The impacts of logging on carbon stocks, tree diversity and forest recovery in Vietnam

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The candidate confirms that the work submitted is her own, except where work which has formed part of jointly authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others.

Chapter 2

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Abstract

Tropical forests are important carbon (C) sinks and biodiversity hotspots. Yet, forests are seriously threatened by anthropogenic disturbances including logging, which is the main driver of forest degradation in Southeast Asia. Here, I analyse the impacts of logging intensity on C stocks, tree diversity and forest recovery in seasonal evergreen broadleaf lowland forests in north-central Vietnam. I established, measured and after two years re-measured twenty-four plots that vary in historical logging intensity, nine years after logging. Logging intensity, estimated by combining Landsat analysis and participatory mapping, was the main factor explaining the variability in aboveground carbon (AGC). Heavily logged forests stored only half the amount of AGC in stems ≥10 cm diameter as lightly logged forests, mainly due to a reduction in large trees. AGC of small trees, dead wood and belowground C stored small fractions of total C stocks, while the topsoil of 0–30 cm depth stored roughly half of total C stocks. Tree diversity did not differ significantly with logging intensity. Some compositional differences were observed with logging intensity, although the challenge of sampling at the species level sufficiently complicates interpretation. Community wood density decreased with logging intensity and timber species only occurred in small numbers in our plots, especially in the heavily logged sites. Vital rates and associated AGC fluxes did not differ with logging intensity during the two-year period. The forests suffered from typhoon-induced tree mortality and damage, with ~40% of plots experiencing a net annual reduction in AGC. This study shows the critical importance of reducing logging intensities to maintain high forest C stocks and community wood density and to prevent the depletion of timber species. Recent storm impacts retard the post-logging AGC recovery of these forests.

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Abbreviations

AGB	Aboveground biomass
AGC	Aboveground carbon
B.A.	Basal area
BGB	Belowground biomass
BGC	Belowground carbon
BIC	Bayesian Information Criterion
C	Carbon
CCBA	Climate, Community and Biodiversity Alliance
CEC	Cation exchange capacity
CWD	Coarse woody debris
Dbh	Diameter at breast height
FIPI	Forest Inventory and Planning Institute
FSC	Forest Stewardship Council
HD	Height-diameter
IDH	Intermediate disturbance hypothesis
KNT	Khe Nuoc Trong forest
KW	Kruskal-Wallis rank sum test
Lmm	Linear mixed effects model
Mlm	Multiple linear regression model
NBR	Normalized Burn Ratio
NDVI	Normalized Difference Vegetation Index
NIR	Near Infrared
NMDS	Non-metric multidimensional scaling

PCA	Principal component analysis
POM	Point of measurement
REDD+	Reducing Emissions from Deforestation and forest Degradation, and promoting conservation, sustainable forest management and enhancement of forest carbon stocks
RIL	Reduced impact logging
SOC	Soil organic carbon
SWIR	Shortwave Infrared
Wsg	Wood specific gravity

Chapter 1. General introduction

1.1. Tropical forests and ecosystem services

Tropical forests are of global importance and play a crucial role in carbon (C) storage and climate regulation (Bonan, 2008; Malhi and Grace, 2000), maintenance of biodiversity (Brummitt et al., 2020; Myers et al., 2000), water supply and regulation (Ellison et al., 2017; Sheil, 2014) and other important ecosystem services. Tropical forests store approximately 25% of terrestrial C and can sequester annually large amounts of C (Bonan, 2008). Living biomass (above- and belowground) encompasses roughly half of total C stored in tropical forests, while the remaining C is stored in the soil, dead wood and litter (Pan et al., 2011). Further, global concentrations of terrestrial biodiversity are highest in tropical forests (Connell, 1978; Dirzo and Raven, 2003), including plant diversity (Brummitt et al., 2020; Leigh Jr. et al., 2004). Yet, tropical forests are seriously threatened by anthropogenic disturbances (Curtis et al., 2018; Lewis et al., 2015; Peres et al., 2006; Phillips, 1997), leading to large scale forest loss globally (Hansen et al., 2013).

Deforestation and forest degradation and the subsequent loss of ecosystem services is of global concern. Deforestation involves the conversion of forest to another land-use. Forest degradation implies subtle changes in forest structure rather than wholesale land-use change and is often defined as a human-induced decrease in the capacity of forests to provide ecosystem services (Putz and Redford, 2010; Sasaki and Putz, 2009; Thompson et al., 2013). By deforesting and degrading forests, large amounts of carbon dioxide are released to the atmosphere, thereby contributing to climate change (Le Quéré et al., 2018). Aware of this, nations agreed in 2015 to include a paragraph on the conservation of forests in the Paris global climate agreement, in the form of a mechanism called REDD+ (UNFCCC, 2015; Article 5). REDD+ provides financial incentives for Reducing Emissions from Deforestation and forest Degradation, and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries.

1.2. Deforestation and forest degradation in Southeast Asia

This thesis focusses on Southeast Asia, where about 15% of the world's tropical forests are located (FAO, 1995). However, much of this region has experienced exceptionally high deforestation rates (Achard et al., 2002; Estoque et al., 2019; Hansen et al., 2013; Miettinen et al., 2011; Stibig et al., 2014). The total tropical forest cover of Southeast Asia dropped from 268 million ha in 1990 to 236 million ha in 2010 (Stibig et al., 2014). Another study estimated that Southeast Asia lost approximately 80 million ha forests between 2005 and 2015, of which the majority of the losses (62%) occurred in Indonesia (Estoque et al., 2019)**.** A global analysis showed that C losses resulting from deforestation and forest degradation exceeded C gains from forest growth in tropical America, Africa and Asia, with highest average C loss per area in Asia (Baccini et al., 2017). For the tropics as a whole, forest degradation and disturbances accounted for most C losses from forests, i.e. 69% of the total (Baccini et al., 2017). In Asia, forest degradation was estimated to form 46% of total C emissions from forests across the region (Baccini et al., 2017). Further, Southeast Asia has an exceptional conservation value compared to other tropical regions. This region has the highest proportion of threatened vascular plant, reptile, bird, and mammal species and contains the highest mean proportion of country-endemic bird and mammal species (Sodhi et al., 2010). The devastating effects of habitat loss on Southeast Asian biodiversity is of extreme concern (Sodhi et al., 2010, 2004).

The main causes of tropical forest cover loss in Southeast Asia are the conversion of forest to cash crops plantations, including oil palm, followed by logging and the replacement of natural forests by forest plantations (Stibig et al., 2014). A cross-continental analysis in 46 tropical and sub-tropical countries showed that logging is the main driver of forest degradation and accounted for more than 80% of total degradation in Asia (Hosonuma et al., 2012). Globally, logging is also of major concern. At least 20% of the global humid tropical forest biome has been selectively logged between 2000 and 2005 and selective logging is still increasing in the tropics (Asner et al., 2009). Therefore, understanding the impacts of logging on forest ecology is of major importance.

1.3. Selective logging and changing local environments

Selective logging in the tropics generally targets specific tree species above a certain stem diameter, typically around 40–60 cm diameter but depending on the species (Edwards et al., 2014). This species- and size-specific selection leads directly to the removal of predominantly large trees with high wood density (Macpherson et al., 2012). This implies removing specific high-value timber species (Macpherson et al., 2012), part of the reproductive tree population (Bagchi et al., 2011) and large aboveground carbon (AGC) stocks (Sist et al., 2014; Slik et al., 2013). Logging thins the stand density, thereby reducing competition and creating opportunities for other plants, i.e. the availability of space and resources. Light is often a major limiting factor for plants in tropical forests (Chazdon, 1988; Chazdon et al., 1996). Logging creates forest canopy gaps and changes the light environment in the forest, leading to increased irradiance for understorey plants (Chazdon et al., 1996; Yamada et al., 2014), which can stimulate tree growth and regeneration (Delcamp et al., 2008; Sist and Nguyen-The, 2002). At the same time, surrounding trees experience negative impacts from logging due to collateral damage and tree mortality (Sist et al., 1998; Sist and Nguyen-The, 2002). Often, heavy machinery is used to extract and transport logs and this causes soil compaction (Greacen and Sands, 1980; Pinard et al., 2000), which in turn can limit tree recruitment (Pinard et al., 1996). Careful pre- and post-logging guidelines, so-called reduced impact logging (RIL) practices, can be implemented to reduce damage to the remaining forest stand (Pinard and Putz, 1996; Sist et al., 2003b). However, conventional (unplanned) logging is still the most common logging practice in the tropics (Putz et al., 2000a). Logging can thus have important impacts on forest ecology and dynamics.

1.4. Impacts of logging on forest ecology

Meta-analyses and a review showed that selectively logged tropical forests can retain substantial C and timber stocks, biodiversity values and associated ecosystem functions, despite the damage caused by logging (Edwards et al., 2014; Gibson et al., 2011; Putz et al., 2012). Being the region with the largest area of tropical forests, most studies on logging have been conducted in the Amazon (e.g. Asner et al., 2006; Blanc et al., 2009; Hérault et al., 2010; Keller et al., 2004; Mazzei et al., 2010). Most large-scale regional analyses on the impacts of logging on forest ecology were also conducted for tropical South-American forests (Berenguer et al., 2014; Piponiot et al., 2016; Rutishauser et al., 2015; Sist et al., 2014). In tropical Southeast Asia, most logging studies have been concentrated in Dipterocarp equatorial forests in Borneo (e.g. Pinard et al., 2000; Pinard and Putz, 1996; Rozak et al., 2018; Sist et al., 2003a, 1998). Studies on logging in mainland tropical Southeast Asia and in non-Dipterocarp Asian tropical forests are rare.

Big trees of particular species are usually targeted during selective logging. As big trees store a large fraction of total AGC stocks (Rozak et al., 2018; Sist et al., 2014; Slik et al., 2013), forests typically loose a considerable fraction of their AGC after logging (Berry et al., 2010; Putz et al., 2008). A meta-analysis found that selective logging in forests from different climatic regions reduced AGC stocks globally by 43% (Zhou et al., 2013). A limited amount of studies have assessed C pools beyond AGC in logged forests in Southeast Asia (Nam et al., 2018; Rozak et al., 2018; Saner et al., 2012). Logged forests usually contain more necromass than unlogged forests (Gerwing, 2002; Keller et al., 2004), but this necromass declines over time as decomposition proceeds. Dead wood can store significant fractions of total C stocks in logged forests in Southeast Asia (Pfeifer et al., 2015; Rozak et al., 2018). BGC and soil organic carbon (SOC) stocks were assessed in a few logged forests in Southeast Asia, i.e. in Vietnam, Indonesia and Malaysia (Nam et al., 2018; Rozak et al., 2018; Saner et al., 2012).

Global studies observed a significantly lower tree species richness (Clark and Covey, 2012) and diversity of plants (Gibson et al., 2011) in selectively logged forests compared to primary tropical forests. However, many individual studies found a higher or similar tree diversity in logged than unlogged forests (Berry et al., 2010, 2008; Cannon et al., 1998; Ding et al., 2012; Hall et al., 2003; Imai et al., 2012; Sheil et al., 1999). Usually, the majority of species that are present before logging persists at decreased densities after logging and logging creates additional opportunities for the establishment of immigrant species, leading to higher diversity in disturbed forests (Sheil et al., 1999). In studies in southern China and the Central African Republic, logged and unlogged forests had a similar tree species composition (Ding et al., 2012; Hall et al., 2003), while the tree species composition in Borneo differed between logged and unlogged forests (Berry et al., 2008). A meta-analysis that compared logged with primary forests found that logging may enhance the species richness of early successional trees, but it also tends to reduce the richness of late successional trees (Clark and Covey, 2012). Further, logging-induced compositional changes can change community wood density (Verburg and Van Eijk-Bos, 2003) and valuable timber species of harvestable size can become depleted after logging (Berry et al., 2008; Richardson and Peres, 2016).

Studies in French Guiana and Indonesia found that tree growth and recruitment in logged forests was generally higher than in primary forests (Delcamp et al., 2008; Hérault et al., 2010; Sist and Nguyen-The, 2002). The first few years after logging, the death of trees that were injured during the logging operations often led to higher mortality rates in logged forests compared to unlogged forests, but thereafter similar mortality rates were observed in logged and unlogged forests (Blanc et al., 2009; Sist and Nguyen-The, 2002; Yamada et al., 2013). Various studies assessed AGC dynamics after logging in Southeast Asian forests (Berry et al., 2010; Do et al., 2018; Nam et al., 2018). Shortly after logging, forests often experience net AGC losses due to high mortality of damaged trees, before returning to net AGC sequestration (Blanc et al., 2009; Mazzei et al., 2010). AGC net change rates in logged forests are usually higher than in unlogged forests (Berry et al., 2010; Roopsind et al., 2017), likely because remaining trees experience less competition (Villegas et al., 2009). It can take decades to more than a century for logged forests to recover their pre-logging AGC stocks (Blanc et al., 2009; Pinard and Cropper, 2000; Rutishauser et al., 2015).

Logging can thus lead to significant changes in C stocks, tree diversity, species composition and demographic processes (growth, recruitment and mortality). While most studies compared logged with unlogged forests, few studies specifically considered variations in logging intensity.

1.5. The role of logging intensity

Logging intensity is often quantified as the number of harvested trees or volume of extracted wood per hectare. The logging intensity in selectively logged forests usually varies from 1 to 20 trees ha^{-1} (Putz et al., 2008) and 1 to 220 m^{3} wood logged ha^{-1} (Martin et al., 2015; Putz et al., 2012). Several meta-analyses identified that logging intensity was the major parameter in determining post-logging AGC stocks (Martin et al., 2015; Zhou et al., 2013), tree species richness (Martin et al., 2015) and AGC recovery (Piponiot et al., 2016; Rutishauser et al., 2015) in tropical forests.

AGC typically declines with logging intensity (Gerwing, 2002; Martin et al., 2015; Rozak et al., 2018; Zhou et al., 2013). In forests in Brazil and Indonesia, the amount of necromass increased with logging intensity (Gerwing, 2002; Rozak et al., 2018). In Indonesian forests, belowground carbon (BGC) decreased with logging intensity, but no relationship was observed between logging intensity and SOC (Rozak et al., 2018).

A meta-analysis in logged forests found that tree species richness increased at low logging intensities but decreased at higher intensities (Martin et al., 2015), consistent with predictions of the intermediate disturbance hypothesis (Connell, 1978). In Vietnam, tree composition varied between heavily and lightly logged forests (Hoang et al., 2011). Usually, heavily disturbed forests harbour more pioneer species and species of low conservation value, while shade-tolerant species and valuable timber trees are most common in lightly disturbed sites (Bongers et al., 2009; Carreño-Rocabado et al., 2012; Hoang et al., 2011). In a tropical forest in Bolivia, the community changed towards species with low wood density with an increase in disturbance intensity, caused by logging and silvicultural treatments (Carreño-Rocabado et al., 2012). Timber trees are generally more abundant in lighter logged forests than in heavily logged sites (Hoang et al., 2011, 2008).

Vital rates, i.e. the growth rates of surviving trees and tree population recruitment and mortality rates, all increased with logging intensity soon after (2-4 years) and long after (18 years) logging in forests in Borneo, probably related to enhanced resource availability and more injured trees at high logging intensity (Lussetti et al., 2016; Sist and Nguyen-The, 2002). After an initial period of AGC losses directly after logging, AGC recovery rates increased with logging intensity in forests in the Central African Republic and Central Guyana (Gourlet-Fleury et al., 2013; Roopsind et al., 2018). In Amazonian forests, logging intensity was the major variable related to the speed of post-logging AGC recovery (Piponiot et al., 2016; Rutishauser et al., 2015). However, remarkably, no studies have yet assessed the impacts of logging intensity on AGC recovery in Southeast Asian forests, in spite of the potential importance of this in terms of managing national C emissions and sequestration for countries in this region. Especially little research has been carried out in logged forests in mainland Southeast Asia. This study focusses on Vietnam, a large tropical country in the region which covers ~331,000 km^2 and was once almost entirely forested. Natural forests here have experienced extensive deforestation and forest degradation over recent decades.

1.6. Forests in Vietnam

In 2012, forests covered 41% of Vietnam's land area, representing a total forest area of 13.9 million ha, which consists for 10.4 million ha of natural forests and 3.4 million ha of planted forests (MONRE, 2014). Vietnam's most typical vegetation is monsoon tropical forest and its vegetation can be grouped into eight main types (Averyanov et al., 2003; Fig. 1.1). Plant diversity in Vietnam is high, containing at least 11,373 vascular plant species and ~3% of the plant genera is endemic (MONRE, 2008). Vietnam's richness of biological diversity results from various factors. Firstly, Vietnam's elongated shape covers a wide range of latitudes, which encompasses a large range of climates (Sterling and Hurley, 2005). Secondly, ~75% of the land area of Vietnam is hilly and mountainous (Averyanov et al., 2003; MONRE, 2014) and this topography impacts the distribution of species and biotic communities by mediating temperature and humidity (Sterling and Hurley, 2005). The Truong Son Range (Annamite

Range), Vietnam's major topographic feature, runs from north to south along the Vietnamese border with Laos and Cambodia and into south-central Vietnam (Averyanov et al., 2003). It forms an important barrier between the moist uplands of Vietnam and the drier monsoon forests of Laos and Cambodia and passes through the transition zone between the subtropical northern and tropical southern climates of Vietnam (Sterling and Hurley, 2005). Further, Southeast Asia's complex geological and climatic past also contributed to Vietnam's diversity (Sterling and Hurley, 2005). Forests in Vietnam play also an important role in C storage. A recent large-scale study including 52 locations in undisturbed old-growth forests throughout Vietnam showed that country-wide averages of AGC of stems ≥10 cm diameter were equal to 108 Mg C ha^{-1}, with ~80% of the plots between 50 and 140 Mg C ha^{-1} (T.V. Do et al., 2019).

All forests in Vietnam have been classified based on timber reserves of standing trees, i.e. poor (10–100 m^3 ha^{-1} timber volume), medium (101–200 m^3 ha^{-1}) and rich forests (201–300 m^3 ha^{-1}) (MARD, 2009). According to management purposes, forests in Vietnam can either be production, protection or special use forests (MARD, 2018). Production forests are designated for timber supply. Protection forests include sites with important protection functions, such as watershed and coastal areas. Special use forests include sites that are important for biodiversity conservation, such as national parks and protected areas.

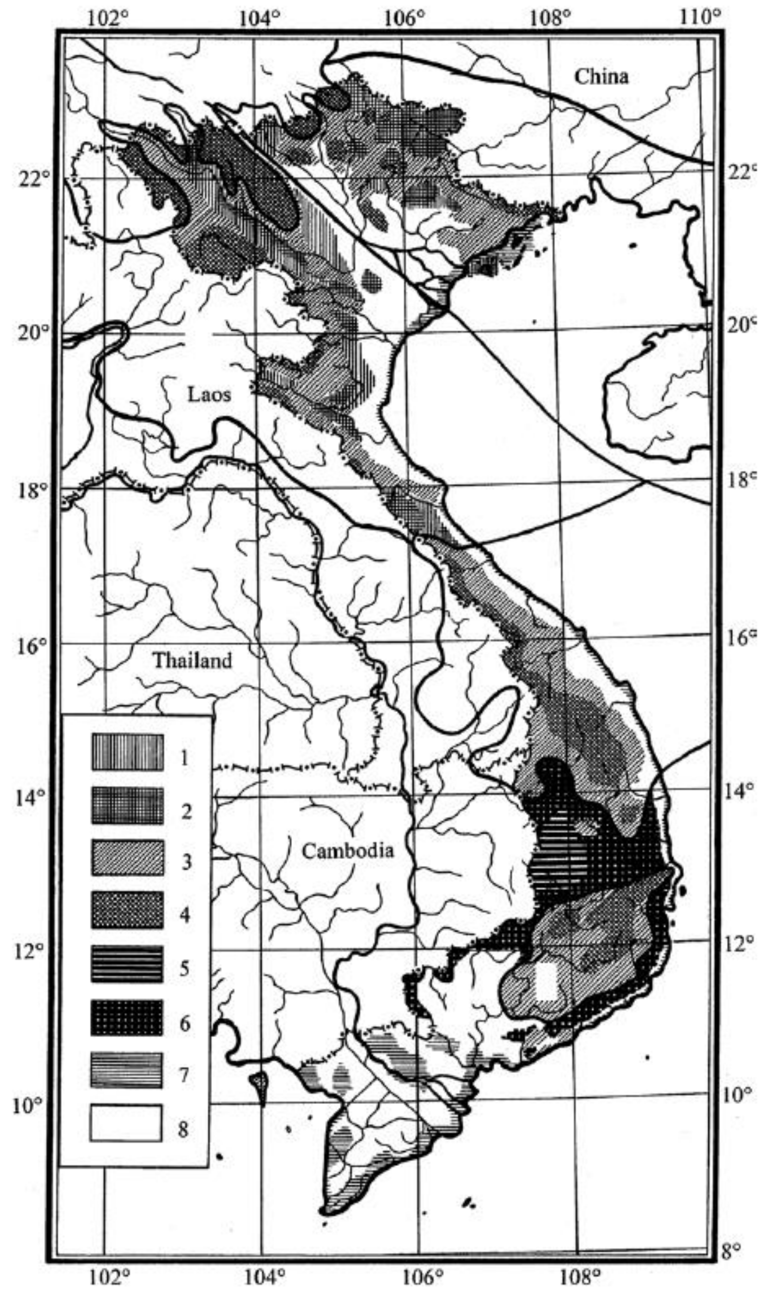


Figure 1.1. Map of the main vegetation types in Vietnam.

