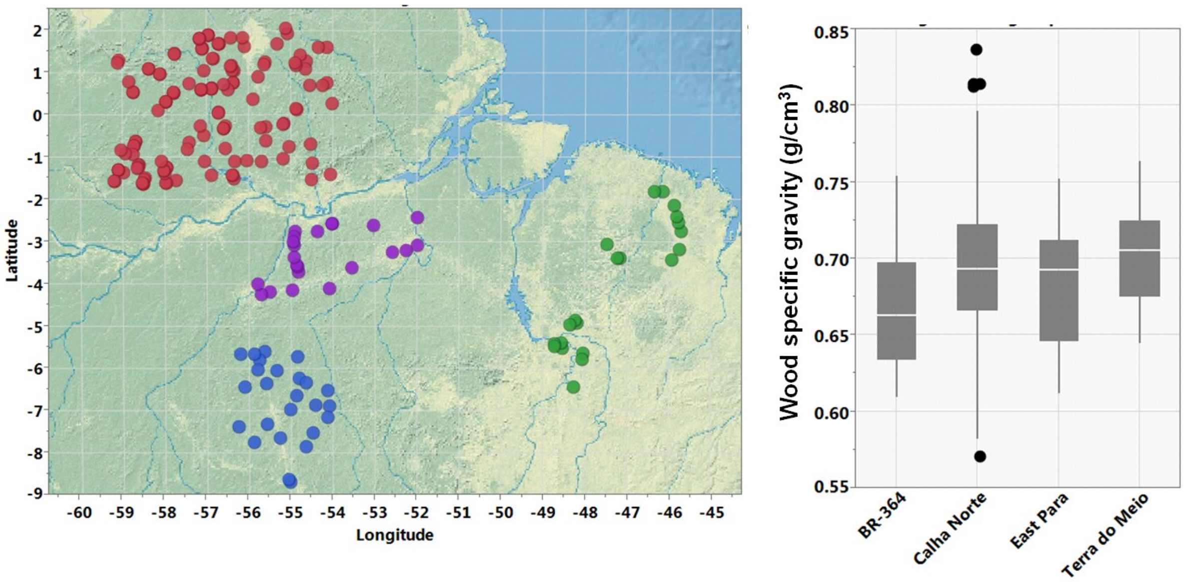

Figure S2.2. Spatial distribution of 1-ha RADAMBRASIL forest plots inventoried prior to large-scale mechanized logging in the early 1970s. These are colour-coded according to the four major eastern Amazonian logging frontiers examined in this study (East Pará: green; Terra do Meio: purple; Calha-Norte: red; and BR-163 Highway: blue). Right panel shows boxplots describing the mean wood specific gravity (WSG, often referred to as wood density) per canopy tree in those plots, further indicating that the pre-logging WSG profile of trees within different logging frontiers was similar, and that plots in the currently most depleted frontier (East Part) was comparable to less depleted frontiers in their functional profile of canopy tree species.

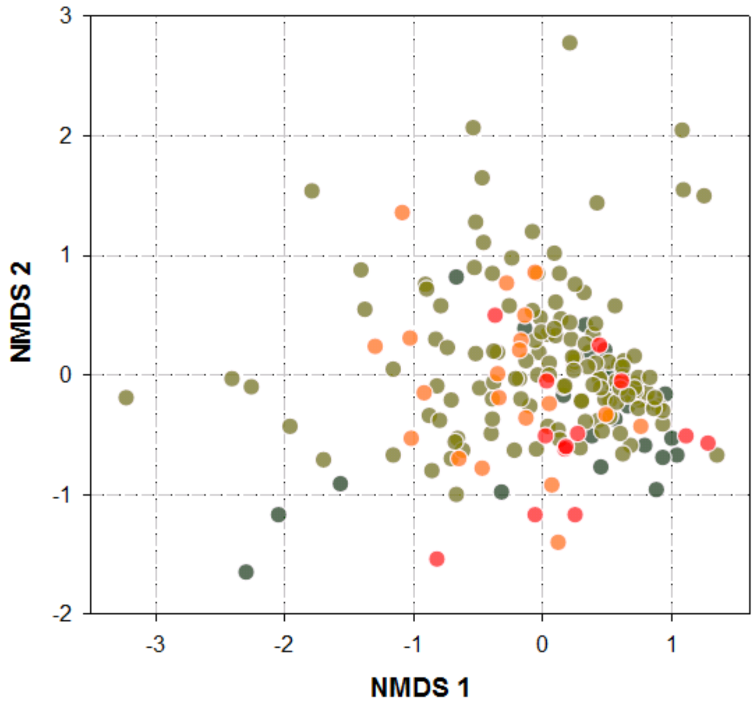

Figure S2.3. Nonmetric multidimensional ordination of RADAMBRASIL forest plots based on the volumetric contributions of timber species broken down into different timber price classes (A, B, C and D) across the four major logging frontiers examined in this study. Symbols are colour-coded according to logging frontiers: East Pará: red (2) Terra do Meio: orange; Calha Norte: light green; and (4) BR-163 Highway: dark green. There were no significant differences across forest plots grouped by frontiers in the multivariate structure of the abundance of timber species contained according to timber price categories (ANOSIM, 999 permutations, *P* =0.327).

Table S2.1. Response and explanatory variables considered in this study and their respective data sources*.

Variable abbreviation	Variable description	Unit	Source
Reais per ha	Total gross revenue expected for the management plan per hectare of net area of authorised land.	R$/ha	SEMA/PA
J-evenness	Pielou's J–evenness of Rank Abundance Distribution curves (by ranked species value) per AUTEF	Numerical	Derived from SEMA/PA using 'Vegan' Package in R
$NMDS_1$	Ordination scores along the first axis of Nonmetric Multidimensional Scaling (NMDS) ordination based on the total volumetric abundance of different timber species as declared in AUTEF plans	Numerical	Derived from SEMA/PA using Primer
Municipio	Municipality the landholding is found in within PA state	Categorical	SEMA/PA
Forest type	Forest type (plantation or natural)	Binary	SEMA/PA
Total reais	Total revenue expected for a given concession. The sum of all species specific values per cubic metre	Brazilian Reais	SEMA/PA and DOEPA 2010
Xlong/ylat	Location of concession centroid	Decimal degrees	SEMA/PA
p.landharvested	Proportion of concession area in relation to total landholding area	Percentage	SEMA/PA
lhsize_ha	Total landholding area	Hectares	SEMA/PA
Concession_area	The authorised area for logging	Hectares	SEMA/PA
Net_concession	Concession area excluding set asides for biodiversity protection under Brazilian law	Hectares	SEMA/PA
vol_perha	Volume of timber offtake per hectare of Concession_area	Cubic metres per hectare	SEMA/PA
Total_vol	Total volume of timber offtake per AUTEF	Cubic metres	SEMA/PA
no.spplogged	The total number of species logged per AUTEF	Numerical	SEMA/PA
no.genera_logged	The total number of genera logged per AUTEF	Numerical	SEMA/PA
pdeforest	The proportion of deforestation in the AUTEF circular polygon	Percentage	PRODES 2011
pforest	The proportion of forest cover in the AUTEF circular polygon	Percentage	PRODES 2011
pwater	The proportion of water bodies in the AUTEF circular polygon	Percentage	PRODES 2011
pnonforest	The proportion of non-forest in the AUTEF circular polygon	Percentage	PRODES 2011
buffer.pdeforest	The proportion of deforestation in the AUTEF's 10km buffer	Percentage	PRODES 2011
buffer.pforest	The proportion of forest cover in the AUTEF's 10km buffer	Percentage	PRODES 2011
buffer.pwater	The proportion of water features in the AUTEF's 10km buffer	Percentage	PRODES 2011
buffer.pnonforest	The proportion of non-forest in the AUTEF's 10km buffer	Percentage	PRODES 2011
HPD_km	Human population density as a weighted average of the census data within each AUTEF polygon.	Number of inhabitants per squared kilometre	IBGE 2011
buffer_HPD_km	Human population density as a weighted average of the census data within the AUTEF's 10km buffer	Number of inhabitants per squared kilometre	IBGE 2011
Centroid_ndist_AllRds	The AUTEF centroid nearest distance to all extant roads	Metres	IBGE 2011
Centroid_ndist_Heavyrds	The AUTEF centroid nearest distance to heavy-traffic roads. These are defined as those with traffic above and including 1000 heavy vehicles per day	Metres	IBGE 2011
Centroid_ndist_Rivers	The AUTEF centroid nearest distance to all water bodies	Metres	PNLT 2010
Buffer_ndist_AllRds	The AUTEF 10km buffers edge nearest distance to all extant roads	Metres	PNLT 2010
Buffer_ndist_HighFluxRds	The AUTEF 10km buffers edge nearest to heavy traffic roads. These are defined as those with traffic above and including 1000 heavy vehicles per day	Metres	PNLT 2010
Buffer_ndist_Rivers	The AUTEF 10km buffers edge distance to all water bodies	Metres	PNLT 2010
Frontier Age	Logging Frontier Age	Years	INCRA 2013 and Pereira et al. (2010)
BA	Basal area estimate	Squared metres	RADAM-BRASIL

*SEMA/PA: The State Environmental Secretariat of Pará

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Table S2.2. Final averaged models

Averaged coefficient estimates (β), unconditional standard errors (SE), P-value, and relative importance (Σ w_i) of averaged coefficients calculated over all models retained in the final candidate set for the patterns of timber species composition (NMDS1), species selectivity (J') and total estimated timber revenue (R$/ha) from trees available in AUTEF stands across 446 logging concession plans in Pará, Brazil.

Predictors	β	SE	P-value	No. models[a]	Σ w_i
Timber revenue (R$/ha)[b]					
Distance to heavy-use roads (km)	4.663e-07	6.129e-08	< 2e-16 ***	7	1.00
Matrix forest cover (%)	1.262e-03	4.852e-04	9.48e-3 **	7	1.00
Frontier age (years)	-2.610e-03	1.349e-03	0.05360	6	0.87
Basal area (m^2)	6.489e-03	5.474e-03	0.23710	2	0.27
Distance to nearest river (m)	4.093e-07	4.688e-07	0.38393	2	0.22
Log10 Concession area (ha)	-2.076e-02	2.590e-02	0.42416	2	0.12
Human population density ($/km^2$)	1.112e-03	2.791e-03	0.69109	1	0.10
Species selectivity (J') [c]					
Log10 Concession area (ha)	3.429e-02	9.858e-03	5.22e-4 ***	3	1.00
Distance to heavy-use roads (km)	-1.930e-07	2.765e-08	< 2e-16 ***	4	1.00
Frontier age (years)	1.257e-03	5.307e-04	01.81e-2 *	4	1.00
Basal area (m^2)	-3.654e-03	2.023e-03	0.071599	3	0.81
Human population density ($/km^2$)	8.507e-04	1.083e-03	0.433422	1	0.21
Matrix forest cover (%)	-1.176e-04	2.006e-04	0.558849	1	0.18
Timber species composition (NMDS1) [d]					
Log10 Concession area (ha)	-2.261e-01	4.052e-02	< 2e-16 ***	6	1.00
Distance to heavy-use roads (km)	1.716e-06	1.065e-07	< 2e-16 ***	6	1.00
Frontier age (years)	-8.129e-03	2.272e-03	3.58e-4 ***	6	1.00
Distance to nearest river (m)	1.688e-06	7.445e-07	0.023733 *	6	1.00
Human population density ($/km^2$)	-6.313e-03	4.421e-03	0.154420	3	0.50
Matrix forest cover (%)	-7.491e-04	7.641e-04	0.328227	2	0.27
Basal area (m^2)	-7.985e-03	8.089e-03	0.324903	2	0.27

*Significance codes: 0 '***' 0.001 '**' 0.01 '*' 0.05*

[a] Number of models containing each predictor variable over all models retained in the final candidate set.

[b] r-squared estimate from the full model (which equals that of the top model): 0.27

[c] r-squared estimate from the full model (which equals that of the top model): 0.33

[d] r-squared estimate from the full model (which equals that of the top model): 0.66

Chapter 3: Timber extraction in Amazonian forests can only be sustainable with economies of scale

3.1 Abstract

The relative contribution of smallholdings to overall deforestation in Amazonia remains contentious, deforestation attributed to smallholders across studies ranges from 18%, to 96%. However, the degree to which smallholders are able to manage tropical forest timber stocks sustainably remains virtually unknown. We examined the role of smallholders in the context of the Amazonian logging industry and discuss how does their timber offtake decisions or productivity compare against that of large landowners. We report data on legally planned logging of ~9.6 million cubic meters of timber across 314 tree species extracted from private and community-based concessions between 2009 and 2012. Using data from 824 government-approved concession management plans, we document patterns of timber offtake by volume, species composition, and monetary value along a gradient of property sizes and logging frontiers of eastern Amazonia. Our data suggest that smallholders appear to exert stronger high-grading pressure upon the high-value hardwood species available in their landholdings, thereby accruing higher gross revenue productivity per unit area, at least in the short term. Only large properties (~3,000 ha) were most likely to be issued concession areas of 30% or less of their land. Moreover smallholders were more likely to overestimate the minimum areas of forest set-asides as required by law, with all mini smallholders (≤100 ha) and 99% of smallholders (101 – 400 ha) inconsistently reporting their own forest set-asides. These findings are further discussed in the context of fluctuating government policies that can exert complex and perverse incentives for actor-specific forest-set aside compliance and deforestation. Our results illustrate that greater positive incentives for actor-specific responsibilities would be effective in supporting

more sustainable forest management and ensuring long-term viability of reduced impact logging among smallholders of Amazonia.

3.2 Introduction

Most rural properties worldwide are small, with FAO (The Food and Agriculture Organization of the United Nations) data showing that 85% of all farms are smaller than 2 ha (Nagayets 2005). Global scale inequality was one of the root causes of recent financial crises and it remains higher than inequality within any given country (Lysandrou 2011). Land distribution inequality within Brazil remains high; smallholders comprise 84% of all landholdings but only 24% of the overall land area (IBGE 2006). In developing countries, large corporations in the private sector often dominate local political life, often becoming providers of otherwise unavailable public services, including building local roads, charging local taxes, and co-funding public schools with local governments (Raup 1969). Smallholders generally have limited political voice, education, and access to information, rural credit and secure land tenure (Poulton et al. 2005; 2010). Given the stark inequalities in land distribution and market opportunities, there is widespread societal support towards smallholders (Schumacher 1974; Monbiot 2008).

Smallholders often have advantages in local knowledge and low labour costs, especially if self-motivated family members are engaged (Wiggins 2009). On the other hand, large property owners can have higher management costs but benefit from lower unit transaction costs (Raup 1969). Other factors that favor large landholdings include greater financial flexibility, better access to niche markets, rural credit, and other financial services, marketing outreach, information on technology and markets, product traceability, and quality assurance (Poulton et al. 2005; 2010). Largeholdings also often take advantage of perverse government subsidies, often undercutting smallholders by selling commodities below productions costs (Godfrey 2002).

Growing concerns over future global food security have fueled debates over the role of smallholders and their productivity efficiency compared to largeholders and agro-corporations in meeting food demands. Nobel economist Amartya Sen first described evidence that small farmers in India were more productive per unit area than largeholders (Sen 1962). Similar evidence has since been reported in various agricultural systems worldwide, including Turkey, Nigeria, Burkina Faso, Ghana, Niger, Mali, Pakistan (Heltberg 1998; Ünal Gül 2008; Okoye et al. 2009; Wiggins 2009) and Brazil, where differences in agricultural outputs could not be attributed to soil fertility (Cornia 1985). In agriculture, measureable economies of scale are often reported, whereas the reverse is more difficult to interpret (Raup 1969).

It has been suggested that increasing farm sizes and growing dominance by few large companies in the USA, for example, may result from external factors including information systems, financial constraints and research agendas, rather than evidence of economies of scale (Hallam 1991). For instance, wage-rental ratio, the ratio of labour to the rental of land or capital, is influenced by opportunity costs outside agriculture, which increase the price of labour and induce machinery innovations. Farm sizes in the USA decreased during the energy crises of the mid 1970s when the wage-rental ratio also decreased (Kislev & Peterson 1991). Smallholders can thrive when labour is abundant and land expensive, but once a diversified economy raises labour costs, farm sizes tend to increase. This size transition co-occurs with development as GDP correlates closely with land costs and labour (Hazell 2005).

There is also debate as to whether smallholders or large landholdings outperform one another across multiple land uses in reducing environmental impacts. With privatisation, resources are often parcelled into smaller competing units that are often incompatible with much larger ecological boundaries, generating externalities that can aggravate environmental damage (Hanna 1996). Some attribute most environmental degradation in developing countries to the cumulative effects of smallholders, which often lack the appropriate capacity for impact assessments and regulation (Repetto 1987).

The relative contribution of smallholdings to overall deforestation in Amazonia remains contentious. Deforestation attributed to smallholders range from lows of 18%, 30% and 47% (Alencar et al. 2004; Fearnside 1993; Pacheco 2005, respectively) to highs of 64% and 96% (Walker et al. 2000; Aldrich et al. 2006). Godar et al. (2012) highlighted four main shortcomings in these studies. Firstly, coarse large-scale data are often used to achieve fine scale results because georeferenced data at the property scale are rare and expensive to obtain, and studies represent snapshots in time of complex systems that can change over time. Moreover, basin-wide assumptions of different actor-contributions to deforestation are difficult to interpret because of a complex combination of different regional colonisation histories, livelihood and production strategies, and land tenure systems. Lastly, deforestation fails to represent the full scope of environmental impacts in tropical forests, which also include forest degradation through wildfires, fragmentation, edge effects, and changes in species composition (Laurance & Peres 2006). A recent analysis using 13,303 sub-municipal census districts across the Brazilian Amazon showed that aggregated deforestation in 2004-2011 attributed to properties ≥ 2,500 ha declined by 63% from a peak in 2005, whereas that of smallholders (≤ 100 ha) increased by 69% (Godar et al. 2014).

Between 2000 and 2005 at least 20% of all tropical forests worldwide were selectively logged (Asner et al. 2009). In 2012, global scale roundlog, sawn wood and plywood production from tropical forests reached 239.3 million m^3 and Brazil accounted for 85% of the roundlog production in Latin America/Caribbean region with a total harvested volume estimated at 30.8 million m^3, which takes no account of the poorly quantified illegal trade (ITTO 2012). Globally, market demands for Amazonian timber stocks will only increase as Asian and African tropical hardwood supplies are exhausted (Verissimo et al. 1998; Laporte et al. 2007). The 125 Mha State of Pará (the second largest in Brazil) has experienced the oldest history of logging across Amazonia spanning three centuries, but still retains vast untapped timber stocks in remote unlogged forests. Native timber has become the mainstay of the Pará economy since the first road linking the state to southern Brazil was built in 1970, with sawmills producing a 8 million m^3 yr^{-1} of sawn timber in the early 1990s (Uhl et al. 1997). In 2009, Pará accounted for 47% of the roundlog production, 44% of the gross timber revenue (~US$1.1 billion), and 45% of all direct and indirect jobs in the wood-related sector across the entire Brazilian Amazon (Pereira et al. 2010).

Almost 60% of all smallholders (≤ 200 ha) in western Amazonia do not have land tenure, have no know-how in forest management for sustainable timber yields, and have little access to appropriate markets, so that forest conversion to agriculture remains their best financial option (Vosti et al. 2003). Yet there is growing governmental support for smallholder timber extraction at both community and family levels from governmental funding and technical assistance programs (such as the PMCF, *Programa Federal de Manejo Florestal Comunitário e Familiar*, Decree Nº 6.874 of June 5th 2009). However, the degree to which smallholders are able to manage tropical forest timber stocks sustainably remains virtually unknown. Indeed, what is the role of smallholders in the context of the Amazonian logging industry? How does their timber offtake decisions or productivity compare against that of large landowners?

Here, we present the first assessment to our knowledge of the economics and general indicators of sustainability of legally sanctioned private and community-based logging concessions representing 824 small, medium, and large landholdings. These landholdings were officially authorised to extract nearly 10 million cubic meters of timber between 2009 and 2012. In particular, we explain broad patterns of available timber stocks and timber species composition within pre-logging forest stands. We then aim to understand the relationship between concession size and; (1) estimate gross revenues flowing from those stocks; (2) estimate the degree of high grading or species selectivity observed in offtakes; (3) examine to what degree landowners have complied with legal frameworks that require permanent riparian forest set-asides and at least 80% of landholdings to be maintained as native forest reserves. We also (4) discuss the implications of landholding size in the context of rural development policies governing logging concessions and the global timber industry.

3.3 Methods

3.3.1 Study areas and AUTEF management plans

Mandatory legal approval of forest management plans within the eastern Amazonian state of Pará must be issued to all timber extraction enterprises, including those in communal lands, small to medium private properties, and largeholdings controlled by logging companies, in the form of a 'Forest Exploitation Permit' (Autorização de Exploração Florestal; hereafter, AUTEF). The State Environmental Secretariat of Pará (SEMA) issues AUTEF plans, a legal requirement under both SIMLAM (Brazilian Integrated Environmental Licensing and Monitoring System) and SISFLORA (Forest Product Trade and Transport System) for planned timber harvests of any forest site at any spatial scale. We extracted and digitized data from a total of 824 AUTEF plans across Pará sanctioned between 2006 and 2012. These included the name of the rural entrepreneur, community, landholder, or company carrying out each logging operation, the municipal county, geographic coordinates of each concession, forest type (planted or natural), the total standing volume authorized for extraction, the total landholding size, the net size of areas authorized for logging (excluding legally protected riparian forest set-asides where logging is not permitted).

At the time the AUTEF plans were issued the Brazilian forest code legislation (nº 4.771 of 15th September 1965) made it a legal requirement of landowners to also retain 'permanent protection' areas along riparian forests and high-slope upland forests as *Areas de Protecão Permanente* (APPs). From the total of 824 AUTEF plans, a subset of 678 AUTEF plans provided APP areas and additional set-aside areas within the landholding defined as 'Legal Reserves' according to the Brazilian Forest Bill (No. 12.727, http://www.planalto.gov.br/ccivil_03/_Ato2011-2014/2012/Lei/L12727.htm). A further more detailed subset of 446 AUTEF plans (issued between 2009-2012) also included the total volume (m^3) of inventoried timber per tree species per concession to be extracted, we therefore use either one of these data sets depending on the nature of the analysis.

Because timber species were identified *in situ* within concession areas by experienced tree parataxonomists hired to support management plans, we converted vernacular names into their corresponding Latin nomenclature and then removed species-level synonymia whenever necessary based on a comprehensive checklist of timber species of Central and Eastern Amazonia compiled from multiple sources (Silva et al. 1977; Parrotta et al. 1995; Ribeiro 1999; Lorenzi & Flora 1998; Lorenzi 2002; Lorenzi 2008).

AUTEF plans granted for exotic tree monocultures, which included eucalyptus (*Eucalyptus*), teak (*Tectona*), and pine (*Pinus*) plantations, were excluded from the analyses. Although Paricá (S*hizolobium amazonicum*) plantations were reported in landholdings (20%), this species is native to Amazonia and was therefore retained in those AUTEF plans defined as 'natural' forests. All AUTEF applications to SEMA referred to a unique forest stand of known size based on GPS fixes of property boundaries, although a few exceptionally large landholdings controlled by a logging company may have included more than one AUTEF for different logging compartments exploited in different years (APUs, Annual Production Units).

The Brazilian government defines smallholders as 1-4 fiscal modules, the size of which varies by region. In heavily developed states such as São Paulo, a fiscal module is often 5 ha, but as large as 100 ha in more remote Amazonian regions. Property size classes were divided as followed; 'mini smallholders' (≤ 100 ha), smallholders (>100 but ≤400 ha, i.e. the upper size limit of 4 fiscal modules in Amazonia); 'medium' landholders (>400 and ≤1500 ha, i.e. the upper size limit size is equivalent to 15 fiscal modules and used programs such as *Terra Legal*); large landholdings (i.e. all those properties >1500) and very large landholdings (>5000 ha).

3.3.2 Timber price data

Given that many species accrue significant value along different supply chains and export market prices are affected by complex international demands, we used regional scale logwood prices per timber species in Brazilian Reais (R$ per m^3 of lumber) available from an official source for the state of Pará that serves as a benchmark for timber merchants (DOEPA 2010). This reflects the dominant domestic market, which consumed 95% of all timber produced in Brazil in 2011 (ITTO 2012), and best reflects realistic transaction prices of unprocessed timber expected by loggers at sawmills or other points of timber sales. Timber prices (R$/$m^3$) are grouped by DOEPA (2010) into four categories, with gradually fewer timber species commercialized under increasingly higher price brackets: Class A (11 species, 6 genera): > R$75.0/$m^3$; Class B (18 species, 12 genera): R$45.0/$m^3$ - R$74.0/$m^3$; Class C (40 species, 31 genera): R$25.0/$m^3$ - R$44.0/$m^3$; and Class D (all other 245 species within 157 genera): R$1.0/$m^3$ - R$24.0/$m^3$. The logwood price data we used are deliberately conservative compared to other sources, which may take into account valuation along supply chains (Stone 1998; Bacha & Rodriguez 2007).

Data on local timber extraction costs were unavailable so our analyses are based on estimates of gross expected revenues. However, extraction costs should scale to the total volumes of timber removed and extent of logging areas exploited, which are taken into account here. Alternative sources of income that may be available to different landholders may also affect local economies of scale and timber species selectivity but are beyond the scope of this study. These may include sales of non-timber forest products and residual dead wood derived from collateral damage at logging clearings (e.g. to meet the high charcoal demand for smelting iron ore in eastern Amazonia), and value-added through timber processing capacities.

3.3.3 Geographic data

Because exact landholding boundaries of logging concessions were unavailable from AUTEF management plans as spatially explicit polygons, circular buffers of sizes corresponding to each known landholding area (range = 26 – 844,021 ha), which had been reported in all 824 AUTEF plans, were projected around their geographic coordinates using ESRI ArcMap 10.2.2. Each of these buffers was then assigned an additional 10-km radius external buffer to represent the approximate landscape structure of the forest/nonforest matrix surrounding each AUTEF landholding.

The proportion of forest and deforested areas, natural savannahs (*cerrado*), and water bodies were calculated for both internal landholding projections and external buffers using 30-m resolution data from the Brazilian Space Agency PRODES project (Table S1 of Chapter 1). Deforestation areas under cloud pixels (for which the deforestation year was unknown) were excluded from any landholding projection, but these amounted to only <3% of all pixels.

Approximate dates of logging frontiers follows Pereira et al. (2010), but these were further refined by accounting for the official onset of any INCRA agrarian settlement within a 75-km buffer of each AUTEF geographic centroid (Table S1 of Chapter 1). These government-sanctioned agrarian settlements typically mark the arrival of first settlers into new forest frontiers as they rapidly take advantage of new roads into previously inaccessible areas (Peres & Schneider 2012). In addition, earlier cycles of logging in eastern Amazonia typically occurred within 25 km of major roads (Uhl et al. 1991), so dating of logging frontiers corresponding to each AUTEF site was further verified by accounting for the completion year of all major paved and unpaved roads (or road segments) built in previously remote forest regions based on a comprehensive compilation of historical records.

Baseline data on forest structure and composition prior to any large-scale timber extraction were unavailable for logging concession sites. However, we use data from the RADAMBRASIL forest inventories (Brasil 1978), which were conducted by the Brazilian government from the late 1960s to the early 1970s to map timber resources across Brazilian Amazonia, to estimate the plot-scale aggregate basal area (BA, m^2/ha) and wood specific gravity (wood density, g cm^{-3}) under pre-logging conditions for each AUTEF site. We considered all tree species within each timber price bracket. RADAMBRASIL is the most extensive network of forest plots ever undertaken across the entire Brazilian Amazon, and included at least 2,345 one-hectare plots surveyed across the region (/<ftp://geoftp.ibge.gov.br/>). This was done using an ordinary krigging interpolation of total forest BA from all 1-ha plots, within which a total of 128,433 canopy trees ≥ 31.8cm in diameter at breast height (DBH) [or ≥100cm in circumference at breast height (CBH)] were sampled. We also used RADAMBRASIL data to test for proportional differences in total basal area (m^2)

and total volumes (m^3) of high vs low value timber (classes A - B and C – D, respectively) across the four major logging frontiers of varying histories containing the logging concessions examined here. These frontiers are, from the oldest to the most recently exploited: (1) East Pará, primarily along the Belém-Brasília Highway (BR-010) and the main State Highway of Pará (PA-150); (2) Terra do Meio region, along the Transamazon Highway (BR-230); (3) the Calha Norte region of northwestern Pará; and (4) along the Cuiabá-Santarém Highway (BR-163) of southwestern Pará. For a comprehensive profile of different geographic and historical variables by landholding class sizes, see Table 3.1.