Map constructed by Averyanov et al. (2003). 1. Evergreen broadleaf plane forests on alkaline soils. 2. Evergreen and semi-deciduous broadleaf, mixed and coniferous limestone mountain forests. 3. Evergreen lowland forests on silicate rocks at 0-1000 m a.s.l. 4. Evergreen montane and highland forests on silicate rocks at 1000-3000 m a.s.l. 5. Semi-deciduous dry lowland forests. 6. Deciduous dry lowland forests and savanna-like woodlands. 7. Coastal vegetation, lowland wetlands and mangrove thickets. 8. Secondary, weed and agricultural plant communities, timber and industrial plantations. All types of native primary vegetation are adversely affected by anthropogenic disturbances or converted over most of the country (Averyanov et al., 2003).

1.7. Historical forest cover changes in Vietnam

Between 1943 and 1993 much of Vietnam's forests were cleared, with forest cover declining from an estimated 43% to 28% (MARD, 2018; McElwee, 2004). Extensive areas of forests were severely damaged during the Vietnam War between 1955 and 1975, in which massive amounts of chemicals were used to defoliate forests and destroy crops in large parts of Vietnam (Orians and Pfeiffer, 1970). The greatest decline in forest cover in Vietnam happened between 1976 and 1990: starting in the 1970s, wood was largely needed for war materials and after the war it was demanded as reconstruction material to rebuild houses (McElwee, 2004). In addition to the domestic demand, exporting wood was one of the rare opportunities to earn money in the post-war years (McElwee, 2004). The state nationalization of all forest resources from 1954 to the 1980s contributed considerably to illegal logging, because all locally-used forests were considered as national property (McElwee, 2004). During this period, there was a generalized "free-for-all" on the forests by both the state and local communities, thereby contributing to a massive breakdown of local tenure rules and resource allocation. Started in the late 1980s, many of the state-owned logging reserves were dissolved and this land was distributed to local communities, but deforestation and forest degradation still continued (McElwee, 2004).

In the 1990s, Vietnam's forest cover increased significantly and Vietnam experienced a forest transition: a national-scale shift from net deforestation to net reforestation (Meyfroidt and Lambin, 2008a). A combination of internal political, socioeconomic and land-use processes and the beginning of large-scale reforestation programs contributed to this reforestation (MARD, 2018; Meyfroidt and Lambin, 2008b). The most recent major initiative to restore Vietnam's forest cover was the "5 million ha reforestation project", implemented between 1998 and 2010 (de Jong et al., 2006). This reforestation project focussed mainly on establishing new plantations instead of supporting natural regeneration and has replaced diverse, often degraded, native flora by mono-plantations of exotic trees (McElwee, 2009). Although the forest cover has increased during recent decades, the quality of the forests has decreased: the area of degraded forests has increased and, even though new forests have been planted, forest density overall has declined (Pham et al., 2012). Thus, only increasing the forest area through reforestation is not sufficient to restore forest ecosystem functions and services, especially C stocks (Pham et al., 2012). From 1993 to 2013, forest cover changes varied considerably among regions (Cochard et al., 2017). During this period, natural forest regrowth was highest in Vietnam's northern mountain provinces, while deforestation continued in the Central Highlands and Southeast Region. Highest increases in forest plantations were observed in mid-elevation provinces of the Northeast and Central Coast Regions (Cochard et al., 2017).

Historical logging in Vietnam has included both legal exploitation of natural forests by government-licensed logging operations and illegal logging (MARD, 2018). In 1955, the first forest harvesting regulation was introduced in the north of Vietnam (Nam, 2017). Since then, selective logging regulations in Vietnam have been revised multiple times (Nam, 2017). To reduce the exploitation of forests, the state enacted an export ban on raw logs from Vietnam during the 1990s, but without much effect (McElwee, 2004). Since 2014, all logging in natural forests in Vietnam is illegal (MARD, 2018; pers. comm. Trai Trong Le, 2019). However, illegal logging is a major driver of forest degradation in Vietnam (McElwee, 2004) and was previously estimated at 1 million m^3 $year^{-1}$ in total (Ogle et al., 1998). While the state blames local people for illegal logging in Vietnam, various problems, i.e. land tenure insecurity, poverty, ineffective state management, poorly enforced logging bans and endemic corruption, have all contributed to the problem (McElwee, 2004). A major problem is that the private market for logs has created incentives for both large-scale and small-scale illegal logging operations throughout the country (McElwee, 2004). Further, displacement of deforestation to neighbouring countries remains a large problem in Vietnam (Meyfroidt and Lambin, 2009).

1.8. REDD+ in Vietnam

To reduce deforestation and forest degradation, Vietnam has been extensively involved in international discussions and preparations to implement a REDD+ scheme (Hoang et al., 2019; Pham et al., 2012). The nation was identified as one of the first pilot countries under the United Nations REDD (UN-REDD) programme and the World Bank's Forest Carbon Partnership Facility (FCPF), the world's best-known proponents of REDD+ (Pham et al., 2012). Nowadays, Vietnam has a national REDD+ action programme that runs until 2030 and several REDD+ projects have been implemented at the local level (Hoang et al., 2019). Vietnam's emission reductions programme of the FCPF encompasses the entire north-central coast region, spanning six provinces and representing approximately 16% of the land area and 12% of the total population (MARD, 2018). This region was selected based on its critical biodiversity importance and socio-economic status: it includes five important conservation corridors and has the country's highest rates of poverty per capita (MARD, 2018).

For effective implementation of REDD+ projects, it is essential to have a thorough understanding of the drivers, extent and spatial distribution of deforestation and forest degradation. Vietnam's main direct drivers of deforestation and forest degradation are land conversion for agriculture, infrastructure development, logging and forest fire (Pham et al., 2012). Important indirect drivers are the pressure of population growth and migration, the state's weak forest management capacity and the limited funding available for forest protection (Pham et al., 2012). Drivers and their impacts vary between regions and over time, suggesting that the design and implementation of individual REDD+ programmes need to be adjusted locally (Pham et al., 2012). Forests are not evenly distributed across Vietnam, neither are deforestation and forest degradation. Between 2000 and 2010, deforestation and forest degradation were larger in the provinces with larger areas of forests and at higher elevations (Khuc et al., 2018). During that time, the largest area of deforestation and forest degradation was in the north-central region of Vietnam, followed by the northeast, central highland, northwest and south-central regions (Khuc et al., 2018). A recent analysis in a protected forest in north-central Vietnam suggests that activities to reduce forest degradation are likely to be financially viable through a REDD+ program (Ngo et al., 2020). Further, REDD+ projects require detailed data on the impacts of deforestation and forest degradation on forest ecosystem services such as carbon storage and harbouring biodiversity. Given that Vietnam has a long history of state and illegal logging and that logging is a key driver of forest degradation in Vietnam's REDD+ focus region, i.e. the north-central coast (MARD, 2018), I focus in this thesis on the impacts of logging on forests in north-central Vietnam.

1.9. Impacts of logging on Vietnamese forests

Various allometric equations have been developed to estimate tree aboveground biomass (Huy et al., 2016a, 2016c, 2016b; Kralicek et al., 2017; Nam et al., 2016) and belowground biomass (Kralicek et al., 2017; Nam et al., 2016) for evergreen broadleaf forests and Dipterocarp forests in Vietnam. For Vietnamese old-growth forests, AGC stocks were estimated in large-scale studies for evergreen broadleaf forests and dry Dipterocarp deciduous forests throughout the country (Con et al., 2013; T.V. Do et al., 2019). Further, AGC stocks were estimated for poor, medium and rich forests that experienced different degrees of degradation from various disturbance types, including logging, in evergreen broadleaf forests and dry open Dipterocarp forests in the Central Highlands (Hai et al., 2015; Luong et al., 2015). Only one study quantified AGC stocks in logged Vietnamese forests, i.e. in evergreen broadleaf forests in the Central Highlands (Nam et al., 2018). Carbon pools beyond AGC have been rarely assessed for Vietnamese forests. Nam et al. (2018) is the only study that estimated C in dead wood, BGC and SOC in logged forests in Vietnam.

A range of studies assessed tree diversity and composition in evergreen broadleaf forests, various types of Dipterocarp forests and other forest types throughout Vietnam (Blanc et al., 2000; Con et al., 2013; Do et al., 2017; Van and Cochard, 2017), of which only a few were conducted in logged forests (H.T.T. Do et al., 2019; Hoang et al., 2011; Tran et al., 2005). Vital rates and associated AGC dynamics at the population level have been very rarely studied in Vietnam. The only work on this in logged Vietnamese forests has been performed in evergreen broadleaf forests in the Central Highlands (Do et al., 2018; Nam et al., 2018).

Vietnam has a long history of state and illegal logging. Several studies assessed C stocks, tree diversity and AGC dynamics in old-growth Vietnamese forests, yet only limited research has been performed in logged forests. This thesis and a study derived from this research (Stas et al., 2020) is the first work to evaluate the impacts of the intensity of logging on C stocks, tree diversity and forest recovery in Vietnam. This is also one of the few studies that has been conducted in logged evergreen broadleaf forests in Vietnam.

1.10. Study site

Khe Nuoc Trong forest (KNT) is located in Le Thuy District in Quang Binh Province in north-central Vietnam (17.0°N, 106.7°E). Quang Binh is one of the six provinces that is included in

Vietnam's emissions reductions programme of the FCPF (MARD, 2018). KNT covers approximately 20,000 ha of evergreen tropical forests in the Annamese mountain range and is part of a larger natural forest network of approximately 500,000 ha (Fig. 1.2) (Department for agriculture and rural development, 2010). KNT contains one of the last remaining extensive and relatively pristine lowland evergreen tropical forests in Vietnam (Birdlife International, n.d.). KNT is a key biodiversity area: previous research listed 987 vascular plant species and 227 terrestrial vertebrate species, with a significant amount of threatened species (Department for agriculture and rural development, 2010) and the Annamese lowlands are known as an endemic bird area (Birdlife International, n.d.). Elevation in KNT ranges from 120 m to 1220 m, with the majority of the area (90%) <700 m altitude on hilly terrain (Department for agriculture and rural development, 2010). Climate conditions are similar to the neighbouring Bac Huong Hoa Nature Reserve, where the area has a tropical monsoon climate with hot summers and relatively cold winters and storms from June to September (Mahood and Hung, 2008; pers. comm. Trai Trong Le, 2019). The hot season is from March to June, when the area receives a hot, dry wind originating from Laos ("Foehn"), with average temperatures of 29 °C in June and July and frequent temperatures up to 39 °C when the Foehn is blowing (Mahood and Hung, 2008). In December and January, temperatures can drop to 15 °C (>500 m elevation). Total annual rainfall varies between 2400 and 2800 mm with highest rainfall from August to November. The drier period lasts only for a few months per year, i.e. between December and March (Mahood and Hung, 2008). KNT is surrounded by five communes and villages within three communes have easy access to KNT (Fig. 1.2) (Ngo et al., 2020).

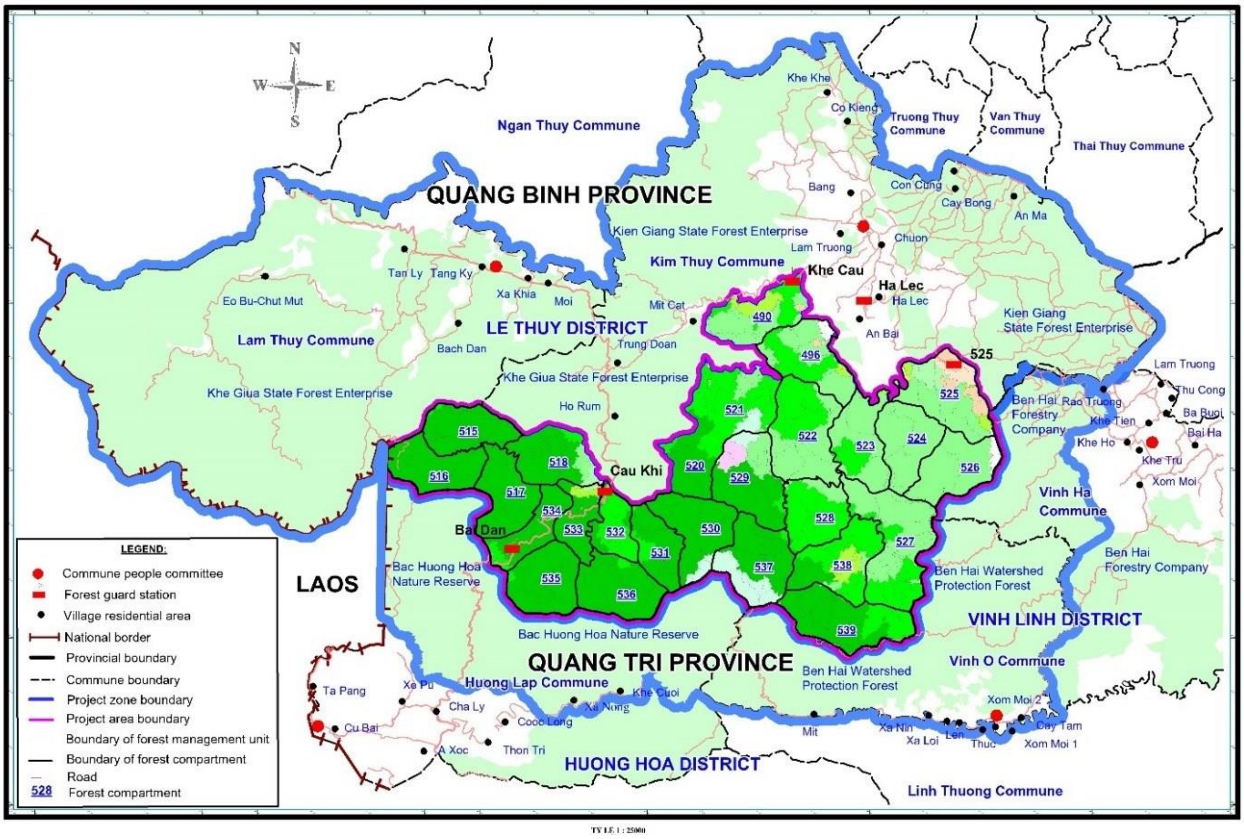


Figure 1.2. Map of the project area in north-central Vietnam.

Map produced by Ngo et al. (2020). Khe Nuoc Trong forest is highlighted in green, the shading represents the timber volumes according to the national classification system (rich: dark green, medium: bright green, poor: light green). The proposed extended boundary for the future KNT nature reserve has been shown, together with a part of the neighbouring natural forest network. Villages are marked with black circles.

KNT was selectively logged by the state between 1982 and 2007. Since 2007, the forest has been officially protected as a watershed protection forest and logging is forbidden, but widespread illegal logging still occurs (illegal logging also happened during the state logging period). In recent years, logging contributed more than half of the total income that forest-adjacent communities derived from natural forests in KNT, resulting in ongoing forest degradation (Ngo et al., 2020). The forest contains areas with various levels of forest degradation, from relatively undisturbed to heavily degraded forests. KNT consists for 40%

(8030 ha) of rich forests; 25% (4930 ha) of medium forests and 31% (6188 ha) of poor forests (forest classifications have been described in Chapter 1.6; Birdlife International, n.d.). KNT has not been affected by the defoliants during the Vietnam War. To enhance protection of KNT's valuable forests, it was proposed to designate KNT as a nature reserve in the national special use forest system (Birdlife International, n.d.; Department for agriculture and rural development, 2010). This proposal is currently under discussion.

1.11. Objectives and thesis outline

The main objectives of this study are to assess the impacts of logging intensity on C stocks, tree diversity and forest recovery in Khe Nuoc Trong forest in north-central Vietnam. The specific objectives are:

1) To evaluate the spatial pattern of historical logging extent and intensities in KNT.
2) To quantify forest C stocks in KNT and determine the extent to which AGC correlates with logging intensity.
3) To assess whether tree diversity, tree species composition, community wood density and availability of timber species differ with logging intensity.
4) To analyse whether vital rates and associated AGC dynamics differ with logging intensity and evaluate if the forests, nine years post-logging, are recovering.

In Chapter 2, I present an approach in which we combine Landsat analysis and participatory mapping to estimate historical logging intensities for KNT. I quantify the main C pools, i.e. living aboveground carbon, dead wood, belowground carbon and soil organic carbon, in logged forests in KNT. Further, I explore correlations between logging intensity, soil, topography and current estimates of AGC stocks.

In Chapter 3, I assess the impacts of logging intensity on tree diversity and composition in KNT. Further, I analyse whether community wood density and the availability of timber species differ with logging intensity.

In Chapter 4, I assess the impacts of logging intensity on vital rates and associated AGC dynamics in KNT nine years after logging, assessed over a two-year period. This allows me to evaluate if the forests are currently recovering from past disturbances.

In Chapter 5, I summarize the main findings of this thesis, discuss the implications for forest management and propose directions for future research.

Chapter 2. Logging intensity drives variability in carbon stocks in lowland forests in Vietnam

Abstract

Forest degradation in the tropics is generating large carbon (C) emissions. In tropical Asia, logging is the main driver of forest degradation. For effective implementation of REDD+ projects in logged forests in Southeast Asia, the impacts of logging on forest C stocks need to be assessed. Here, we assess C stocks in logged lowland forests in north-central Vietnam and explore correlations between logging intensity, soil, topography and living aboveground carbon (AGC) stocks. We present an approach to estimate historical logging intensities for the prevalent situation when complete records on logging history are unavailable. Landsat analysis and participatory mapping were used to quantify the density of historical disturbances, used as a proxy of logging intensities in the area. Carbon in AGC, dead wood, belowground carbon (BGC) and soil (SOC) was measured in twenty-four 0.25 ha plots that vary in logging intensity, and data on recent logging, soil properties, elevation and slope were also collected. Heavily logged forests stored only half the amount of AGC of stems ≥10 cm dbh as lightly logged forests, mainly due to a reduction in the number of large (≥60 cm dbh) trees. Carbon in AGC of small trees (5-10 cm dbh), dead wood and BGC comprised only small fractions of total C stocks, while SOC in the topsoil of 0-30 cm depth stored ~50% of total C stocks. Combining logging intensities with soil and topographic data showed that logging intensity was the main factor explaining the variability in AGC. Our research shows large reductions in AGC in medium and heavily logged forests. It highlights the critical importance of conserving big trees to maintain high forest C stocks and accounting for SOC in total C stock estimates.

2.1. Introduction

The world's forests play a crucial role in carbon (C) storage and regulating the climate (Bonan, 2008; Malhi and Grace, 2000). By deforesting and degrading forests, a substantial amount of carbon dioxide is released to the atmosphere, thereby contributing to global warming (Le Quéré et al., 2018). With increasing awareness of the importance of forests in mitigating climate change, a mechanism was developed in which financial incentives are provided for Reducing Emissions from Deforestation and forest Degradation, and promoting conservation, sustainable forest management and enhancement of forest C stocks (REDD+) in developing countries (Angelsen et al., 2009).

Forest degradation and disturbances account for 69% of total C losses from tropical forests (Baccini et al., 2017). Degradation can be broadly defined as a human-induced decrease in the capacity of an ecosystem to provide services (Putz and Redford, 2010; Sasaki and Putz, 2009; Thompson et al., 2013). Cross-continental analyses of 46 tropical and sub-tropical countries showed that degradation mainly results from timber extraction and logging, forming over 80% of all sources of forest degradation in Asia (Hosonuma et al., 2012).

Logging can have considerable negative impacts on the environment. Selective logging consists of harvesting the largest, and often oldest, trees of certain species. The felling and extraction of those large trunks generates substantial collateral damage to the surrounding stand (Sist et al., 1999; Sist and Nguyen-The, 2002), thereby affecting soil (Pinard et al., 2000), tree regeneration (Bagchi et al., 2011; Fredericksen and Mostacedo, 2000) and overall C stocks (Rozak et al., 2018). Yet, selectively logged forests can still contain substantial C and timber stocks, biodiversity and associated ecosystem functions (Putz et al., 2012; Edwards et al., 2014).

In several meta-analyses, the logging intensity, i.e. the number of harvested trees or volume of extracted wood per hectare, appeared to be the most important parameter in determining post-logging biomass stocks (Martin et al., 2015; Zhou et al., 2013) and biomass recovery (Piponiot et al., 2016; Rutishauser et al., 2015). Aboveground living carbon (AGC) typically declines with logging intensity (Gerwing, 2002; Martin et al., 2015; Rozak et al., 2018; Zhou et al., 2013). The amount of necromass and trees damaged by logging usually temporarily increases with logging intensity and can represent significant amounts of total C stocks in logged forests (Gerwing, 2002; Rozak et al., 2018; Sist et al., 1998). Logging intensity was also found to be the main factor explaining the variability in belowground living carbon (BGC) stocks (i.e. roots) (Rozak et al., 2018).

While most studies have focused on AGC, living C (above- and belowground) encompasses about half of total C stored in tropical forests (Pan et al., 2011). Other important C pools such as necromass, litter, BGC and soil organic carbon (SOC) (IPCC, 2006; Pan et al., 2011), are often not included in forest C stock assessments. Only a few studies in Southeast Asia have investigated the effects of logging on C pools beyond AGC (Nam et al., 2018; Pfeifer et al., 2015; Rozak et al., 2018; Saner et al., 2012).

Vietnam experienced intense state logging after the end of the Vietnam War (referred to as American War in Vietnam; McElwee, 2004). Between 1943 and 1993 much of the country's forests were cleared, with forest cover declining from 43% to 28% (MARD, 2018). Since then, forest cover has increased due to the expansion of plantations and natural forest regeneration. At the same time, remaining natural forests have become increasingly degraded through ongoing logging (Cochard et al., 2017; de Jong et al., 2006; Meyfroidt and Lambin, 2008a).

This shift from net deforestation to net reforestation in Vietnam in the 1990s is the so-called "forest transition" (Meyfroidt and Lambin, 2008a).

Vietnam has pledged strong commitment to REDD+ (MARD, 2018; Pham et al., 2012), requiring a sound understanding of how logging impacts forest C stocks. Several forest biomass and allometric model studies have been conducted in Vietnam (e.g. Con et al., 2013; Huy et al., 2016a, 2016b, 2016c; Kralicek et al., 2017), but only a few were carried out in logged forests (Hai et al., 2015; Luong et al., 2015; Nam et al., 2018, 2016). Successive logging bans implemented by the Vietnamese government means that since 2014 all logging in natural forests in Vietnam is illegal (MARD, 2018; pers. comm. Trai Trong Le, 2019). Limited knowledge of the extent of logging and the impacts on forest C stocks in Vietnam hinders proper implementation of REDD+ activities, notably a reduction in forest degradation through better forest management.

In Vietnam and most tropical forest countries, detailed information on logging history is often lacking. Satellite data can be used to identify historical forest disturbances resulting from selective logging. Although this is challenging, recent work has shown it is possible with high to medium spatial resolution (5-30 m) satellite images (Miettinen et al., 2014). Images from the Landsat satellites have been widely used to identify forest degradation associated with selective logging (Asner et al., 2004a; Hirschmugl et al., 2014; Langner et al., 2018; Souza et al., 2013). Although it is not possible to discriminate individually harvested trees using the 30 m resolution of Landsat images, the image resolution is sufficient to identify disturbances associated with selective logging such as logging roads, access tracks, skid trails and log decks (Asner et al., 2004a).

Here, we assess C stocks in four main pools, namely AGC, aboveground necromass, BGC and SOC, in logged lowland forests in north-central Vietnam. We present a novel approach, which combines Landsat analysis and participatory mapping, to estimate historical logging intensities for the prevalent situation when records on logging history are unavailable. Finally, we explore possible correlations between past logging intensities, topography, soil and current estimates of AGC stocks.

2.2. Material and methods

2.2.1. Site description

Data were collected in the logged forests of Khe Nuoc Trong (KNT), located in north-central Vietnam. The site has been described in detail in Chapter 1.10.

2.2.2. Field measurements

2.2.2.1. Plot establishment

Twenty-four permanent plots of 0.25 ha each (50 m x 50 m on the ground), 6 ha in total, were established in lowland forests (<700 m elevation) between April 2016 and June 2017. As the poor, medium and rich lowland forests cover similar percentages of area in KNT, sample plots were equally and spatially stratified by the three forest types (n=8 per class) (Fig. 2.1). Within each stratum, sampling was directed towards locations that cover the gradient of living aboveground biomass (AGB) values estimated by Baccini's pantropical biomass map (Baccini et al., 2012; access data via http://whrc.org/publications-data/datasets/pantropical-national-level-carbon-stock/) and constrained to accessibility. Within the selected pixels on Baccini's map, plots were randomized in the field.

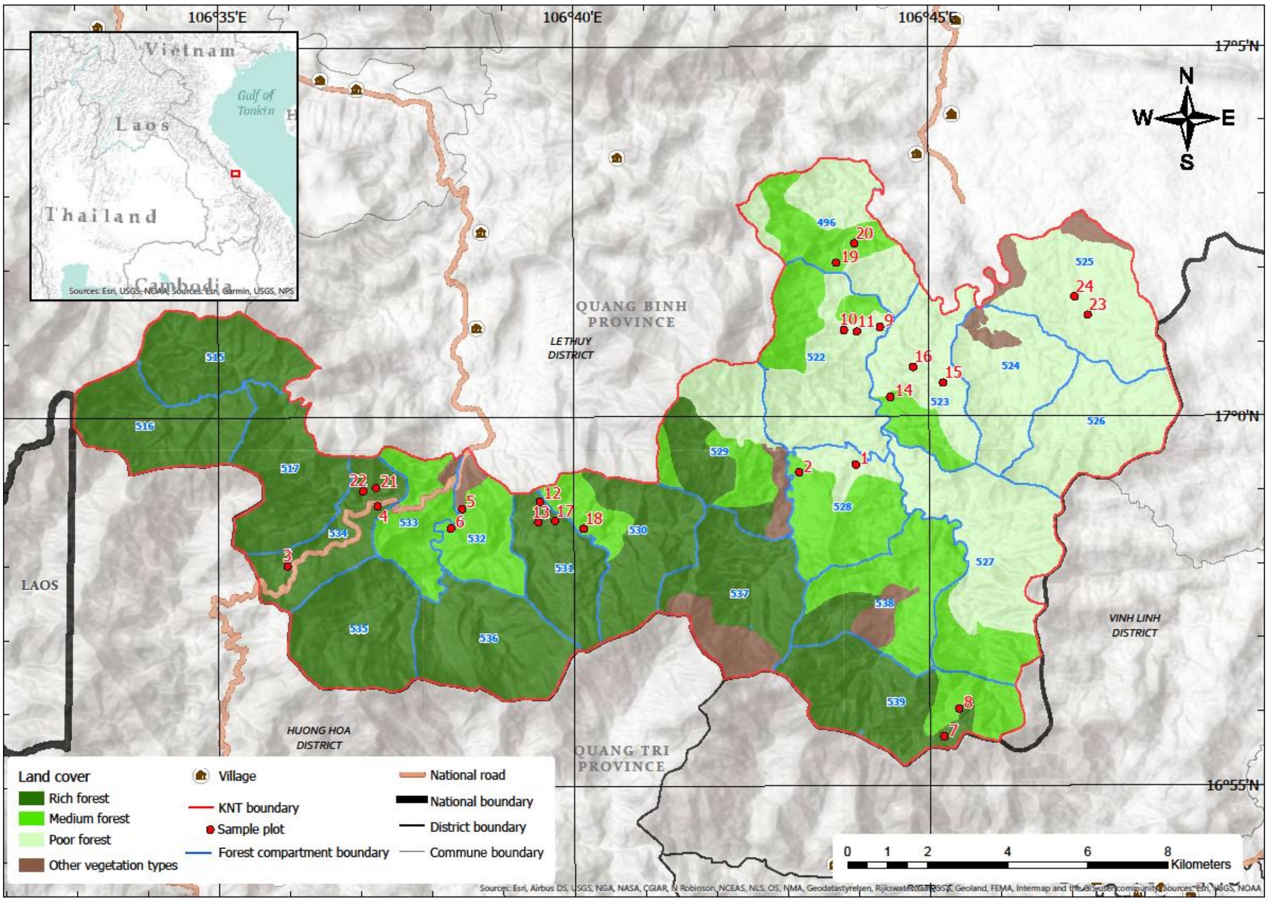


Figure 2.1. Vegetation map of KNT with the locations of the 24 sample plots.

The green shading represents the timber volumes according to the national classification system (rich: dark green, medium: bright green, poor: light green). The red line shows the boundary of KNT. Sample plots are marked with red circles. Map produced by the Forest Inventory and Planning Institute (FIPI), 2010.

A previous study in logged forests in Vietnam showed that AGC and BGC of woody stems 1 cm ≤ dbh < 10 cm stored 1.3% of total C stocks, while shrubs (including seedlings with stem <1 cm dbh) stored only 0.2% of total C stocks (Nam et al., 2018). In logged forests in Vietnam and Indonesia, litter comprised only 0.2-1.6% of total C stocks (Nam et al., 2018; Rozak et al., 2018) and did not differ with logging intensity (Rozak et al., 2018). For these reasons, we focus on the dominant C stocks: AGC in woody stems ≥5 cm dbh, C in dead wood, BGC and SOC.

2.2.2.2. Living stems

Measurements in the plots were conducted according to the RAINFOR and GEM protocols (Marthews et al., 2014; Phillips et al., 2016; http://www.rainfor.org/en/manuals). Streamlining our data collection methods with frequently used plot sampling protocols enables us to include our plot data in global databases of forest plots, such as ForestPlots.net (Lopez-Gonzalez et al., 2011). Such networks facilitate large-scale analyses, which are crucial for improving our understanding of regional and global ecological dynamics. In each 0.25 ha plot, living woody stems (trees, shrubs and stranglers, but excluding lianas) ≥10 cm diameter at breast height (dbh; i.e. diameter at 130 cm height or above buttresses or stem deformities, but below major branching points) were tagged and their dbh measured. A botanist identified each stem with Vietnamese and scientific species names in the field and, when needed, collected botanical vouchers for further identification in the herbarium of FIPI in Hanoi, Vietnam. Total living tree heights were measured with a Haga altimeter. Heights of deviating stems, i.e. broken, fallen, leaning, bended, rotten, hollow, resprouted, forked ≤1.3 m height, fluted stems and stranglers, were measured for each individual separately. For trees without these deviations, i.e. “intact” trees, heights were measured from a subset of trees to construct a local height-diameter (HD) model. For each plot, we randomly selected ~10 intact trees in each of the following diameter classes (if this amount of trees was present): 5-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, ≥40 cm dbh and the biggest stems. Living saplings, i.e. stems 5-10 cm dbh, were measured in a belt transect of 4 m x 50 m, running from north to south through the centre of each plot. The dbh, height and species were measured and identified as described above.

2.2.2.3. Dead wood

All fallen and standing aboveground dead wood, i.e. woody stems and branches ≥10 cm dbh (including logged stumps), were measured in each 0.25 ha plot. For each fallen piece (often referred to as coarse woody debris; CWD), the diameter at both extremities, length, shape, a visual estimation of % wood (subtracting hollow spaces) and degree of wood decay was recorded (Baker and Chao, 2011). For fallen pieces that were too heavy to lift to measure the diameter with a diameter-tape around the stem or branch, the length of two perpendicular sides was recorded. The degree of wood decay was classified in the field into one of five decomposition classes, based on simple wood characteristics (Appendix 2.A1; Baker and Chao, 2011). Measurements were only made to the point where the piece was at least 10 cm diameter and length measurements were made to the plot borders. For standing dead wood, the same measurements were done.

2.2.2.4. Soil collection

In each plot, soil samples were collected for 1) chemical analyses and 2) bulk density, both in two soil layers: 0-10 cm depth and 10-30 cm depth. Each sample used for chemical analyses was collected with a soil auger from five sampling points within the plot and equal amounts were mixed to obtain a representative plot sample of at least 600 g. In a few cases, the soil was too stony to collect the samples with a soil auger and samples were collected by digging a hole and taking equal amounts of soil over the total depth. Bulk density samples were collected close to the centre of each plot by placing a core of 100 cc volume vertically in the 0-10 cm and 10-30 cm depth layers. Within a few days after collection, samples for the chemical analyses were air-dried. The soil samples were analysed for the following properties: pH, SOC, total N, total P, available P, total K_2O, exchangeable bases (K^+, Na^+, Ca^{2+}, Mg^{2+}), cation exchange capacity (CEC), exchangeable H^+ and Al^{3+}, texture and bulk density. Methods of the soil laboratory analyses are described in Appendix 2.B.

2.2.2.5. Topography

For each plot, the mean elevation was calculated from the four plot corners and the centre, recorded with a GPS device. The 50 m x 50 m plot was divided into subplots of 10 m x 10 m and the slope of each 10 m side of a subplot was measured. Subsequently, the maximum absolute slope of the north-south and east-west measurement was calculated for each subplot. The mean slope of the plot was calculated by averaging the maximum absolute slope values of the subplots.

2.2.3. Logging assessment

State logging activities started in 1982 and observed AGC stocks are the result of past disturbances and subsequent forest recovery. Turnover rates in tropical forests are high and mean decomposition rates of coarse woody debris in tropical forests vary between 0.11 and 0.46 year^{-1} (Baker et al., 2007), suggesting that evidences of historical logging could rapidly vanish in the field. To understand better historical logging activities at each site, we interviewed (ex-) employees working for the state logging company and contacted responsible government offices. Unfortunately, no records of historical logging or maps were available anymore for our sites, making it impossible to gather spatial information about logging intensities in KNT at plot-level. Therefore, we assessed logging in three ways: 1) measuring logged stumps in the plots to assess recent logging levels; 2) analysis of satellite images to evaluate the spatial pattern of historical forest disturbances associated with selective logging; and 3) participatory mapping to verify historical logging activities.

2.2.3.1. Logged stumps

Logged stumps $\geq$10 cm diameter were measured in each 0.25 ha plot. The diameter of the stumps was measured at the highest point (when buttresses were present, the inner round part of the stem was measured to estimate the diameter).

2.2.3.2. Remote sensing analysis

Previous studies have identified forest disturbances using different vegetation indices, including the Normalized Difference Vegetation Index (NDVI) and the Normalized Burn Ratio (NBR). NBR was first used to identify and map forest fires, but has recently been used for the identification of tropical forest canopy disturbances caused by selective logging (Grogan et al., 2015; Langner et al., 2018; Lima et al., 2019; Shimizu et al., 2017). NBR values vary between -1 and +1, with larger positive values indicating closed canopy cover and smaller positive or negative values for open canopy (Langner et al., 2018). NBR is particularly suitable for detecting forest disturbance, due to its sensitivity to bare soil and non-photosynthetic vegetation created by forest disturbance events, and the lower contamination from atmospheric haze compared to NDVI (Grogan et al., 2015; Langner et al., 2018).

We used the Earth Explorer and the USGS ESPA ordering system to retrieve all available Landsat images for Path/Row 126/48 and 125/48 with less than 70% land cloud cover between 1988 and 2016 inclusive. A total of 439 Landsat 4/5, 287 Landsat 7 and 65 Landsat 8 surface reflectance images were obtained. All images were then corrected to remove any remaining cloud, cloud shadows and poor quality pixels resulting from unequal atmospheric conditions using the ‘pixel_qa’ layer. We calculated NBR as:

$$\mathrm{NBR} = \frac{\mathrm{NIR} - \mathrm{SWIR}}{\mathrm{NIR} + \mathrm{SWIR}}$$

where NIR is the Near Infrared band (Band 4 in Landsat 4,5 and 7 and Band 5 in Landsat 8) and SWIR is the Landsat Shortwave Infrared band (Band 7). We identified disturbance as pixels exhibiting a reduction in NBR, with two consecutive images showing NBR <0.5. In this way, we were able to distinguish between recently disturbed canopy cover and naturally open canopy (Langner et al., 2018). Pixels without tree cover throughout the study period (i.e. rivers) and pixels with plantation forestry (identified through large-scale disturbance at 3-7 year intervals) were removed from the analysis. The Ho Chi Minh highway in the western part was not counted as disturbance (the road was not developed as a logging road).

Previous work found that the area of skid trails (Broadbent et al., 2006) or the area damaged by logging (i.e. skid trails and tree fall gaps; Panfil and Gullison, 1998) followed a quadratic increasing function of harvest intensity. Therefore, we assumed a positive relationship between the density of disturbances and the logging intensity, and used the disturbance intensity as a proxy of logging intensity. Previous studies have suggested that the majority of timber extraction occurs within 700-1000 m of a logging road (Bryan et al., 2013; Gaveau et al., 2014). The logging intensity per plot was estimated by summing the number of disturbed pixels within 1000 m of each plot centre, with disturbed pixels within 500 m weighted by a factor 2. If after sufficient reduction in the NBR of a pixel (i.e. a disturbance), the NBR recovered and reduced significantly again in a different year, this was counted as another disturbance. Plots 1-18 were established in 2016 and disturbances were summed from 1988-2015. Plots 19-24 were established in 2017 and disturbances were summed from 1988-2016.

2.2.3.3. Participatory mapping

To further assess the intensity of logging across different regions of KNT, and confirm that the identified pixels with disturbance on satellite images represent mainly logging events, interviews were conducted in Kim Thuy commune (Le Thuy District, Quang Binh Province). Interviews were conducted with commune and village leaders as well as with households (both men and women) within two villages in March 2018. During the interviews, participants developed participatory maps identifying the level of logging prior to 2010 within different regions of KNT (logging activities after 2010 were likely still evident in the field by the presence of logged stumps).

First, maps of KNT showing key landmarks (rivers, roads, villages) were discussed and the participants familiarised themselves with the map. After discussion and agreement amongst the participants, each area was allocated a logging classification (light, medium, heavy) and a period when the majority of the logging occurred. After the interviews, the individual maps were combined to create a single map of KNT. Consistent responses from households within both villages and from the commune and village leaders provide confidence in the assigned logging classifications within the broad regions. The forest plots were then allocated a logging classification based on the map. More details on the communities and livelihood surveys can be found in Ngo et al. (2020).