3.3.4 Data analysis

Patterns of timber tree species volumetric abundance and dominance within AUTEF management plans were examined using data from *in situ* forest inventories reported for each logging concession area authorized by SEMA. Timber tree species within each concession were rank-ordered in terms of their overall stock value (R$), defined as their total volumetric stock (m^3) multiplied by the species-specific reported timber price per m^3 to examine the offtake distribution of species-specific timber values (Σ R$ • m^3). Using the *vegan* package in R, we then constructed Rank Abundance Distribution (RAD) curves (McGill et al. 2007) in timber stock values (on a log-scale) to derive the evenness J' (Pielou 1975) in timber revenues across all co-occurring species exploited at a concession. This provides additional insights into the degree to which loggers could maximize harvesting selectivity by focusing on high-value timber to most efficiently meet their maximum legal quota of 30 m^3/ha, as required in approved AUTEF plans. Pielou's J' evenness could thus be defined as a measure of high-grading of an assemblage of coexisting tree species within a concession area. We selected this evenness measure because it is the most widely used in ecology, and is an excellent species-abundance predictor of species richness in tropical forests (He & Legendre 2014). J' values range from 0.0 to 1.0, with larger values representing more even species distributions in reported volumetric offtakes in relation to market values of timber tree species, or a wider offtake portfolio of timber species by value suggesting lower species selectivity. Conversely, steeper RAD slopes represented by lower J' values indicate high-grading, or timber revenues disproportionately concentrated on only a few highly profitable timber species.

To examine multivariate patterns of species composition we used Nonmetric Multidimensional Scaling (NMDS) ordination based on the total volumetric abundance of different timber species declared in each AUTEF management plan. Vernacular identification of Amazonian trees is often ambiguous at the species level for several tree morphospecies but sufficiently robust at the genus level (Higgins & Ruokolainen 2004). However, this is a lesser problem for the much smaller subset of commercially important timber species identified by experienced para-botanists in the field. NMDS ordinations thus used an abundance-based (Bray-Curtis) similarity matrix including 153 tree genera surveyed across the 446 concessions for which a timber tree inventory was available, once the raw data had been standardized and sqrt-transformed. Stress values at two-dimensional scaling was 0.23 or lower and ordination scores along the first axis ($NMDS_1$) are defined as an additional descriptor of the concession-scale composition of timber species in terms of their declared volumetric abundance. Linear regression models were constructed in R version 2.15.1 to examine the relationship between concession size and timber revenues, timber species selectivity (J' values), and the declared volumetric composition of timber offtakes ($NMDS_1$).

At the time the AUTEF plans were issued the Brazilian forest code legislation (nº 4.771 of 15th September 1965) made it a legal requirement of landowners to retain APPs, in addition, private properties must retain 80% of their landholding as 'Legal Forest Reserves', (*Reserva Legal*, RL). The total set-aside area is therefore the sum of both APPs and RLs. The proportional area within a landholding polygon that had been authorized for timber extraction is hereafter referred to as 'concession area'. The proportion of land left within a landholding that is unknown/undesignated according to each AUTEF plan (i.e. not allocated as a logging concession area or as forest set asides) is defined as 'residual land', which was calculated as the total landholding area minus the sum of the concession and set aside areas. Residual land values must therefore fall between 0% and 100% of the total property size. To understand the degree to which landholders complied with legal set asides, we examined the proportion of residual land within AUTEF-authorized landholdings (*N*=678).

3.4 Results

In total, 9,568,249 m^3 of timber across 314 native tree species were legally harvested in the State of Pará between 2006 and 2012 from 446 private and community-based logging concessions for which timber species composition was available. Total timber volumes were also available for a further 378 concessions exploited over the same period. Absolute timber offtake per AUTEF plan ranged between 77 and 298,611.63 m^3 of lumber (mean ± SD = 20, 966.39 ± 27, 292 m^3, *N*=824) depending on sizes of authorized areas, although timber offtake rates per unit area was highly invariant (27.2 ± 4.5 m^3 ha^{-1}, *N*=824) and below the maximum legally required quota of 30 m^3 ha^{-1}.

Timber management plans were associated with highly variable landholding sizes ranging from 26 to 910,307 ha (13,810.0 ± 71,719.3 ha, *N*=824), the distribution of which was visibly clustered by size classes across the State of Pará depending on frontier history (Fig. 3.1). Smallholders are mostly distributed in the Terra do Meio region, along the Transamazon Highway (BR-230). Moreover it is suggested (Fig. 3.1) that 'medium to large' landholders are the leading property sizes found in the new frontier expansions of the Calha Norte region of northwestern Pará and along the Cuiabá-Santarém Highway (BR-163) of southwestern Pará. Over a third of all 824 concessions were granted to 'mini smallholders' and 'smallholders' combined (22% and 35% respectively, *N*=292), 19% to 'medium' landholders (*N*=153), 33% to large landholdings and 13% to very large landholdings (> 5000 ha). In terms of land distribution, we find the vast majority of legally approved aggregated concession areas of Pará are found in large and very large properties (combined sum of 85% of concession areas, Table 3.1). For our subset of AUTEF plans granted to landholdings with available data on species-specific volumetric offtakes (*N*=446) the distribution of size classes was similar and as followed; mini smallholders 24%; smallholders 13%; medium properties 22%; 32% large landholdings and 10% very large properties.

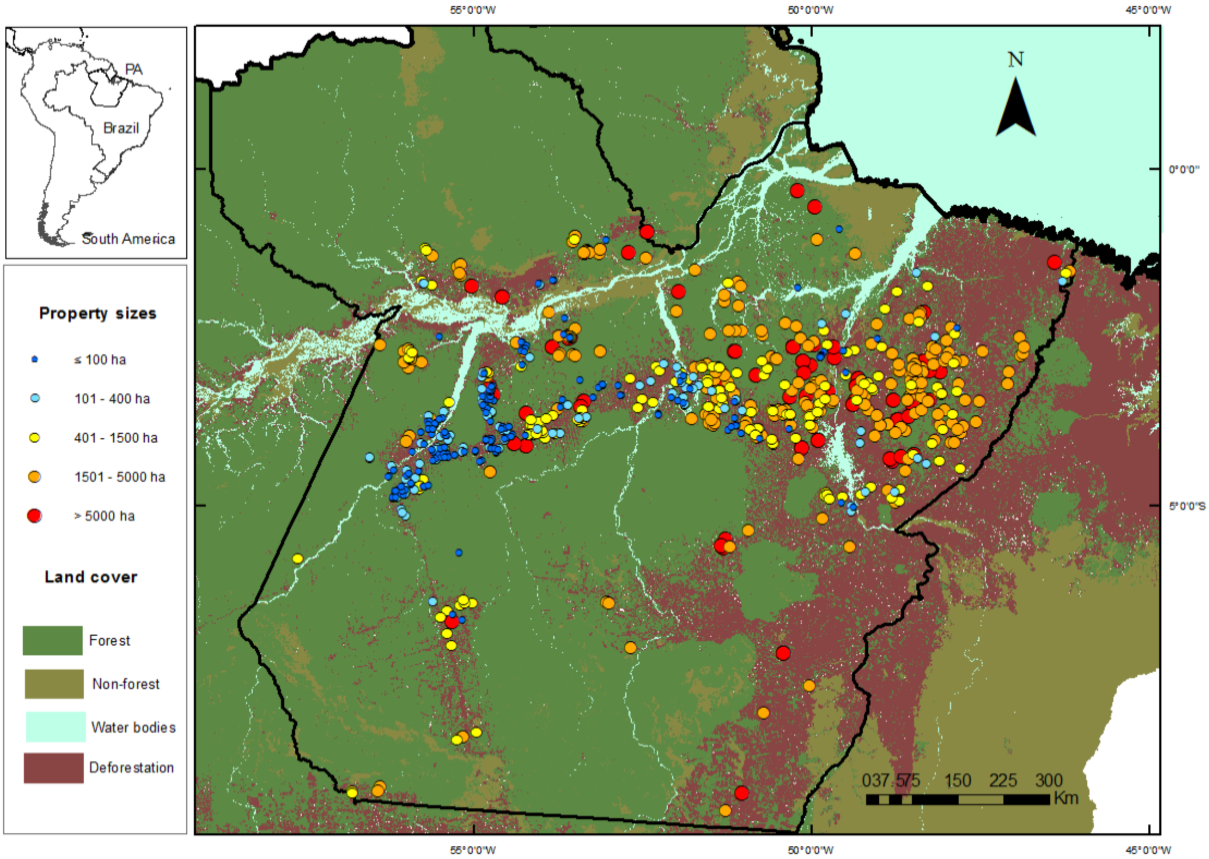

Figure 3.1 Major classes of land cover within the Brazilian state of Pará showing the spatial distribution of 824 private and community-based AUTEF forest management plans approved by SEMA between 2006 and 2012 by property size. Deforested areas as of 2012 are indicated in brown; non-forest areas refer to natural vegetation types, including Amazonian cerrados, outside the closed-canopy forest domain.

Forest concession areas authorized for timber extraction ranged widely, from 20.4 to 9,971.5 ha (775.1 ± 1,027.2, *N*=824, Fig. 3.2) and was a nonlinear function of the total landholding size, reaching an asymptote at properties around 1700 ha. We carried out a linear piecewise regression to estimate the asymptote breakpoint using 'segmented' package in R and after 9 iterations it was estimated at 1,798 ha (R^2_{adj} = 0.86). The total number of authorized taxa for extraction per concession site ranged from only 1 to 70 (22.3 ± 10.6) species, 1 to 57 (20.9 ± 9.1) genera, and 1 to 27 (12.8 ± 4.5) families, and encompassed a taxonomic spectrum of 314 tree species representing 153 tree genera and 38 families. Timber prices per cubic meter ranged from R$ 0.1 m^{-3} to R$ 86.5 m^{-3} (24.3 ± 16.7). Considering the 446 timber concessions for which the species composition of the timber offtake was known, higher gross timber revenues could be derived from larger concession areas (R^2_{adj} = 0.54, P = < 0.001). However, expected timber revenues per ha were weakly but significantly lower in larger concessions (R^2_{adj} = 0.07, P = < 0.001, Fig. 3.3a). Larger landholdings extracted a marginally higher diversity of tree species (R^2_{adj} = 0.009, P = 0.03, Fig.

3.3b), which likely reflected their significantly higher Pielou's J' offtake selectivity values (R^2_{adj} = 0.15, P = < 0.001, Fig. 3.3c) and significant trends in the multivariate composition of timber species by volume, as summarized by the first NMDS axis (R^2_{adj} = 0.28, P = < 0.001, Fig 3.3d). These data suggest that smallholders (lower J' values) are more likely to disproportionately concentrate timber revenues on only a few highly profitable timber species.

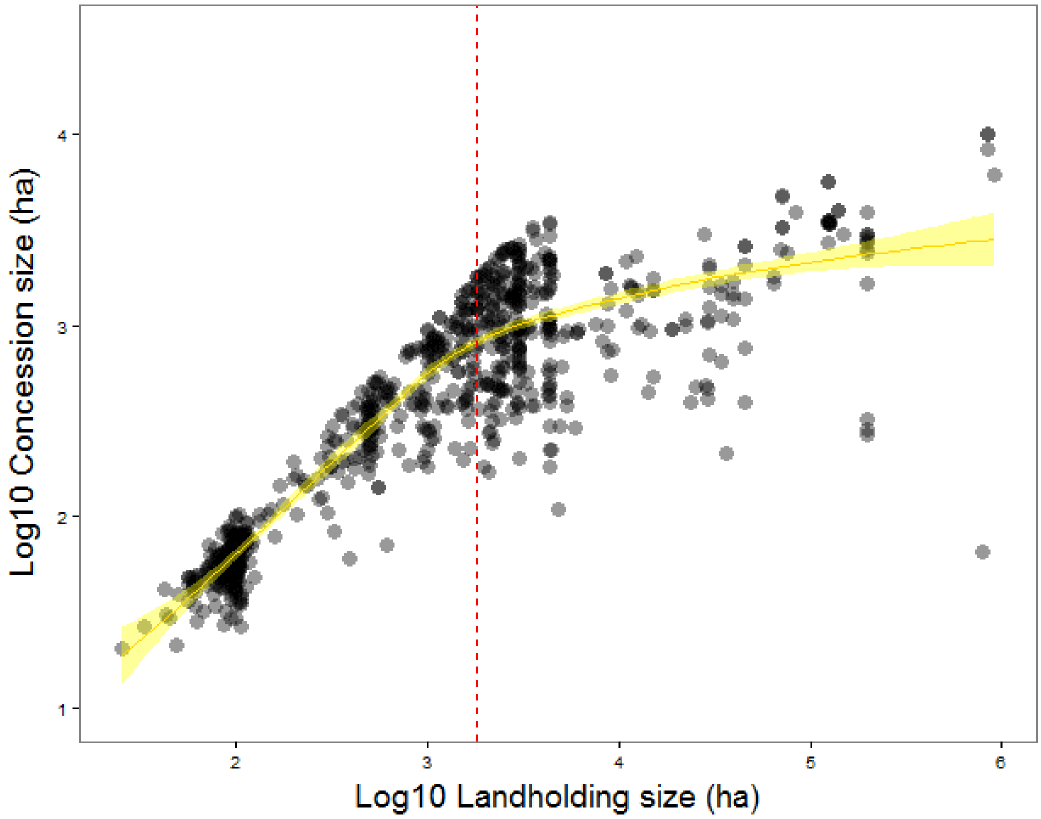

Figure 3.2 Total landholding and concession areas ($\log_{10}$ ha) of 824 private and community-based AUTEF forest management plans approved by SEMA. The yellow solid line represents a smoother fitted through plan means and shaded areas the 95% confidence interval regions. The vertical dashed line (red) represents the estimated asymptote breakpoint (at properties of 1,798 ha) using a piecewise linear regression (R^2_{adj} = 0.86).

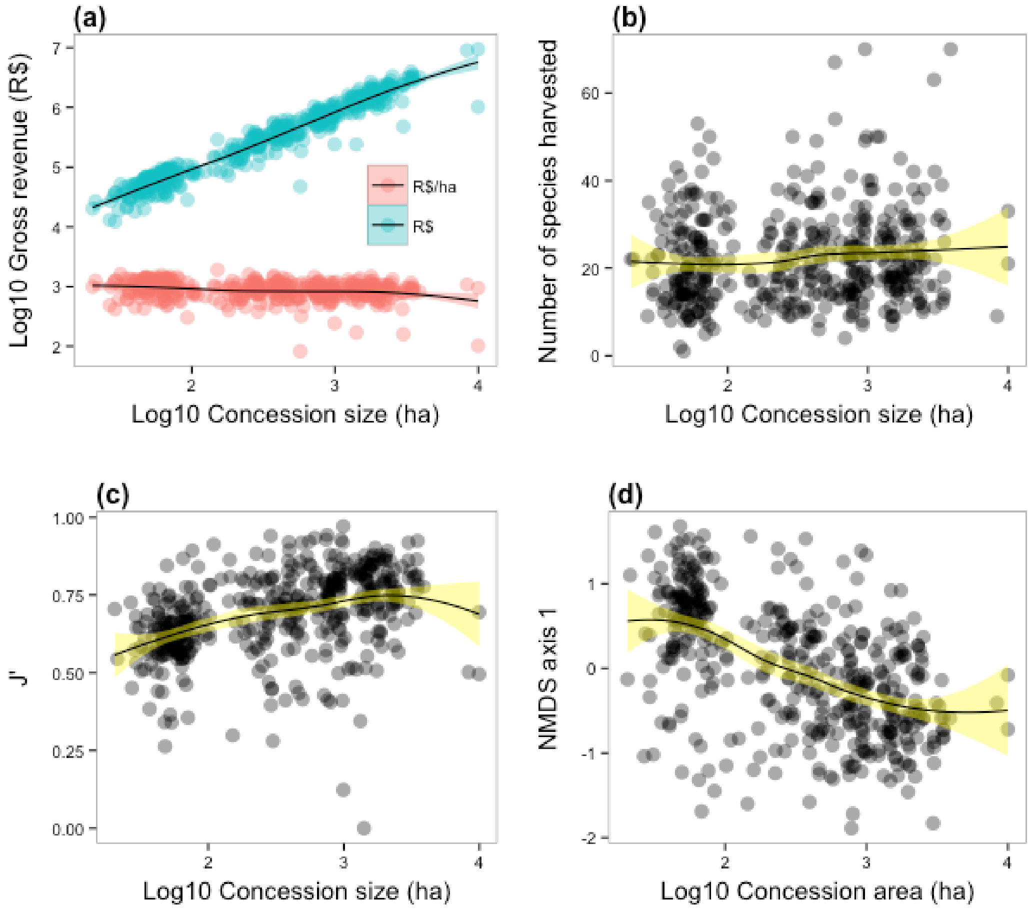

Figure 3.3. (a) Gross expected revenue ($\log_{10}$ R$) and revenue per unit area ($\log_{10}$ R$ ha^{-1}) across 446 private and community-based AUTEF forest management plans by their declared concession areas ($\log_{10}$ ha). The relationship between concession area (*N*=446) and **(b)** the number of species harvested, **(c)** Species selectivity represented by Pielou's J' evenness values (0-1) and **(d)** the volumetric composition of offtakes ($NMDS_1$). The yellow solid line represents a smoother fitted through plan means and shaded areas the 95% confidence interval regions.

The total expected gross monetary revenue (Σ R$ • m^3) predicted across all AUTEF forest management plans was 13.2% for class A timber, 41.3% for class B, 19.2% for class C, and 26.3% for class D. However, stand-scale contributions of these timber price classes were highly variable across variable-sized concessions. Predicted revenues broken down by timber price classes are dominated by class B timber tree species, and scale to concession size for all timber price brackets, but more slowly for the most commercially valuable timber species (Fig. 3.4), i.e. smaller concessions may exert greater financial dependence on class A species. In addition to lower timber selectivity (J') values, we find further evidence that smaller properties appear to exert stronger high-grading pressure on forest stands, compared to larger properties. After class B timber species, total revenues attributed to class A timber dominated smallholder concessions up until around the threshold of ~100 ha. At this size threshold, revenue composition shifted, with class A timber species being overtaken by class C and then class D. This shift becomes more pronounced as property sizes increased from medium through to very large properties (Fig. 3.4).

In relation to total property sizes, proportional concession areas ranged from 0.008 to 100% (49.8 ± 27.3 %, N=824, Fig. 3.5a). Proportions were similar for mini smallholders (66.0 ± 14.1 %), smallholders (62.5 ± 16.0 %), and medium properties (60.1 ± 20.9 %). Only large properties applied for logging concessions areas under half their property sizes (45.2 ± 26.3 %), which further declined to only 6.0 ± 5.3% in very large properties.

Most AUTEF forest management plans (65%) complied with the minimum forest set-aside of 80% of their land, at least on paper. The proportions of landholdings dedicated to Legal Reserves ranged from 2.1 to 165.2% (80.7 ± 8.3 %, N=678) and varied little across property size classes (Table 3.1). In particular, 80% of all properties (N=678) reported inconsistent sizes for either RL or APPs in relation to the total property size, thereby resulting in negative values for residual land. We carried out a linear piecewise regression to estimate the asymptote breakpoint using 'segmented' package in R and after 4 iterations it was estimated at 759 ha. We find that only very large properties (> 5,000 ha) were likely to report consistent sizes of required forest set asides (Fig. 3.5b, Table 3.1).

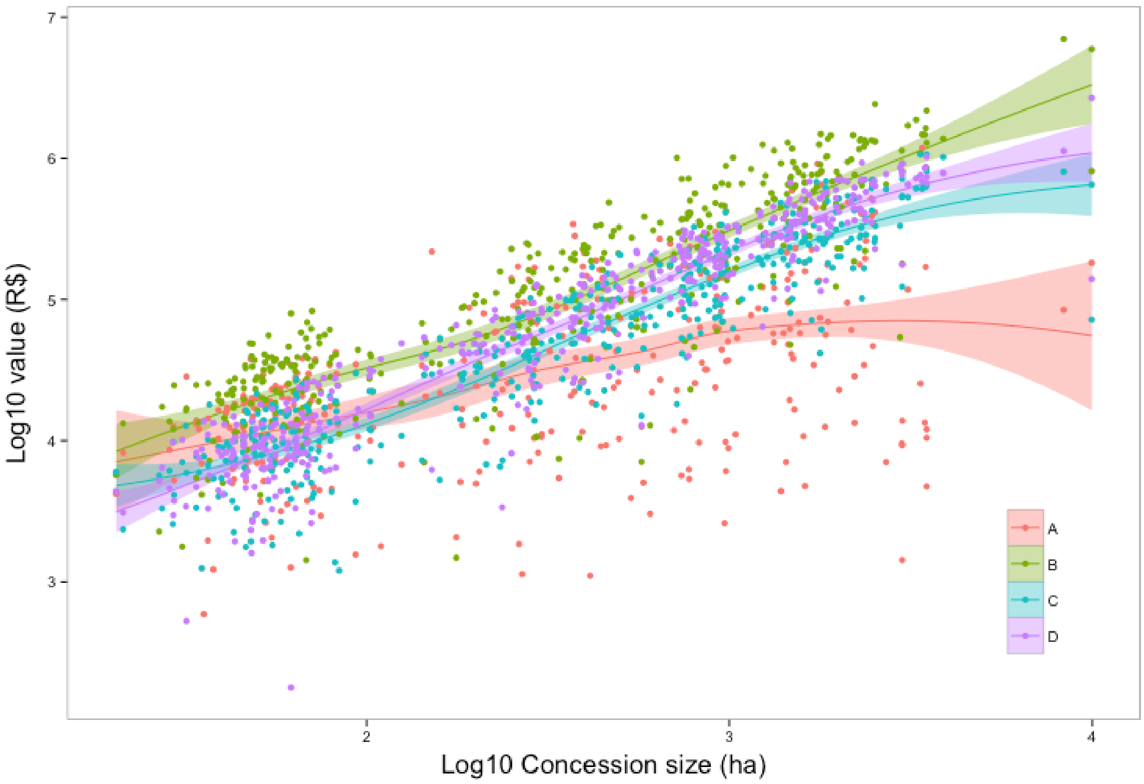

Figure 3.4. The relationship between concession size ($\log_{10}$ ha) and the sum of total expected gross revenue ($\log_{10} \Sigma$ R$ • m^3) for each contrasting market price class in each AUTEF plan (*N*=446). Data points are colour-coded by price classes and distributed as followed; A in red (> R$75.0/m^3); B in green (R$45.0/m^3 - R$74.0/m^3); C in blue (R$25.0/m^3 - R$44.0/m^3); and Class D in purple (R$1.0/m^3 - R$24.0/m^3). Colour-coded smoother lines are also represented for each price class, and shaded areas the 95% confidence interval regions. Class B timber (green line) dominates the total value of timber offtakes, regardless of concession size. As concession size increases, there is a decreasing contribution of high value class A timber species to the sum of total expected gross revenues. Notably the smallest concessions (especially smaller than 100 ha), may exert greater financial dependence on class A species (red line intersection).

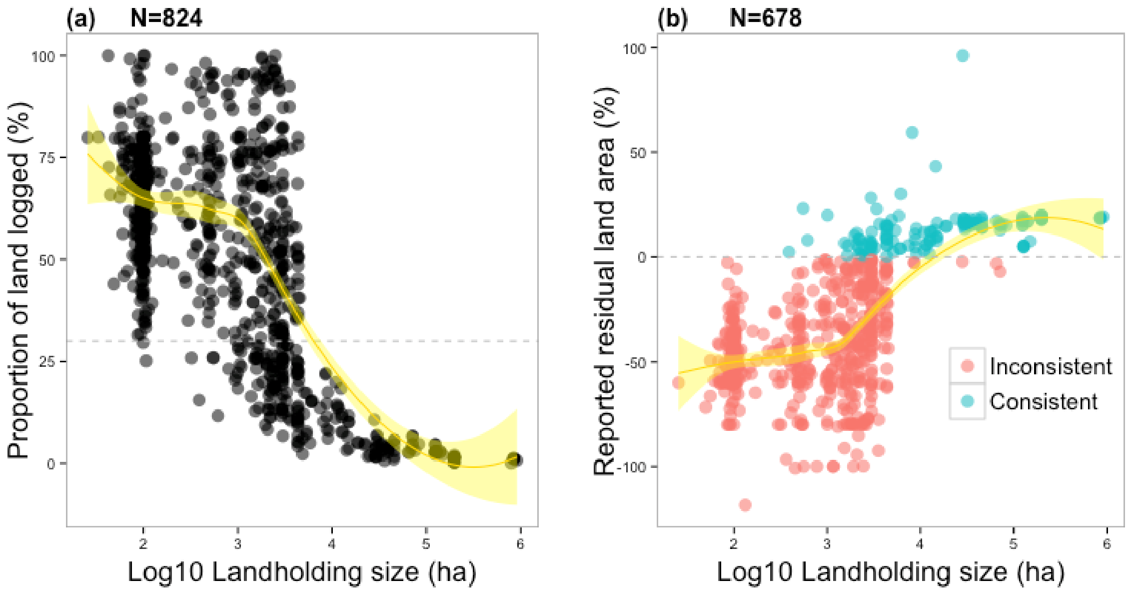

Figure 3.5. (a) The relationship between total landholding size ($\log_{10}$ ha) and the proportion of the landholding declared for logging (concession area). The grey horizontal dashed line represents concession areas that make up 30% of total property sizes. The yellow solid line represents a smoother fitted through plan means and shaded areas the 95% confidence interval regions. **(b)** Total landholding size ($\log_{10}$ ha) and the amount of residual land left in landholdings after legal forest set asides (Legal Reserves and APPs) and concession areas were accounted for. The grey horizontal dashed line crosses the y axis at 0%, i.e. where the sum of legal forest set-asides areas (RL and APP's) and concession areas are equal to the whole property size. Properties below this line have negative residual land % and thus reported inconsistent information (data points coloured red). Landholdings on or above the dashed line report consistent areas in terms of legal forest reserves, concession or property sizes and may have surplus areas (positive residual land %).

Table 3.1. A summary of our reported variables by property size class across all 824 AUTEF plans and our two subsamples of 678 and 446 plans. Key to column heads; N, number of properties surveyed in each size class; mean landholding size within each size class in ha (mean ± SD); aggregate concession area (ha) i.e. sum of all concession areas within each class of property size; proportion of land logged (mean ± SD %); Proportion of total volumetric offtake across all AUTEF plans (% of 17.3M m^3); RL legal forest-set aside compliance (mean ± SD %); residual land (mean ± SD %) is the proportion of undesignated land in relation to the entire property size (i.e. total property size minus the sum of legal forest set asides plus concession areas); the proportion of AUTEF plans with inconsistent (negative residual land); expected revenue from class A timber species (mean ± SD %); estimated forest cover remaining within a 10km buffer (mean ± SD %); frontier age (mean ± SD years); and basal area estimates (mean ± SD m^2).