2.2.4. Data analysis

2.2.4.1. Estimation of forest carbon stocks

The BIOMASS package in R (Réjou-Méchain et al., 2017) was used to develop a local HD-model, extract wood density (wood specific gravity; wsg) values and calculate plot-level living AGB. For the subset of intact trees from which heights were measured (n=1065), several HD-models were fitted and the model with lowest residual standard error (a Weibull function) was chosen. As the linear relationship between logarithmic height and logarithmic diameter did not differ significantly with logging class, the intact trees from all plots were grouped to develop one HD-model. This model was applied to estimate heights of intact trees without height measurements. Heights of deviating trees were assigned from the individual height measurements in the field. Wood densities were assigned using the global wood density database (Chave et al., 2009; Zanne et al., 2009). If the species was present in the database, the global species-level average was assigned. Wood densities of missing species were attributed by, in subsequent order, the global genus- or plot-level average. Tree AGB was computed using the generic allometric equation 4 of Chave et al. (2014) including dbh, height and wsg as input parameters (Réjou-Méchain et al., 2017), and summed per plot to retrieve plot-level AGB, expressed in Mg ha^{-1}.

The volume of standing dead trees and fallen pieces of dead wood was calculated using the formula of a conical frustum. Aboveground necromass was calculated by multiplying these dead wood volumes with the estimated wood density for each decomposition class and summed per plot. For each plot, the wood density per decomposition class was estimated by multiplying the mean plot-level wood density of living trees ≥10 cm dbh with a proportional reduction in wood density that was estimated by Chao et al. (2017) for tropical lowland forests in Taiwan (see Appendices 2.A1 and 2.A2).

Living below ground biomass (BGB), i.e. root biomass, for stems ≥10 cm dbh was estimated using the allometric equation of Nam et al. (2016), developed for trees >10 cm dbh in logged forests in central Vietnam and using dbh and wsg as input parameters. SOC (in Mg C ha^{-1}) was calculated by multiplying the fractional SOC with the bulk density and soil volume of the depth layer. Living AGB, necromass and living BGB were converted into C stocks using a default value of 47% C content (IPCC, 2006). Finally, living AGC ≥5 cm dbh, C in dead wood, living BGC and SOC were summed by sample plot to retrieve total C stocks.

2.2.4.2. Testing differences in forest structure between logging classes

Plots were ordered by the number of disturbances identified on the Landsat images and equally divided over three logging classes (n=8 plots per logging class): light, medium and heavy logging intensity. The logging classes derived from the Landsat analysis and participatory mapping were compared. In the analyses we used the logging intensities calculated from the Landsat analysis, where possible the continuous data or otherwise the derived three logging classes. Differences in forest structures were tested amongst logging classes using ANOVA or, when distributions within groups were not normally distributed, the Kruskal-Wallis rank sum test. For parameters that showed differences among logging classes, a Tukey test (for parametric tests) or Dunn test (for non-parametric tests) was performed to identify which logging classes differ.

2.2.4.3. Explaining variances in AGC

Linear mixed effects models (lmm; using package lme4 in R; Bates et al., 2015) and multiple linear regression models (mlm) were fitted, in which the AGC ≥10 cm dbh was predicted by various fixed effects (for lmm) and independent parameters (for mlm) and their interactions, i.e. logging intensity, elevation, slope, basal area (B.A.) of logged stumps and soil properties. In the lmm's, the site, i.e. plots located in the same area, was treated as a random effect to reduce pseudo-replication. Soil parameters that explain most of the variances in the soil dataset were selected through a principal component analysis (PCA), in which both depth layers were combined. The selected soil parameters were averaged over the 0-10 cm and 10-30 cm depth layer and added to the mlm's and lmm's. Models were generated and the best-fitted models were selected using model averaging based on Bayesian Information Criterion (BIC). The importance of each parameter in the initial model was calculated, showing the proportional frequency that each parameter was selected in fitted models. Package relaimpo in R (Grömping,

2006) was used to calculate the relative importance of the selected parameters in explaining the variance in AGC in the final model. All analyses were carried out using R (version 3.5.3; R Core Team, 2019) and results are expressed per hectare.

2.3. Results

2.3.1. Logging intensities in KNT

Fig. 2.2 shows the map of deforestation and forest degradation in KNT for the period 1988-2016. Over this whole period, there is evidence of forest disturbance in the eastern portion of KNT. Most clearly identified is the development of logging roads in the east of KNT starting during 1988-1994, adjacent to the state logging company Kien Giang, and extensive disturbances and associated logging trails in the south of the reserve starting during 2005-2009. A periodic pattern of forest disturbance roughly every 3-7 years was observed along the north-eastern boundary, which is likely caused by development and management of *Acacia* plantations. The development of the road through the western section of the reserve beginning in 2001-2002 was easily identified. In contrast, with the exception of the road construction, there is little evidence of logging in the western portion and all lightly logged plots are located in the west of KNT (see Appendix 2.C1 for logging classifications per plot).

No disturbed pixels were detected within either of the 24 forest plots (Fig. 2.2). We calculated the disturbance intensity for each plot as the sum of disturbed pixels within 1000 m of the plot, with disturbances within 500 m of the plot weighted by a factor 2 (see Material and methods; 2.2.3.2). At least three disturbed pixel were identified within 1000 m of all plots, with summed weighted disturbance varying by a factor 65 between the most heavily disturbed and most lightly disturbed plots. For most plots, disturbances within 1000 m occurred on multiple different years (mean = 8 different years, maximum 14 different years), suggesting ongoing disturbance rather than isolated harvesting years. Plots were classified into light, medium and heavy logging intensity based on the calculated disturbance intensity. Overall, there was good agreement between the logging classifications from the remote sensing analysis and participatory mapping (Spearman's ρ = 0.66), with a broadly similar distribution of plots identified as light, medium and heavy logging intensity (classifications were the same or maximum one level difference in intensity; Appendix 2.C1). This agreement provides additional confidence that our remote sensing analysis is able to quantitatively assess historical logging activities in this region.

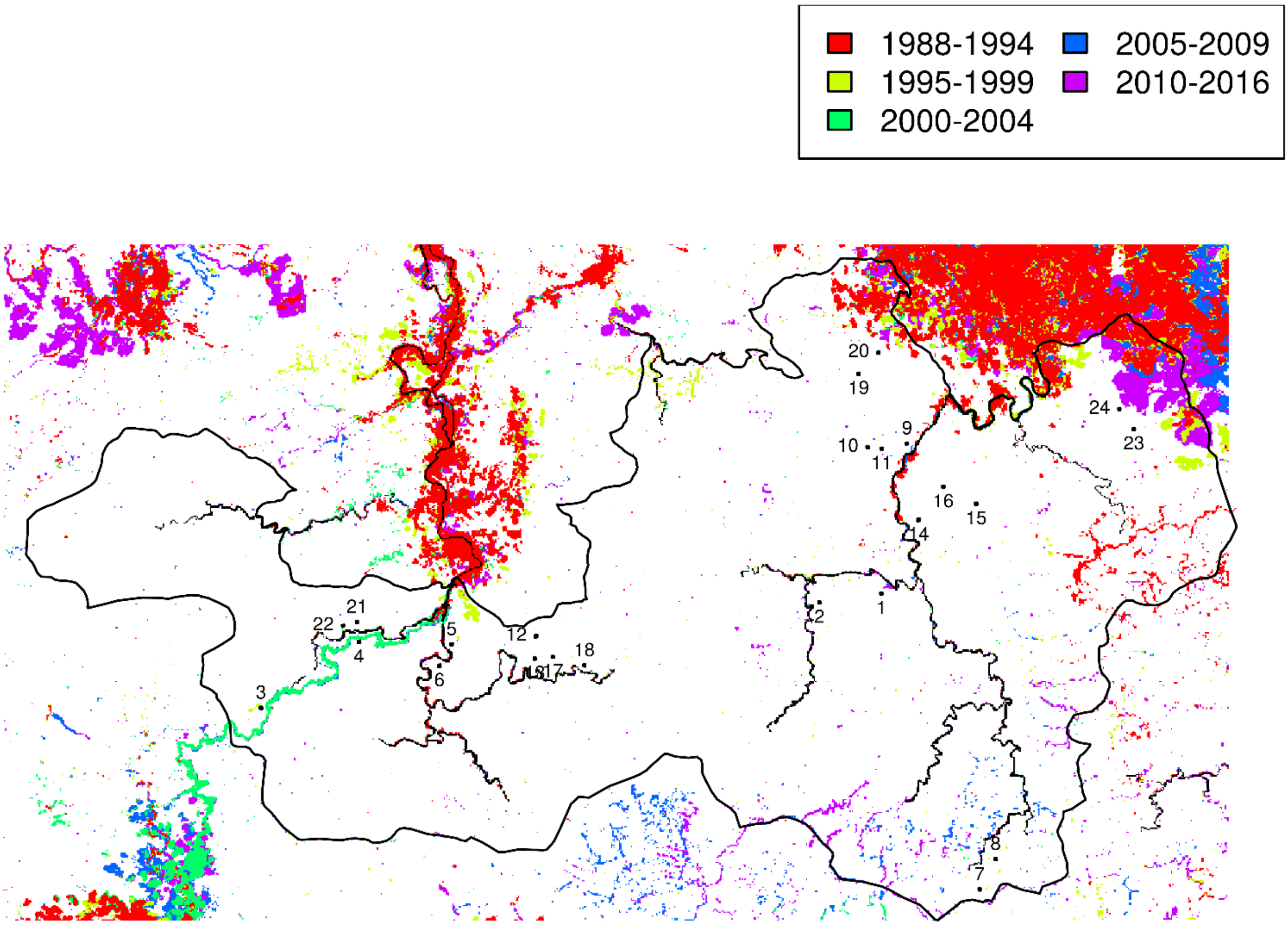


Figure 2.2. Map showing the disturbances identified on Landsat images between 1988 and 2016, indicating deforestation and forest degradation in KNT.

The locations of the 24 plots are shown. The colour indicates the earliest date of any identified forest disturbance. Black outline indicates the KNT boundary and inner lines indicate big rivers. For plots 1-18 disturbances were counted from 1988-2015, for plots 19-24 from 1988-2016 (see Material and methods; 2.2.3.2).

2.3.2. Forest C stocks for different logging classes

Table 2.1 compares the forest plots according to the logging classifications derived from the Landsat analysis (Appendix 2.C2 shows details for each plot). The B.A. of logged stumps still identifiable in the plots, indicating the amount of logging in recent years, was significantly higher in medium logged compared to lightly logged forests (Table 2.1). Maximum stem diameter, B.A. of living stems and AGC of stems ≥10 cm significantly differed among logging classes, with bigger trees, higher B.A. and higher AGC stocks in lightly logged plots compared to medium and heavily logged plots. AGC of stems ≥10 cm dbh in lightly logged plots was double that in heavily logged plots (Fig. 2.3a), with consistent results using logging classifications derived from participatory mapping (Fig. 2.3b). For the further analyses, the logging intensity according to the Landsat analysis was used. The AGC of regenerating stems of 5-10 cm dbh, C in dead wood and SOC (both 0-10 cm and 10-30 cm depth) did not differ between the logging classes (Table 2.1). BGC and total C were significantly higher in lightly logged areas compared to forests logged with medium and heavy logging intensity.

Table 2.1. Parameters describing the forest structure in lightly, medium and heavily logged forests.

Logging intensity was classified according to the results of satellite analysis. Total C is a summation of the C stocks in AGC of stems ≥5 cm dbh, C in dead wood, BGC and SOC 0-30 cm depth. Values represent the mean ± standard deviation (n = 8 for each logging class). Significant p-values are annotated with *, different letters within a given row indicate significant differences between logging classes. KW = Kruskal-Wallis rank sum test.

Parameter	Specification	Logging intensity			p	Test
		Light	Medium	Heavy		
B.A. logged stumps (m^{2}ha^{-1})	≥10 cm dbh	1.0 ± 1.5^{a}	3.6 ± 2.6^{b}	2.0 ± 1.5^{ab}	0.046*	KW
Max. dbh (cm)		94.6 ± 12.0^{a}	59.4 ± 13.7^{b}	53.7 ± 12.3^{b}	<0.001*	ANOVA
# living stems (ha^{-1})	5-10 cm dbh	613 ± 237^{a}	819 ± 373^{a}	725 ± 401^{a}	0.499	ANOVA
	≥10 cm dbh	497 ± 71^{a}	565 ± 45^{a}	681 ± 241^{a}	0.061	ANOVA
B.A. living stems (m^{2}ha^{-1})	≥10 cm dbh	31 ± 5^{a}	20 ± 3^{b}	21 ± 6^{b}	<0.001*	ANOVA
C stocks (Mg C ha^{-1})	AGC stems 5-10 cm dbh	3 ± 1^{a}	4 ± 2^{a}	4 ± 3^{a}	0.811	KW
	AGC stems ≥10 cm dbh	108 ± 27^{a}	56 ± 9^{b}	52 ± 17^{b}	<0.001*	KW
	C dead wood	4 ± 4^{a}	6 ± 3^{a}	3 ± 2^{a}	0.100	KW
	BGC	15 ± 3^{a}	10 ± 1^{b}	10 ± 3^{b}	0.002*	KW
	SOC 0-10 cm	33 ± 5^{a}	33 ± 7^{a}	30 ± 4^{a}	0.432	ANOVA
	SOC 10-30 cm	66 ± 9^{a}	63 ± 10^{a}	63 ± 10^{a}	0.812	ANOVA
	Total C	229 ± 33^{a}	173 ± 21^{b}	163 ± 29^{b}	<0.001*	ANOVA

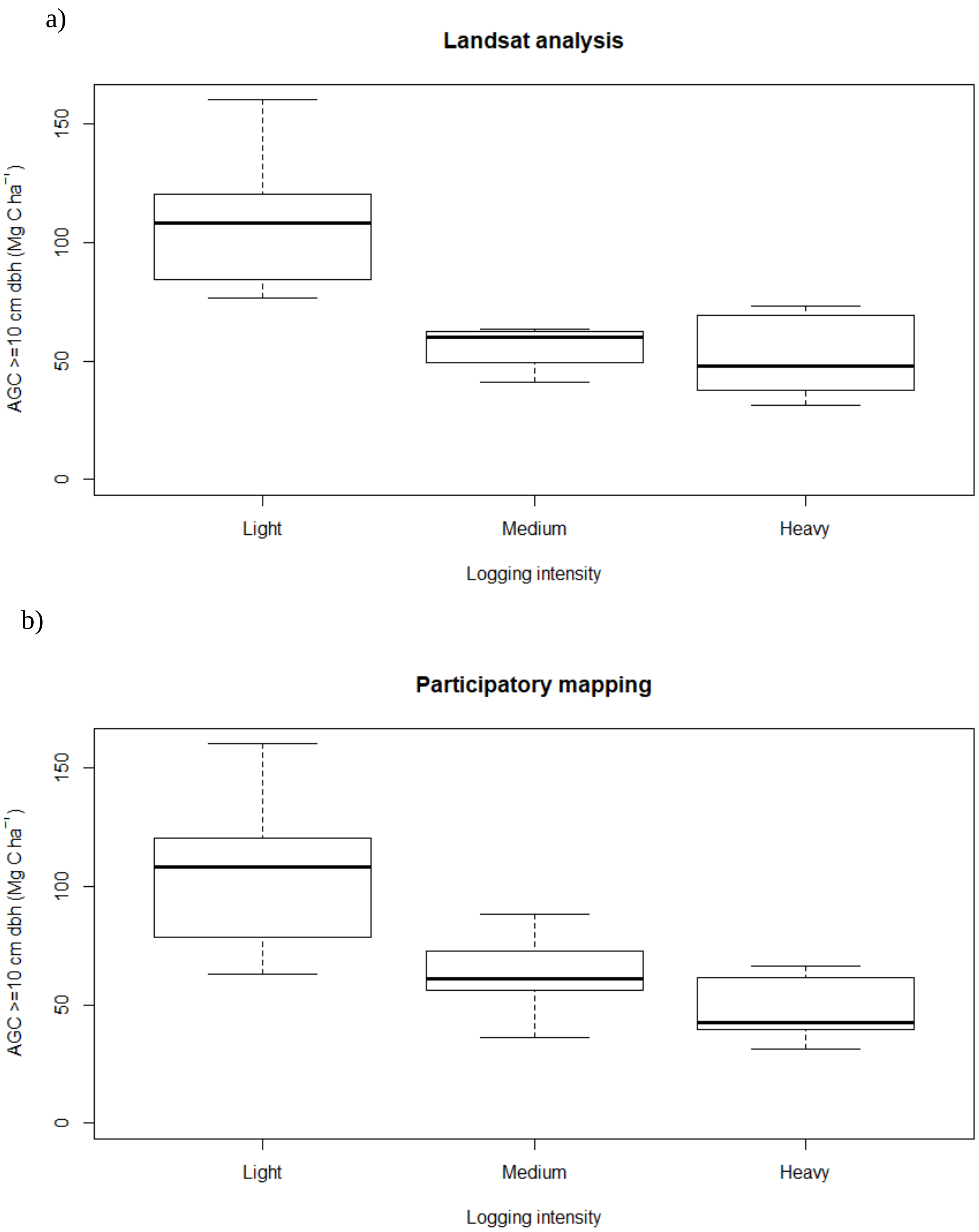
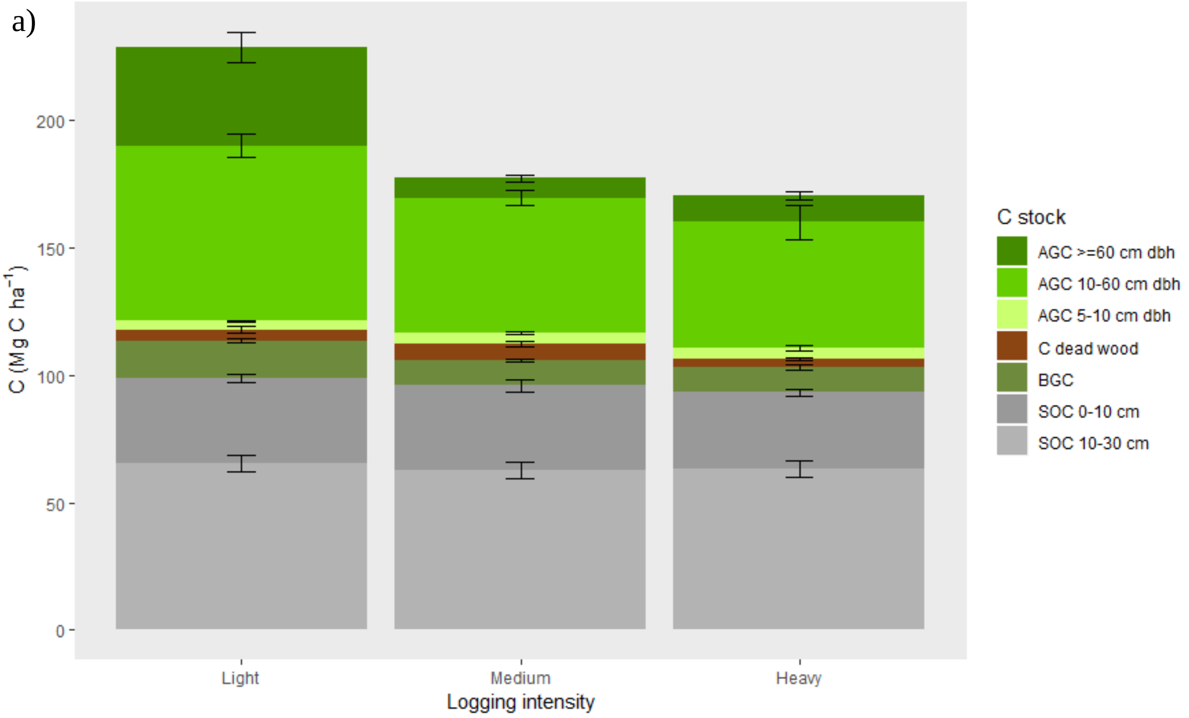


Figure 2.3. Boxplots of the AGC of stems ≥10 cm dbh in lightly, medium and heavily logged forests, classified according to the Landsat analysis (a) and participatory mapping (b).

Each logging class contains eight plots.

Average total C stocks equalled 229, 173 and 163 Mg C ha^{-1} in lightly, medium and heavily logged forests, respectively (Table 2.1). The main difference in C stocks among logging classes was found in AGC stocks ≥10 cm dbh, i.e. heavily logged forests stored half the amount of AGC as lightly logged forests. This difference is largely attributed to the amount of AGC stored in stems ≥60 cm dbh: this C pool represents 17 % of total C stocks in lightly logged forests, while these large stems were almost absent in medium and heavily logged forests (4 and 6% of total C stocks, respectively; Fig. 2.4a and 2.4b. See Appendices 2.D1 and 2.D2 for absolute and proportional C stocks per plot). Small amounts of C were stored in AGC of regenerating trees (5-10 cm dbh), dead wood and BGC. SOC in the top layer (0-30 cm depth) stored a large fraction of total C, representing respectively 43%, 54% and 55% in lightly, medium and heavily logged forests.



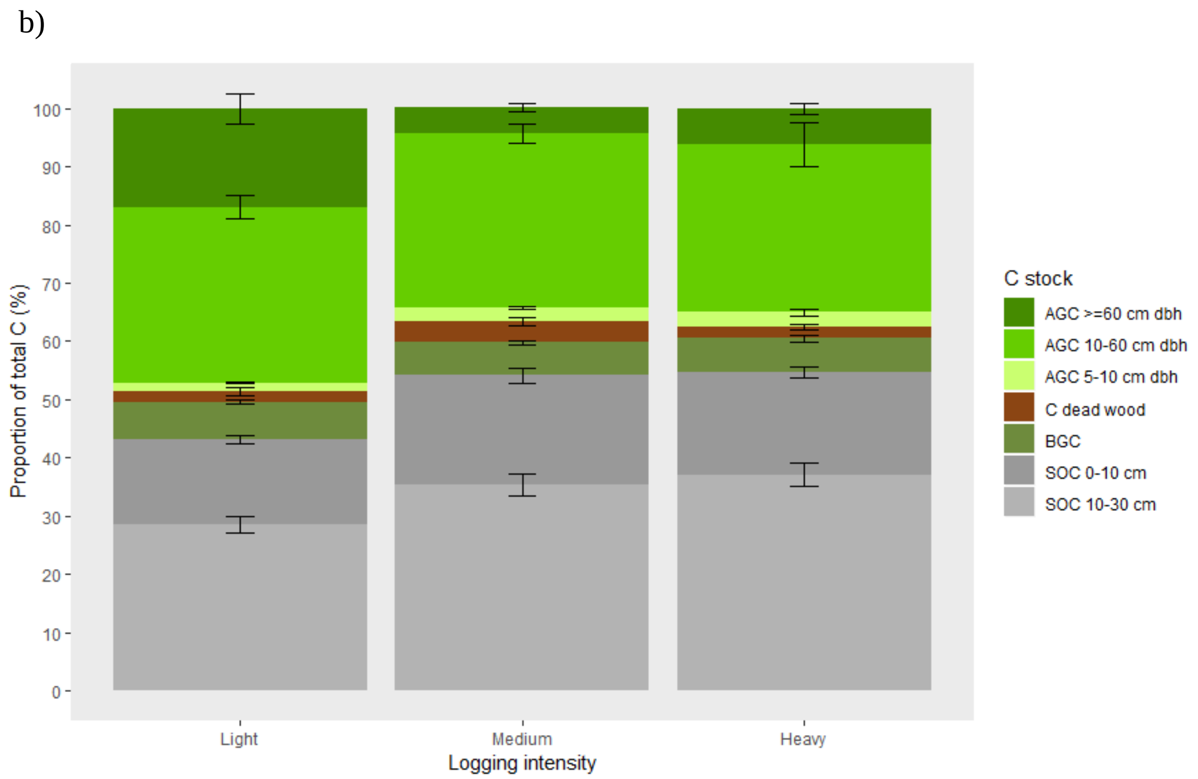


Figure 2.4. The absolute (a) and proportional (b) mean C pools ± standard error in each logging class.

While AGC of stems ≥10 cm dbh and BGC differed with logging intensity, AGC of stems 5-10 cm dbh, C in dead wood and SOC did not vary significantly with logging intensity (Table 2.1). Hereafter, we explore the impacts of logging intensity, topography and soil on AGC of stems ≥10 cm dbh and assess the importance of logging intensity in predicting AGC stocks.

2.3.3. Explaining variances in AGC

We used a PCA analysis to select the soil parameters that explain most of the variances in the soil dataset. The six most dominant soil parameters were included in the models, i.e. % clay, exchangeable H^+, exchangeable K^+, exchangeable Ca^{2+}, % sand and available P, however these and the next highest ranked soil parameters were almost all equally dominant (Appendix 2.E).

To explore which parameters explain the variability in AGC of stems ≥10 cm dbh, linear mixed effects models (lmm) and multiple linear regression models (mlm) were fitted according to the following initial model, with the random effect "site" only in the lmm's:

AGC ≥10 cm dbh = log(logging intensity)*elevation + log(logging intensity)*slope + elevation*slope + B.A. stumps + % clay + exchangeable H^+ + exchangeable K^+ + exchangeable Ca^{2+} + % sand + available P (+ site) [Eq. 2.1]

The disturbance parameters and topography values per plot are listed in Appendix 2.C1, the soil values in Appendix 2.F. The multiplication in Eq. 2.1 indicates that the interaction between parameters is also included.

The best lmm predicted AGC by logging intensity and elevation (fixed effects) and site (random effect) (R^2=0.81, BIC=211). The best mlm predicted AGC by logging intensity and elevation (R^2=0.80, p <0.001, BIC=207). The mlm model had the lowest BIC and was therefore chosen to predict AGC. The model was fitted as follows:

AGC ≥10 cm dbh = 127.10 - 18.76 * log(logging intensity) + 0.05 * elevation [Eq. 2.2]

From the 24 mlm's that were fitted, logging intensity was selected in each model (1.0) and elevation in 19 models (0.83) (Fig. 2.5). The fitted AGC (using Eq. 2.2) versus observed AGC shows a good relationship (Fig. 2.6).

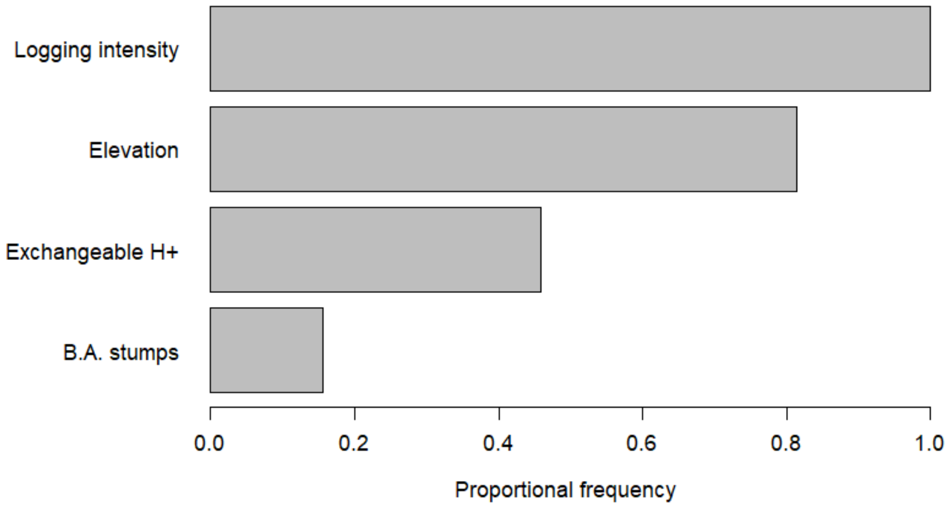


Figure 2.5. Proportional frequency of the parameters that were selected in fitted mlm's.

Parameters with proportional frequency ≥0.1 are shown.

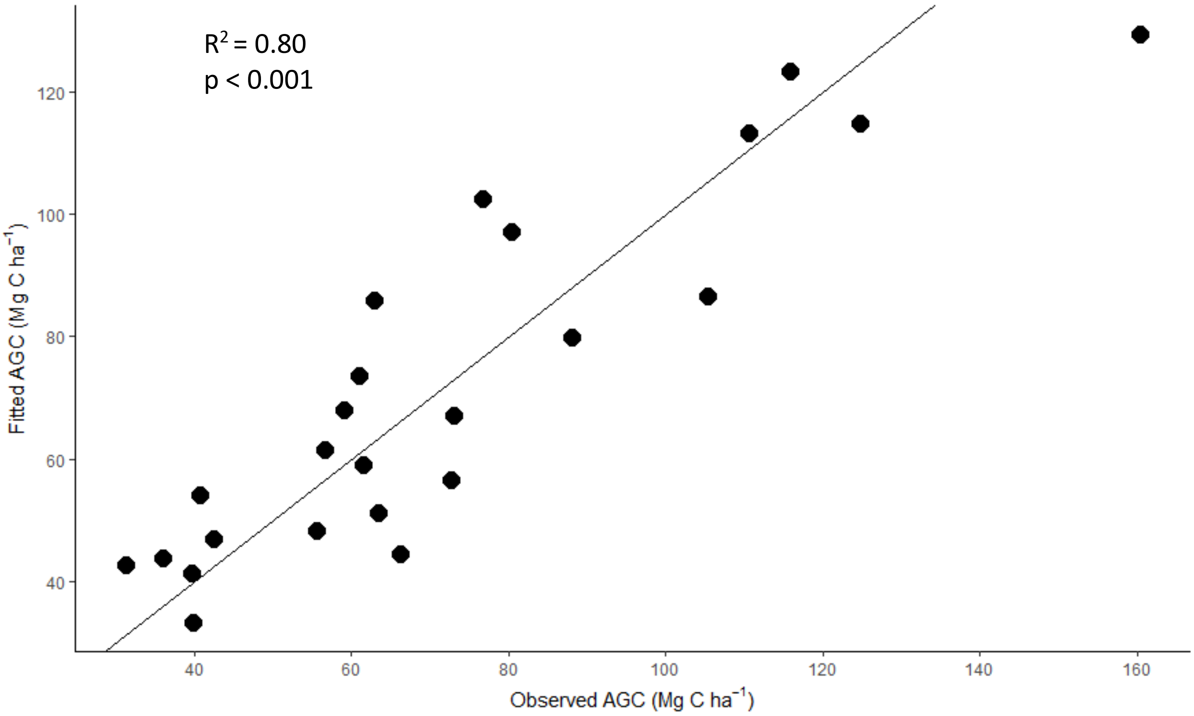


Figure 2.6. Fitted AGC (using Eq. 2.2) versus measured AGC of stems ≥10 cm dbh.

The best mlm (Eq. 2.2) explained 80% of the variance in AGC of stems ≥10 cm dbh, of which 77% was explained by logging intensity and 23% by elevation. AGC ≥10 cm dbh declined significantly with logging intensity (Fig. 2.7).

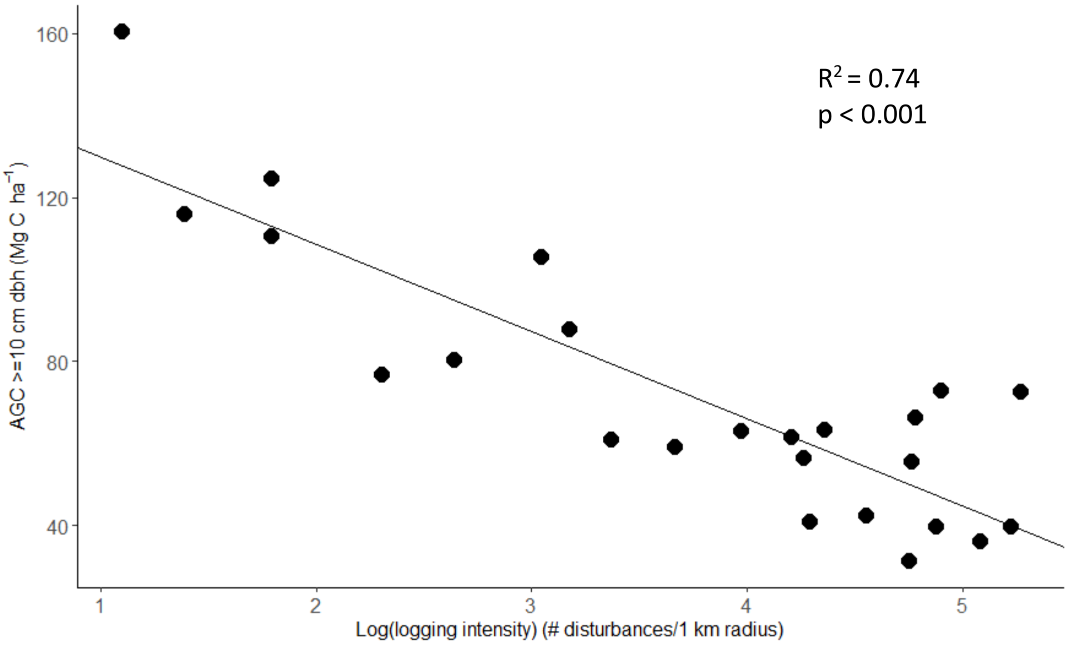


Figure 2.7. The relationship between AGC ≥10 cm dbh and the logarithmic logging intensity.

2.4. Discussion

While most Vietnamese forests experienced logging activities in the past, detailed information on logging extent and intensity is often lacking. Here, we propose a novel approach to trace back historical logging activities and estimate logging intensities for sites where these data are absent, using a combination of Landsat analysis and participatory mapping. In this study, we assessed the effects of logging intensity on main forest C pools in a lowland forest in Vietnam. We found lower AGC of stems ≥10 cm dbh in medium and heavily logged compared to lightly logged forests, due to the reduction of stems ≥60 cm dbh, which were likely harvested. SOC in the 0-30 cm top layer stored up to 55% of total C stocks and is therefore an important C pool at our site. Fitting various models in which AGC of stems ≥10 cm dbh was related to logging intensity, topography and soil showed that logging intensity was the main factor explaining the variability in AGC between plots.

2.4.1. Logging intensities

Logged stumps were observed in 21 of 24 plots, indicating that most areas have been recently logged. No relationship was observed between the B.A. of these recently logged stumps and the AGC of stems ≥10 cm dbh, suggesting that recent logging has not caused the difference in AGC stocks and that logging in earlier periods may have been important. Although it is challenging to distinguish natural forest disturbances from human-induced disturbances such as logging on remote sensing analyses, disturbed locations identified with the Landsat analysis are mostly located along rivers and near the border of KNT, or are organised into linear patterns, suggesting a human origin. This, in combination with the high similarity in logging classifications based on the participatory mapping, are likely evidences that most disturbances identified on Landsat images are due to logging events.

Most plots in the west of KNT have been lightly logged, while the plots in the east have been logged with medium or heavy logging intensity. The lightly logged plots were located at slightly higher elevation (mean elevation of 370 m) compared to the plots logged with medium and heavy intensity (mean elevation of 233 and 254 m, respectively). *Erythrophleum fordii*, *Sindora tonkinensis* and *Sindora siamensis*, the most valuable timber species, usually grow in areas with lower elevation (pers. comm. Trai Trong Le and Anh Tuan Pham, 2018). More intense logging in the east can be explained by the fact that the state logging company was located just outside the eastern border of KNT and the area is more easily accessible due to the flatter terrain, lower elevations and the presence of big rivers.

Testing various models showed that the logging intensity and elevation are the most important predictors for plot-level AGC of stems ≥10 cm dbh, with highest AGC values in lightly logged forests and forests at higher elevation. The relationship between logging and AGC is consistent with the results of meta-analyses, which showed that logging intensity was the best predictor for post-logging changes in AGC and changes in AGC were negatively correlated with logging intensity (Martin et al., 2015; Zhou et al., 2013). A study in Indonesia, similar to our study, tested the effects of logging intensity, topography and soil variables on several main forest C stocks and also found that logging intensity was the main factor in explaining the variability in AGC of trees >20 cm dbh (Rozak et al., 2018).

2.4.2. C stocks in logged forests

Differences in total C stocks between logging classes were largely found in the AGC stock of stems ≥10 cm dbh. Big trees (≥60 dbh) were largely reduced in medium and heavily logged forests, most likely removed by logging. Big trees store large amounts of AGC and their density and survival after logging largely determines C dynamics (Rozak et al., 2018; Sist et al., 2014; Slik et al., 2013).

The mean AGC of trees ≥10 cm dbh in our study varied from 108 Mg C ha^{-1} in lightly logged, 56 Mg C ha^{-1} in medium logged to 52 Mg C ha^{-1} in heavily logged forests. Our AGC stocks are similar to other logged forests in Vietnam and Malaysia (Hai et al., 2015; Luong et al., 2015; Nam et al., 2018; Saner et al., 2012) and old growth forests in Vietnam (Con et al., 2013), but considerably lower than AGC stocks in logged Dipterocarp forests in Indonesia (Rozak et al.,

2018) (Table 2.2). Trees from the Dipterocarpaceae family consist of large canopy and emergent trees (Ashton, 1982; Whitmore, 1984) and are important C stocks, but Dipterocarps did not occur in our plots. In our study, the AGC of stems ≥10 cm dbh in medium and heavily logged forests was, respectively, 48% and 52% lower than the AGC in lightly logged forests. Several studies found a considerable decline in AGC with logging intensity, but the exact reduction varies between studies, which is likely caused by variations in logging intensity, logging method and time since logging. For instance, forests logged with medium and high logging intensity in Indonesia stored, respectively, 9% and 33% less AGC of stems >20 cm dbh than lightly logged sites (Rozak et al., 2018). In Brazil, the AGC of trees ≥10 cm dbh of an intact forest decreased by 23% after logging with moderate intensity and by 43% after logging with heavy intensity (Gerwing, 2002). AGC stocks of stems >10 cm dbh in a forest 22 years post-logging were 28% lower compared to an unlogged forest in Malaysian Borneo (Saner et al., 2012). Overall, a meta-analysis found that selective logging reduced AGC stocks on average by 43% (Zhou et al., 2013). Small trees of 5-10 cm dbh stored only small amounts of AGC in our sites, i.e. 3-4 Mg C ha^{-1}, and did not differ with logging intensity. Similar values were found in logged forests in Vietnam and Malaysia (Nam et al., 2018; Saner et al., 2012; Table 2.2). Like our study, no correlation was observed between the logging intensity and AGC of small stems (5-20 cm dbh) in Indonesia (Rozak et al., 2018). However, in forests in Brazil, the AGC of small trees (>2 m tall and <10 cm dbh) decreased in heavily logged forests (Gerwing, 2002).

Logging can cause a temporal increase in necromass, which reduces over time when decomposition proceeds. Mean C stocks in dead wood at our sites ranged from 3-6 Mg C ha^{-1} and did not differ with logging intensity. A similar amount of C in necromass was found elsewhere in Vietnam (Hai et al., 2015; Nam et al., 2018; Table 2.2). Considerably higher values of C in necromass were observed in logged forests in Malaysia (Pfeifer et al., 2015) and

Indonesia (Rozak et al., 2018) (Table 2.2). Further, in forests in Indonesia and Brazil, the amount of necromass increased with logging intensity (Gerwing, 2002; Rozak et al., 2018). Previous work showed that necromass is sensitive to wood density values of living trees, i.e. necromass is positively related to plot-level average living wood density (Chao et al., 2009). In our study, mean wood density of living stems $\geq$10 cm dbh was low in each plot, ranging from 0.505 to 0.604 g cm^{-3}, and the majority of the logging occurred more than nine years before field measurements. Therefore, it is likely that much of the necromass caused by logging has already decayed, which contributed to low necromass values and no significant differences in necromass with logging intensity in our forests.