	N=824				
Property size classes (ha)	N	Landholding size (ha)	Aggregate concession area (ha)	Proportion of land logged (%)	Proportion of total volumetric offtake (% of 17.3M m^3)
Mini smallholders (≤ 100)	186	87.1 ± 15.3	10,604	66.0 ± 14.1	1.6
Smallholders (101 – 400)	106	163.5 ± 91.5	10,828	62.5 ± 16.0	1.7
Medium landholders (401 – 1,500)	154	837.2 ± 333.5	76,232	60.1 ± 20.9	12.1
Large landholders (1,501 – 4,999)	271	2,781.3 ± 879.1	321,173	45.2 ± 26.3	51.9
Very large landholders (≥ 5,000)	107	97,787.6 ± 178,166	219,842	6.0 ± 5.3	32.7

	N=678			N=446				
Property size classes (ha)	Compliance with RL (%)	Residual land (%)	AUTEF plans with inconsistent areas reported (%)	Revenue from class A timber (%)	Revenue from class D timber (%)	Forest in 10km buffer (%)	Frontier age (years)	Basal area estimate (m^2/ha)
Mini smallholders (≤ 100)	79.8 ± 3.0	-50.8 ± 13.8	100	19.4 ± 14.1	20.2 ±15.9	76.6 ± 26.7	24.6 ± 11.9	13.4 ± 2.7
Smallholders (101 – 400)	80.9 ± 6.4	-48.2 ± 16.9	99	19.2 ± 16.2	24.7 ±16.7	53.7 ± 33.4	24.5 ± 14.8	11.4 ± 3.5
Medium landholders (401 – 1,500)	79.8 ± 6.7	-43.7 ± 24.5	98	15.7 ± 15.2	27.5 ±16.8	58.8 ± 31.7	25.4 ± 10.8	11.3 ± 3.0
Large landholders (1,501 – 4,999)	82.1 ± 9.7	-30.2 ± 28.9	84	6.7 ± 8.9	30.2 ± 16.0	80.4 ± 25.3	26.9 ± 9.2	11.7 ± 2.4
Very large landholders (≥ 5,000)	79.1 ± 11.0	14.7 ± 12.0	0.05	5.7 ± 10.4	28.1 ±10.2	71.9 ± 29.4	30 ± 12.7	11.1 ± 2.2

3.5 Discussion

On the basis of 824 Amazonian timber concessions, we provide strong evidence that smallholders in tropical forests exert stronger high-grading pressure upon the high-value hardwood species available in their landholdings, thereby accruing higher gross revenue productivity per unit area, at least in the short term (cf. Sen 1962). To the best of our knowledge this is the first large-scale assessment of this kind on timber productivity from natural forests in the tropics across a wide range of rural property sizes. Mini, small, and medium size landholders selectively logged more than 60% of their landholding areas, whereas large landholdings (>1,500 ha), in particular those larger than 3,000 ha, were most likely to be issued concession areas of 30% or less of their land, which was further reduced to only ~6% in very large properties (> 5,000 ha).

Smallholders were also more likely to overestimate the minimum areas of forest set-asides as required by law, with all mini smallholders (≤100 ha) and 99% of smallholders (101 – 400 ha) inconsistently reporting their own forest set-asides, resulting in negative residual land values that were either incorrect or dishonest. This is line with newly settled small farmers in southern Amazonia, where properties below 150 ha retained far less upland and riparian forest set-asides than largeholdings above this threshold (Michalski et al. 2010). This is consistent with the growing per capita contribution of smallholders to overall deforestation across Amazonian agricultural frontiers (Godar et al. 2012; Schneider & Peres 2015), not least because small farmers inherently lack sufficient economies of scale to both sustain an average-size family and retain most of their forest.

In response to record-high deforestation levels in 1995, RL set asides were increased in 1996 from 50% to 80% of private property sizes. In practice, these requirements were largely unattainable, and millions of landholders across Brazil struggled to meet those legal requirements, which failed to restrict deforestation as planned (Soares-Filho et al. 2014). Producers in previously forested regions of Mato Grosso, Brazil faced opportunity costs of up to US$3–5.6 billion in net present land value, which led to widespread non-compliance of legal obligations sanctioned by the Brazilian Forest Code (FC; Stickler et al. 2013). Amendments to the FC in 2012 have now reversed RL requirements from

80% to 50% of each landholding for all municipal counties within Amazonia and APP areas can now be declared as part of the RL reserve quota (Law 12.651, 25 March 2012). Additional legislative revisions have also provided amnesty for any illegal deforestation in smallholdings (which in Amazonia are defined as 4 fiscal modules or <440 ha) prior to 2008. Under these new guidelines, some 90% of all rural properties throughout Brazil have been legally absolved of any deforestation violation, whereas this does not apply to largeholders (Soares-Filho et al. 2014). Large landholdings therefore face a top-down legal incentive to comply with forest set-asides, which is consistent with our findings that only very large properties almost always reported positive residual areas (99.95% of landholdings ≥ 5,000 ha).

AUTEF forest management plans are self-declared legal documents and we are aware that many landholders may deliberately fabricate total property sizes or forest set-asides on paper, adding as much as >80% of the total landholding area. Studies overlaying AUTEF data with high-resolution spectral mixing analysis of logging-induced forest disturbance have shown documental inconsistencies such as inflated concession areas or authorized logging occurring in areas that had already been heavily degraded or deforested. Inconsistencies were reported in 37% of all AUTEF plans sanctioned in 2007-2008 but this was improved to 10% (2009 – 2010); 11% (2010 – 2011) and 13% (2011 – 2012) (Monteiro et al. 2008; 2010; 2011; 2012). On average, 65% of 678 AUTEF plans that we examined adhered to the Forest Code obligations (on the basis of the ≥80% property size threshold) at the time these plans were granted. Godar et al. (2012) found the same average compliance level based on the revised 2012 RL requirement of 50% and georeferenced property boundaries across four municipal counties along the Transamazon Highway. The authors found that smallholders (up to 100 ha) were more compliant with RL and deforested 38% of their properties in comparison to 42% in medium landholdings (100 – 600 ha) and 30% in large landholdings (> 600 ha). When discussing patterns of deforestation how one distinguishes between small and large landholders is crucial, the majority of Godar et al. (2012) 'medium' landowners may be defined as smallholders in our study (101 - 400 ha) and although we don't investigate deforestation directly, if the proportion of concession area is a proxy for forest degradation we too find that these same small-medium property sizes (after mini smallholders) expect to have the highest levels of forest degradation and that proportional degradation decreases with increased property size class.

Modern Amazonian colonisation has been an ongoing dynamic process; in the state of Pará, 452,493 ha were allocated between 2011 and 2013 to 10,235 families across 41 settlements (INCRA 2015). Smallholders remain in old logging frontiers long after the large commercial mills close and move to new unlogged forest regions (Sears et al. 2007). Analysing AUTEF plans by property size fails to incorporate the breadth of deforestation and colonisation histories that different actor types may have experienced. Landowners are likely carrying out logging activities in conjunction with other activities such as extraction of non-timber forest products, and production of charcoal, soy or cattle. Cooperatives can also provide cost-sharing opportunities for smallholders such as machinery investments (Markelova et al. 2009). However, we do not know the extent to which AUTEF landholders engaged in other forms of land use.

Central government programs such as the PMCF (*Programa Federal de Manejo Florestal Comunitário e Familiar*, Decree Nº 6.874 of June 5th 2009) have greatly incentivized small-scale logging by smallholders, including isolated families and households in sustainable-use forest reserves. A study commissioned by the program between 2009 and 2010 found 1,213 cases of forest management by local communities or families across six Brazilian Amazonian states (Pinto et al. 2010). It was estimated that 27% of these cooperatives also harvested non-timber forest products with a total aggregated contribution worth 72% to regional GDP (gross domestic product across the Brazilian states of Acre, Amapá, Amazonas, Maranhão, Pará and Rondônia).

Because smallholders are presumably forced to exert higher selectivity and dependence towards high value class A timber species, we assume this enables them to accrue higher gross revenues per unit area, although this fails to consider timber exploitation costs and access to markets. It has been suggested that adding sustainable timber yields to smallholder cash flow would delay deforestation by about a decade, but also adds about two years of negative cash flow after investments in log hauling and processing equipment (Vosti et al. 2003). High discount rates in tropical countries often encourages harvests of forest stands to maximise net present values, thereby allowing profit reinvestments rather than waiting for slower-timber growth (Winterhalder & Smith 2000; Rice et al. 1997). It may be unrealistic to expect smallholders to fulfill additional stringent requirements beyond reduced impact logging that ensure more sustainable timber yields (Zarin et al. 2007). This is further aggravated by excessive bureaucracies and fluctuating government policies, which were incompatible with the long-term viability of reduced impact logging to livelihoods in a community forest management system in the state of Amazonas, Brazil (Waldhoff & Vidal 2015). Recent regulations now require all concession areas to retain at least 15% of all large trees (≥ 50cm DBH) for each target timber species within their Annual Production Units, which also cannot violate a population density threshold of four large trees per 100 ha of every species extracted (Normative Ruling no. 01 of 12th February 2015). We raise concerns that at present, smallholders in Pará may be unable to adhere to these requirements. Largeholders can afford greater financial flexibility, and their comparatively vast gross returns allow them to comply with best-management practices and carry out selective logging using more stringent tree selection, mechanized access, and roundlog removal techniques. However, we acknowledge that the greater spectrum of inventoried timber species licensed for logging in larger landholding is partly induced by positive species-area relationships; offtakes across all the concessions we examined added another 2 species for every order of magnitude increment in concession size. Largeholders may also be able to afford higher quality forest inventories employing qualified parataxonomists and forest engineers who can ensure that a higher proportion of the most desirable species are available for a second harvest.

We did not find consistent patterns in the proportion of forest cover within 10-km buffers around our concession sites, but census districts dominated by smallholders across the Brazilian Amazon have forests of higher integrity in terms of fewer edges, more core forest, and less degradation (Godar et al. 2014). These authors highlight that smallholders are more likely to dominate forested frontiers whilst large properties are more prevalent in consolidated frontier areas with access to markets, and have financial capital to build private infrastructure such as roads. On the other hand, smallholders are likely to depend on public roads and may have an additional livelihood dependence on secondary forests that may buffer edge effects. These patterns are consistent with our findings in Fig. 3.1.

Between 1995 and 2013 the Brazilian government degazetted a total of 2.5 Mha from 38 protected areas comprised of federal and state-level forest reserves and indigenous territories across all nine Brazilian Amazonian states (Martins et al. 2014). In 74% of cases, the alleged reason to downsize or nullify these reserves was illegal occupation or land grabbing; whereas state-level governments prefer to expropriate lands than to compensate and evict settlers. Complex and perverse incentives still exist for speculative land grabbing, a major historical driver of regional violence and deforestation in the Brazilian Amazon (Borras et al. 2012).

The rural territorial tax (*Imposto Territorial Rural, ITR*) is meant to burden large unproductive landholdings and inhibit the illegal occupation of public forests. In reality, the ITR has created a perverse incentive for deforestation, with one study estimating ITR tax evasions in the order of R$ 270M per year (~115M US dollars at the mean commercial exchange rate of 2014, data from IPEA, Instituto de Pesquisa Econômica Aplicada, available at http://www.ipeadata.gov.br/), just in the state of Pará (Silva & Barreto 2014). ITR taxes are levied by property size, whereby smallholdings <200 ha that were often allocated through the agrarian reform program are exempted of this tax. Because landholders are not charged for forest areas on their land, the aim is to increase productivity in previously deforested areas. Productive smallholders pay very small land taxes, with those between 200 and 500 ha required to pay only 0.1% of their land value whereas very large landholdings (>5,000 ha) using 30% or less of their lands are required to pay 20%. The Brazilian federal government still nominally controls 38 M ha of undesignated public lands in Amazonia but speculative 'land grabbers' frequently deforest these areas to show evidence of *de facto* occupation (Silva & Barreto 2014). However, the government rarely prosecutes ITR tax evaders. Hence, contrary to the original intentions of the ITR, medium-sized landholdings that deforest illegally (up to 15 fiscal modules in Amazonian properties <1,500 ha) are perversely fast-tracked in any attempt to obtain legal land tenure through the, Legal Land Programme (*Programa Terra Legal* Law 11.952/2009, Brito & Barreto 2011). These policies might explain the asymptote breakpoints in property sizes described in Fig. 3.5 and in particular Fig. 3.2; concession areas as a function of total landholding size increases nonlinearly and decreases at landholdings larger than 1700 ha.

Comparisons between the logging and agricultural sectors must be made with caution. Timber extraction is not analogous to farming and there is widespread evidence that at present, most of tropical timber extraction is far from sustainable (Wadsworth & Zweede 2006; Sist & Ferreira 2007; Zarin et al. 2007; Putz et al. 2008; Peña-Claros et al. 2008; Schulze et al. 2008; Macpherson et al. 2012). In Asian markets, 'peak timber' are already evident, as the region fast approaches a typical symmetric 'Hubbert Curve' logistic distribution seen in many overexploited non-renewable extractive industries (Shearman et al. 2012). Merry et al. (2009) estimated that 4.5 ± 1.35 billion cubic meters of commercial timber are available across Brazilian Amazonia, 1.2 billion m^3 of which currently profitable to harvest, resulting in an estimated total stumpage value of US$15.4 billion.

Significant implementation of sustainable forest management in Amazonia is further compounded by pervasive illegal logging activities, which directly compete with lower-impact legal logging and account for 50–90% of all pantropical forestry products worth US$ 30 – 100 billion yr^{-1} or 10–30% of the global wood trade (Nellemann 2012). Agricultural opportunity costs vary by actor type and local soil conditions, but evidence suggests that REDD+ smallholder compensations in Sumatra might be more expensive than previously thought (Cacho et al. 2014). Rural properties or enterprises of different sizes have inherently different comparative advantages at different stages in the business cycle, and the socially optimum size of an enterprise can be influenced by both consumer demands and public policies (Stigler 1958). Our results clearly illustrate that greater positive incentives for actor-specific responsibilities would be effective in promoting more sustainable forest management among a burgeoning number of smallholders across Amazonia.

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Chapter 4: Defining and quantifying collateral damage induced by reduced-impact logging in Amazonian forests

4.1 Abstract

Selective logging of tropical forests, in particular reduced impact logging (RIL), has long been suggested as a benign compromise between profitable forest land use and biodiversity conservation. Globally, only 35% of natural forests are reported to be primary forest. To date, a quarter of all primary pantropical forests have been selectively logged and this will continue to increase both in extent and harvest intensity. The selective removal of exceptionally large trees in forest stands leads to several forms of collateral damage to the residual stand. Gaps are created from the canopy to forest floor within felled-tree impact zones, crushing smaller stems, and substantially increasing greenhouse gas emissions as fallen or severely damaged stems are committed to mortality. This degradation is poorly quantified or understood despite representing the second 'D' of REDD+ (Reduced Emissions from Deforestation and Degradation) and recognised by both the UN Framework Convention on Climate Change and the Intergovernmental Panel on Climate Change. We reviewed 73 studies on selective logging impacts in tropical forest fauna to assess the extent to which they at least mention or attempt to quantify collateral damage to the residual stand. Our findings indicate that ~90% failed to do so. We also estimated the collateral damage associated with 248 harvested trees in a certified industrial-scale RIL operation of Eastern Brazilian Amazonia (harvest intensity 30m^3 ha^{-1}) and report data from 137 logging gaps where 3,256 damaged trees (≥10 cm DBH, diameter at breast height) were inventoried. Mean logging gap area (which may include >1 felled tree) was 891.1 m^2 resulting in a mean gap size of 492.1 m^2 per felled tree. Over a third (35.9%) of our total sampling area of logged forest was cleared by felled-trees alone in terms of the aggregated areas of all logging gaps. For each tree harvested, we estimated an average loss of 11.7 damaged

stems, and for every $1m^2$ of timber basal area removed, $1.5m^2$ was lost in damage. Stem DBH was the main predictor of damaged stem survival, but mortality was indiscriminate across stems smaller than ~23 cm DBH. Our results indicate that the effects of collateral damage even in well planned RIL operations are substantial and remain overlooked by the tropical forest conservation community. Without a standardised method or currency for quantifying collateral damage, logging impact studies will remain poorly comparable, ultimately hindering our understanding of how timber extraction impacts tropical forest ecosystems, forest wildlife, and forest carbon fluxes.

4.2 Introduction

Globally, only 35% of natural forests are reported to be 'primary forest', the remaining 65% are described as 'other naturally regenerated forest' (FAO 2015). The Amazonian rainforest contains approximately a quarter of all terrestrial species and the region is paramount to tropical biodiversity conservation (Dirzo & Raven 2003). There are hundreds of pantropical logging impact studies, a meta-analysis of faunal impact studies suggests that logging-induced disturbance on forest fauna tends to be more severe in the Neotropics than in the Indomalayan or African tropics (Burivalova et al. 2014).

Collateral damage on the other hand, i.e. damage to the residual stand of selectively logged forests, has been largely overlooked in tropical forest disturbance ecology. Selective logging degrades tropical forests, initially through the removal of target harvest trees. When large timber trees fall to the ground, they often crush and remove crowns and branches of surrounding non-target trees under their crown and bole. Damage is often extensive and creates forest gaps from the canopy to forest floor, and remaining standing trees can become particularly vulnerable to toppling over from wind blasts (Uhl & Vieira 1989). Structural damage also occurs at the point of roundlog removal as extraction vehicles (bulldozers or tractors) manoeuvre felled logs out of the forest through dense networks of skidding trails and logging roads. In large commercial operations, additional logging decks are opened so timber can be stockpiled and transported to sawmills. Selectively logged forests are more susceptible to wildfires (Holdsworth & Uhl 1997; Lindenmayer et al. 2009) and between 1999 and 2002 selective logging created more forest fragmentation (forest edges) than deforestation across the Brazilian Amazon (Broadbent et al. 2008).

The UN Framework Convention on Climate Change (UNFCCC) at its 13th Conference of the Parties acknowledged and incorporated the need to quantify carbon losses resulting from forest degradation via the REDD+ mechanism (Reducing Emissions from Deforestation and forest Degradation, UNFCCC 2008). Additionally, the IPCC (Intergovernmental Panel on Climate Change) have yet to establish clear guidelines for measuring carbon emissions from forest degradation, including selective logging operations (IPCC 2006; Pearson et al. 2014). Compared to deforestation, climate regulation and carbon storage capacity of selectively logged forests and other forms of anthropogenic forest degradation have been poorly addressed by both civil society and governments (Berenguer et al. 2014).

Timber extraction methods to reduce collateral damage have been designed as early as the 1950s (Nicholson 1958). Initially, studies focused on sustaining timber yields of the most coveted high-value species (Putz et al. 2000). Focus shifted from the 1980's onwards in support and promoting reduced impact logging techniques (hereafter, RIL) over conventional logging (hereafter CL, Putz et al. 2008). Less damaging management techniques to extract roundlogs were developed and now include the use of wheeled tractors over crawler tractors, cutting vines and climbers prior to logging, directional felling, road and skid trail planning, and maintaining buffers of unlogged forest cover along watersheds to protect against runoff and erosion (Fox 1968; Froehlich et al. 1981; Pinard & Putz 1996; Vidal et al. 1997; Putz et al. 2000).

Impact studies of both conventional and selective logging on wildlife across several taxa have been reported since the 1980s (Johns 1985). A literature review of 75 logging impact studies on topical forest fauna revealed that most studies (88%) failed to conduct a pre-logging baseline assessment, 65% failed to report on the type of management, and 45% failed to report on logging harvest intensities, either as basal area removed (m^2 ha^{-1}), volumetric removal (m^3 ha^{-1}), or the number of trees harvested (trees ha^{-1}), significantly limiting comparability among studies (Laufer et al. 2013). Burivalova et al. (2014) conducted a meta-analyses on 48 tropical studies to understand broad patterns in species richness across a gradient of logging intensity. Predictors of species richness included logging intensity but damage to the residual stand was not evaluated and most likely unavailable (of 200 potential studies, only 48 met quality criteria for analyses). We argue that failing to quantify residual damage to remaining forest stands is a lost opportunity to appropriately link structural and compositional degradation of tropical forests through selective logging to any observable impacts on forest structure and composition, and forest wildlife.

Although approaches to evaluate collateral damage are highly variable, two broad methods have prevailed: area-based and felled-tree based methods (Parren & Bongers 2001). Area-based methods often examine plot scale pairwise differences in forest structure (pre- and post-logging) and express damage as a percentage of undamaged forest. Tree-based methods attribute damage to individual felled trees, usually expressed as a ratio of damaged to harvested trees (Picard et al. 2012). Scaling between these different methods to compare impact studies can be difficult. Picard et al. (2012) conducted a meta-analysis of published literature in order to construct a pantropical equation relating logging intensity (CL) and damage. They highlight several limitations in comparing collateral damage across studies, damage is often scale dependent and influenced by biased calculations, i.e. differences in units, minimum diameter thresholds in assessing damaged trees and time elapsed since logging. In area-based approaches, proportional damage can be vastly affected by whether whole concession area or concession Annual Production Units (areas sanctioned for logging in a given year, hereafter APUs) are considered. Additionally, the focus of studies determines what is measured. For instance, carbon-centric studies on forest biomass recovery can render the level of felled tree impact damage difficult to deduce.

Beyond existing requirements of the IPCC and REDD+ programmes, residual damage estimates have been applied to equations that govern and set logging intensities to granted forest concessions in Central Africa (Picard et al. 2012). There is evidence that current industrial scale logging is demographically unsustainable and may lead to biological depletion or extinction of commercial timber species across the tropics (Chapter 1, Shearman et al. 2012). Without standardizing the terminology and metrics of damage, it would be impossible to fully understand the impacts of selective logging on tropical fauna and flora and carbon stocks. A standard set of methods to estimate residual damage is therefore urgently needed.

We review the tropical forest literature and assess the extent to which collateral damage is measured across selective logging impact studies on forest fauna. We then provide a simple but comprehensive rapid method to quantify *in situ* structural collateral damage associated with a RIL operation in Eastern Amazonia, Brazil. We propose a hybrid approach encompassing both area- and tree-based methods, where damage can be apportioned to felled trees or as a percentage of unlogged areas. Instead of using plot surveys or line transects we survey damage in each logging gap and aggregate damage in gaps where multiple trees were harvested. Moreover, we quantify damage on multiple vegetation layers from ground level damage to understorey vegetation to tree and canopy damage. Finally, we explore residual damage at the species composition level in a neotropical forest (see Medjibe et al. 2011 for impacts of selective logging on species richness in Africa).

4.3 Methods

4.3.1 Study site

Sampling was conducted at the Fazenda Rio Capim landholding of CIKEL Brasil Verde group (hereafter, CIKEL) in the eastern state of Pará, Brazil (3°32'S, 48°49'W, Fig. 4.1). This ~140,000-ha landholding encompasses large areas of natural primary (12,000 ha) and logged *terra firme* forests (110,000 ha) in addition to abandoned pastures (18,000 ha), which may contain plantations of fast-growing tree plantations [paricá (*Schizolobium amazonicum*) and eucalyptus (*Eucalyptus spp.*)]. Hunting is prohibited across the landholding. Mean annual rainfall is 1800 mm, and the topography is mostly flat with a mean elevation of 20m a.s.l. (Sist & Ferreira, 2007). CIKEL has been harvesting Forest Stewardship Council (FSC) certified timber since 2001. RIL techniques include a minimum cutting diameter of 55cm for all commercial species, a cutting cycle of 35 years, and in accordance to Brazilian forest management laws (Normative Ruling 05 of 2006), a maximum extraction intensity of $30m^3 ha^{-1}$. Some of the most commonly harvested species include *Manilkara huberi, Hymenaea courbaril, Astronium lecointei, Parkia pendula, Couratari oblongifolia, and Pouteria bilocularis*.

4.3.2 Logging offtake data

Each harvested tree stump was identified to species level and georeferenced. The angular direction (degrees) of each tree fall was estimated *in situ* using the direction of any residual portions of their bole and crown as a guide with the aid of one of our field assistants, a long-term logger and parataxonomist with over 30 years of experience at this site. Individual tree identification number tags at the time of tree felling were also recorded and later cross-referenced with our database to obtain information including positional references, species, DBH (diameter at breast height, cm) and volume (cubic metres). We hypothesized that gaps where harvested trees fell in different directions, rather than sharing the same broad impact zone, imparted greater collateral damage. We therefore expressed the directional variance of tree-falls across gaps where more than one tree was logged (i.e. multiple tree gaps) as the circular standard deviation ($SD_ø$) of angular direction of tree-falls, which was obtained using the 'circular' R package (Agostinelli & Lund 2013). The mean distance between logged trees within each logging gap was analysed in ArcGIS 10.2.2 using the suite of tools within the spatial statistics package and our own GPS georeferences obtained in the field.

4.3.3 Logging gaps and canopy fracture

Data were collected over two sampling periods (April-May 2012 and May-June 2013, Fig. 4.1), at canopy gaps generated by RIL tree-falls after a period of 9 to 10 months post-harvest to ensure we did not underestimate collateral damage, which becomes increasingly difficult to detect over time (Picard et al. 2012). We define logging gaps as open areas created by the impact of felled-trees (including both bole and crown) within selectively logged forest stands surrounded by undamaged standing trees. Ellipsoid gap areas were estimated from field measurements using a 50-m measuring tape of the longest straight axis (between farthest undamaged trunks) and its corresponding perpendicular width. Collateral damage associated with canopy tree felling and removal was sampled across 137 logging gaps comprising a total aggregate gap area of 122,083 m^2 or ~12.2 ha.

Hemispherical digital photographs were georeferenced and taken at the approximate centroid of each gap using a NIKON Coolpix camera model E8800, and a NIKON FC-E9 fish eye converter lens on an automatic exposure setting. Images were stored in JPEG format, 3264 x 2448 pixels in dimension. A standardized camera set up included placing the top of the camera (and image) aligned with the magnetic north, at a height of 1.5m with the aid of a tubular spirit level, and always during rainless fully overcast weather conditions to prevent image glare from direct sunlight (Rich 1990). Digital images (Fig. S4.1) were transformed into binary black and white pixels using an automatic threshold algorithm to separate canopy and sky by edge detection using SideLook 1.1.01 (http://www.appleco.ch, Nobis 2005), thus reducing some subjectivity as a source of error (Nobis & Hunziker 2005). Canopy openness was estimated using Gap Light Analyzer version 2.0 (http://www.ecostudies.org/gla/, Frazer 1999).

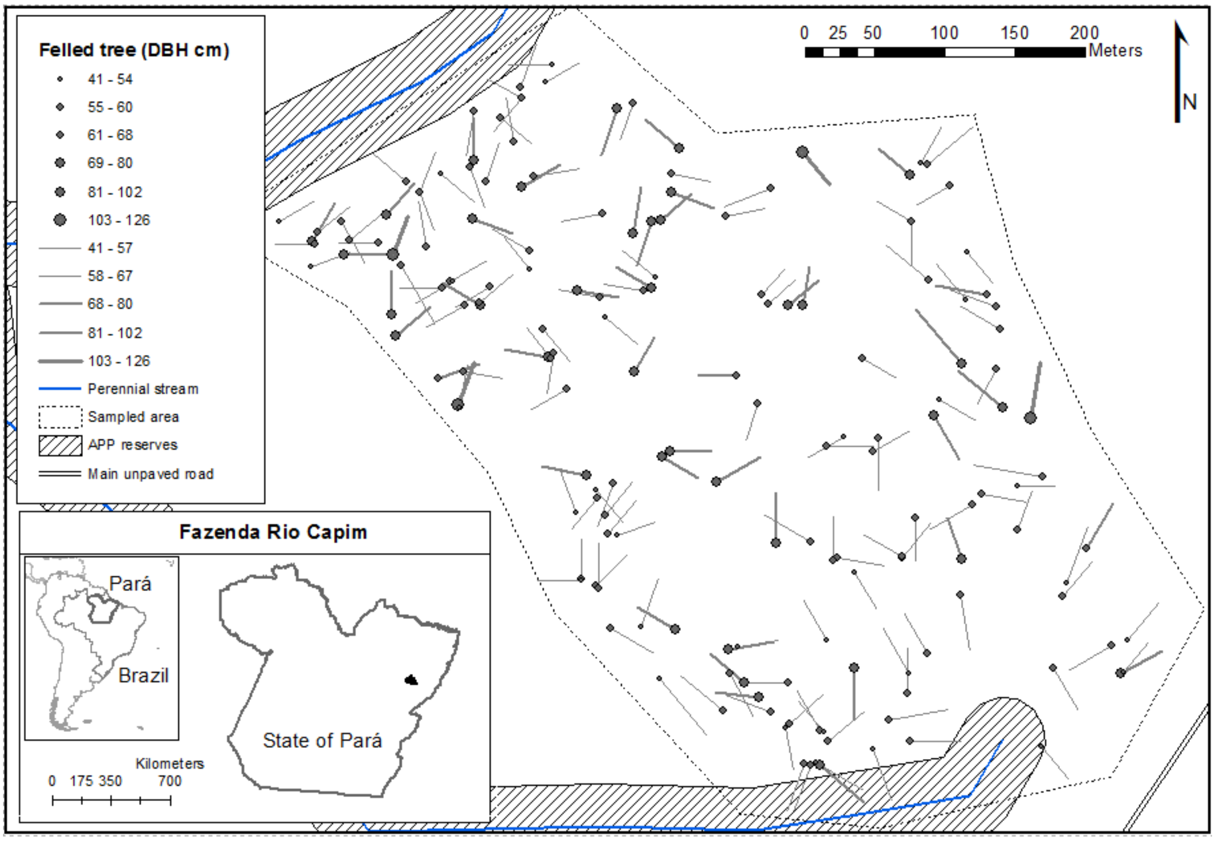

Figure 4.1 Location of the study area (Fazenda Rio Capim) within the eastern Amazonian state of Pará, Brazil, showing the spatial distribution and angular direction of tree-falls of a sample of harvested timber trees. Length of boles in logged trees (straight lines) and tree girths (circles) are proportional in size to actual measurements of the length (m) and DBH (cm) of tree trunks. The extensive network of skid trails generated by bulldozers are not shown. Hatched areas show nominally protected riparian forest strips along perennial forest streams, which were spared from logging.

4.3.4 Residual damage to the stand

All trees within the boundaries of each gap with DBH equal or greater than 10cm were recorded according to their damage category (ranked from 0 to 6) and species vernacular name. Physical damage to individual stems was assigned to moderate damaged (classes 1 – 3) if they were likely to survive, or severe damage (classes 4 – 6, Fig. 4.2) if they were either dead or dying, herein referred to as committed to mortality or committed necromass (Sist & Ferreira 2007). Using these data, we calculated the overall basal area (m^2) of both damaged and committed to dying trees per logging gap. All damaged stems like the harvested tree stumps were identified *in situ* to the level of species by a highly experienced tree parataxonomist employed by CIKEL, who had worked in the study area for over 30 years. We then converted vernacular names into their corresponding Latin nomenclature based on a comprehensive checklist of timber species of

Central and Eastern Amazonia compiled from multiple sources (Silva et al. 1977; Parrotta et al. 1995; Ribeiro 1999; Lorenzi & Flora 1998; Lorenzi 2002; Lorenzi 2008). We were able to match vernacular names to Latin nomenclature to 95% of all damaged trees, which represented 124 species, 92 genera, and 39 families.

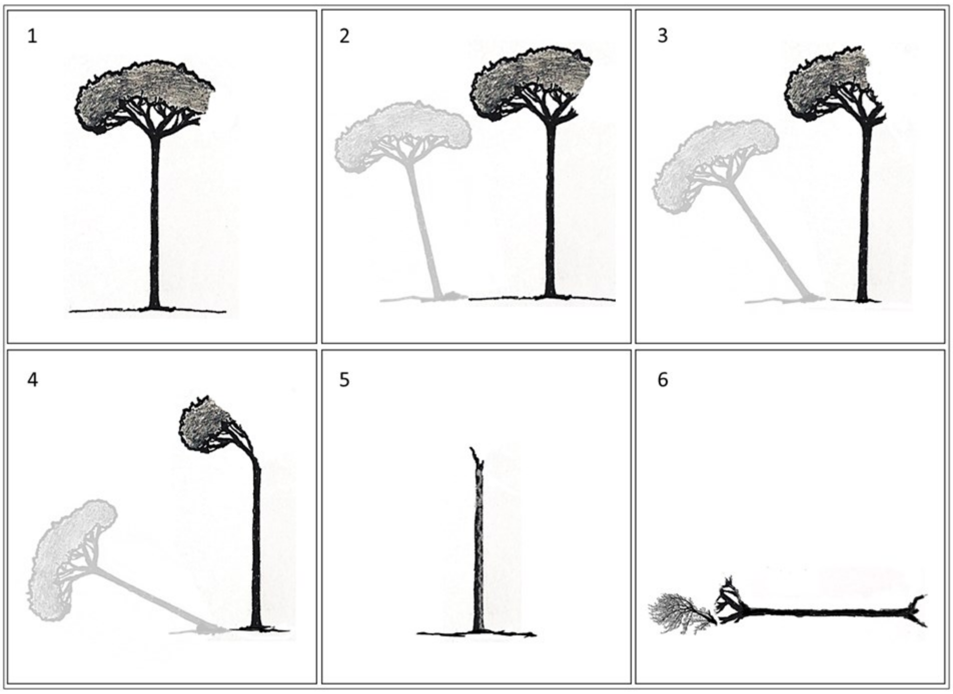

Figure 4.2 Schematic representation of damaged trees as a result of logging activities, classified into six categories of increasing damage severity. Damage categories 4, 5 and 6 were lethally damaged and are therefore defined as either dead or committed to mortality. Trees affected by low to moderate damage (categories 1 – 3) are defined as post-logging survivors. Physical damage categories are ranked according to damage severity as following: (1) few branches removed or small areas of the bark removed; (2) less than one third of the crown removed or angular tree inclination following impact lower than 20°; (3) approximately half of the crown removed or tree inclination between 20° and 45°; (4) over two thirds of the crown removed, or tree inclination greater than 45°, or large areas of the bark beyond the cambium layer damaged; (5) Broken-tops: standing boles without any of the crown; (6) Down wood as a result of tree-felling impact or trees that had been completely uprooted.

To summarise the intensity of damage incurred to the residual stand within logging gaps as a result of timber extraction, two aggregate damage variables were constructed. First, a simple damage score was calculated for each logging gap by assigning individually damaged trees a score equivalent to their level of damage and then aggregating these scores at the level of gaps (i.e. $[\sum 1] + [\sum 2] + [\sum 3] + [\sum 4] + [\sum 5] + [\sum 6]$). Second, we calculated a weighted damage score, defined as the sum of basal areas (BA) of damaged trees per gap $\sum$ m^2 by each damage class weighted by their level of damage from 1 to 6 (i.e. $[\sum m^2$ damage 1

• 1] + [$\sum$m^2 damage 2 • 2]…). Damage class 0 refers to down wood from natural tree and branch falls, so was excluded from the analysis.

4.3.5 Understorey density within skid trails

Using a transect line running 2.5 km into unlogged primary forest within the Fazenda Rio Capim landholding (approximately 0.6km east of Rio Capim river and 5km south of the centroid of all logging gaps), we sampled the understorey vegetation density 1.5 km into the transect line to reduce any road or edge effects. At every 60-m interval and within a perpendicular distance of 10m from the transect line, lateral understorey photographs were taken in all four compass directions (N, E, S, W) using a double-sided white sheet (150cm x 190cm) placed at a distance of 3m from the observer and 0.75m above ground. This was carried out both to the left and right of the transect line at 50 locations, totalling 200 photographs. Furthermore, along the adjacent skid trails of all sampled logging gaps, photographs were also taken of each side of the white sheet. All understorey photographs were taken using a Canon PowerShot SX230 HS. Understorey shrub density was defined as the percentage of black pixels in bichromatic photographs, and fractal dimension was used as a proxy of understorey vegetation complexity (Marsden 2002) processed using the SideLook 1.1.01 software (Zehm et al. 2003). Photographs containing large swathes of high-contrast shadows considered as vegetation black pixels were discarded from analyses. Widths of primary and secondary roads, and skid trails adjacent to each logging gap were measured in situ every 10m using a 50-m tape.

4.3.6 Quantitative review on collateral damage

We reviewed all available formal literature to understand the extent to which selective logging impact studies in tropical forests either qualitatively or quantitatively describe physical damage to the residual stand. Articles were reviewed to extract the following information: (1) Geographic location and coordinates (if available); (2) Logging intensity [as numerical (stems ha^{-1}), basal area (m^2 ha^{-1}) or volumetric damage (m^3 ha^{-1})]; (3) Percentage-based metrics as a proportion of damaged: area (m^2); no. of stems; basal area (m^2) or aboveground biomass (AGB, carbon, Mg ha^{-1}) to intact forest/sampled area/concession size/landholding size; (4) If damage severity was taken into consideration; (5) Area-based metrics of damage to the residual stand (no. of stems ha^{-1}; m^2 ha^{-1}; m^3 ha^{-1}; Mg ha^{-1}); (6) Metrics of residual damage based on felled trees (no. of stems ha^{-1}; m^2 ha^{-1}; m^3 ha^{-1}; Mg ha^{-1}); (7) If canopy fracture was measured (e.g. using LiDar, hemispherical photographs or any other ground-based sampling technique); and (8) Percentage, area, and felled tree based ratios of logging intensity to damage. These values were calculated if they were not readily available in the text, but could be deduced from other information provided in the article or by the author(s). Whenever information was provided on damage severity, we calculated the damage ratio of harvested trees to those stems which had been likely committed to mortality as described above (Fig. 4.2) or described as 'severely damaged'.

We define a third broad category of damage as percentage-based estimates. All area-based methods calculated percentage-based damage density (at least m^2 ha^{-1}) but not all percentage-based studies provided area-based collateral damage information. For instance, percentage-based only studies also included those citing the proportion of damage reported in other studies in the same study region or percentage of damaged stems from an assumed undisturbed forest baseline.

4.3.7 Data analysis

To calculate our total areas sampled (in 2012 and 2013, respectively), automated minimum convex polygons joining all our GPS data points (harvested stumps, gap centroids and skid trail start/ends) were created in ArcGIS 10.2.2. Species-specific wood density or wood specific gravity (WSG) measurements (g cm^{-3}) were compiled from a variety of published compilations from the State of Pará Association of Wood Exporters (AIMEX, Associação das Indústrias Exportadoras de Madeiras do Estado do Pará) and the global wood density database (AIMEX 2013; Zanne et al. 2009). We prioritised species-specific WSG data (83%), but if these were unavailable we used the corresponding regional-scale genus average based on data from anywhere in the Amazon basin (12%). For unknown species (5%) we used the mean WSG across taxa found in our data.

All further data analyses were conducted using R version 3.3.1. We tested whether the size of damaged trees (DBH) committed to mortality differed significantly from those of logged trees and natural down wood using two-sample Kolmogorov-Smirnov tests (K-S test). Data normality was tested using Shapiro-Wilk tests. Because understorey vegetation complexity and understorey density were highly correlated (Spearman's rho = 0.8, P = <0.001, N=174), we considered only understorey density in any further analyses.

We ran a Generalized Linear Model (GLM) with a binomial error distribution and a logit link function to examine the effects of tree size (DBH) and genus identity on the probability of damaged stems either surviving or dying (including stems that were both dead and committed to mortality). Because variation in WSG is already represented at the genus level, we excluded WSG from analyses. We first filtered genera so that only those with counts >10 trees (across all 137 logging gaps) were included and genus was found to be not significant. We then used a Generalized Linear Mixed Model (GLMM) with a binomial error distribution and a logit link function to examine the effects of DBH and wood density on the probability of damaged stems surviving or dying, after examining the correlation between these fixed effects (Pearson's r = 0.07, P = <0.001, N=3256). To account for any discrepancies in sampling effort and ensure a balanced design, we randomly selected a subset of stems committed to mortality (351 out of 2905) to match the sampling effort across all trees likely to survive. These data were then used to construct a global model using the 'lme4' package (Bates et al. 2014). Assuming a nested sampling design (damaged trees within logging gaps) we specified logging gap as a random factor and fitted the GLMM using the Laplace approximation inference method (Bolker et al. 2009). We tested the variance inflation factor (VIF) within our global model, with all VIF values ≈1.0, or below our preselected threshold of 3 (Zuur et al. 2010), and overdispersion was not found (Crawley 2002). We subsequently used the 'MuMIn' package (Bartón 2015) to test all possible combinations of variables and ranked them according to the AIC difference between the lowest AIC model and model i (ΔAIC). Where model sets 'best' model had an Akaike weight < 0.9 and ΔAIC < 2, model averaging was used to estimate coefficients (Burnham & Anderson 2002).

4.4 Results

Individual logging gap sizes ranged from 157.1 to 4,849.8 m^2 (mean ± SD, 891.1 ± 805.3 m^2, N = 137). We sampled an estimated area of ~14 ha (137,493 m^2) in 2012 and ~20 ha (199,333 m^2) in 2013. This resulted in 35.9% (12.2/34 ha) of combined ground damage (i.e. aggregated areas of all logging gaps) within our total sampling area of logged forest. A total of 248 trees belonging to 55 species, 39 genera and 17 families were extracted from these gaps. The number of trees logged per gap ranged from 1 to 6 (1.8 ± 1.2 stumps) or 0.2 to 2.6 m^2 in basal area (0.6 ± 0.5 m^2). An additional 347 trees of down wood from natural tree-falls were found across the 137 gaps, with a basal area ranging from 0 to 2.5 m^2 per gap (0.3 ± 0.3 m^2) amounting to a total basal area of 33.7 m^2 across all gaps. In total 2,905 stems were defined as dead or severely damaged to be deemed committed to mortality (Fig. 4.2). Damage was recorded across 3,256 individual stems, with an aggregate basal area of 128.5 m^2.

Aggregate volumetric offtake from the 137 gaps was 793 m^3, ranging between 0.7 and 12.0 m^3 per harvested tree (3.2 ± 1.7 m^3, N = 248). The mean size of damaged trees committed to mortality, which ranged from 7.5 to 173.4 DBH (19.4 ± 11.3 cm N = 2,905), was significantly smaller than both the mean size of harvested trees (65.5 ± 14.3 cm, range = 40.7 to 125.7 cm, N = 248; K-S test Z = 0.97, P = <0.001) and natural down wood which ranged between 10.0 and 173.0 cm (29.4 ± 19.2 cm, N = 347, K-S test Z = 0.33, P = <0.001) (Fig. 4.3). Damage scores were highly correlated with both weighted damage score per logging gap (Pearson's r = 0.9, P = <0.001), and the number of damaged stems committed to mortality (r = 0.9, P = <0.001). We therefore use the number of damaged stems in all further analyses.

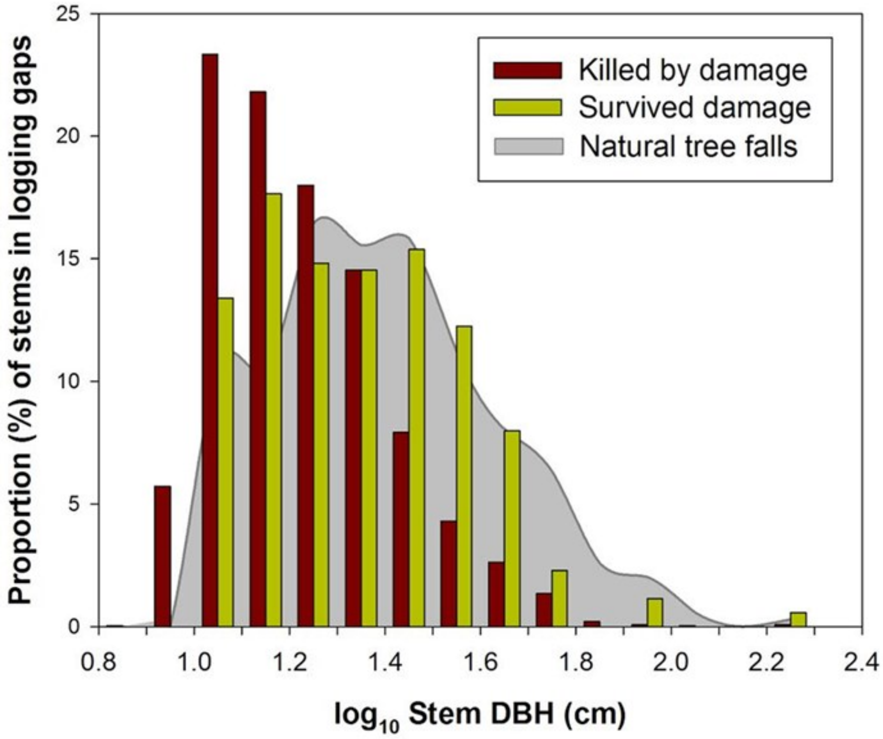

Figure 4.3 Diameter at breast height (DBH) distribution of either damaged but surviving stems (green bars) or damaged stems that were either dead or committed to mortality (red bars). This is compared against the DBH distribution of natural dead wood resulting from natural tree-falls (background shaded area) inventoried within 137 logging gaps.

The top ranking GLMM model explaining tree survivorship within logging gaps had an Akaike weight of 0.65 and contained only DBH as a predictor. The alternative model was our full model (containing both predictors DBH and wood density) and had an Akaike weight of 0.35 (ΔAIC 1.2, cumulative weight w_i =1). Our analyses suggest that DBH is the main determinant of stem survival probability (β = 6.216, P = <0.001), whereas WSG was not significant (β = 0.900, P = 0.368).

Within logging gaps, aggregate basal area and volume of harvested trees explained 53% and 51% of the variation in the overall number of dead stems and those committed to mortality (P = <0.001). The aggregate basal area or volume of harvested trees further explained 47% and 44%, respectively, of the variation in logging gap size as measured on the ground (P = <0.001). These variable also explained 40% and 39%, respectively, of the variation in the proportion of canopy openness (P = <0.001). Larger gap areas clearly resulted in greater canopy openness (Pearson's r = 0.6, P = <0.001) but gap area explained only 35% of the variation in canopy openness (P = <0.001).

Across the aggregate area of 122,083 m^2 of our logging gaps, the total loss in basal area from the residual stand included 103.1 m^2 from severely damaged stems in addition to 87.4 m^2 from harvested trees. We estimated a ratio of 11.7 stems ≥ 10 cm DBH committed to mortality for every harvested tree (Fig. 4.4a, 4b); and 1.5 m^2 of basal area committed to mortality for every 1 m^2 of harvested trees (Fig. 4.4c). Logging gap area was positively correlated with the number of stems committed to mortality (Pearson's r = 0.7, P = <0.001) and explained 56% of the variaiton (P = <0.001). The number of stumps per gap was positively correlated with both logging gap area (Pearson's r =0.6, P=<0.001) and the number of stems committed to mortality (Pearson's r = 0.7, P=<0.001, see Fig. S4.2). The number of stumps in each gap explained 38% of the variation in gap area (P = <0.001) and 48% of the variation in the number of stems committed to mortality (P = <0.001).

The number of damaged stems committed to mortality was positively correlated with species richness committed to necromass (Pearson's r = 0.4, P = <0.001), but only explained 12% of the variation (P = <0.001, Fig. S4.3).

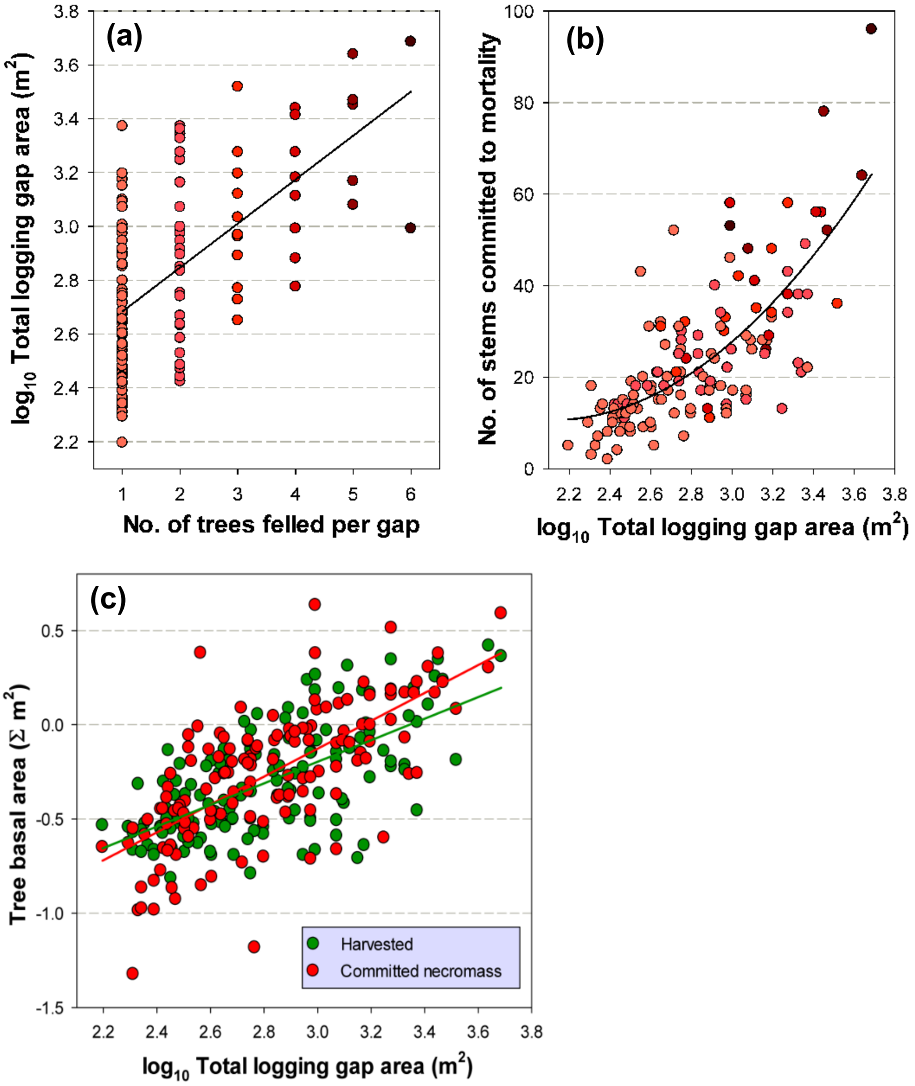

Figure 4.4 (a) The total number of trees felled per logging gap and total gap area ($\log_{10}$ m^2, N = 137). **(b)** The total number of damaged stems committed to mortality and their corresponding logging gap areas ($\log_{10}$ m^2, N = 137). Data points (a-b) are colour-coded (from light to dark red) according to the number of felled trees per logging gap. Data suggests that for every harvested tree a ratio of 11.7 damaged stems are committed to mortality. **(c)** Logging gap area (m^2) and the aggregate basal area (∑ m^2) of harvested trees (Harvested, green circles) and aggregate basal area (∑ m^2) of damaged stems as committed to mortality (Committed necromass, red circled). For every 1m^2 of timber harvested, ~1.5m^2 of basal area was lost as collateral damage. Coloured lines represent linear fits and the crossover suggests that aggregate basal area of the committed necromass tends to exceed that of harvested trees in logging gaps larger than 500m^2.

Mean density of vegetation in regenerating skid trails ranged from 0.0 to 96.9 (14.0 ± 16.0%, *N* = 95 data points) in logged areas and was significantly lower than that of primary forest which ranged from 10.9 to 93.2% (41.4 ± 15.6%, *N* = 79, K-S test, Z = 0.72, P = <0.001; Fig. 4.5). Skid trail width (6.1 ± 2.3 m, *N*=195) was negatively correlated with understorey density within them (Pearson's r = -0.25, P = <0.001) but explained only 5% of the variation in understorey density (P = 0.02).

Self-reported data from CIKEL indicate that on average 4.8% of the APUs are cleared for skid trails (mean density of 117m per ha); 1.5% for primary and secondary logging roads combined (mean density of 0.26m per ha) and 0.67% for logging decks (weighted mean deck size=749.52 m^2, N = 227). Data was sampled to estimate the overall extent of damage as part of the obligatory post-logging report to the State Environmental Secretariat of Pará (SEMA). We expect that these figures are somewhat underestimated. CIKEL reported a mean skid trail width of 3.63 m, half of our mean width from measurements in the same APU.

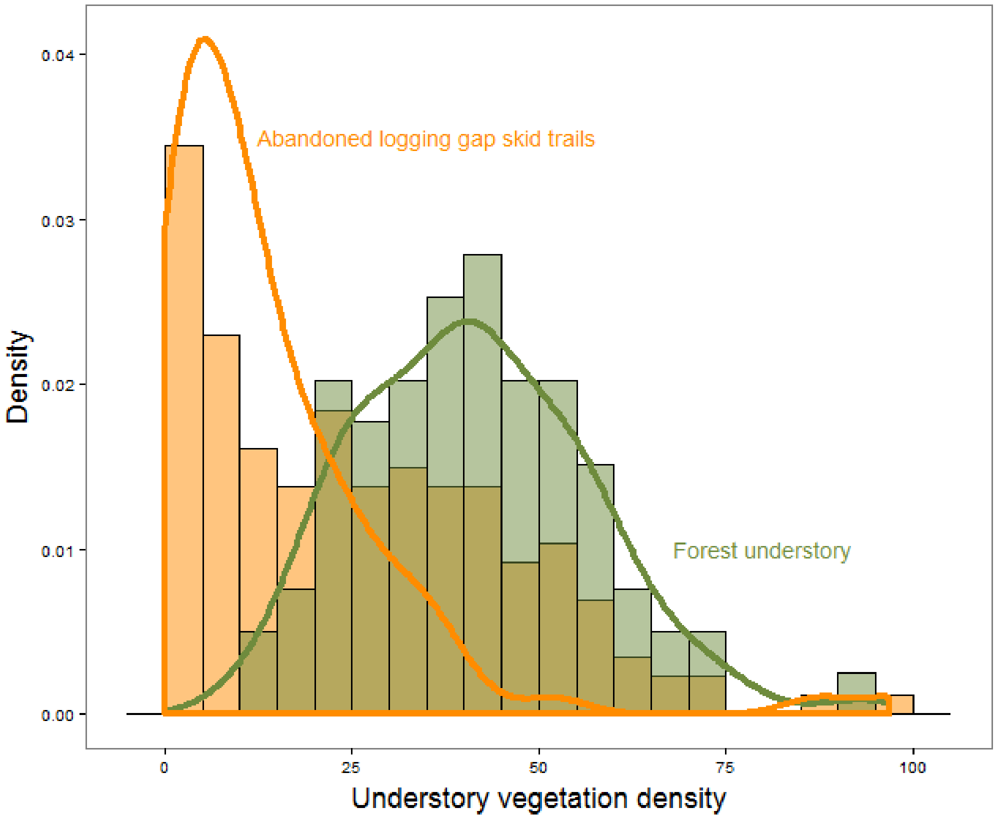

Figure 4.5 The relative proportion (density) of sampled points in both unlogged primary forest and abandoned and regenerating skid trails and understory vegetation (%). Orange and green bars (and density kernels) represent data points sampled within skid trails and in the understory of unlogged primary forest, respectively.

4.5 Discussion

For every harvested tree ~12 stems are committed to mortality and 492.1 m^2 of the forest stand is cleared. Over a third (35.9%) of our total sampling area of logged forest was cleared by felled-trees alone (i.e. aggregated areas of all logging gaps). Our conservative estimates suggest that to fully understand impacts and effects of tropical selective logging, collateral damage cannot be ignored. Most of the disturbance generated by logging operations results from collateral damage to the residual stand, yet this is scarcely measured in logging studies in tropical forests (Fig. 4.6). Our review of 73 studies on faunal impacts of selective logging in tropical forests shows that 90% (N = 73) of them failed to quantify or even mention collateral damage. The vast majority of studies that quantified damage based on felled trees focused on different features of selective logging per se (this study included) or compared conventional logging (CL) versus RIL (71%, N=17, Appendix 3). Only 4 studies from our original set of 73 impact studies provided empirical felled-tree based ratios, i.e. the number of stems damaged per harvested tree (Fig. 4.7). Moreover, this figure is likely underestimated because our percentage-based category also included papers that referred to or extrapolated damage intensities from other studies to their own study areas. We thus presume that a similar proportion of other ecological studies on selective logging impacts also fail to properly quantify collateral damage.

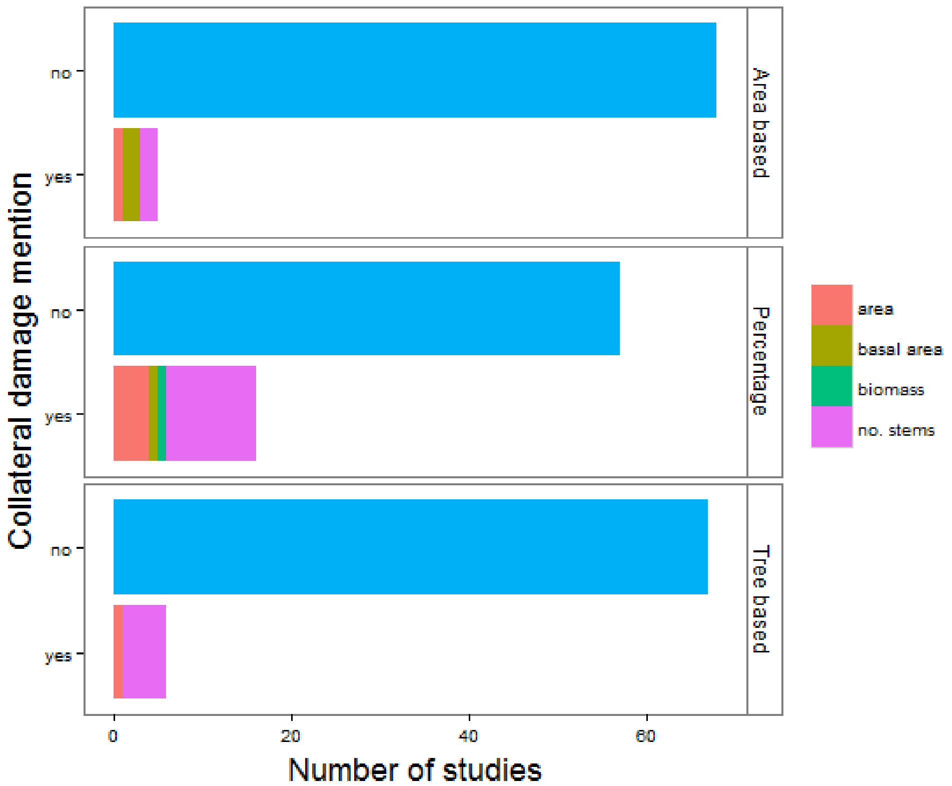

Figure 4.6 The proportion of studies on faunal responses to selective logging in tropical forests that at least mention or attempt to quantify collateral damage to the residual stand. The different units used to measure collateral damage [area (m^2); number of trees; tree basal area (m^2); volume (m^3); aboveground biomass (AGB) or carbon (Mg ha^{-1}) and their proportional use in the literature. See Appendix 2 for references.