BGC stocks of our forests ranged from 10 to 15 Mg C ha^{-1} and are similar to values in other Southeast Asian forests (Nam et al., 2018; Saner et al., 2012; Table 2.2). Higher BGC stocks were found in Dipterocarp forests in Indonesia (Rozak et al. 2018; Table 2.2), in which BGC was estimated using an allometric model based on dbh. Mature Dipterocarp trees often have large diameters, which likely explains the high BGC values found in these forests. In our study, BGC significantly decreased with logging intensity. We calculated BGC using an allometric equation with dbh and wsg as input parameters and the reduction of large trees in medium and heavily logged forests likely explains the reduced BGC values in those forests. Consistent to our results, the BGC of trees >20 cm dbh decreased with logging intensity in Indonesian forests (Rozak et al., 2018).

Our plots are located in one geographical area, which resulted in similar soil properties across plots and none of the selected soil parameters contributed to explain sufficient variation in AGC. SOC ranged from 93 to 99 Mg C ha^{-1} in the top 0-30 cm layer, did not differ significantly between the logging classes and comprised more than half of total C stocks in our medium and heavily logged forests. Several studies estimated SOC values in Southeast Asia, but the depth layer varies per study (Table 2.2). SOC values in our plots are similar to other studies in Vietnam (Hai et al., 2015; Nam et al., 2018; Sang et al., 2013) and larger than most soils in tropical Southeast Asia (Rozak et al., 2018; Saner et al., 2012) (Table 2.2). This suggests that Vietnamese forests in general have higher SOC storage, possibly because the forests are in the sub-tropics rather than tropics, leading to slower decomposition rates. In Indonesian forests, no relationship was observed either between logging intensity and SOC (Rozak et al., 2018).

Table 2.2. Values of the various C pools in our study in comparison to studies in Southeast Asia.

The bold numbers are the values found in our study.

C stock	C (Mg C ha^{-1})	Details	Disturbance	Country	Reference
AGC larger stems	**52-108**	≥10 cm dbh	9 years post-logging	Vietnam	Our study
	154	≥10 cm dbh	30-32 years post-logging	Vietnam	Nam et al., 2018
	64-147	AGC and BGC of stems ≥5 cm dbh	Poor, medium and rich forests*	Vietnam	Hai et al., 2015
	47-150	>5 cm dbh	Poor, medium and rich forests*	Vietnam	Luong et al., 2015
	50-181	≥10 cm dbh	Old growth forests	Vietnam	Con et al., 2013
	160-227	>5 cm dbh	16 years post-logging	Indonesia	Rozak et al., 2018
	92	>10 cm dbh	22 years post-logging	Malaysia	Saner et al., 2012
AGC small stems	**3-4**	5-10 cm dbh	9 years post-logging	Vietnam	Our study
	3	1-10 cm dbh	30-32 years post-logging	Vietnam	Nam et al., 2018
	5	<10 cm dbh and >2 m height, seedlings (<2 m height), woody vines	22 years post-logging	Malaysia	Saner et al., 2012
C dead wood	**3-6**	CWD, standing dead trees ≥10 cm dbh	9 years post-logging	Vietnam	Our study
	6	CWD, standing dead trees	30-32 years post-logging	Vietnam	Nam et al., 2018
	4-6	CWD, standing dead trees, litter	Poor, medium and rich forests*	Vietnam	Hai et al., 2015
	11-30	CWD, standing dead trees >10 cm dbh	16 years post-logging	Indonesia	Rozak et al., 2018
	15-34	CWD, standing dead trees, hanging dead wood ≥10 cm dbh	Various logging history	Malaysia	Pfeifer et al., 2015
BGC	**10-15**	Living trees ≥10 cm dbh, coarse roots (>2 mm diameter)	9 years post-logging	Vietnam	Our study
	20	Living trees ≥10 cm dbh, coarse roots (>2 mm diameter)	30-32 years post-logging	Vietnam	Nam et al., 2018
	38-57	Living trees >5 cm dbh, coarse roots (≥5 mm diameter)	16 years post-logging	Indonesia	Rozak et al., 2018
	17	Living trees >10 cm dbh, coarse roots (>2 mm diameter)	22 years post-logging	Malaysia	Saner et al., 2012
SOC	**93-99**	0-30 cm depth	9 years post-logging	Vietnam	Our study
	116	0-50 cm depth	30-32 years post-logging	Vietnam	Nam et al., 2018
	62-64	0-30 cm depth	Poor, medium and rich forests*	Vietnam	Hai et al., 2015
	51-61	0-30 cm depth	Secondary forests	Vietnam	Sang et al., 2013
	43-47	0-100 cm depth	16 years post-logging	Indonesia	Rozak et al., 2018
	22	0-30 cm depth	22 years post-logging	Malaysia	Saner et al., 2012

* Poor, medium and rich forests refer to the classification used by the Vietnamese government and is based on timber reserves of standing trees. The forests in the study of Hai *et al.* (2015) varied from 10-30 years post-logging and/or shifting cultivation to low disturbed old growth forests. Pfeifer et al. (2015) studied lightly or illegally logged forests and twice-logged forests (~10 years after the last logging cycle).

2.4.3. Implications for forest management

Our results suggest that historical logging has reduced the C storage of forests in a protected natural forest in Vietnam. Large areas of natural forests in Vietnam have been degraded by logging and are likely to store less C than unlogged forests. If no further logging occurs in these forests, they will recover and sequester C, providing a substantial potential for Vietnamese forests to mitigate climate change. However, ongoing illegal logging threatens the recovery of AGC in our forests and is likely to be an issue more widely across Vietnam. Efforts to reduce illegal logging whilst supporting livelihoods of local communities are now required. Vietnam is starting to establish a REDD+ programme and is one of the countries selected to participate in the Forest Carbon Partnership Facility. By certifying the project activities under these programmes or through project-based schemes such as the Climate, Community and Biodiversity Alliance (CCBA; CCBA, 2013), it is possible that further forest degradation will be prevented and natural recovery favoured.

2.5. Conclusions

We presented an approach to estimate historical logging intensities by combining Landsat analysis and participatory mapping, which could be more widely applicable for sites that lack historical logging data. Heavily logged forests stored only half the amount of AGC of stems ≥10 cm dbh as lightly logged forests, mainly due to a reduction in large (≥60 cm dbh) trees. Conserving big trees thus remains important to maintain high forest C stocks. SOC in the 0-30 cm top layer stored around half of total C stocks in these forests, showing the importance of accounting for this C stock. While topography and soil properties can influence AGC stocks, our study found that logging intensity was the most important factor explaining the variability in AGC of stems ≥10 cm dbh in lowland forests in Vietnam. Timber demand sharply increased after the Vietnam War, which led to intense state logging in forests throughout the country. Our study contributes to the understanding of the effects of long-term logging activities on forest C stocks in north-central Vietnam. These data are crucial to develop REDD+ approaches to prevent further degradation and allow eventual recovery of Vietnamese logged forests.

Chapter 3. The impacts of logging intensity on tree diversity and composition in lowland forests in Vietnam

Abstract

Tropical forests are key biodiversity areas, yet they are seriously threatened by anthropogenic disturbances including logging. Logging alters the abundance and distribution of species and can thereby affect tree diversity and composition. In this chapter, I assessed the impacts of logging intensity on tree diversity, composition, community wood density and availability of timber species in a lowland forest in north-central Vietnam. Trees were identified in 24 sample plots that vary in logging intensity. No significant relationships were observed between tree diversity and logging intensity. Some compositional differences were observed with logging intensity, although the challenge of sampling at the species level sufficiently in these high diversity systems complicates interpretation. Community wood density differed with logging intensity, with a shift towards lighter wood density species observed in heavily logged forests. Timber species only occurred in small numbers in our plots, with highest abundance in lightly and medium logged forests and most trees from harvestable size in the lightly logged plots. My analysis shows that reducing logging intensities is important to maintain high community wood density values, and hence to conserve forest carbon, as well as to prevent the depletion of valuable timber species after logging.

3.1. Introduction

Tropical ecosystems contain the world’s highest concentrations of species diversity, yet they are seriously threatened by habitat destruction (Dirzo and Raven, 2003). Logging contributes to this and is the major driver of forest degradation in the tropics and sub-tropics (Hosonuma et al., 2012). Selective logging targets specific species and size classes and alters the abiotic environment, such as changes in temperature, humidity and solar radiation (Ramírez-Marcial, 2003), which in turn affects demographic processes (growth, recruitment and mortality) of plant communities and interactions among species (Putz et al., 2000b).

The impact of human activities on biodiversity is highly variable (Putz et al., 2000b). Meta-analyses showed that selectively logged tropical forests can retain high biodiversity values, despite the damage caused by logging (Gibson et al., 2011; Putz et al., 2012). Global analyses that conducted pairwise comparisons of biodiversity values in human-disturbed and primary tropical forests found that selectively logged forests had a significant reduction in tree species richness (Clark and Covey, 2012) and diversity of plants (Gibson et al., 2011). In contrast, many individual studies found that logged forests had higher or similar tree diversity compared to unlogged forests (Berry et al., 2010, 2008; Cannon et al., 1998; Ding et al., 2012; Hall et al., 2003; Imai et al., 2012; Sheil et al., 1999). Most species that are present before logging usually persist at reduced densities after logging. Additionally, due to the increased heterogeneity of forest microhabitats caused by logging, opportunities are created for the establishment of immigrant species, which often leads to higher diversity in disturbed areas (Sheil et al., 1999).

Various hypothesis have been proposed to explain local species diversity (Connell, 1978). One of them, the intermediate disturbance hypothesis (IDH), predicts that local species diversity is maximal at an intermediate level of disturbance (Bongers et al., 2009; Connell, 1978; Sheil and Burslem, 2003). Indeed, a meta-analysis performed in logged forests found that tree species richness increased at low logging intensities but decreased at higher intensities (Martin et al., 2015). However, changes in species richness do not provide any information about the conservation value of individual species (Sheil et al., 1999). For instance, most of the additional immigrant species that can establish after logging are good dispersers and generalists that depend on disturbances (Sheil et al., 1999). Therefore, it is important to assess the impacts of logging on compositional changes.

Logging can affect species composition by changing the abundance and distribution of species (Putz et al., 2000b). Some studies found that logged forests had a similar tree species composition compared to unlogged forests (Ding et al., 2012; Hall et al., 2003), while others found that tree species composition varied between logged and unlogged forests (Berry et al., 2008) and between heavily and lightly logged forests (Hoang et al., 2011). Generally, more early successional tree species and species of low conservation value are observed in (heavily) logged or disturbed forests, while vulnerable species and valuable timber species are most common in lightly logged or undisturbed forests (Bongers et al., 2009; Carreño-Rocabado et al., 2012; Clark and Covey, 2012; Hoang et al., 2011).

Compositional changes can alter community-level traits such as wood density. Previous work in Borneo found that logging increased the fraction of softwood stems in small diameter classes, while a large decline of softwood emergent stems was observed in the largest diameter classes (Verburg and Van Eijk-Bos, 2003). Another study found that the wood density in the community was lower at high disturbance intensity, caused by logging and silvicultural treatments (Carreño-Rocabado et al., 2012). First harvests in old-growth tropical forests can be highly selective, in which often higher-value, shade-tolerant and emergent species with high wood density are targeted, while later logging events rely more on longer-lived, lower-value pioneer species with low wood density (Macpherson et al., 2012). Previous work did not find evidence that the post-logging timber species composition and overall value of forest stands recovered beyond the first logging round, which suggests that the most valuable timber species are likely becoming rare in old logging areas (Richardson and Peres, 2016). Timber trees are generally more abundant in lighter logged forests than in heavy logged sites (Hoang et al., 2011, 2008). Further, logging gaps tend to be rapidly colonized by competing plants such as pioneer species and lianas, which strongly limits the regeneration of timber species after logging (Fredericksen and Mostacedo, 2000; Sist and Nguyen-The, 2002).

Vietnam has remarkable levels of species richness and endemism of vascular plants (MONRE, 2014; Sterling and Hurley, 2005). This high diversity can be partly explained by Southeast Asia’s complex geological and climatic past, Vietnam’s wide range of latitudes covering a transition zone from subtropical to tropical and the relatively hilly and mountainous topography in Vietnam (Sterling and Hurley, 2005). Several studies assessed the tree diversity and composition in Vietnamese forests (Blanc et al., 2000; Con et al., 2013; Do et al., 2017, 2010; Van and Cochard, 2017), of which a few in logged forests (H.T.T. Do et al., 2019; Tran et al., 2005), but only one study (Hoang et al., 2011) considered variations in logging intensity.

Khe Nuoc Trong forest (KNT) in north-central Vietnam is one of the last remaining extensive lowland forests in the Annamese mountain range and harbours species of exceptional conservation value (Department for agriculture and rural development, 2010). KNT experienced various levels of state logging in the past and ongoing illegal logging (Chapter 2). Previous work showed that variations in above-ground carbon (AGC) stocks across KNT could be explained by differences in logging intensities (Chapter 2). To improve biodiversity conservation, it is important to understand the impacts of historical logging intensities on floristics in KNT. Here, I investigate the impacts of historical logging intensities on tree diversity, species composition, community wood density and availability of timber species in KNT.

3.2. Material and methods

3.2.1. Site description

Data were collected in the logged forests of KNT in Quang Binh Province, Vietnam. KNT was logged by the state with different logging intensities between 1982 and 2007 (Chapter 2). Since 2007, logging is forbidden, but illegal logging is still ongoing. At least 14 tree species are extracted for timber in KNT, of which *Erythrophleum fordii*, *Sindora siamensis* and *Sindora tonkinensis* are the most valuable timber species (Viet Nature Conservation Centre, pers. comm., 2018)**.** More site details are described in Chapter 1.10.

3.2.2. Field measurements

We established 24 plots of 0.25 ha each, 6 ha in total, in lowland forests (<700 m elevation) between April 2016 and June 2017 (Fig. 2.1). Plots varied in the degree of degradation due to historical logging. Living woody stems ≥10 cm diameter at breast height (dbh; i.e. diameter at 130 cm height) were tagged and their dbh measured. A botanist identified each stem with Vietnamese and scientific species names in the field and, when needed, collected botanical vouchers for further identification in the herbarium of the Forest Inventory and Planning Institute (FIPI) in Hanoi, Vietnam. The dbh and species of stems 5-10 cm dbh were measured and identified in a belt transect of 4 m x 50 m, running through the plot centre. The measurements have been described in more detail in Chapter 2.2.2.

3.2.3. Data analysis

3.2.3.1. Logging assessment

Historical logging records were lacking for our site. Therefore, we estimated historical logging intensities by combining remote sensing analysis with participatory mapping. Forest canopy disturbances were identified on Landsat 4/5, 7 and 8 surface reflectance images between 1988 and 2016 inclusive. We identified disturbance as pixels exhibiting a reduction in the Normalized Burn Ratio (NBR), with two consecutive images showing NBR <0.5. The disturbance density, a proxy of logging intensity per plot, was estimated by summing the number of disturbed pixels within 1000 m of each plot centre, with disturbed pixels within 500 m weighted by a factor 2. Disturbances were summed from 1988-2015 (plots 1-18) and from 1988-2016 (plots 19-24). Participatory mapping with commune and village leaders as well as with households was performed to confirm logging levels at the sites. The logging intensities derived from the Landsat analyses was used for the analyses. Plots were ordered by increasing number of disturbances as identified on the Landsat images and equally divided over three logging classes ($n=8$ plots per logging class): light, medium and heavy logging intensity. More details and the rationale behind this approach on assessing historical logging intensities are described in Chapter 2.2.3.

3.2.3.2. Tree diversity

For all analyses in this chapter, the number of stems and species ≥5 cm dbh were counted per plot, meaning a summation of the number of stems ≥10 cm dbh in the 0.25 ha plot and the number of stems 5-10 cm dbh in the 0.02 ha belt transect (without scaling area sizes). All analyses were conducted using R (version 3.5.3) (R Core Team, 2019). Typos in taxonomic names were corrected using the Taxonomic Name Resolution Service (Boyle et al., 2013), incorporated in the BIOMASS package (Réjou-Méchain et al., 2017). A rarefaction curve was constructed using the function specaccum in the vegan package (Oksanen et al., 2019). Function specpool in the vegan package, selecting the first order jackknife, was used to estimate the number of unobserved species in the area and adding them to the observed species richness in the plots (Oksanen et al., 2019). Various diversity metrics were calculated for each plot, i.e. the Shannon index, Simpson's index, Fisher's α, species richness (number of species) and the evenness, all in the vegan package (Oksanen et al., 2019). As diversity typically increases with the number of stems, all diversity metrics were calculated based on the lowest number of total stems in the plots. We calculated mean diversity metrics from 1000 permutations and in each run individuals were randomly selected with replacements, following the same approach as Wunderle et al. (2006). We used ANOVA to test whether the diversity metrics differed between logging classes. Generalized additive models, fitted with function gam in mgcv package (Wood, 2011), were fitted for the relationship between the rarefied Shannon index and logging intensity and between the rarefied species richness and logging intensity.

3.2.3.3. Tree species composition

A non-metric multidimensional scaling (NMDS) analysis was performed to visualize plot dissimilarity in terms of tree species composition, using the metaMDS function in the vegan package with Bray-Curtis Dissimilarity index (Oksanen et al., 2019). I used relative species abundance, using the decostand function, and created a distance matrix, using the vegdist function with method Bray-Curtis, both in the package vegan (Oksanen et al., 2019). Further, I identified the five most abundant tree species for each logging class.

3.2.3.4. Wood density and availability of timber species

Wood densities were extracted from the global wood density database (Chave et al., 2009; Zanne et al., 2009), incorporated in the BIOMASS package (Réjou-Méchain et al., 2017). If the species was present in the database, the global species-level wood density average was assigned. Following usual methods (Baker et al., 2004; Chave et al., 2006; Slik, 2006), wood densities of missing species were attributed by, in subsequent order, the global genus- or plot-level average. Community wood density of stems ≥5 cm dbh was averaged by plot and an ANOVA was used to test if wood density differed with logging intensity.

At least 14 tree species are extracted for timber in KNT (Viet Nature Conservation Centre, pers. comm., 2018)**.** I measured all logged stumps ≥10 cm diameter in the plots (Chapter 2). Logged stumps were observed with diameter as low as 10-15 cm. During interviews, local communities living around KNT told that they log timber trees of at least 30 cm diameter (Viet Nature Conservation Centre, pers. comm., 2018). Official selective logging regulations in Vietnam have been revised multiple times throughout the years and the minimum cutting diameter has varied through time, for different type of wood species and with region, but has been as low as 30 cm diameter (Nam, 2017). I assume a minimum legal cutting size of 30 cm dbh. I assessed the abundance of timber species in the plots, both below and above the legal cutting limit.

3.3. Results

The NMDS analysis with 24 plots showed a clear distinction in tree species composition between plots 1-18 and plots 19-24 (Appendix 3.A). The majority of species that are most abundant in plots 19-24 did not occur in plots 1-18. Plots 19-24 were spatially distributed over the study site, so locational bias seems an unlikely explanation for these differences. Species in plots 1-18 and 19-24 were identified by different botanists. Due to this clear distinction in species composition, I assume that species were not identified uniformly. I therefore restricted my analyses to plots 1-18, which were all identified by the same botanist. Hereafter, all results are presented for 18 plots.

3.3.1. Tree diversity

For the 18 plots combined, the botanist identified most of the trees up to species level (83%), 17% up to genus level and <1% up to family level or unidentified (see Appendix 3.B for proportional frequencies of the levels of tree identifications for each plot). In total, the 2919 stems ≥5 cm dbh in the 18 plots represented 167 species (Fig. 3.1), which is 78% of the estimated total species present in the area (215 species; estimated using the first order jackknife).

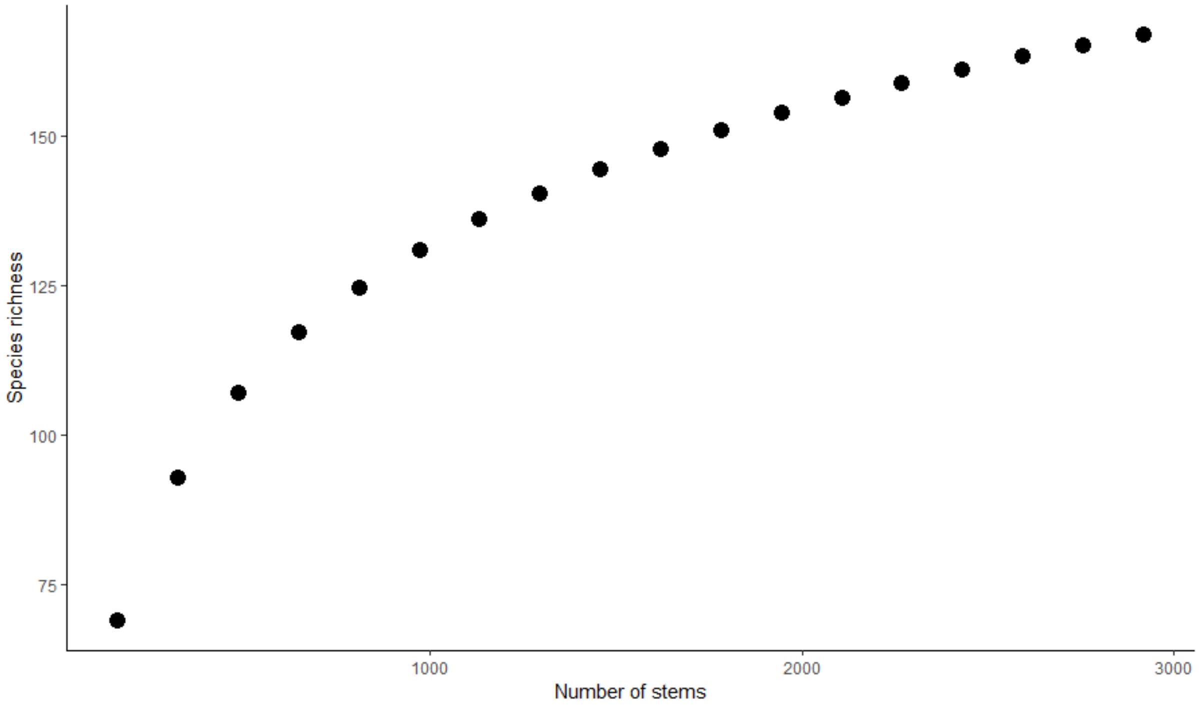


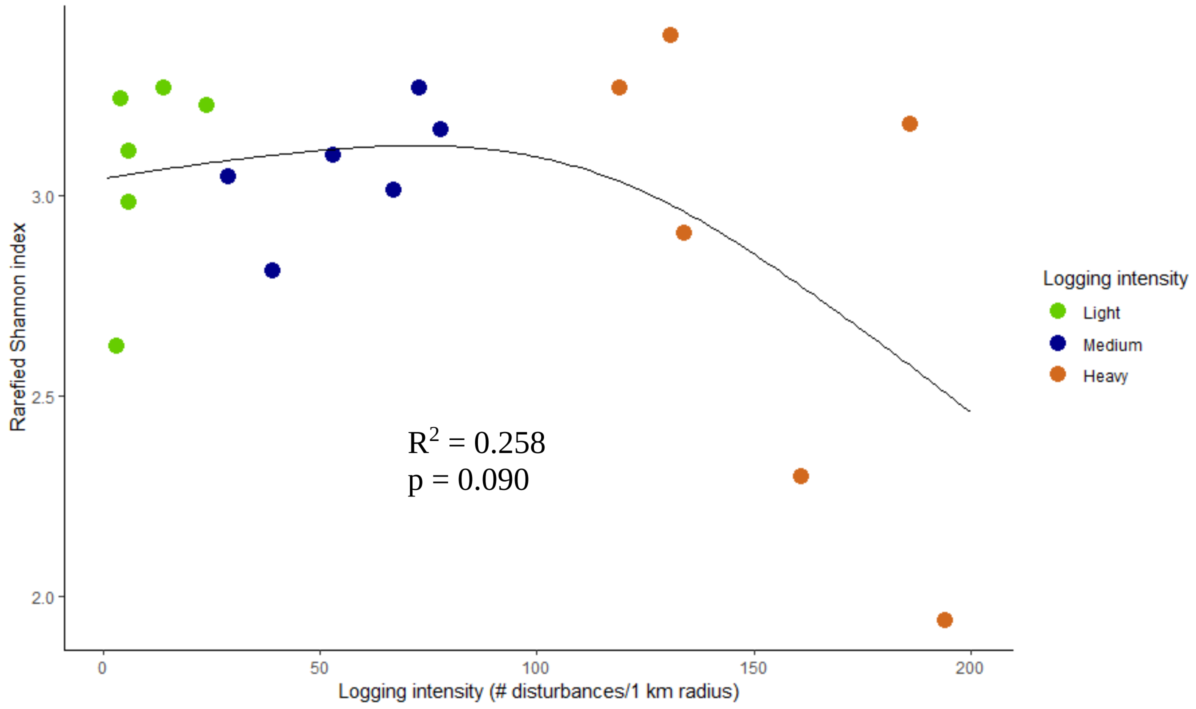
Figure 3.1. Rarefaction curve for stems ≥5 cm dbh for all 18 plots grouped.

The number of stems added in each step is equal to the average number of stems per plot, derived from the 18 plots.

The total number of stems ≥5 cm dbh increased with logging intensity: summing the amount of stems in the plots per logging class resulted in 828 stems in lightly logged, 963 stems in medium logged and 1128 stems in heavily logged forests. As diversity typically increases with the number of stems (Fig. 3.1), the following diversity analyses were calculated by standardized numbers of stems, i.e. the lowest number of stems found in the plots.

Generalized additive models showed no significant relationships between the rarified plot-level Shannon index and logging intensity (Fig. 3.2a) and between the rarified species richness and logging intensity (Fig. 3.2b), likely due to the high variability in diversity in heavily logged plots. When the logging intensity was classified into light, medium and heavy logging classes, we did not observe significant differences in diversity metrics between classes, also due to the large variance in diversity metrics between plots (Table 3.1).

a)



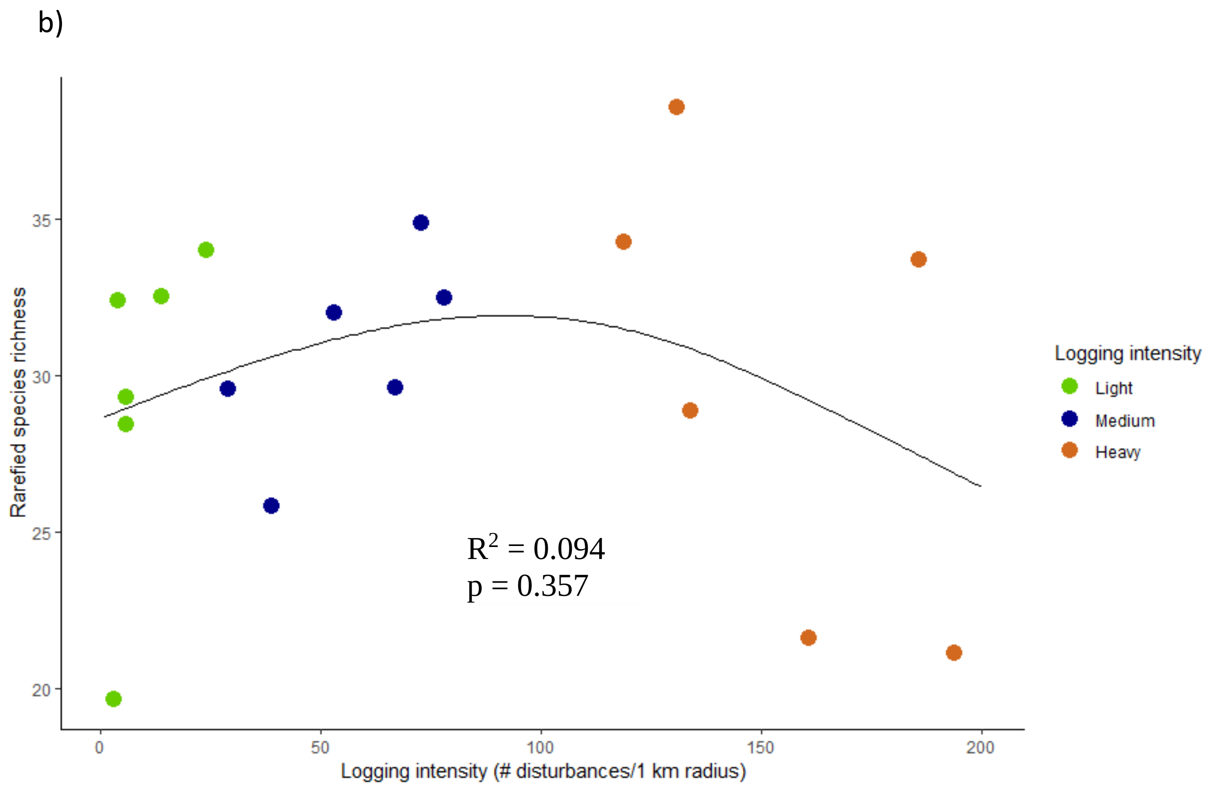


Figure 3.2. The relationship between the rarefied plot-level Shannon index (a) and species richness (b) for stems ≥5 cm dbh and logging intensity. Both diversity indexes have been standardized by the lowest number of stems in the 18 plots and the mean values shown are calculated based on 1000 iterations.

Table 3.1. Mean ± standard deviation of various diversity metrics for each logging class. Values have been calculated from the mean diversity values per plot, which were standardized by the lowest number of stems ≥5 cm dbh in the plots and 1000 permutations (plot-level diversity metrics are shown in Appendix 3.C). Bold numbers represent the highest mean value for each diversity metric.

Diversity index	Logging intensity			p	Test
	Light	Medium	Heavy		
Shannon	**3.08** ± 0.24	3.07 ± 0.15	2.83 ± 0.59	0.465	ANOVA
Simpson	**0.94** ± 0.02	0.94 ± 0.01	0.88 ± 0.11	0.200	ANOVA
Fisher's α	15.00 ± 4.14	**15.96** ± 2.82	15.68 ± 6.34	0.930	ANOVA
Species richness	29 ± 5	**31** ± 3	30 ± 7	0.906	ANOVA
Evenness	**0.91** ± 0.02	0.90 ± 0.02	0.84 ± 0.12	0.159	ANOVA

3.3.2. Tree species composition

The NMDS analysis showed that the species composition in lightly and medium logged plots was fairly similar, while the species composition in heavily logged forests was more dissimilar and differed largely between heavily logged plots (Fig. 3.3). Especially plots 5, 7 and 8, all heavily logged plots, had a distinct species composition. These three plots showed also distinct low diversity values (Appendix 3.C). Further, deviations in forest structure were observed in these plots: plot 5 contained the lowest number of stems ≥5 cm dbh, i.e. 95 stems in the plot, and had a large number of banana trees. Plots 7 and 8 had the highest numbers of stems ≥5 cm dbh, with most stems in the dbh-class of 10-20 cm (67% for plot 7, 71% for plot 8). The total number of stems ≥5 cm dbh in plots 7 (251 stems) and 8 (311 stems) was considerably higher than the mean number of stems in the 18 plots, i.e. 162 stems (all plots, excluding plots 5, 7 and 8, contained between 109 and 189 stems ≥5 cm dbh). In general, plots located in the same geographical area were more similar in terms of species composition (see Fig. 2.1 for the spatial distribution of the plots).

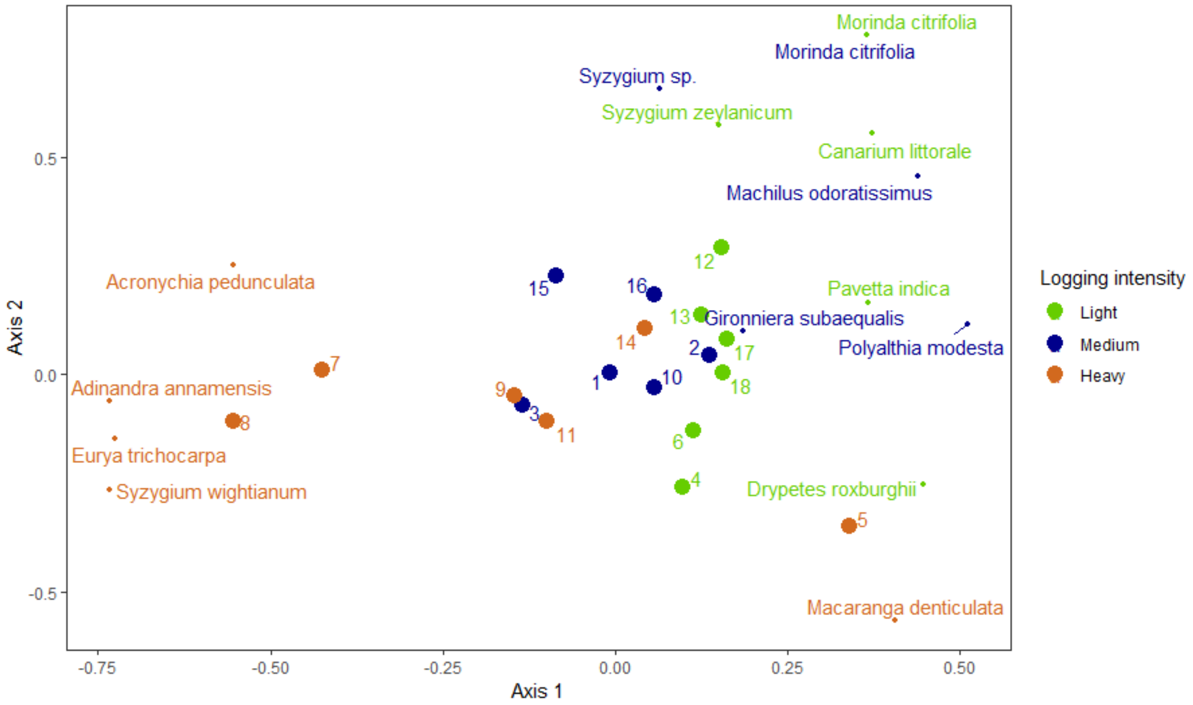


Figure 3.3. NMDS plot showing the dissimilarity in tree species composition of stems ≥5 cm dbh between the 18 plots and the five most abundant species for each logging class. The big dots represent the plots, the small dots the species.

The five most abundant species differed for each logging intensity, only *Morinda citrifolia* was an abundant species in both the lightly and medium logged forests (Table 3.2). Four out of the five most abundant species in the heavy logging class were strongly correlated with plots 7 and 8, especially *Eurya trichocarpa* was very abundant (Table 3.2) and strongly determines the first axis in the NMDS plot (Fig. 3.3). *Macaranga denticulata*, also among the most abundant species in the heavy logging class, was frequently present in plot 5. The most abundant species in the lightly and medium logged forests were more closely grouped.

Table 3.2. The five most abundant species and their abundances for each logging intensity. Stems ≥5 cm dbh are included and species are ordered by decreasing abundance per logging intensity.

Logging intensity	Species	Abundance
Light	*Pavetta indica* L.	57
Light	*Drypetes roxburghii* (Wall.) Hurus.	56
Light	*Canarium littorale* Blume	47
Light	*Morinda citrifolia* L.	40
Light	*Syzygium zeylanicum* (L.) DC.	32
Medium	*Morinda citrifolia* L.	64
Medium	*Gironniera subaequalis* Planch.	45
Medium	*Machilus odoratissimus* Nees	38
Medium	*Syzygium sp.*	37
Medium	*Polyalthia modesta* Finet & Gagnep.	36
Heavy	*Eurya trichocarpa* Korth.	206
Heavy	*Adinandra annamensis* Gagnep.	67
Heavy	*Acronychia pedunculata* (L.) Miq.	42
Heavy	*Macaranga denticulata* (Blume) Müll. Arg.	39
Heavy	*Syzygium wightianum* Wight & Arn.	39

3.3.3. Wood density

Although >80% of the stems was identified to species level, not all species were present in the global wood density database. From the 2919 stems ≥5 cm dbh, wood densities (wood specific gravity; wsg) were assigned for 971 stems to species level, for 1647 stems to genus level and for 301 stems to plot level. The wsg of individual stems ≥5 cm dbh ranged from 0.310 to 0.780 g cm^{-3} for each logging class. Community wsg of stems ≥5 cm dbh decreased significantly with logging intensity (Table 3.3).

Table 3.3. Mean ± standard deviation community wood density (g cm^{-3}) for stems ≥5 cm dbh for each logging intensity.

Logging intensity			p	Test
Light	Medium	Heavy		
0.574 ± 0.020^{a}	0.556 ± 0.024^{a}	0.523 ± 0.013^{b}	0.002*	ANOVA

Most stems in the lightly and medium logged forests had a wsg in between 0.5 and 0.7 g cm^{-3} (Fig. 3.4). In the heavily logged forests, 54% of the stems had a wsg between 0.5 and 0.6 g cm^{-3}, with a reduced number of stems in the wsg-class of 0.6-0.7 g cm^{-3}.

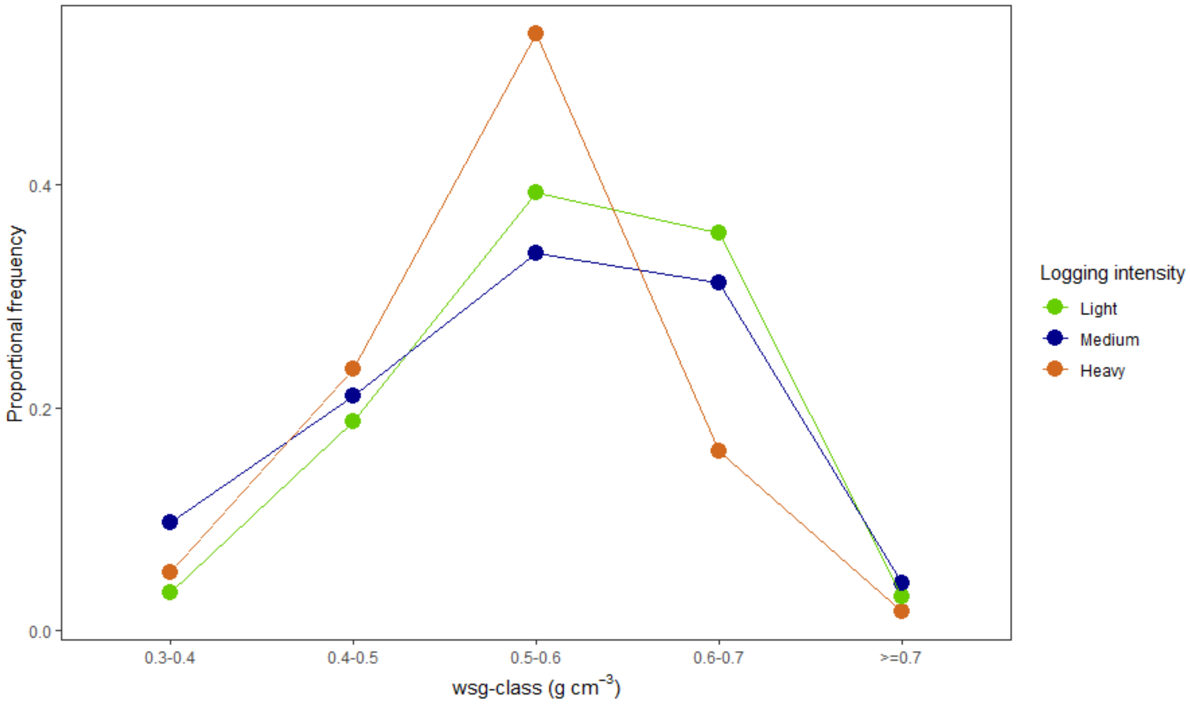


Figure 3.4. Proportional frequencies of stems ≥5 cm dbh per wood density class for each logging intensity.