Given the rapid forest regeneration in gaps following logging activities, we acknowledge that quantifying damage over time becomes increasingly difficult (Cannon et al. 1994). We argue, however, that when assessing and interpreting short-term ecological impacts of selective logging, relying solely on proxies such as harvest intensity can be deceptive. Whenever possible, it is imperative that any observable impacts on forest structure, forest composition or forest wildlife are appropriately linked to quantitative measures of damage to the residual stand. We find that the number of felled trees and the aggregate volume of harvested trees explained 48% and 51%, respectively, of the variation in the number of damaged stems committed to necromass ($P = <0.001$). Thus we expect that volumetric harvest intensities (m^3 ha^{-1}, Fig. 4.7b) are likely better predictors of collateral damage than harvest intensities expressed simply as the number of trees felled per ha (Fig. 4.7a).

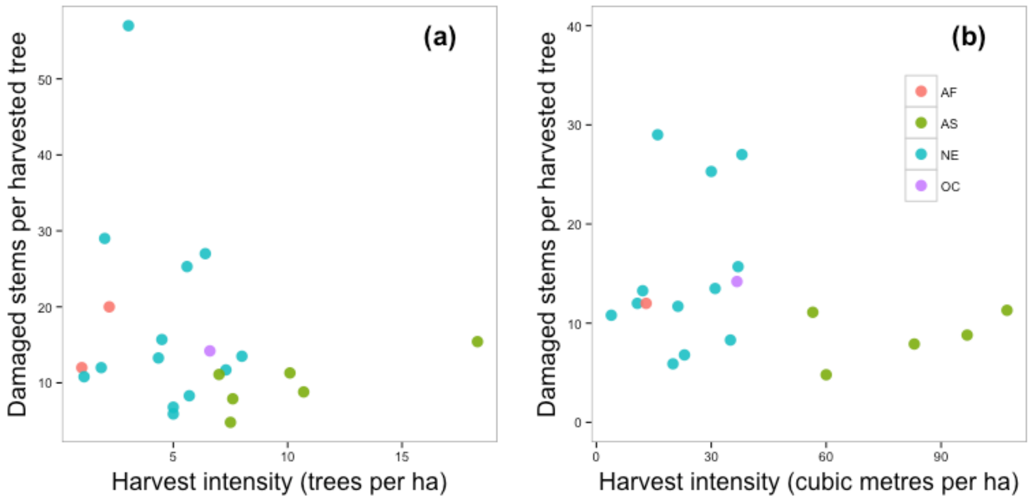

Figure 4.7 The ratio in the number of damaged stems per harvested tree across different tropical forest studies in all major land masses. Units to measure harvest intensity include (a) number of trees harvested per ha and (b) the mean cubic volume of harvested timber (m^3 ha^{-1}). Geographic locations of studies are represented as following; AF, Africa; AS, Asia; NE, Neotropics; and OS, Oceania. See Appendix 2 for data sources.

Our data also suggest that only about 50% or less of the variation in damage (expressed either as the number of stems committed to mortality, gap size or canopy openness) can be explained by harvest intensity alone (either as basal area or volumetric offtake). The rest of the source of variation remains unknown. Overall basal area damaged was found not to be proportional to the timber volume harvested in the same region of Eastern Amazonia (Paragominas, Pará) as this study (Veríssimo et al. 1992). Selectively logged forests vary widely in their natural and anthropogenic disturbance dynamics, including those caused by natural tree-falls, large patches of blowdowns, El Niño induced wildfires, habitat fragmentation and hunting (Siegert et al. 2001; Peres 2001; Espírito-Santo et al. 2014). The history and intensities of these anthropogenic pressures can vary even at regional scales (Berenguer et al. 2014). As such, logging disturbance may often scale to logging intensity but this relationship is unlikely to be linear given the many confounding factors (Panfil & Gullison 1998, but see Picard et al. 2012).

There are several methods used to assess percentages of ground disturbance, consisting of any combination of damage; in primary or secondary roads; logging decks; skid trails and canopy gaps opened by target felled trees. The apportioned stem or ground area damaged can be reported in relation to baselines, control areas, sampled areas or total property areas, making comparability among studies challenging (Picard et al. 2012). In felled-tree methods, collateral damage can be expressed as basal area or the aggregated biomass loss, but we focused on the number of damaged stems because forest biomass and carbon fluxes are largely affected by vegetation regrowth and the former units make no distinction at the level of individual stems or species loss.

Most damage within a logging operation occurs at the point of impact, i.e. logging gaps (Uhl et al. 1991; Feldpausch et al. 2005) and gap size is proportional to tree-fall size (Tyrrell & Crow 1994). Therefore, studies assessing logging damage through remote sensing of logging decks alone will not provide an accurate assessment of damage (Asner et al. 2004). Literature reviews of ground disturbance suggest that RIL may cause less damage than CL (Fig. 4 of Pereira et al. 2002). However, at high harvest intensities, or when exceptionally large trees fall, the proportion of ground disturbance resulting from both management techniques is similar (Fig. 5 of Feldpausch et al. 2005). Notwithstanding skidder damage, RIL does little to reduce per capita damage from felled trees compared to conventional logging, in particular at harvest intensities exceeding 8 trees per ha (Sist et al. 1998). In Borneo, the percentage of stems damaged by skidder machinery was positively correlated with harvest intensity (felled trees ha^{-1}) in RIL but not in CL, and RIL caused less damage than CL (Sist et al. 2003). However when only the damage caused by felled trees was considered (logging gaps), there was no obvious benefit of either management approach, and harvest intensities were uncorrelated with levels of damage (Fig. 2 of Sist et al. 2003). However some studies have found average gap sizes in CL larger than those in RIL (Hendrison 1990; Holdsworth & Uhl 1997), but this is likely due to excessive manoeuvring by skidders around felled trees in CL.

Mean logging gap sizes throughout the literature range from 131 to 1,022 m^2 (Hendrison 1990; Uhl et al. 1991; Johns et al. 1996; Parren & Bongers 2001; Jackson et al. 2002), yet most logging impact studies are restricted to treefall gaps where a single target tree is felled. Our mean gap size of 891 m^2 is higher than most studies because we measured *in situ* actual canopy gaps generated by *n* felled trees. If we consider only single-felled tree gaps, our mean gap size is reduced to 563 m^2. Similarly, Jackson et al. (2002) found a smaller mean gap size for single-felled trees (591 m^2) than gaps generated by multiple felled trees (1,022 m^2). Damage created by neighbouring individual target trees does not accumulate linearly (Parren & Bongers 2001) partly because impact zones of nearby felled trees can partially overlap and damage by shared skid trails to remove *n* trees is usually lower than if they had been logged at different sites (Picard et al. 2012). Accordingly, we find that the number of damaged stems committed to mortality (y) as a function of the number of stumps or felled trees (x) increases at a rate of $y = 7.4 + 9 \bullet x$ ($R^2 = 0.49$, P = <0.001, Fig S4.2).

Our results are consistent with other studies (Pinard & Putz 1996; Bertault & Sist 1997; Panfil & Gullison 1998; Alder & Silva 2000; Parren & Bongers 2001) in terms of increasingly larger stems succumbing to lower mortality rates (Fig. 4.3). At the CIKEL landholding, the break-even size threshold between trees either dying or surviving a treefall impact was approximately 22.4 cm in DBH. Lethal damage was, however, indiscriminate across stems in the smallest size classes and all species were equally affected.

4.5.1 Variation in damaged tree to felled tree ratios

Our literature review shows that the number of severely damaged stems to every harvested tree varies between 4.8 and 57 across the humid tropics (mean ± SD, 15.5 ± 11.3, Fig. 4.7). Our observed mean ratio of dead or dying stems to felled trees (11.7) is within the range of other studies even though our figure is likely an underestimation of severe damage because we only measured damaged stems within logging impact zones and ignored damage from logging infrastructure such as logging roads, skid trails and loading decks. Moreover, some additional trees may have been missed underneath the coarse woody debris of felled tree crowns (Parren & Bongers 2001) but we feel our sampling effort was robust enough to keep these to a minimum. Conversely, a minority of the stems killed during the process of opening skid trails or bulldozer manoeuvers during skidding may have been accounted for if they fell within or were piled into areas defined as a gap.

Exceptionally large ratios such as 1:57 logged to damaged trees at relatively low harvest intensities of 3 trees ha^{-1} in French Guiana (Thiollay 1993) may be explained by smaller size thresholds in measuring damaged stems ≥ 5cm DBH whereas this and most other studies use a damage tree size cut-off of ≥ 10cm DBH. Moreover, Thiollay (1993) failed to distinguish between damage severities so the number of trees "*killed or damaged (uprooted, bent down, scarred, broken)"* were added to all damaged stems found along skid trails opened to drag roundlogs to the nearest part of the logging track network. Likewise, the low ratios found by Sist et al. (2003) in both RIL (4.8) and CL treatments (7.9) in Indonesian Borneo can be attributed to their use of a high size threshold of ≥ 20cm DBH to measure damage. Standardizing damage criteria, including minimum tree size cut-offs and spatial distribution of damaged stems, is therefore crucial to enhance comparability of logging disturbance across studies.

Because selective logging (both RIL and CL) typically removes the largest trees in the stand, a significant amount of forest basal area is lost through harvested trees. However, we found that this ratio almost doubled once a conservative number of stems damaged by the logging operation is considered (i.e. 1.5 m^2 lost in damage to every 1m^2 lost in harvested trees). Similar volumetric damage ratios have been found elsewhere in the state of Pará (e.g. 1.6 and 1.9 m^3: Uhl et al. 1991; Veríssimo et al. 1992). Other studies in Australia and Belize, have found similar ratios, ranging between 1 and 1.8 (Crome et al. 1996; Lewis 2001). White (1994) observed an exceptionally high ratio in basal area loss of 11.2 in Gabon but they included both damaged stems and lianas (≥ 10cm DBH).

Heavy skidder machinery can disturb an area of up to 70% of felled tree gaps, but incur less damage to large trees (20%) because these can be avoided by the operator (Feldpausch et al. 2005). Heavily compacted soils of skid trails become exposed which leads to erosion and impedes regeneration, even 10 years post logging (Thiollay 1997). We found significantly lower (25%) understorey vegetation density in skid trails than that of primary forest (Fig. 4.5). CIKEL's estimate of 4.8% of the APU area opened due to skidders is in line with the ~5% recorded in the literature (Pereira et al. 2002; Feldpausch et al. 2005).

Selective logging impacts are not limited to structural damage, floristic simplification of the residual forest stand has also been observed. For example, 31% of all saplings inventoried in logged stands elsewhere in Eastern Amazonia belonged to *Jacaranda copaia*, a fast-growing pioneer species which comprised only less than 1% of saplings in unlogged forest stands (Gerwing 2002). Wood density and growth rates are inversely related to tree mortality in Amazonian forests, where timber trees are often emergent species with high wood density, and low natural rates of mortality and recruitment (Nascimento et al. 2005). In regenerating selectively logged tropical forests of China, species richness recovers faster than the background pattern of species composition in unlogged forests (Xu et al. 2015). After half a century of regeneration, community similarity between selectively logged and old growth forests decreased and basal area did not recover, even in the absence of further anthropogenic impacts (Xu et al. 2015). Similarly, 45 years of regeneration following selective logging in Uganda has shown that rates of stem, species composition and functional traits do not decline through time in logged forests, suggesting that successional models that assume recovery to pre-disturbance community structures may be inadequate (Osazuwa-Peters et al. 2015). A long-term study at the Tapajós National Forest (Pará) that monitored residual stands prior to and after logging over 30 years indicate that changes in species composition among commercially-valuable timber trees included average losses of 18 old-growth canopy species per treatment area, but no overall decreases in tree species diversity (A.L. de Avila, personal communication).

Gerwing (2002) also describes a loss of 25 to 28 $m^3\ ha^{-1}$ of commercially valuable timber species through collateral damage. Moreover, Veríssimo et al. (1992) suggests that 85% of the volume of damaged species following selective logging either had commercial value (49% as sawn timber) or other local wood applications such as house construction. Because lethal damage is indiscriminate and all species in smaller sizes classes are equally affected, species–area relationships are partially responsible for the high significance of the correlation between stems committed to mortality and species richness committed to necromass ($P = <0.001$, Fig. S4.3). Species respond differently to newly available light conditions and maximum growth rates are often negatively correlated with survival in the most resource-limited environments (Nascimento et al. 2005;

Dent & Burslem 2009). Because we sampled damage only one year post logging composition shifts were likely still in their initial stages, and long-term observations may likely reflect these species-specific trade-off dynamics.

Selective logging increases the amount of coarse woody debris of forests stands, committed necromass (≥10cm DBH) can account for over half of the aboveground biomass (Pfeifer et al. 2015). This additional amount of phytomass loss severely compromises the carbon storage and sequestration capacity of forests stands. Moreover, to meet the East Amazonian export demand for iron-ore, this increased amount in coarse woody debris of selectively logged stands is also frequently removed from the residual stand to become charcoal and used in the smelting of pig-iron (Fearnside 1989). This dead and dying phytomass however comprises a significant pool of nutrients which are slowly released through decomposition (Krankina et al. 1999; Vitousek & Sanford 1986). Necromass also plays an important role in maintaining forest ecosystem functions and interacts with disturbance events like fire (Pyle et al. 2009). Deadwood has also been found to facilitate seedling regeneration through associated fungi (Fukasawa 2012), provides structural habitat for many taxa including birds (Gibbs et al. 1993) and insects across all major orders, in particular coleopterans and dipterans (Grove 2002). Once removed from forest stands, all of the nutrients and habitat structure and ecosystem functions of collateral damage stems are permanently lost from the ecosystem.

4.5.2 Conclusions

These findings indicate that at present, there is still no consensus on how best to sample or report logging collateral damage. Logging intensities and extraction techniques vary within and between tropical forests (Picard et al. 2012) as do stand-level floristic composition (ter Steege et al. 2006). Moreover, because densities of commercially viable species are also widely variable, logging intensities can be patchy even at a landscape scale (Uhl & Vieira 1989). As such, logging intensities alone cannot accurately predict collateral damage.

As logging continues to expand, understanding the value of logged forests to maintaining ecosystem services is urgently needed (Edwards et al. 2014). Our results provide further evidence that unless collateral damage is quantitatively measured it becomes difficult to successfully avoid degradation through REDD+ financing for instance, or for valuable ecological impact studies to effectively assess logging impacts on biodiversity, or guide forest management. Without a common unit for quantifying collateral damage, logging-induced degradation of forest stands remains undermined, underreported and ignored by both the tropical forest conservation community and policy makers.

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4.6 Supporting information

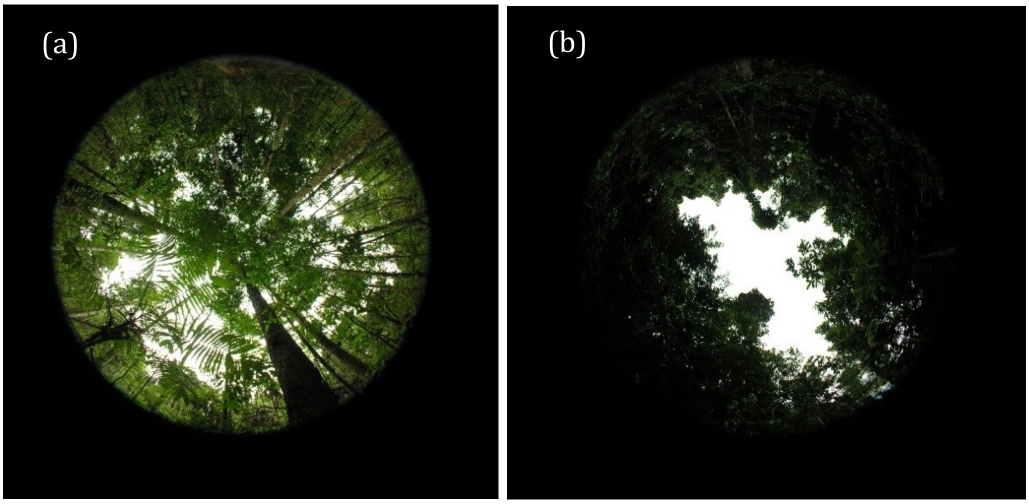

Figure S4.1. Coloured hemispherical photographs of an area of **(a)** primary forest and **(b)** a selective logging gap within a reduced impact logging concession of Eastern Amazonia.

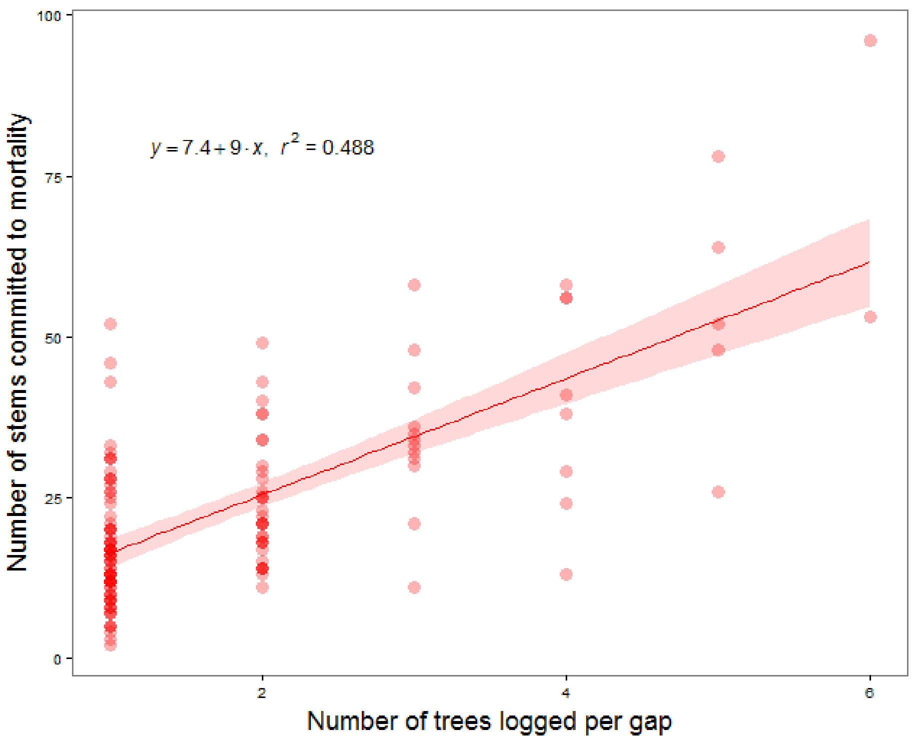

Figure S4.2. The number of trees logged (stumps) per logging gap (N =137) and the number of stems committed to mortality inventoried in each logging gap.

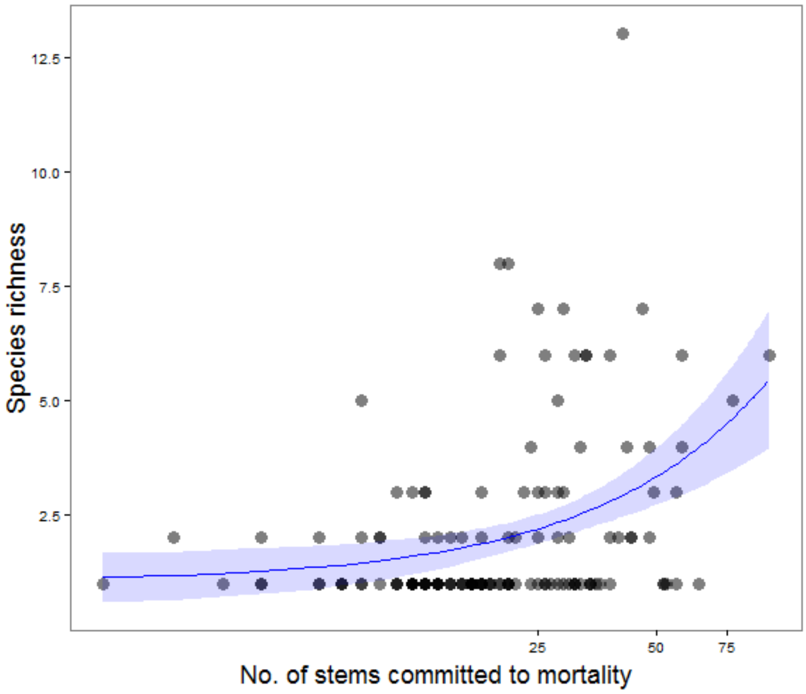

Figure S4.3. Species richness and total number of damaged stems committed to mortality inventoried in 137 logging gaps.

4.7 Appendix 2

Reference list of the tropical selective logging fauna impact studies reviewed and included in Figure 4.6.

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Chapter 5: Carbon value in sparing logging-induced degradation of Amazonian forests rivals their net timber revenues

5.1 Abstract

The spatial extent of tropical forest degradation far exceeds that of tropical deforestation. In Amazonia, forest carbon storage capacity is severely compromised by primary forest degradation. Selective logging, even using best-practice reduced impact logging methods, causes significant and underreported collateral damage to residual forest stands. The biomass loss associated with logging-induced degradation is poorly addressed by both the tropical forest conservation community and climate change researchers. The UNFCCC (United Nations Framework Convention on Climate Change) and the IPCC (Intergovernmental Panel Climate Change) have both highlighted the need to quantify carbon losses resulting from forest degradation, in particular for REDD+ (Reducing Emissions from Deforestation and forest Degradation) mechanisms. However, clear guidelines for measuring carbon emissions from forest degradation, including selective logging operations, are yet to be established. We examine the extent of collateral damage resulting from both felled-tree impacts and infrastructure damage of a large reduced impact logging concession in Eastern Amazonia. Over 11 years, a total of 344,485 trees were harvested, equalling ~1.5 million m^3 of timber (harvest intensity of $31 m^3 ha^{-1}$, or 5.0 and 8.9 trees per ha) across 48,178.9 ha of tropical forest (482 km^2). We estimated that less than 41% of the selectively logged forest has remaining tree cover. Logging gaps comprised the greatest ground disturbance in terms of cleared area (~50%) followed by skid trails (7%), logging roads (2%), and log yards (0.8%). Mean above ground biomass loss was 92.1 Mg ha^{-1} or 51.9 Mg C ha^{-1} in committed emissions (including estimates of both above- and below-ground necromass). We estimated a 2.7 ratio in biomass loss resulting from all logging

damage to harvested roundlogs. Our analyses provide evidence that the benefits accrued from sparing primary forests from logging-induced degradation could rival financial revenues from timber extraction through carbon financing schemes such as REDD+, even if we use a conservative value for carbon stocks (US$ 8 per Mg C) and consider only harvested trees and their associated collateral damage.

5.2 Introduction

Brazil is estimated to hold 836 Mha of intact forest landscapes or 64% of the world's total forested land (Potapov et al. 2008). The vast majority, 72% of all non-plantation forests of Brazil are found in Amazonia (SFB 2013a). Brazil also holds the largest live above and below-ground carbon stocks of any country in the tropics (62 petagrams of Carbon, Pg C), roughly equivalent to all of sub-Saharan Africa (Saatchi et al. 2011). Globally, deforestation has been estimated to be responsible for almost 20% of all anthropogenic greenhouse gas emissions (Gullison et al. 2007).

Due to deforestation and land use changes, including the abandonment of agricultural lands, the vast carbon sink of Amazonia has been estimated to fluctuate between a net source and a net sink of carbon dioxide, with an interannual variability of ± 0.2 Pg C yr^{-1} (Houghton et al. 2000). The Amazonian net carbon balance has been estimated to range between a net loss of 0.48 ± 0.18 Pg C yr^{-1} in an anomalously dry year (2010) and negligible net changes in a wet year (2011: 0.06 ± 0.1 Pg C yr^{-1})(Gatti et al. 2014). The authors highlight that the key driver of net carbon losses in 2010 was the suppression of photosynthesis due to feedback mechanisms between emissions from forest fires and drought, exacerbated by an El Niño episode.

Droughts increase the likelihood of wildfires. Just a single forest fire event triggers an irreversible cascade of both structural and functional impoverishment of Amazonian forest ecosystems including a shift from a closed-canopy forest dominated by old-growth tree species to a more open degraded forest dominated by short-lived pioneers (Barlow & Peres 2008). Large trees account for most of the forest biomass (Clark & Clark 1996), so the delayed post-burn mortality of large trees (DBH ≥ 50cm) alone can induce a 67% loss in live forest biomass some 3 years after a surface fire (Barlow et al. 2003). The magnitude of threats posed by fire are further aggravated through positive feedback mechanisms; fires strengthen the effects of regional droughts, spiralling into secondary fires and further degradation that is facilitated through expanding road networks, particularly in newly accessible logging frontiers (Nepstad et al. 2001).

Across the tropics, logging-induced degradation further exacerbates forest flammability by opening up the canopy, adding to the ground fuel load, and desiccating the understorey (Nepstad et al. 1999; Cochrane 1999; Siegert et al. 2001; Nepstad et al. 2004). Like wildfires, selective logging also leads to structural and functional impoverishment including lower canopy height (Asner et al. 2010), shifts in tree community composition (Phillips et al. 2004), and a severe reduction in forest biomass. The largest trees (DBH ≥ 60cm) removed through selective logging represent less than 10% of total tree density but they store up to >50% of the total stand phytomass (Sist et al. 2014). The extent to which this forest biomass is eroded through logging depends on the baseline stand density, species composition, harvest intensities, extraction methods, and the overall amount of collateral damage involved, including impacts of felled-trees and areas cleared for logging infrastructure. Selective logging has been estimated to reduce tropical forest carbon stocks by 20 to 56% at harvest intensities between 35 and 154 m^3 ha^{-1} (Pinard & Putz 1996; Gerwing 2002; Berry et al. 2010; Putz et al. 2012).

The extent of tropical forest degradation is inherently difficult to measure (Peres et al. 2006). Worldwide estimates suggest that degradation is increasing, but in Brazilian Amazonia this is exacerbated by the weakened Forest Code legislation and other policies including the degazetting and downsizing of protected areas, and renewed investments in infrastructure development projects, including hydroelectric dams, mining and agricultural land conversion (Ferreira et al. 2014; Marques & Peres 2014; Soares-Filho et al. 2014). An area equal to twice the size of northern Borneo's intact forest has been logged (Gaveau et al. 2014), and 60-123% of all deforested areas in the Brazilian Amazon have been degraded in the same regions through logging (Asner et al. 2005). According to the International Tropical Timber Organisation (ITTO), 53% (403 million ha) of the global scale 'permanent tropical forest estate' (public lands officially designated for either production or protection), are currently being logged (Blaser et al. 2011).

There is a high degree of congruence between biodiversity value and carbon stocks in species-rich tropical biomes like Amazonian forests (Strassburg et al. 2010). Compared to deforestation, the climate regulation and carbon storage capacity of selectively logged forests and other forms of anthropogenic forest degradation have been poorly addressed by both civil society and governments (Berenguer et al. 2014). The UNFCCC (United Nations Framework Convention on Climate Change) and the IPCC (Intergovernmental Panel Climate Change) have both highlighted the need to quantify carbon losses resulting from forest degradation for REDD+ mechanisms (Reducing Emissions from Deforestation and forest Degradation), but have yet to establish clear guidelines for measuring carbon emissions from forest degradation, including selective logging operations (IPCC 2006; UNFCCC 2008). As such, a small number of studies have measured biomass and committed emissions from logging and associated collateral damage across tropical forests (Keller et al. 2004; Feldpausch et al. 2005; Mazzei et al. 2010; Pearson et al. 2014).

The notion of promoting and financing Reduced Impact Logging (hereafter, RIL) as a carbon sequestration offset programme had already been suggested some 20 years ago (Putz & Pinard 1993; Putz et al. 2012), well before the onset of REDD+. In 2015, the global value of different carbon pricing mechanisms was worth just under US$50 billion (Kossoy et al. 2015). Brazil is the second most popular country for voluntary offset supply locations [39.5 million metric tonnes of carbon equivalent (M MgC), worth US$233 million; Hamrick & Goldstein 2014], after the USA (136 M MgC). This is largely due to a first-of-its-kind non-market based carbon project commenced in 2013 in order to provide bridging finance until a REDD finance mechanism is agreed upon within the United Nations negotiation process. A multilateral agreement between German development agency KfW, the Norwegian government and the Brazilian state of Acre (REDD+ Early Movers, REM programme hereafter). The REM project is not an offset project as such but provides performance-based payments (US$50 M) for demonstrated emission reductions from avoided deforestation (8 M MgC) in Acre (Hamrick & Goldstein 2014).

Tropical primary forests are irreplaceable for biodiversity conservation (Gibson et al. 2011), yet the carbon market share of REDD+ programs continues to be modest but has increased 500% between 2009 and 2011 (Linacre et al. 2011). If we are to successfully implement REDD+ schemes to avoid tropical forest degradation and biodiversity loss, a clearer understanding of different land-use opportunity costs is urgently needed. Here, we estimate the net revenues accrued from harvested timber over a decade of reduced-impact logging in a vast forest management concession of Eastern Amazonia, where nearly 345,000 harvested timber trees were identified and mapped across a legal logging concession area of 48,179 ha. We also estimate the overall carbon value of all harvested timber trees and the associated necromass resulting from stand-scale collateral damage. Finally, we re-evaluate the potential of REDD+ funding mechanisms in sparing undisturbed tropical primary forests from committed greenhouse gas emissions resulting from logging-induced forest degradation.

5.3 Methods

5.3.1 Study landscape

Sampling was conducted at Fazenda Rio Capim, CIKEL Brasil Verde Ltda group in Eastern Pará state of Brazil (3°32'S, 48°49'W). This landholding of ~140,000 ha encompasses large areas of natural undisturbed (12,000 ha) and RIL-managed upland (terra firme) forests (110,000 ha) in addition to abandoned pastures (18,000 ha) containing tree plantations of fast-growing paricá (*Schizolobium amazonicum*) and eucalyptus (*Eucalyptus spp.*). Unplanned timber extraction and hunting are prohibited across the landholding. Mean annual rainfall is ~1,800mm and the lowland topography is mostly flat with a mean elevation of 20m a.s.l. (Sist & Ferreira, 2007). CIKEL has been harvesting Forest Stewardship Council (FSC) certified timber since 2001. RIL techniques included a minimum cutting diameter of 55cm for all commercially valuable species, a cutting cycle of 35 years in accordance to Brazilian forest management law, and a maximum extraction intensity of 30 m^3 ha^{-1} (Normative Ruling no. 05 of 2006). Most harvested species include *Manilkara huberi, Manilkara paraensis, Hymenaea courbaril* and *Astronuim lecointei*, which account for 35% of all the volume harvested (~1.5 million m^3) across 344,485 trees.

5.3.2 Harvested tree biomass

Data consisted of completed stand-scale spatially explicit inventories of all broadleaf trees ≥ 30cm DBH (diameter at breast height) across 14 Annual Production Units (APUs) exploited between 2000 and 2011. APU is a term used to annually label different parts of a landowner's property that was logged in different years. APU's vary in size and are both temporally and spatially distinct. Note however that APU's 1, 2 and 3 were logged in 2000; 4 and 5 in 2001 and APU 8 was logged over 2004 and 2005. All trees earmarked for harvesting were mapped and coded as *cut*, whereas the remaining tree management categories were coded as either *remaining* and *stock* stems. All trees were identified by a team of parataxonomists, led by one of our field assistants (Chapter 4), a long-term logger and parataxonomist with over 30 years of experience at this site. The corresponding Latin nomenclature of all harvested tree species was converted from locally identified vernacular names based on a comprehensive checklist of timber species of Central and Eastern Amazonia compiled from multiple sources (Silva et al. 1977; Parrotta et al. 1995; Ribeiro 1999; Lorenzi & Flora 1998; Lorenzi 2002; Lorenzi 2008). Only 6% of all species could not be identified but these represented less than 1% of all harvested trees. We calculated above-ground biomass (AGB) of committed necromass from damaged trees ≥10cm DBH using the following equation developed by Chave et al. (2005) for moist forests:

$$ABG = 0.0509 \times wsg \times dbh^2 \times H$$

Harvested tree height may be crucial in determining the magnitude of collateral damage found in logging gaps (Feldpausch et al. 2005). Moreover, adding tree height to the model improved model performance, as tropical moist forests may have a similar diametric structure but vary substantially in canopy height (Chave et al. 2005). We therefore used a clinometer and a 50-m measuring tape along secondary logging roads throughout our selectively logged forest, to measure *in* *situ* a random set of 250 broadleaf tree heights (maximum height range = 11.0 – 77.2). A stand-specific diameter-height allometric relationship (linear regression) between DBH and maximum height (H) was then derived ($R^2 = 0.67$) and then used to estimate the mean total height of all trees damaged on site (Fig. S5.1). Heights were measured in metres and wood specific gravity (WSG) in grams per cubic centimetre. Resulting AGB estimates from the equation are expressed in kilograms. Species-specific WSG data was obtained from the global wood density database (Zanne et al. 2009). If any given species was not found in the database, its mean regional genus-level WSG was used instead. In Amazonian forests, there is high variability of below-ground biomass (BGB); between 10 and 50%, and a mean of 17% (Brown & Lugo 1982). We therefore opted for a conservative estimate of BGB using the equation (1) from Mokany et al. (2006), resulting approximately 13% of our AGB.

5.3.3 Cleared-area estimates

To assess the extent of forest damage along skid trail networks, we used GIS data released by CIKEL on the total length of skid trails per APU and applied a 165% margin of error based on our own replicated field measurements of skid trail width from the same concession (mean ± SD; 6.1 ± 2.3 m, *N*=195, Chapter 4). CIKEL measured total skid trail lengths across two comprehensively sampled areas (75.0 and 88.8 ha, respectively) within two APUs that were selectively logged in 2010 and 2011. Skid trail widths were measured from the edge of bulldozer wheel grooves along the trail (CIKEL, pers. comm.) rather than from undamaged forest edges as we carried out *in situ* in 2012-2013 (Chapter 4). As such, mean skid trail width estimated by CIKEL by combining both primary and secondary skid trails was severely underestimated (4.28 m and 3.63 m for the areas assesses in 2010 and 2011, respectively) compared to our mean of 6.1m, which was 165% wider on average. We applied the mean skid trail density of 11.66 km/ km^2 across the two sampled areas for all APUs for which we had spatially explicit offtake data and multiplied these lengths by our own mean skid trail width to estimate the total area cleared along skid trails. Brazilian legislation (Forest Code 2012) requires riparian corridors, hilltops, steep slopes and stream headwaters to be set aside as Areas of Permanent Preservation (APPs, Forest Code 2012). All logging concession sizes within APUs therefore exclude APP areas, thereby ensuring that the density of skid trails and other sources of damage were not overestimated.

CIKEL provided road density data for four APUs, on the basis of which we calculated a mean road density (2.89 km per km^2), which was then applied to other APUs lacking road density data. As above, we similarly applied a correction factor based on our own field measurements to CIKEL's self-reported mean primary and secondary logging road width of 5.4 m. To estimate the extent of deforested areas within log yards each year, we used a weighted mean of CIKEL's reported aggregate areas for 185 and 42 log yards, which were individually mapped and measured in 2010 and 2011 respectively, resulting in a mean log yard size of 749.52 m^2. We also had data on the total number of log yards opened across four APUs, resulting in a mean density of 10.84 log yards per km^2, which was then extrapolated to the remaining APUs with missing data to estimate their total numbers and aggregate areas of log yards.

To estimate the total logging gap area opened within each impact zone of felled-trees, we applied our own data on the basal area of harvested trees (BA_H, m^2) and the ellipsoid size (m^2) of 137 logging gaps measured in situ using a 50-m tape (see Chapter 4). This was then used to predict the area of logging gaps across all APUs for which logging gap size data were missing (Fig. S5.2), based on the following equation:

$$Gap\ size\ (m^2) = 196\ + (1090 \bullet BA_H)$$

Finally, once all above logging infrastructure and logging gap areas had been accounted for, all remaining areas (ha) within each APU was assumed to be undamaged primary forest.

5.3.4 Committed necromass from collateral damage

Data from chapter 4 included detailed sampling of 137 logging gaps where 248 trees were harvested. Aggregate damage in these felled-tree impact zones (logging gaps) comprised 2,905 stems that were individually measured and identified to species, and deemed so severely damaged they were defined as committed to mortality, eventually adding to the forest understorey necromass. These data were used to construct the following equation between aggregate harvested tree basal area per gap and its associated community-wide collateral damage, where AGB_{CD} is the AGB (kg) of the aggregate collateral damage per gap, and BA_H (m^2) is the total basal area of harvested trees (mean=0.6 ± 0.5 m^2, range=0.2 to 2.6 m^2; Fig. 5.S3).

$$AGB_{CD} = 10^\wedge(3.9 + \left(0.76 \times log_{10}(BA_H)\right))$$

This equation was applied to all harvested trees for each APU exploited between 2000 and 2014.

The total AGB of logging-induced committed necromass generated in areas cleared for infrastructure AGB_I (rather than from direct tree felling) was estimated using the following equation between AGB_{CD} and ellipsoid size of logging gaps (Chapter 4). Here, however, we exchanged logging gap sizes for the total area cleared by logging access and infrastructure for each APU (*INFRA*), as following (Fig. S5.4):

$$AGB_I = 10^\wedge(1.5 + \left(0.78 \times log_{10}(INFRA)\right))$$

The carbon fraction of above- and below-ground biomass was estimated using prevailing factor of 0.5 (Brown & Lugo 1982; Brown & Lugo 1992; Higuchi & Carvalho 1994; Malhi et al. 2004) and expressed in metric tonnes of Carbon (MgC). We restrict our estimates of carbon fluxes to forest biomass loss associated with both harvested trees and their collateral damage. We adopt the IPCC Tier 1 assumption that all carbon is emitted from harvested trees at the time of the event but acknowledge some may be stored in other flux chains including long-term timber products. We do not estimate net forest fluxes, which should include all biomass prior to logging activities and also account for forest regeneration within cleared and damaged areas and natural tree mortality. Additional GHG emissions that are also beyond the scope of this study include those from timber extraction vehicles and machinery, and other activities carried out on the CIKEL landholding such as industrial-scale charcoal production and cattle ranching.

5.3.5 Timber prices

Given that many species accrue significant value along different supply chains and export market prices are affected by complex international demands, we used regional scale logwood prices per timber species in Brazilian Reais (R$ per m^3 of lumber), which are most relevant to local timber revenues. These were available from an official source for the state of Pará that serves as a benchmark for timber merchants (DOEPA 2010). This reflects the dominant domestic market, which consumed 95% of all timber produced in Brazil in 2011 (ITTO 2012), and best reflects realistic transaction prices of unprocessed timber expected by loggers at sawmills or other points of timber sales. Timber prices (R$/$m^3$) were grouped by DOEPA (2010) into four categories, with gradually fewer timber species commercialized under increasingly higher price brackets: Class A (11 species, 6 genera): > R$75.0/$m^3$; Class B (18 species, 12 genera): R$45.0/$m^3$ - R$74.0/$m^3$; Class C (40 species, 31 genera): R$25.0/$m^3$ - R$44.0/$m^3$; and Class D (all other 245 species within 157 genera): R$1.0/$m^3$ - R$24.0/$m^3$. The logwood price data we used are deliberately conservative compared to other sources, which may take into account valuation along supply chains (Stone 1998; Bacha & Rodriguez 2007).

Alternative sources of income available to CIKEL may also affect economies of scale and timber species selectivity but are beyond the scope of this study. These include sales of residual dead wood derived from collateral damage at logging clearings (e.g. to meet the high charcoal demand for smelting iron ore in eastern Amazonia), sales of *Eucalyptus* and *paricá* from fast-growing tree monocultures, and value-added through timber processing capacities. Our analyses do not attempt to represent net business profits but illustrate the relative gross value of timber vs forest carbon under realistically priced voluntary agreements such as REDD+.

For all harvested trees we calculated the Pearson correlation coefficients r between volume (m^3) and price (R$/$m^3$) across all APUs to estimate the level of species selectivity or high grading occurring over time.

5.3.6 Timber extraction cost data

Comprehensive timber extraction costs incurred by a RIL operation in eastern Amazonia typically includes those from conducting pre-harvest forest inventories, planning and building all necessary logging infrastructure (roads, skid trails and log yards), hiring tree-climbers, cutting, staff wages, state environmental audits and machinery operational costs. Barreto et al. (1998) estimated a total cost of US\$72 ha^{-1} so we applied the mean commercial conversion rate of 1997 (R\$/US\$ 1.078, IPEA 2015) to estimate total average operational cost of timber extraction at 77.62 BRL ha^{-1}. This was then used to calculate net timber revenues from gross timber revenues.

5.3.7 Carbon pricing

The price of carbon through emission tax schemes varies between different jurisdictions, from <1 to 130 USD per Mg C (Kossoy et al. 2015). However, the global averaged voluntary market price declined to just 3.8 USD per Mg C in 2014 (Hamrick & Goldstein 2014). Excluding public sector agreements, 81% of all REDD offsets were contracted for 3-9 USD per Mg C and averaged only 4.3 USD in 2014 (Hamrick & Goldstein 2014). We therefore opted for a conservative approach in using the contracted proxy price pledged by the Norwegian government and German Development Bank (KfW)to the Amazonian state of Acre in the REDD Early Movers Program (REM) of 5 USD per Mg C.

Macroeconomic impacts of the global financial crises of 2007 led to the depreciation of the US dollar and appreciation of emerging market economies such as the Brazilian real (R\$). Exchange rates in 2010 (US\$1 = R\$1.76) were still much lower than the R\$2.23 mean over the last 10 years (http://www.ipeadata.gov.br/). Most timber in Brazil is consumed by the domestic market (only 5% is estimated to be exported: ITTO 2012), Brazil is the only ITTO producer country that did not show any major vulnerabilities following the financial crises, which had negligible impacts on sustainable forest management area, forest governance, policy development and plantation development (Maplesden et al. 2013). We therefore used the mean commercial conversion rate of 2014 (R\$2.353, IPEA 2015) to estimate forest carbon valuation scenarios in both Brazilian Reais and US Dollars. The 2014 exchange rate is not atypical given our recent study period (2000-2014), and reflects the exchange rate at which carbon agreements were made during the pilot 2014 REDD+ REM programme.

5.4 Results

In total we report data from 344,485 trees harvested across 48,178.9 ha total concessions area, which ranged in size between 2385.0 and 7125.2, over 11 years. The proportion of all inventoried trees (DBH ≥ 30cm) allocated for harvesting each year ranged widely between 25% and 68% (mean ± SD, 46.1 ± 13.7%, N=11, Fig. 5.1). On average, DBH across all harvested trees was 74.7 ± 17.7 cm (range = 50.3 - 294.4 cm). Logging intensity varied little between years, from 28.5 to 37.4 m^3 ha^{-1} (31.3 ± 2.9 m^3 ha^{-1}), although this was below the maximum legal quota of 30 m^3 ha^{-1} within APUs of most years, suggesting reasonable compliance with Brazilian law (Fig. S5.5). This equated to a mean offtake of 7.0 ± 1.3 trees ha^{-1} (range = 5.0 - 8.9 trees ha^{-1}) or mean basal area of 3.13 m^2 ha^{-1}.

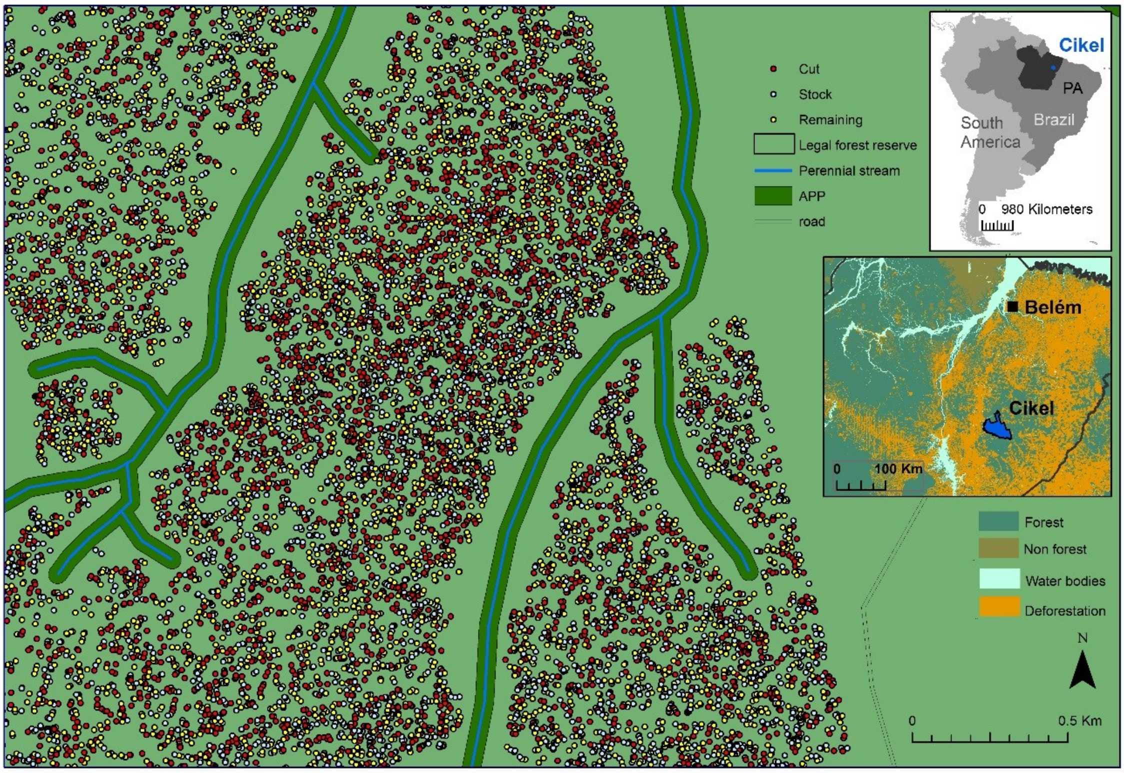

Figure 5.1 Map of the study landscape (Fazenda Rio Capim) within the eastern Amazonian state of Pará, Brazil, illustrating the spatial distribution of a sample of mapped inventoried trees earmarked as harvest trees (red), future stock (blue), and remaining in the stand (yellow). Logging gaps created by fallen tree impacts, skid trail networks generated by bulldozers and log yards are not shown. Dark green areas show nominally protected riparian forest strips that are legally set aside along perennial forest streams (APP).

The total area of ground disturbance across all APUs amounted to 28,456.5 ha, or 59% of their aggregate areas, excluding unlogged forest set-asides. Logging gaps comprised the largest component of ground disturbance in terms of overall cleared area (49%), totalling 23,659.3 ha within the aggregate APU area, or 83% of all ground disturbance recorded (Fig. 5.2). Skid trail networks represented 12.0% of ground disturbance followed by roads and log yards (3.5% and 1.3%, respectively; Fig. 5.2). The proportion of the total logged area (excluding unlogged forest set-asides) cleared by skid trails was 7.1% compared to only 2.1% for logging roads and 0.8% for log yards.

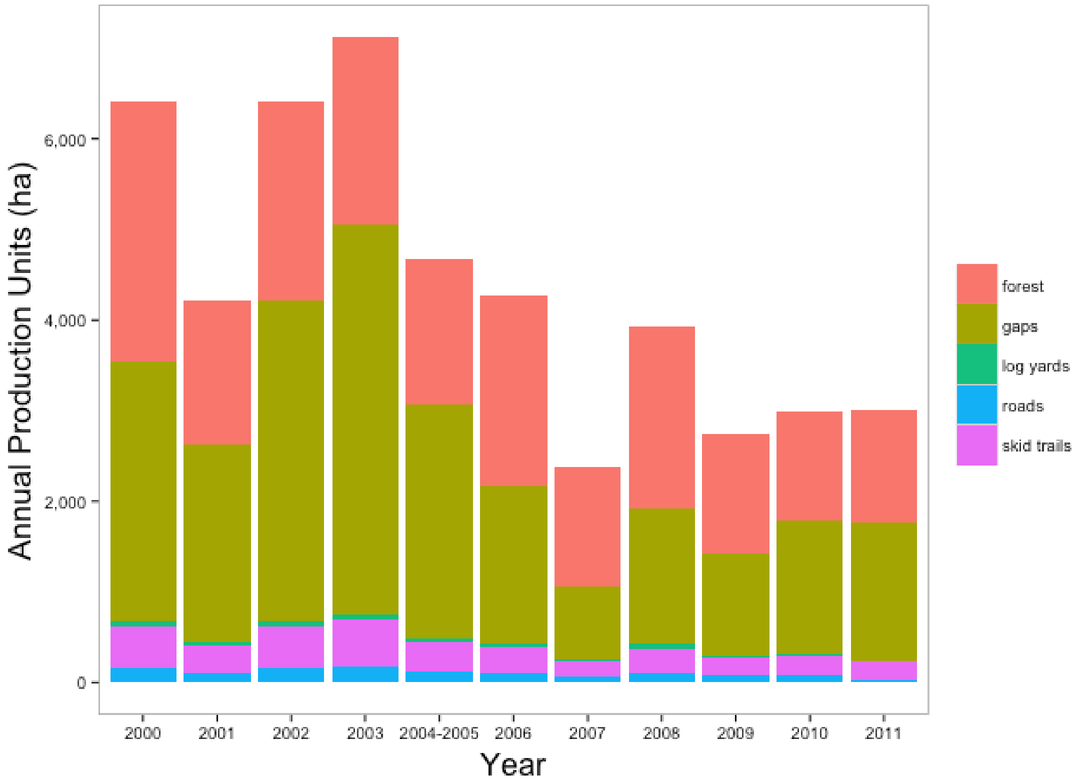

Figure 5.2 Total ground disturbance (cleared areas in ha) due to logging infrastructure (log yards, logging roads and skid trails); felled-tree collateral damage (gaps), and remaining unlogged primary forest across a cumulative logging concession area of 48,179 ha exploited within different Annual Production Units (APUs) over a period of 11 years.

We found that the aggregate loss of AGB ranged between 66.0 and 115.8 Mg ha^{-1} (mean ± SD, 92.1 ± 16.3 Mg ha^{-1}) across all 14 APUs exploited at CIKEL. The total stem removal and mortality resulting from collateral damage accumulated over the 11 years represented 2,600,064 Mg C in committed carbon emissions across all APUs (Fig. 5.3). Over this period of study, the committed carbon loss per unit area, including all removed roundlogs combined with the damage necromass (AGB and BGB), varied between 37.7 and 64.7 Mg C ha^{-1} (51.9 ± 8.9 Mg C ha^{-1}, Fig. 5.4). In Brazil, extracted roundlogs reaching sawmills account for an estimated average of 43% of the overall felled tree biomass (Pearson et al. 2014). We therefore estimated a 2.7 ratio in biomass loss from all residual logging damage relative to that of roundlogs.

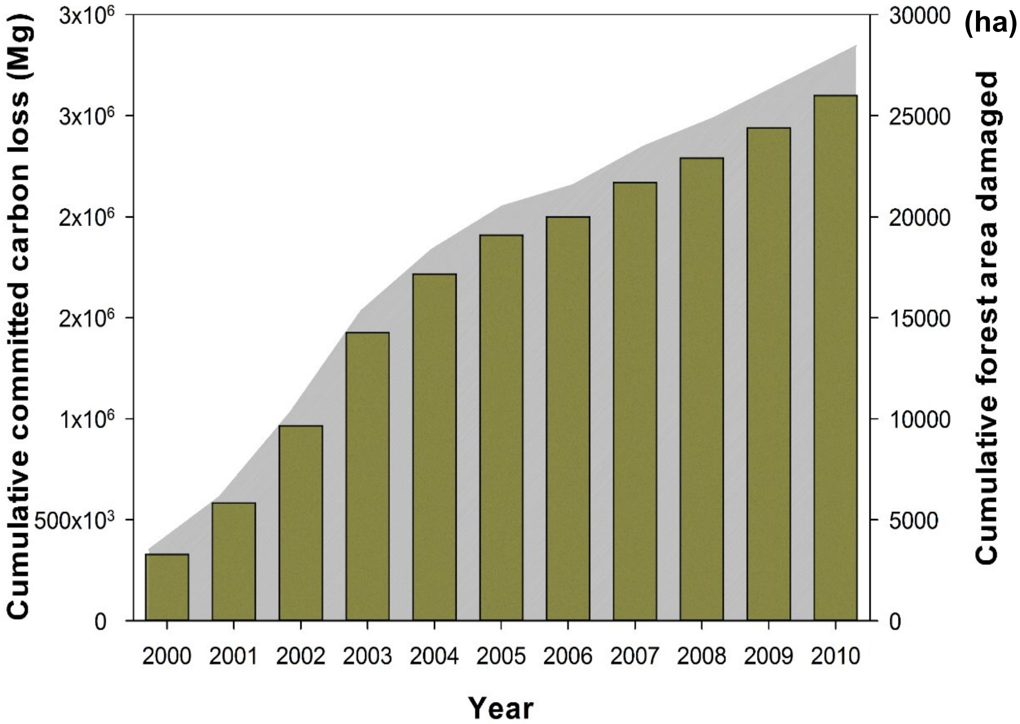

Figure 5.3 Total estimated cumulative committed carbon losses (brown bars) from above (AGB) and below-ground biomass (BGB) from harvested trees, and stem mortality induced by collateral damage in both logging gaps and infrastructure areas cleared for skid trails, log yards and logging roads within 14 APUs logged over a period of 11 years in an Eastern Amazonian landholding. Background shading (grey background area) shows the cumulative forest area damage in ha.

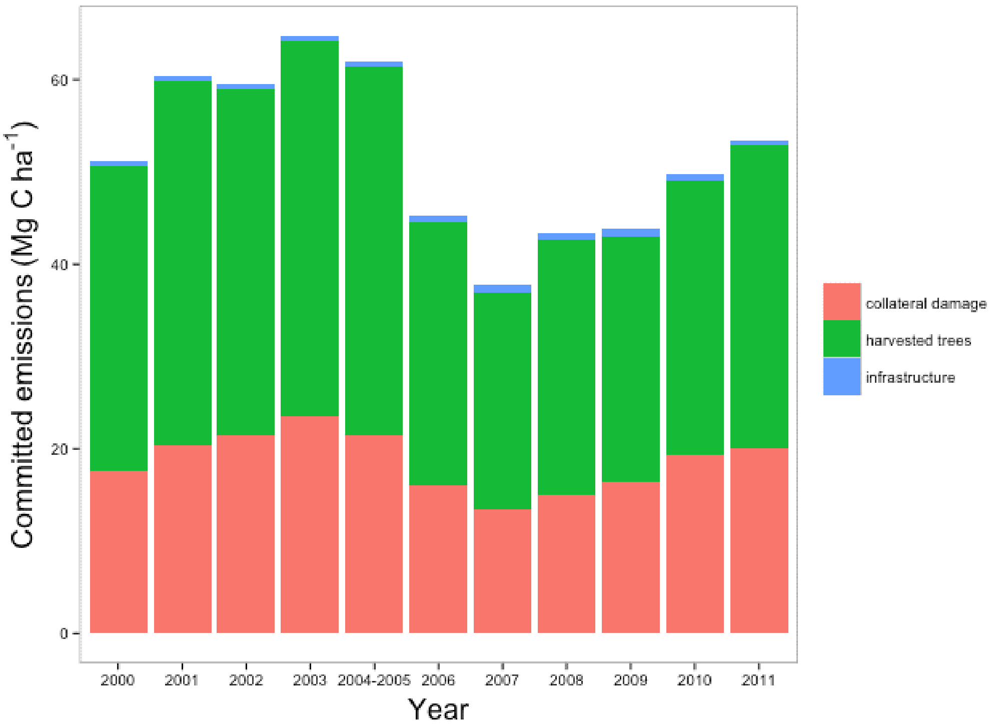

Figure 5.4 Annual estimated committed carbon losses from above and below biomass of harvested trees; collaterally damaged stems in logging gaps; and forest areas cleared for infrastructure in different APUs over a period of 11 years.

Annual gross timber revenues varied across years between R$2.3M and R$8.0M (averaging R$4.5M ± 1.9M yr^{-1}), whilst gross timber revenues per hectare ranged between R$837.2 and R$1254.2 (R$1,003.1 ± 137.5 per ha, Fig. 5.5). Across all APUs, 173 commercially valuable timber tree species representing 99 genera and 39 families were harvested. Pearson correlation values between the overall volumetric offtake (m^3) of each species each year versus the market price (R$/$m^3$) of those species ranged between −0.02 and 0.40 during years of lowest and highest economic efficiency (0.22 ± 0.12 *r*) across all APUs. This time series suggests that CIKEL on average has been proportionally harvesting less valuable timber species over time (Fig. 5.6), presumably as available densities of high-value species declined from the oldest to the most recently harvested APUs.

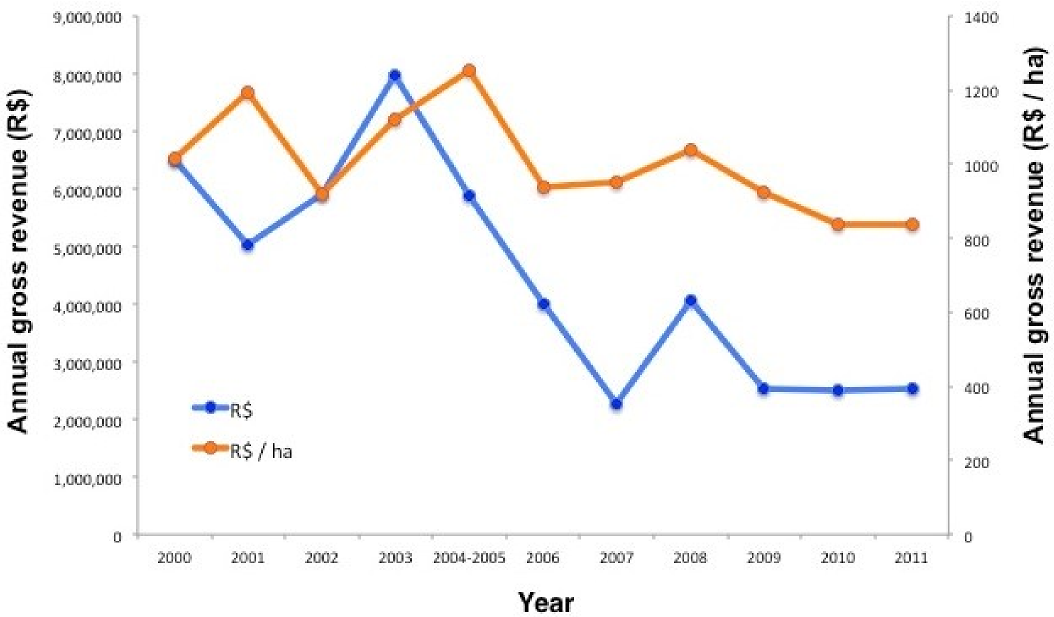

Figure 5.5 Annual estimated gross timber revenues and revenues per unit area (ha) within Annual Production Units (APUs) logged over a period of 11 years within an eastern Amazonian logging concession.

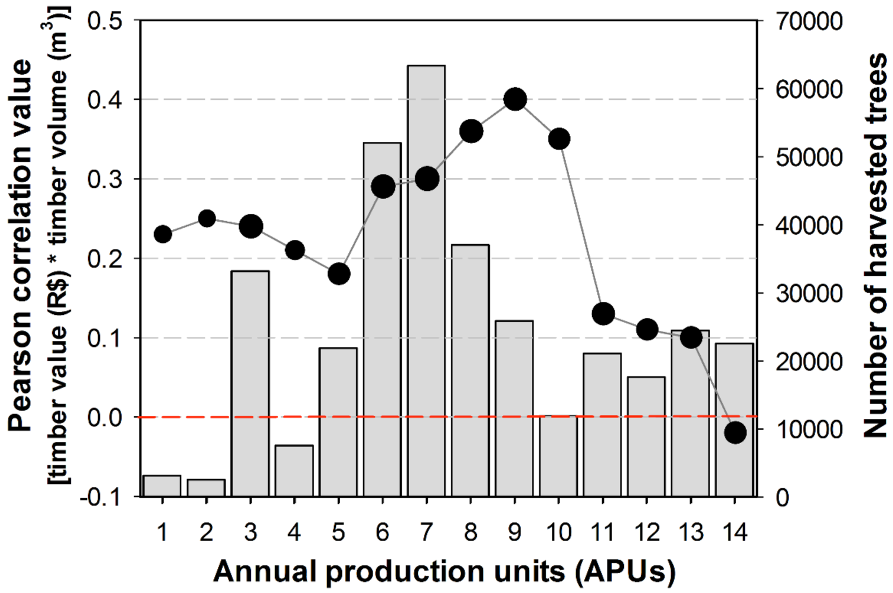

Figure 5.6 Pearson correlation coefficient (r) values (solid circles) between the monetary value per timber volume (R$ m^{-3}) of each timber tree species and their total volumes harvested (Σ m^3), r values represent a measure of high-grading. We consider ~0.34 million timber trees extracted from 14 Annual Production Units (APUs) over 11 years within a large eastern Amazonian logging concession. APUs are spatially and temporally distinct and can be interpreted as a time series (2000-2011). Note however that APU's 1, 2 and 3 were logged in 2000; 4 and 5 in 2001 and APU 8 was logged over 2004 and 2005. There was a sharp declining trend in r values after 2006 (APU 9). Solid circles are scaled to the net sizes of APUs and exclude the area of nominally protected riparian forest strips set aside along perennial forest streams as legally mandated. Red dotted line represents a correlation coefficient of 0.0. Grey bars represent the total number of trees harvested in each APU.

Once reasonable estimates of timber exploitation costs were taken into consideration, net timber revenues varied between R$2.1M and R$7.4M yr^{-1} (R$4.1M ± R$1.8M) and net revenues per unit area between R$759.6 and R$1,176.6 ha^{-1} yr^{-1} (R$925.5 ± R$137.5). In US dollars, this equated to overall yearly revenues of US$0.9M and US$3.1M yr^{-1} (US$1.7 ± 0.8M) or US$322.8 and US$500.0 ha^{-1} yr^{-1} (US$393.3 ± 58.4 ha^{-1} yr^{-1}, Fig. 5.7b). We estimate that the overall carbon value of sparing all harvested trees plus all associated collateral damage was worth an estimated US$187.9 to US$323.3 ha^{-1} yr^{-1} (US$259.2 ± 44.5 ha^{-1} yr^{-1}, Fig. 5.7a). Therefore, our data indicate that carbon finance schemes would be required to pay out forest carbon transaction prices ranging between US$6.0 and US$9.9 per Mg C (US$7.7 ± 1.3 per Mg C) in order to match the opportunity costs associated with primary forest timber extraction in a typical reduced-impact logging operation in Eastern Amazonia.

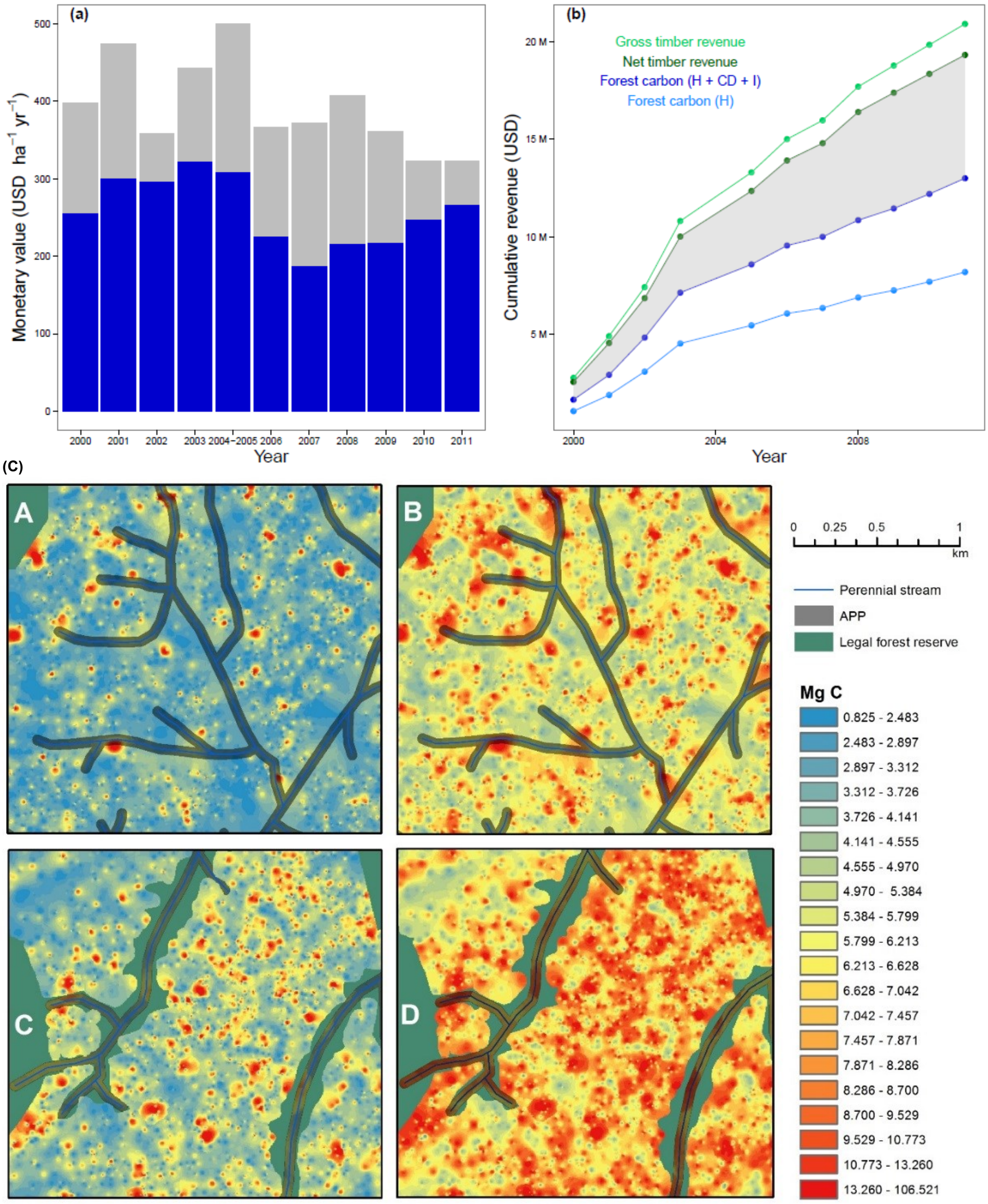

Figure 5.7 (a) Annual net value of timber in USD ha^{-1} yr^{-1} as the additionality (coloured in grey) in projected forest carbon value (calculated at 5 USD per Mg C, coloured in blue) estimated from the *H* + *CD* + *I* scenario of sparing logging-induced forest degradation, i.e. Biomass from harvested trees, collateral damage (in logging gaps) and infrastructure (logging roads, decks, skid trails) are represented by *H, CD,* and *I,* respectively. **(b)** Cumulative revenue (USD) over 11 years of timber extraction showing estimated gross and net timber revenues, and projected forest carbon values both considering (*H* + *CD* + *I*) and excluding carbon loss from logging infrastructure (*H* + *CD*) in sparing logging-induced forest degradation. Grey areas represent the difference between cumulative net values of timber roundlogs versus the projected forest carbon value from *H* + *CD* + *I*. **(c)** Forest biomass (AGB + BGB) maps of two sample areas within APUs 13 (A and B) and 14 (C and D). Panels A and C considered only biomass calculated from *H*; panels B and D considered biomass from both *H* + *CD*. Mg C was interpolated using IDW (inverse distance weighing) at a resolution of 5m in ArcMap 10.2.2. Biomass loss from skid trail networks generated by bulldozers and log yards are not shown. Dark green areas and dark grey shading show legal forest reserves and nominally protected riparian forest strips along perennial forest streams, respectively.

5.5 Discussion

Our analyses provide clear and timely evidence that the costs potentially incurred by carbon financing projects in sparing logging-induced degradation can compete with net timber revenues accrued to logging operations in Amazonian primary forests even if only harvested trees and their associated collateral damage are considered and transaction carbon prices are raised from a modest value of US$5 to US$8 per Mg C. The mean additionality between predicted mean net timber revenues (US$393 ha^{-1} yr^{-1}) versus payments from carbon projects was approximately US$134 ha^{-1} yr^{-1} (Fig. 5.7a and b). This market competitive arena becomes more promising because high-grading timber extraction in previously unlogged primary forest typically follows the economic logic of cashing in on a long-term high-value timber stock windfall that is rapidly degraded after the first cutting cycle (Chapter 2 and Fig. 5.6). Opportunity costs from secondary timber revenues can therefore become even less competitive against any market intervention to prevent further erosion of live forest biomass.

Only 1% of all Amazonian tree species account for half of all forest biomass and carbon sequestration, and the largest half of all species contribute with 82.5% of all biomass (Fauset et al. 2015). These large slow-growing, high wood density species are also preferentially targeted by selective logging operations. Such large harvestable trees also provide a largely overlooked and irreplaceable environmental service by collectively acting as a large-scale fire-break in preventing the forest understorey from breaching a flammability threshold, thereby avoiding further forest degradation. Whilst we support progress and incentives for widespread RIL adoption (Putz et al. 2008), we argue that under certain circumstances, instead of being rewarded for RIL over conventional logging (e.g. Putz & Pinard 1993; Pinard & Putz 1996) primary tropical forest landholders could be awarded conservation payments for avoiding logging-induced degradation in the first place. However, this does not deemphasise the importance of logged and secondary forests to biodiversity conservation (Dent & Joseph Wright 2009; Chazdon et al. 2009; Putz et al. 2012), and preventing degraded forests from further land-use change such as conversion to agriculture are of paramount importance to secure long-term forest protection (Wilcove et al. 2013). The context of these circumstances and implications including conflicting funding allocations are further discussed below.

Keller et al. (2004) estimated total course woody debris (DBH ≥ 2 cm) in undisturbed and both reduced-impact and conventionally logged (CL) Amazonian forests (logging intensity of 25-30 m^3 ha^{-1}) and found that for RIL operations, the same amount of AGB is lost in damage (~75 Mg ha^{-1}) as is exported to sawmills, whereas 2.5 times as much biomass is lost in CL. This compares to our estimated ratio of 2.7, and mean collateral damaged AGB loss of 67.1 Mg ha^{-1} (excluding round logs of harvested trees). Our smaller AGB damage can be partially explained by the much smaller stem diameter threshold in Keller et al. (2004). Our carbon loss ratio between total residual damage and exported logs is consistent with that estimated for stems DBH ≥ 10cm an operation in southern Amazonia (2.4) with a logging intensity of 6.4-15 m^3 ha^{-1} (Feldpausch et al. 2005).

We estimated a total AGB loss (whole harvested tree and associated damage) of 92.1 Mg ha^{-1} under conditions of relatively low mean harvest intensity (31.3 m^3 ha^{-1}). This is highly comparable to an estimate of 94.5 Mg ha^{-1}, from the CIKEL Rio Capim landholding (Mazzei et al. 2010). In the Congo Basin, where old-growth timber trees tend to be larger (mean DBH ≥ 85 cm), Medjibe et al. (2011) estimated an AGB loss of only 34.2 Mg ha^{-1}, but at a much lower logging intensity of 8.1 m^3 ha^{-1}. When we express total AGB and BGB loss as committed C emissions, our mean loss of 51.0 Mg C ha^{-1} is almost identical (51.1 Mg C ha^{-1}) to the estimated carbon emissions of an Indonesia RIL operation with a mean harvest intensity of 39.1 m^3 ha^{-1} (Griscom et al. 2014). Estimated committed emissions can also be expressed relative to timber harvest intensities, which in this study ranged between 1.3 to 2.0 Mg C ha^{-1} m^{-3} across all APUs (1.7 ± 0.2 Mg C ha^{-1} m^{-3}). Other similar estimates include 1.1 - 2.0 in Amazonia (Feldpausch et al. 2005); 1.5 in Indonesia (Griscom et al. 2014); and in a pan-tropical study 0.9 - 2.3 Mg C ha^{-1} m^{-3} (Pearson et al. 2014).

Eastern and central Amazonian forests harbour higher AGB values than western Amazonia (Houghton et al. 2001; Baker et al. 2004). Unlogged stands at CIKEL store a relatively high biomass of 410 Mg ha^{-1} whilst other parts of Amazonia stock a mean total live biomass (from 25 different studies) of approximately 297.5 Mg ha^{-1} (Sist et al. 2014). Our biomass estimates are conservative, however, as we did not consider forest biomass loss from damaged understorey shrubs and vines which generally contribute 1- 3% of live forest AGB (Brown & Lugo 1992; Pinard & Putz 1996). Carbon dynamics associated with forest regeneration trajectories over time were beyond the scope of this study. However, Mazzei et al. (2010) suggest that the net biomass balance one year after logging was still negative (-31.1 Mg ha^{-1}) across our study landscape. Moreover, full post-logging biomass recovery is estimated to take >30 years (Blanc et al. 2009), so we did not fully consider the high mortality of residual trees following logging (Ruslandi et al. 2012), much of which may be considerably delayed (Barlow et al. 2003).

Infrastructure damage from logging roads, skid trails and log yards represented 10% of the total ground surface area of our logging units aggregated across all APUs. Pereira et al. (2002) and Pinard & Putz (1996) estimated infrastructure ground damage at approximately 4.7% and 3.5% in RIL operations versus 8.9-11.2% and 12% in CL (at logging intensities of 23 and 104-154 m^3 ha^{-1}, respectively). Likewise, Infrastructure damage has been estimated at 11.4% for Gabon (White 1994) and 8.2% for the Congo Basin (Medjibe et al. 2011). In our study, canopy gaps created from felled-tree impacts resulted in 83% of all ground damage and this has been widely observed across the tropics (White 1994; Pereira et al. 2002; Feldpausch et al. 2005; Medjibe et al. 2011).

Because most forest damage is inevitable and occurs at the point of felled-tree impact, even the most stringent of RIL operations is unable to reduce its net carbon emissions beyond a certain level. This becomes most relevant during the first cutting cycle, when the largest trees across the landscape tend to be selectively removed (Feldpausch et al. 2005). In Malaysian forests, large trees (DBH ≥ 60 cm) can represent 59% of the entire forest biomass (Pinard & Putz 1996). In our study site, large trees ≥60 cm DBH stocked a mean of 183 Mg ha^{-1} worth about half of all forest biomass (Sist et al. 2014). An offtake cap at 110 cm DBH has been suggested to avoid most of the forest biomass being lost (Mazzei et al. 2010). Additional suggestions include a 50% reduction in logging intensity from 6 to 3 trees per ha, saving an estimated 27.7 Mg C ha^{-1} (Sist et al. 2014).

We recognise that forests under extremely high logging intensities, such as dipterocarp-rich Southeast Asian forests (104 – 154 m^3 ha^{-1}), up to 37 Mg C ha^{-1} (or ~73% of our total carbon loss estimate) could be saved by adopting RIL rather than currently practiced CL (Pinard & Putz 1996). In such contexts, we suggest that there is great potential for REDD+ payments to avoid excessive carbon emissions, compensate loggers for lost profits, and reward the adoption of RIL practices (cf. Putz & Pinard 1993; Pinard & Putz 1996; Sist et al. 2014). However, REDD+ payments to avoid logging-induced forest degradation could also be financially feasible even in Brazilian Amazonia, where legally required logging intensities are capped at much lower harvest rates (30m^3 ha^{-1}).

A 20% reduction in live AGB was observed in the aftermath of logging in an eastern Amazonian forest stand (at a similar moderate logging intensity of 35m^3 ha^{-1}) but this forest subsequently lost 83% of its pre-logging biomass after a surface fire (Gerwing 2002). Barlow et al. (2012) highlighted the urgent need to acknowledge and incorporate a reduction in the risk of fires into REDD+ projects if they are to realistically secure long-term carbon emissions reductions, biodiversity conservation and rural poverty alleviation. The replacement of old-growth forests by degraded forests is more widespread than previously thought and directly undermines forest capacity to provide many other goods and services beyond carbon stocks, including regional to global scale climate regulation, water flow regulation, or moderating vector-borne diseases (Foley et al. 2007).

Pearson et al. (2014) used data on roundwood production from 2005 in the FAO-FRA (2010) to suggest that logging emissions in Brazil are equivalent to only 6% of deforestation emissions. We suggest that this severely underestimates latent emissions from logging disturbance because this study failed to consider the substantial contribution of illegal logging which amounts to at least 75% of all timber harvested in Brazilian Amazonia (Wellesley 2014). Additionally, the Brazilian government has begun promoting widespread degradation through annual bids for logging concessions within the vast network of National Forests of Amazonia (Law 11.284 of 2006). For instance, 9.3 M ha were made available to new concessions between 2013 and 2014 (SFB 2013b). Additional future financing could be used to spare logging-induced degradation, particularly in areas of high biodiversity (Gardner et al. 2012), much like the REDD+ REM programme in Acre (Alencar et al. 2012). Ground surveys of carbon biomass across large areas can be expensive and time consuming, but promising new techniques involving LiDAR, could be implemented on a national scale (Asner et al. 2013; Bustamante et al. in press).

These recommendations, however, are not free of complex feedback mechanisms or shortcomings. If logging activities were discontinued in large working landholdings such as CIKEL's Fazenda Rio Capim, this would incur an immediate regional and individual social cost as many hundreds of workers become unemployed. There is also the risk that financial compensation targeting both large logging operations or state forests can further compete with REDD+ financing that could be directly applied to alleviate rural poverty and improve forest livelihoods. Some carbon offset projects in Brazil involving multinational corporations and smallholders have had significant detrimental socioeconomic consequences, including losses in local income and access to natural resources (Kill 2014). REDD+ projects are also subject to long-term financial risks from inconsistencies in long-term donor support to carbon market volatilities (Phelps et al. 2011). Agricultural intensification has often been suggested as one mechanism to increase 'land sparing' for tropical forest conservation and there are concerns that this increase in productivity also leads to increased agricultural land rents thereby inflating conservation costs (Phelps et al. 2013). Success would thus also require payments to be re-adjusted for inflation, fluctuations in roundlog market prices, and consider the rising opportunity costs associated with alternative land uses.

There are further issues of leakage at regional, national and international scales. If REDD+ payments are indeed successful in avoiding logging-induced forest degradation, they may increase pressure on primary forests elsewhere. Inelastic market demands could thus be met through increased plantation forests, illegal logging, or offtake intensification in other tropical timber producing countries. However, there are clear benefits to this approach. State forests and largeholdings like Fazenda Rio Capim are generally free of land tenure conflicts typically observed in REDD+ projects targeting smallholders (Sunderlin et al.

2014). An elevated price from US$1 to US$8 per Mg C might be optimistic but would not be outside the current range of REDD+ projects (Hamrick & Goldstein 2014). REDD and REDD+ were the most negotiated offset projects in 2014 and almost half of the record 25 million Mg C sold were negotiated under the REM programme (Hamrick & Goldstein 2014). This places Brazil as a global leader in REDD+ agreements with huge potential for conservation. We therefore urge policy makers to place avoided tropical forest degradation, including logging disturbance, firmly onto the climate mitigation agenda to finally honour the second D in REDD+.

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5.6 Supporting information

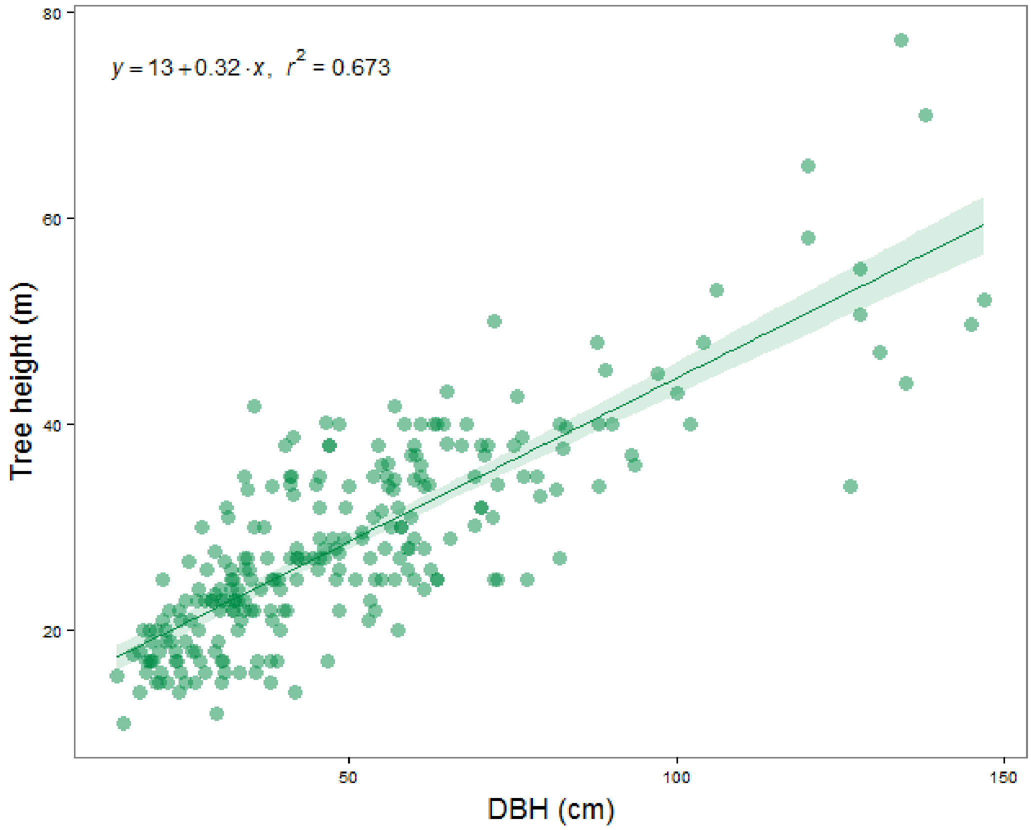

Figure S5.1. DBH (cm) explained 67% of the variation in tree height (metres) among 250 broadleaf trees in an Eastern Amazonian logging concession, based on linear regression (P<0.001).

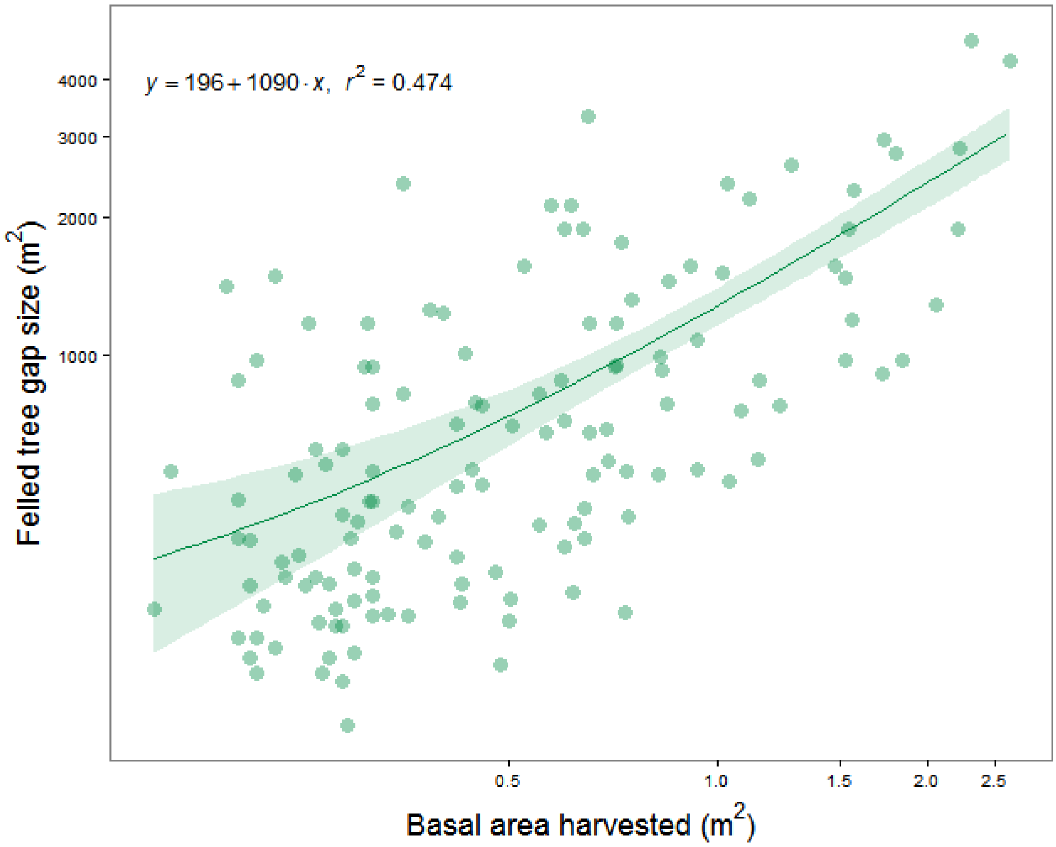

Figure S5.2. Aggregate basal area harvested (m^2) explained 47% of the variation in logging gap size (m^2) based on linear regression (P<0.001).

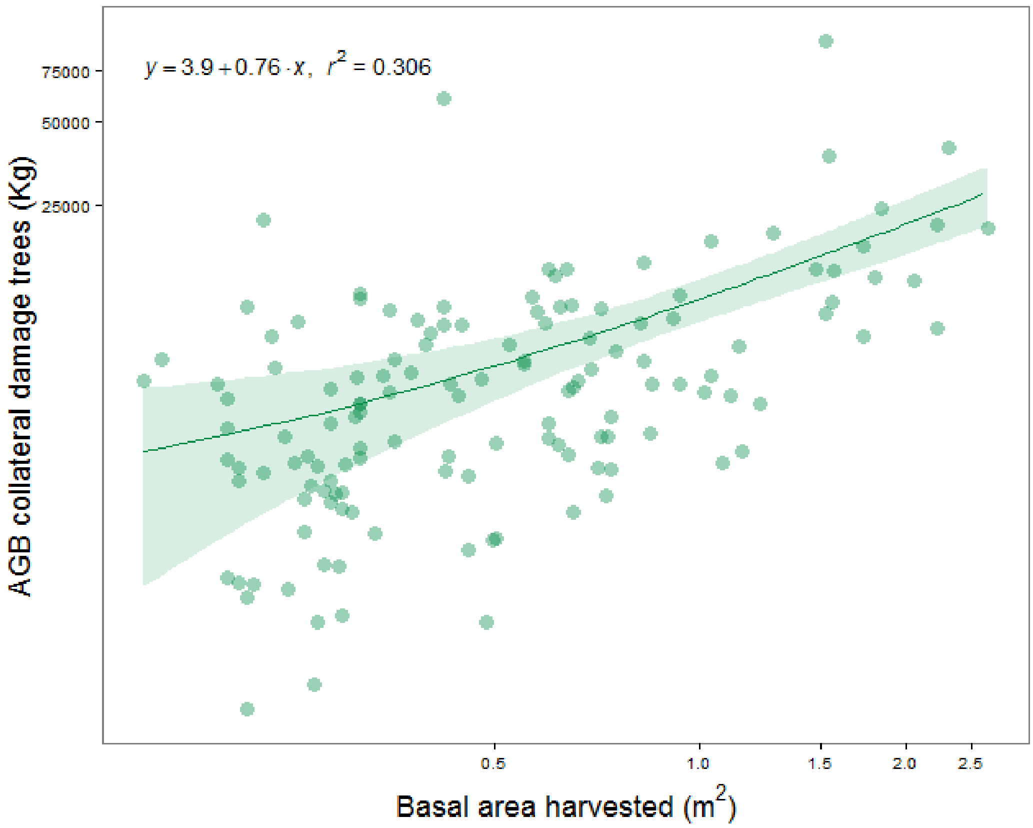

Figure S5.3. Aggregate basal area harvested (m^2) explained 30% of the variation in AGB from committed necromass (kg) resulting from collateral damaged of felled harvest trees, based on linear regression (P<0.001).

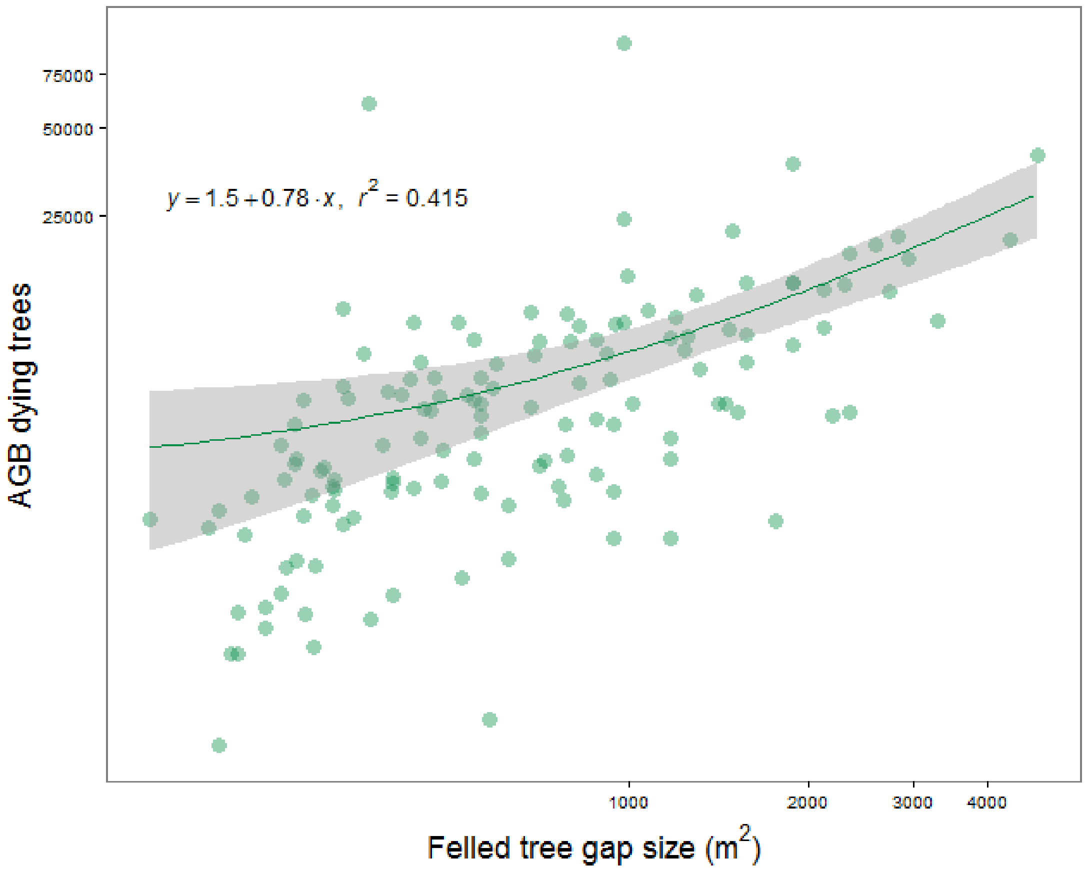

Figure S5.4. Logging gap size (m^2) explained 42% of the variation in committed necromass (kg) from felled trees, based on linear regression (P<0.001).

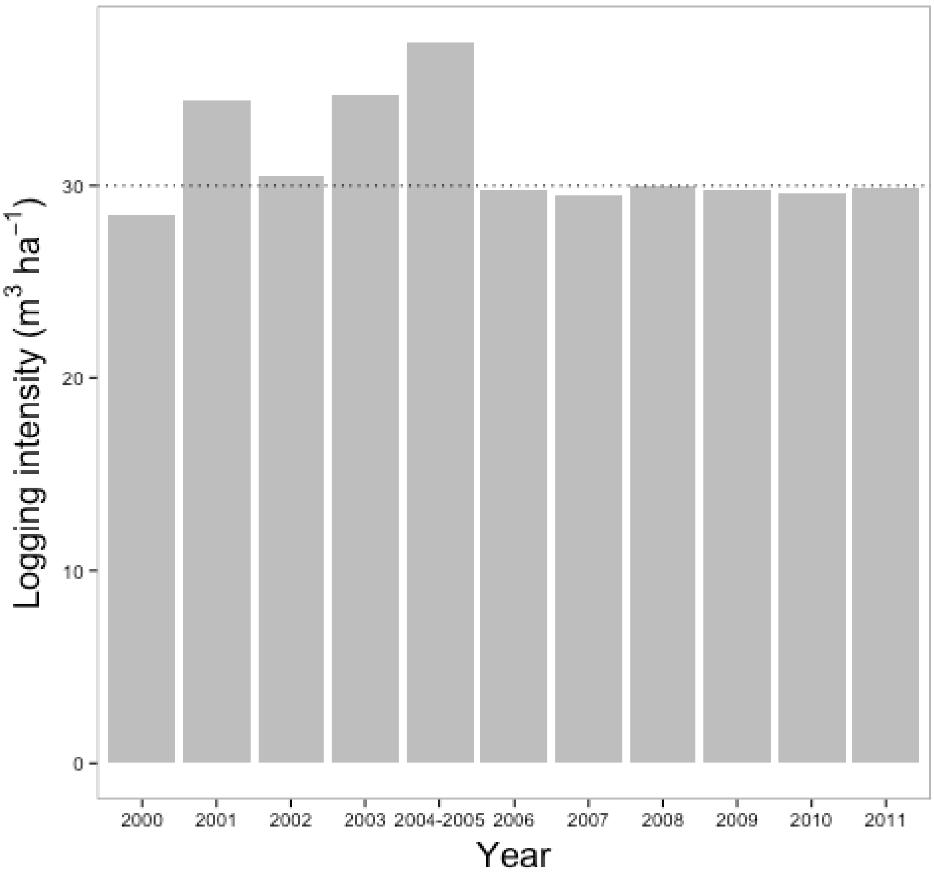

Figure S5.5. Annual logging intensities (m^3 ha^{-1}) of harvested trees in different APUs over a period of 11 years. The grey dotted line represents the Brazilian maximum volumetric harvest limit of $30m^3$ h^{-1} (Normative Ruling no. 05 of 2006).

Chapter 6: Concluding remarks

6.1 Key findings and conservation implications

Only 7.6% (30.6 million ha) of the global permanent tropical forest estate designated for timber production is estimated to be sustainably managed (Blaser et al. 2011). In this doctoral thesis, I have shown that both historical records and the contemporary status of a widespread set of existing forest stands suggest that the most prized and sought after timber species in the Brazilian state of Pará have been repeatedly mined to the point of subregional demographic collapse as a function of local supply/demand conditions mediated by physical access, land-tenure systems, and timber market prices (Chapter 2). The main findings here further support the growing consensus that at present, tropical timber, even under RIL and legal concessions, are being mined from natural forests to the point of demographic collapse (Wadsworth & Zweede 2006; Sist & Ferreira 2007; Zarin et al. 2007; Putz et al. 2008; Peña-Claros et al. 2008; Schulze et al. 2008; Macpherson et al. 2012; Zimmerman & Kormos 2012). Moreover, as we move into the 5th decade since the onset of widespread industrial-scale logging of tropical forests, this study contributes to a body of recent empirical evidence on a pantropical scale that the baseline tree species composition of old-growth forests does not fully recover from selective logging, even following half a century of regeneration, as exemplified by studies in Uganda (Osazuwa-Peters et al. 2015) China (Xu et al. 2015), and Brazil (Chapter 2).

The unsustainable forest mining sequence depicted in Figure 1.4 (Chapter 1, *Source*: Uhl et al. 1997) is described as 'general, unplanned logging' where mechanised extraction of as many as 100 species occurs at harvest intensities of 5-10 trees ha^{-1} and in the absence of any silvicultural treatments. Today's best-case scenario in Amazonian forest logging, represented by landholdings such as CIKEL (Chapters 4 and 5), are fully government sanctioned, use RIL techniques, and carry all privileges and market respectability of FSC certification. Regrettably, however, these RIL operations appear to be performing more like the forest mining sequence illustrated in Uhl et al. (1997), rather than the idealized sustainable forest management sequence. Notwithstanding the application of pre-harvest silvicultural treatments, the CIKEL operation harvested well in excess of 100 species (total of 173 species comprised of 99 genera and 39 families) at an average intensity of ~7.0 trees ha^{-1}. Damage was considerable, with over a third of the total ground sampling area of logged forest being cleared by felled-trees impacts alone, not to mention the additional disturbance generated to access, remove and process roundlogs. It was estimated in Chapter 4 that for every harvested tree, there was an additional loss of ~12 damaged stems. This suggests an estimated ~4 million additional stems were committed to mortality for the 0.34 million trees harvested across the 48,179 ha of old-growth Eastern Amazonian forest examined here (Chapter 5).

6.1.1 Financial sustainability

Although CIKEL was a pioneer enterprise in testing and introducing RIL techniques to Amazonia, and arguably Brazil, they can no longer maintain meaningful profits from timber extraction alone (CIKEL, personal communication). There is evidence in this study that they can no longer fetch high returns; gross expected timber revenues have consistently decreased since 2003, and revenues per unit area (R$/ha) have consistently decreased since 2004-2005 (Fig. 5.5). Notably, the peak expected revenues per unit area (R$/ha) observed between 2004-2005 (Fig. 5.5) also overlaps with the period of highest extraction intensity (37.4 m^3 ha^{-1}, Figure S5.5). New legislation introduced in the following year (Normative Ruling no. 05 of 2006) prohibited offtakes over 30m^3 ha^{-1} and may have limited such high returns in subsequent years. The correlation coefficients between the commercial value (R$/$m^3$) and total volumetric harvest (m^3) of each harvested tree species can be seen as a measure of high grading. This metric peaked in 2006 (APU 9) and has subsequently also been consistently declining until the end of our time series in 2011 (Figure 5.6).

The global financial crises of mid 2007 caused a decrease in tropical logging offtakes, with the highest observed impacts occurring between 2008 and 2009 (Maplesden et al. 2013). However, the first dips in expected revenue and high grading suggested in the large CIKEL landholding (Chapter 5) began just before the onset of the financial crises. This could be because they were limited to harvesting timber in areas that had intrinsically smaller stocks of high-value timber, or in areas that may have already been selectively logged. These effects are compounded and exacerbated by exploiting consistently smaller Annual Production Units (APUs) since 2003 and market competition with illegal logging, which has also been increasing since the global financial crisis (Maplesden et al. 2013; Wellesley 2014). Either way the case study presented in Chapter 5 suggests, in line with Chapters 2 and 3, that the most prized high-value species and individuals are no longer available, even in the most careful and best-managed logging operations. This does not bode well for the vast tracts of Amazonian forests that are either logged illegally, or obtain timber extraction licenses but are too remote for proper enforcement of sustainable logging regulations.

RIL requirements, including normative rulings of 2006, may have rendered best practices financially unsustainable, which has been suggested elsewhere (Rice et al. 1997; Putz et al. 2000; Pearce et al. 2003). Indeed there is evidence that legal natural timber production in Brazil has been decreasing since ~2005 (WWF 2013). These findings bring forward serious consequences for the future of logging in Amazonia. Millions of hectares of public lands in forest reserves of Brazilian Amazonia are currently being handed over to similarly large-scale industrial operations through annual concession bids (Law 11.284 of 2006, SFB 2013b). These areas use similar extraction and management techniques as those of CIKEL, and selective logging may result in second harvests that are demographically and economically unsustainable. Whilst selective logging may be less damaging to residual stands than more predatory alternative land-uses, such as clear cutting and conversion to agriculture, unlogged primary forests continue to be irreplaceable for biodiversity conservation (Gibson et al. 2011).

6.1.2 Largeholders and smallholders

In Chapter 3, this thesis shows that smallholders (≤400 ha) appear to exert stronger high-grading pressure upon the high-value hardwood timber species available in their landholdings, thereby accruing higher gross revenue productivity per unit area, at least in the short term. Moreover smallholders were more likely to overestimate and inconsistently report the minimum areas of forest set-asides required by Brazilian law. Smallholders represented 35% of the 824 AUTEF forest management plans examined here, and 3.3% of the total volumetric roundlog offtake (~17 million m^3, Table 3.1). It may be assumed that most Amazonian smallholders do not have the financial capacity to carry out the stringent inventory planning and monitoring required for the approval of sustainable forest management plans. As such, the sample presented here (Chapter 3) is likely a gross underestimation of the total smallholder contribution to natural logging offtakes in Pará and the wider Amazonian region. Pereira et al. (2010) suggest that roundlogs from smallholders (defined as properties ≤500 ha) accounted for 29% of the total volumetric harvest in the state of Pará in 2009 (and medium- and large-holders for 41% and 31%, respectively) and considerably larger shares in the states of Roraima (78%) and Rondônia (49%).

Smallholders appear to exert more pressure on the landscape by harvesting the largest proportions of their land (over two thirds of their landholdings, Chapter 3), which is entirely understandable given their limited financial elasticity. It may not be coincidental therefore that we find an asymptote in the data; concession size relative to total landholding size increased steeply and then reaches an asymptote at rural properties of around 1,798 ha (Fig 3.2). This figure is close to the property size threshold of 1,500 ha for medium-sized landholdings, as discussed in Chapter 3, that may benefit from rife noncompliance with regional policies, such as the rural territorial tax (*Imposto Territorial Rural,* ITR). In addition, these properties are perversely fast-tracked to obtain legal land tenure through the Legal Land Programme (*Programa Terra Legal* Law 11.952/2009, Brito & Barreto 2011). It may be that fluctuating government policies likely exert complex and perverse incentives for actor-specific forest-set aside compliance and deforestation.

6.1.3 Charcoal production and logging

One cannot discuss Eastern Amazonian logging and the residual down wood resulting from collateral damage without mentioning the issue of charcoal production (Fearnside 1989). Brazil is the largest producer and consumer of charcoal. In 2009 and 2010, 17.4 million tonnes of charcoal were consumed, 85% of which destined for smelting in the pig-iron ore sector to meet mining export demands, primarily for domestic car production in the USA (Magri et al. 2012). The majority of this charcoal (~60%) is sourced from natural native vegetation, rather than plantation forests of fast-growing exotic species such as *Pinus* or *Eucalyptus*, primarily in Amazonia but also from the Cerrado and Pantanal regions (Monteiro 2006). In addition to the collaborative NGO investigation by Magri et al. (2012), a Greenpeace report (Greenpeace 2012) published in May 2012, with an accompanying protest on-board the USA-bound cargo ship Clipper Hope, sparked the first wave of media coverage on the detrimental environmental consequences of pig-iron production in Brazil (http://www.theguardian.com/environment/gallery/2012/may/17/pig-iron-deforestation-brazil). Resolution 411 (2009 available at http://www.mma.gov.br/port/conama/legiabre.cfm?codlegi=604) regulates inspections of operations that process charcoal and residual wood products from natural forests to ensure they do not come from illegally deforested areas, but these inspections are largely not carried out. Both reports (Magri et al. 2012 and Greenpeace 2012) denounce widespread slave labour and illegal harvests of logwood from neighbouring public lands in Amazonia, specifically in the states of Pará and Maranhão. Those lands belong to either Indigenous people (*Awá*, in the Alto Rio Guamá, Alto Turiaçu) or strictly protected areas on paper (Reserva Biológica do Gurupi). Companies purchasing iron-ore smelted from illegally sourced timber include Fiat Ford, General Motors, Volkswagen, BMW, Mercedes, Nissan, and John Deere (Greenpeace 2012).

The production of 1 tonne of pig iron requires the burning of 2.2 m^3 of charcoal which in turn requires 4.4 m^3 of timber (Resolution 411, 2009). Thus to meet the 2-year demand of 17.4 million tonnes (2009 and 2010, Magri et al. 2012), we'd assume ~76.6 million m^3 of timber was consumed, of which ~46.0 million m^3 (60%, Monteiro 2006) was from natural forests, resulting in an annual mean consumption of 23 million m^3. This mean volume of wood required for charcoal production is equivalent in volume to almost 75% of all the natural legal offtake across Brazil in 2012 (30.8 million m^3, ITTO 2012). Charcoal production is thus a largely overlooked but significant driver of deforestation and forest degradation in eastern Amazonia. Furthermore, charcoal production is inherently intertwined with logging companies. Even RIL operations like CIKEL make charcoal, sourced from collateral damage in logged stands, including hollow or discarded portions of target tropical hardwood trees (CIKEL, personal communication). To the best of my knowledge, this has not yet been quantified but re-entering once-logged forest for harvesting this residual wood with skidder machinery and hauling equipment likely crushes any saplings and resprouting vegetation, and creates additional collateral damage to the already severely damaged residual stand (Chapters 4 and 5). Charcoal is produced in highly inefficient artisanal ovens (Fig. 6.1), where most of the energy is lost. It also rests on socially unacceptable methods, including child labour and all child and adult workers are exposed to smoke and hazardous carcinogens (Greenpeace 2012).

As a signatory of the Copenhagen Accord (United Nations Climate Change Conference in Copenhagen in 2009) Brazil stated as part of its Nationally Appropriate Mitigation Actions (NAMAs) that it will phase out and substitute natural forest based charcoal for plantation based charcoal used in the steel industry by 2020, saving an estimated 12 – 15 Mt C yr^{-1} in GHG emissions (Linacre et al. 2011). This is a positive development for forest conservation in eastern Amazonia, even if it reduces revenues from 'best practice' RIL operations including CIKEL.

Figure 6.1 (a) Photo showing some discarded pieces of harvested trees, piled and awaiting removal for charcoal production (Photo: Vanessa Richardson). **(b)** Photo depicting some of the ~600 artisanal charcoal ovens observed in operation during fieldwork in 2013 at the CIKEL landholding of eastern Amazonia (Photo: Vanessa Richardson).

6.2 Recommendations

To avoid further degradation and damage to primary forest areas, fiscal methods for forcing RIL are needed (Putz et al. 2000) and should become more stringent. If the most stringent of RIL practices are only required by voluntary FSC certification, and motivated by financial considerations alone, i.e. if best management practices are not politically enforced, then it renders the system totally market-dependent. This raises serious questions as to whether it is a good idea for a transnational private governance system (such as FSC) to oversee the sustainability and biodiversity of the world's forestry system (Schepers 2010). Outsourcing governmental responsibilities brings problems including uneven geographic representation and the subsequent disadvantages for developing countries; the uncertainty of long-term financial support for private politics; and issues of competing schemes sending contradictory signals toward consumers and policy makers (Pattberg 2005). Because 95% of all natural timber production in Brazilian is consumed domestically (ITTO 2012), international consumer market pressure to boost logging sustainability in Brazil is therefore largely ineffective. Promotion of legally verified timber to the biggest timber consumers, particularly southern Brazil, is therefore urgently needed.

The results from this study suggest that there is an urgent need to standardise biologically sound methods for describing logging damage (Putz et al. 2008). It has become evident that the cutting cycle of 35 years (Normative Ruling no. 05 of 2006 and Barreto et al. 1993) is too short. At current harvest intensities in Amazonia and across the tropics, the cycle should be set to a minimum of 60 years (Kammesheidt et al. 2001; Sist et al. 2003; Mazzei et al. 2010; Zimmerman & Kormos 2012). To avoid large logging gaps across the landscape (Chapter 4), additional RIL measures might include (1) a minimum spacing distance between harvested trees, and (2) not only a minimum, but also a maximum DBH limit for harvest trees (Sist et al. 2003; Mazzei et al. 2010); and (3) reducing harvest intensities from 30 to 20m^3 ha^{-1}. To slow down the compositional shifts (Phillips et al. 2004; Macpherson et al. 2012) resulting from overwhelming proliferation of pioneer vegetation, post-silvicultural treatments such as liberation thinning, vine cutting or enrichment planting should be applied (Zarin et al. 2007; Peña-Claros et al. 2008).

To avoid further degradation from post-logging extraction of residual wood necromass being converted into charcoal, the eastern Amazonian state of Pará should (1) discourage the use of artisanal charcoal ovens for cleaner, more modern systems, (2) ban natural forest based charcoal production, (3) regulate the production of charcoal from plantation forests and (4) ensure the traceability of charcoal by implementing a chain of custody tracking system that excludes charcoal from converted natural vegetation and from operations using slave labour.

Brazilian Amazonia is also home to 24 million people, representing 12% of the Brazilian population (Pereira et al. 2010). Our results illustrate that greater positive incentives for actor-specific responsibilities would be effective in supporting more sustainable forest management and ensuring long-term viability of reduced impact logging among Amazonian smallholders. These might include increasing rural land registry (*cadastro rural*) for smallholders so that their land uses can be regulated. Capacity building and financial or fiscal incentives to support livelihoods based on sustainable harvests of non-timber forest products, instead of timber may also be beneficial. Additionally, further support from regional governments in providing RIL training and capacity building, and setting up cooperatives to share information and technology are indispensable.

Other studies call for more stringent expectations for largeholders and particularly those operating on public forests should adopt reduced harvest intensities and cutting cycles so that species-specific STY can be harvested in the future (Zarin et al. 2007). Additionally, to pave the way to more financially viable secondary cuts logging enterprises should comply with a "primary forest premium," where once logged forests sustain subsequent yields at a previously agreed lower level, either through lowering the frequency or intensity of harvests, or both (Pinchot 1910; Putz et al. 2012). However, instead of securing a second harvest, once-logged public forests such as National Forests (FLONAs) would benefit from being protected to avoid further logging-induced degradation, and to avoid conversion of private lands, lost profits could be mitigated through REDD+ projects (Chapter 5) (Rice et al. 1997; Wilcove et al. 2013; Fisher et al. 2013; Solar et al. 2015).

Lastly, although most natural timber in Brazil is consumed domestically (ITTO 2012), the role of the global tropical timber trade cannot be ignored. As wealthy developed countries in the northern hemisphere import most of their wood to meet their demand, they are free to accrue net increases in domestic forest cover and externalise their deforestation footprint to developing countries (Kastner et al. 2011). Whilst much of the exported wood in Brazil may be plantation-based, plantation forests may also be encroaching into recently converted natural vegetation. Tropical plantation forests increased by 69% over the last 25 years (FAO 2015). Beef sourced from cattle ranches from recently deforested Amazonian forests has already been highlighted (Greenpeace 2009) but the sustainability and ecological footprint of export plantation forests remains poorly understood.

6.3 Future directions

Many unanswered research questions remain. It is largely unknown how far compositional shifts, such as the dominance of light-wooded fast-growing pioneer species (Phillips et al. 2004; Macpherson et al. 2012) and other small-scale edge effects from logging gaps extend into residual forest stands (Panfil & Gullison 1998). Because biodiversity persistence across gradients of forest degradation also depends on both the distance and quality of nearby primary forests patches, a useful research agenda would be to describe the optimal network (size and spacing) of these unlogged wildlife refuges in the context of a logged forest matrix (Johns et al. 1996; Thiollay 1997). The optimal network may differ among taxa and may be interdependent on largely unknown and complex functions and processes including nutrient decomposition, pollination, predation and seed dispersal (Dobson et al. 2006). More data are needed to fully understand these interdependencies, and avoid biodiversity loss and ecosystem function.

Deforestation rates have begun rising once again across Brazilian Amazonia; between October 2013 and October 2014, a total of 24,400 ha were deforested (Fonseca et al. 2014). In relation to rates in 2013, this represents a deforestation increase of 467% and a 1,070% increase in forest degradation which totalled 46,800 ha in 2014 (Fonseca et al. 2014). Additionally, there has been a weakening in the government's determination to tackle illegal logging between 2010-2015 (Wellesley 2014). Increased conflicts over land use have increased pressures on natural forests. This is largely a result of large-scale mining, agriculture, and infrastructure projects including hydroelectric dams which has led not only to deforestation but inequitable land use and conflicts between government agencies (Davidson et al. 2012). Additional high-level organisation between all appropriate government agencies is urgently needed to safeguard equitable land-use planning. This may include scaling up the 'Green Municipal County Initiative' and effective land use zoning to limit agricultural expansion. These measures must also include policies that incorporate and account for bribery incentives (Amacher et al. 2012). At the time of writing, Brazil has yet to submit its proposed Nationally Determined Contribution to the UNFCCC in October 2015 (IETA 2015). Much of the earlier celebrated environmental success depends on political will, which may diminish in the current political climate (Nazareno 2012; Nepstad et al. 2014; Ferreira et al. 2014; Gibbs et al. 2015).

6.4 Policy and research dissemination

Once the main data chapters of this thesis have been submitted for publication, the aim is to write Chapters 2 and 3 into a Portuguese version and make this available to the Brazilian Forestry Service (Serviço Florestal Brasileiro) and the State Environmental Secretariat of Pará (SEMA). Press releases of Chapters 2 and 3 will also be written up and circulated in the Brazilian media. Finally, we will put together a single synthesis paper in Portuguese for Ciência Hoje, which is widely made available to the Brazilian public, including school teachers, and university students.

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