3.3.4. Availability of timber species

At least 14 tree species are generally extracted for timber in KNT (Viet Nature Conservation Centre, pers. comm., 2018), listed in Table 3.4. Only six of these timber species were present in our plots and most of them occurred in small numbers. *Canarium littorale* and *Sindora tonkinensis* were more abundant and occurred in 16 and 8 plots, respectively. Only *Canarium littorale* had a decent amount of stems of harvestable size (≥30 cm dbh), with highest abundance in the lightly logged forests (Table 3.4). The abundance of all timber species combined was highest in lightly and medium logged forests and most trees from harvestable size were observed in the lightly logged plots.

Table 3.4. The abundance of timber species in the plots.

The absolute number of stems ≥5 cm dbh of the 14 most extracted timber species in KNT (Viet Nature Conservation Centre, pers. comm., 2018) present in the plots, both below and above the minimum cutting size (30 cm dbh) and classified by logging intensity.

			# stems 5-30 cm dbh			# stems ≥30 cm dbh		
Species	Vietnamese name	Wsg (g cm^{-3})	Light logging	Medium logging	Heavy logging	Light logging	Medium logging	Heavy logging
Amesiodendron chinense (Merr.) Hu	Trường	0.835	-	-	-	-	-	-
Canarium littorale Blume	Chủa/Trám nâu	0.495	18	25	13	29	9	7
Chukrasia sp.	Chua khét	0.548	-	-	1	-	-	-
Cinnamomum ilicioides A. Chev.	De	0.467	-	-	-	-	-	-
Dysoxylum cauliflorum Hiern	Dạ hương	0.715	-	-	-	-	-	-
Erythrophleum fordii Oliv.	Lim xanh	0.760	-	4	1	-	1	-
Heritiera cochinchinensis (Pierre) Kosterm.	Huỷnh	0.689	-	-	-	-	-	-
Hopea pierrei Hance	Kiền kiền	0.812	-	-	-	-	-	-
Madhuca pasquieri (Dubard) H.J. Lam	Sến	0.668	3	5	3	-	1	-
Michelia sp.	Giổi	0.515	4	-	1	-	-	-
Ormosia laosensis Niyomdham	Ràng ràng lào	0.579	-	-	-	-	-	-
Sindora siamensis Miq.	Gõ	0.716	-	-	-	-	-	-
Sindora tonkinensis K. Larsen & S.S. Larsen	Gụ lau	0.710	5	14	2	2	2	-
Vatica cinerea King	Táu mật	0.707	-	-	-	-	-	-
Sum all timber species			30	48	21	31	13	7

3.4. Discussion

In this study, I assessed whether logging intensities relate to differences in tree diversity, composition, wood density and availability of timber species in KNT. Tree diversity did not differ significantly with logging intensity. The NMDS analysis showed small compositional differences between logging classes and the most abundant tree species differed with logging intensity, although the challenge of sampling at the species level sufficiently complicates interpretation. Stand-level wood density decreased with logging intensity and timber species were rare in my plots.

3.4.1. The impacts of logging on tree diversity

Nine years post-logging, tree diversity did not differ significantly with logging intensity at my site. Due to the high variability in tree diversity in the heavily logged sites, more plots in heavily logged forests would be needed to assess whether tree diversity patterns are consistent with expectations of the IDH, i.e. local species diversity peaks at intermediate disturbance levels. Other work supported the expectations of the IDH (Bongers et al., 2009; Connell, 1978; Sheil and Burslem, 2003) and a meta-analysis also found that tree species richness appeared to increase at low logging intensities but decreased at higher logging intensities (Martin et al., 2015). High logging intensity generally results in a decrease in tree diversity compared to unlogged and lightly logged forests (Lindemalm and Rogers, 2001; Parrotta et al., 2002). Deviating from the IDH, low logging intensity resulted in a decrease in tree diversity compared to unlogged forests in Papua New Guinea 6 years post-logging (Lindemalm and Rogers, 2001).

When comparing logged with unlogged forests in Southeast Asia, tree diversity metrics are commonly higher in logged forests, which was assessed 8 years after logging (Cannon et al., 1998; Sheil et al., 1999), 12-18 years after logging (Berry et al., 2010) and 35–40 years after logging (Ding et al., 2012). The observed increase in richness after logging is likely due to an influx of generalist species that depend on disturbances and hence increase diversity metrices (Sheil et al., 1999). However, no differences in tree diversity were observed between logged and unlogged forests during the first 20 years after logging in Borneo (Verburg and Van Eijk-Bos, 2003)**.** Differences amongst studies can be partly explained by differences in spatial scale (Berry et al., 2008; Imai et al., 2012), logging intensity, time since logging, diversity metrics assessed, whether or not results are standardized by number of individuals and the diameter threshold considered. As diversity metrics are scale-dependent, community (dis)similarity is a more robust metric to indicate the response of tree assemblage to disturbances (Imai et al., 2012).

3.4.2. The impacts of logging on tree species composition

I observed a similar species composition among plots in the lightly and medium logged forests, while the composition of heavily logged sites was more distinct, especially in plots 5, 7 and 8. The most abundant species in the heavy logging class, *Eurya trichocarpa*, was very abundant in plots 7 and 8. This species usually occurs in undisturbed and slightly disturbed sites (Slik, 2009). *Macaranga denticulata* was dense in plot 5 and is a light-demanding and fast-growing tree, usually occurring at forest edges or abandoned shifting cultivation lands and along road sides, which regenerates strongly in full light (FIPI, 1996). After sampling the plots, local people reported that areas of forest near plots 5, 7 and 8 had been cleared and rice had been planted in the past. The remote sensing analysis showed no evidence of complete canopy removal and conversion to agriculture since 1988, therefore if any agricultural disturbances occurred this must have happened before 1988. Besides the compositional differences in these three plots, the presence of bananas, very distinct numbers of stems, high number of stems of

10-20 cm dbh and low plot-level diversity metrics further indicate that these plots have been considerably disturbed in the past and supports my understanding that plots 5, 7 and 8 contain mostly secondary vegetation. Two other abundant species in the heavily logged forest, *Acronychia pedunculata* and *Syzygium wightianum*, are light-demanding trees (FIPI, 1996) and are therefore associated with more open forest canopies. *Morinda citrifolia* was abundant in both the lightly and medium logged forests, but usually this species is found in open secondary forests (Slik, 2009). *Canarium littorale* was among the most abundant species in the lightly logged forests and is an important timber species in KNT. This species is sporadically regenerating at forest edges and in open places (FIPI, 1996). Reduced numbers of this species in the heavily and medium logged forests, especially stems from harvestable size (Table 3.4), is likely due to depletion by logging. Other abundant species in the lightly and medium logged forests are associated with neutral (*Syzygium zeylanicum*; shade-demanding when young, light-demanding when grown up) and high light conditions (*Machilus odoratissimus*; light-demanding, fast-growing tree that occurs in primary and secondary forests and produces a very large number of fruits and seeds) (FIPI, 1996). *Gironniera subaequalis*, the second most abundant species in the medium logged forests, is a fast-growing tree that demands shade when it is young and light when it is mature and tends to develop in a dominant stand in Quang Binh province (FIPI, 1996).

Although the NMDS analysis showed that plots in the lightly and medium logged forests were grouped per logging class, plots located close to each other had also a more similar species composition. Therefore, it is difficult to determine whether observed compositional differences can be attributed to natural variation in tree species or to differences in logging intensity. The challenge of sampling at the species level sufficiently in these high diversity systems complicates interpretation. A study that analysed the effects of logging on compositional diversity in tropical forests found indeed that most studies were pseudo-replicated, leading to biased estimates of change (Ramage et al., 2013).

Some studies observed a similar tree species composition in logged and unlogged tropical forests, i.e. in southern China (35–40 years since logging; Ding et al., 2012) and in the Central African Republic (6 months and 18 years post-logging; Hall et al., 2003). Others found that tree species composition differed between logged and unlogged forests in Borneo (18 years after logging; Berry et al., 2008) and between heavily and lightly logged forests in Vietnam (16 years after logging; Hoang et al., 2011). Also over time, starting just after logging and during the following 20 years, logged forests had much larger changes in tree species composition than old-growth forests in Borneo (Verburg and Van Eijk-Bos, 2003). Generally, more early successional tree species and species of low conservation value are observed in (heavily) logged or disturbed forests, while vulnerable species and valuable timber species are most common in lightly logged or undisturbed forests (Bongers et al., 2009; Carreño-Rocabado et al., 2012; Clark and Covey, 2012; Hoang et al., 2011). This occurs through logged-induced changes in biotic and abiotic processes that regulate resource availability and through size-class and species specific selection common in selective logging (Clark and Covey, 2012). Besides logging, KNT experiences yearly disturbances from typhoons, which has likely further impacted tree diversity and composition at my site. On the Solomon Islands, periodic cyclones appeared to favour the maintenance of differences in species diversity and composition between forest types (Burslem and Whitmore, 1999).

3.4.3. The impacts of logging on wood density and availability of timber species

I observed a shift towards lighter community wood density in heavily logged forests, likely due to a combination of a reduced number of timber species with high wood density in heavily logged forests (Table 3.4) and changed species composition in post-logging regeneration. Similarly, lower community wood density values were observed at high disturbance intensity from logging and silvicultural treatments in Bolivian tropical forests (Carreño-Rocabado et al., 2012). Lighter wood density species are often less suitable for timber purposes.

The 14 reported timber species (Table 3.4) all occur inside KNT. However, only six of these timber species were present in my plots and they only occurred in low numbers, especially in the heavily logged forests, likely due to more intense logging. It is not clear whether the remaining eight timber species did not occur naturally in my plots or that these high-value timber species have been depleted by logging. Scarcity of high-value timber species after logging was seen also elsewhere in Vietnam (Van and Cochard, 2017). In Borneo, more small Dipterocarps (<30 cm dbh), the main timber group in the study, were observed in a logged forest, while more Dipterocarps above harvestable size (>60 cm dbh) were found in an unlogged forest (Berry et al., 2008). The relatively high abundance of Dipterocarps in the understorey of the logged forest suggests that trees of commercial species survived and recruited after logging (Berry et al., 2008). An average of >90% of species recorded in primary forest were also present in logged Dipterocarp forest, including valuable conservation species (Berry et al., 2010). Silvicultural interventions such as thinning, in which the tree density is reduced to eliminate competing trees, can be applied to increase recovery of timber stocks (de Avila et al., 2017).

3.5. Conclusions

In this study, I assessed whether logging intensity caused differences in tree diversity, composition, wood density and availability of timber species in lowland forests in Vietnam. Tree diversity did not differ significantly with logging intensity. More plots in heavily logged forests would be needed to assess whether diversity patterns at my site are consistent with expectations of the IDH. Some differences in tree species composition were observed with logging intensity, although the challenge of sampling at the species level sufficiently complicates interpretation. Community wood density differed with logging intensity, changing towards lighter wood density species in heavily logged forests. Timber species only occurred in small numbers in my plots, especially in the heavily logged forests. Reducing logging intensities is thus important to prevent the dominance of light-demanding pioneer species, to maintain high community wood density values, and thereby conserving forest carbon stocks, as well as to prevent the depletion of timber species after logging.

Chapter 4. The impacts of logging intensity on carbon recovery of Vietnamese forests

Abstract

Selective logging creates forest canopy gaps, which alter light regimes in tropical forests. As light is often a limiting factor for plants in tropical forests, changed resource availability due to logging is expected to impact the growth, recruitment and mortality rates of trees. However, it remains unclear how logging intensity impacts these vital rates and aboveground carbon (AGC) recovery in the evergreen tropical forests of mainland Southeast Asia. Here, I assess the impacts of logging intensity on vital rates and associated AGC dynamics over a two-year period in Vietnamese forests nine years after logging. Averaged over all plots for stems ≥10 cm diameter, tree diameter growth rates were equal to 0.3 cm yr^{-1}, recruitment rates to 1.5% yr^{-1} and mortality rates to 2.3% yr^{-1}. I found evidence across the whole study landscape that stand competition slowed diameter growth, yet there was no clear evidence that logging intensity stimulated tree growth. All vital rates and their equivalent AGC fluxes did not differ with logging intensity. Meanwhile, AGC net changes ranged from -4.4 to 2.6 Mg C ha^{-1} yr^{-1} and averaged 0.1 Mg C ha^{-1} yr^{-1}, with nearly 40% of plots experiencing a negative annual net change in AGC. AGC net changes during the monitoring period were strongly driven by the natural mortality of trees in the plots. Breakage of stems also contributed in some plots to negative AGC net changes by reducing tree heights. Central Vietnam experiences typhoons every year and the forests suffer from considerable storm-induced mortality and damage, which retards post-logging AGC recovery in these forests.

4.1. Introduction

All plants compete for resources, i.e. light, water and nutrients, for their growth, survival and reproduction. In tropical forests, light is often the major environmental factor that limits plant growth, survival and reproduction (Chazdon, 1988; Chazdon et al., 1996). Light availability varies extremely in tropical forests and irradiance values near the forest floor are often less than 1% of the available sunlight (Chazdon et al., 1996). Openings in the forest canopy alter light regimes, which leads to increased irradiance for understorey plants (Chazdon et al., 1996). Selective logging reduces the stand density and creates forest canopy gaps (Asner et al., 2004b), which changes the light environment of the forest (Yamada et al., 2014). It is likely that the changed resource availability after logging impacts diameter growth, recruitment and mortality rates, the so-called vital rates, of trees.

Generally, canopy openings resulting from logging stimulate diameter growth rates of trees: in studies from French Guiana and Indonesia, growth rates in logged forests were considerably higher than in primary forests (Delcamp et al., 2008; Hérault et al., 2010; Sist and Nguyen-The, 2002). In forests in Borneo, tree growth rates in logged plots differed significantly with logging intensity, with the highest growth rates typically in the most heavily logged stands (Lussetti et al., 2016; Sist and Nguyen-The, 2002). Selective logging often removes the biggest trees and thereby part of the reproductive trees in the stand (Bagchi et al., 2011). At the same time, renewed space and resources can provide opportunities for settlement of new recruits. In French Guiana and Indonesia, logging resulted in increased recruitment rates compared to undisturbed populations (Delcamp et al., 2008; Sist and Nguyen-The, 2002). Recruitment rates increased with logging intensity in forests in Borneo (Lussetti et al., 2016; Sist and Nguyen-The, 2002). Further, logging affects the mortality rate of tree populations: directly by harvesting specific timber species and indirectly by increasing the mortality probability of injured surrounding trees shortly after logging (Pinard and Putz, 1996; Shenkin et al., 2015; Sist et al., 2003b; Sist and Nguyen-The, 2002). Typically, mortality rates are higher in logged compared to unlogged forests during the first few years after logging, while thereafter similar rates are observed in logged and unlogged forests (Blanc et al., 2009; Sist and Nguyen-The, 2002; Yamada et al., 2013). Mortality rates increased with logging intensity in forests in Borneo, directly as well as longer after logging, likely due to more damaged trees at high logging intensity (Lussetti et al., 2016; Sist and Nguyen-The, 2002). A study in Malaysia that assessed the long-term impacts of logging found that vital rates of trees in logged and primary forests differed even 50 years after logging (Yamada et al., 2013).

Consequently, changes in tree vital rates affect aboveground carbon (AGC) stocks and dynamics of forests. Various studies assessed AGC dynamics after logging in tropical forests (Blanc et al., 2009; Gourlet-Fleury et al., 2013; Mazzei et al., 2010; Piponiot et al., 2016; Roopsind et al., 2018; Rutishauser et al., 2015; Sist et al., 2014; Vidal et al., 2016; West et al., 2014), with only a few studies in Southeast Asian forests (Berry et al., 2010; Do et al., 2018; Nam et al., 2018). Depending on the magnitude of diameter growth, recruitment and mortality, forests can become a net carbon (C) source or sink after logging. Shortly after logging, forests are often a net C source due to high mortality of damaged trees, before returning to a C sink (Blanc et al., 2009; Mazzei et al., 2010). Usually, AGC net change rates in logged forests are higher than in unlogged forests (Berry et al., 2010; Roopsind et al., 2017), likely because remaining trees benefit from reduced competition (Villegas et al., 2009). Estimates about the time needed to recover pre-logging AGC stocks in logged forests vary from decades to over a century (Blanc et al., 2009; Pinard and Cropper, 2000; Rutishauser et al., 2015).

Studies from the Amazon showed that the speed of post-logging AGC recovery was primarily related to logging intensity (Piponiot et al., 2016; Rutishauser et al., 2015). After the initial period of AGC stock losses after logging, AGC recovery rates increased with logging intensity in the Central African Republic and Central Guyana (Gourlet-Fleury et al., 2013; Roopsind et al., 2018). However, remarkably, no studies have yet analysed the impacts of logging intensity on AGC recovery in Southeast Asian forests, in spite of the importance of this for managing national C emissions and sequestration for countries in this region. Extensive forests in Vietnam have been disturbed by logging over recent decades (McElwee, 2004). Vital rates and associated AGC dynamics at the population level have been very rarely studied in Vietnam. The only work on this in logged Vietnamese forests has been performed in evergreen broadleaf forests in the Central Highlands (Do et al., 2018; Nam et al., 2018). Yet, the impact of logging intensity on forest recovery in Vietnam and the recovery potential of logged Vietnamese forests beyond the Central Highlands remain unclear. To better understand recovery of logged forests in mainland Southeast Asia, I assess here the impacts of logging intensity on vital rates and associated AGC dynamics in north-central Vietnam, nine years after logging and assessed over a two-year period.

4.2. Material and methods

4.2.1. Site description

Measurements were performed in the logged forests of Khe Nuoc Trong (KNT) in Quang Binh Province, north-central Vietnam. The site has been described in detail in Chapter 1.10.

4.2.2. Field measurements

I established 24 permanent plots of 0.25 ha each (50 m × 50 m), 6 ha in total, in lowland forests (< 700 m elevation) in Vietnam. Plots varied in the degree of degradation due to historical logging. In the first census, plots were measured between April 2016 and June 2017. All living woody stems ≥10 cm in the plots were tagged, their diameter measured at breast height (dbh) and their species identified. Total living tree heights were measured for all deviating stems, i.e. broken, fallen, leaning, bended, rotten, hollow, resprouted, forked ≤1.3 m height, fluted stems and stranglers. For "Intact" trees, stems without these deviations, heights were measured from a subset of trees in each diameter class and subsequently a local height-diameter (HD) model was constructed. The dbh, height and species of stems 5–10 cm dbh were measured and identified in a belt transect of 4 m × 50 m, running through the centre of each plot. The measurements in the first census have been described in more detail in Chapter 2.2.2.

All plots were re-measured two years after the initial tree census, i.e. between May 2018 and May 2019. To minimize the effects of seasonal fluctuations on stem expansion and shrinkage (Sheil, 2003, 1995), plots were re-measured as close as possible to the first census date, with a maximum of one month difference. In the second census, the dbh of each living stem was re-measured, it was recorded if a tagged stem died and newly recruited stems (i.e. stems ≥10 cm dbh in the plot and stems 5-10 cm dbh in the belt transect) were measured as described in the first census. If the stem was irregular, damaged, had a buttress or close to a buttress or had other problem at the original point of measurement (POM), a new POM was chosen for the dbh measurement according to the RAINFOR and GEM protocols (Marthews et al., 2014; Phillips et al., 2016; http://www.rainfor.org/en/manuals). The diameter was measured both at the original POM and the new POM. I measured only tree heights of stems with a broken top in the second census. For stems that died, the cause of mortality (natural or logged) was recorded.

4.2.3. Data analysis

4.2.3.1. Logging assessment

We analysed Landsat images between 1988 and 2016 inclusive to evaluate the spatial pattern of historical forest disturbances associated with selective logging. The logging intensity per plot was estimated by summing the number of disturbed pixels (identified in the satellite analysis) within 1000 m of each plot centre. For some analyses, the continuous logging intensity was classified into three logging classes, i.e. light, medium and heavy logging intensity, by ordering the plots by increasing number of disturbances and dividing the plots equally over the logging classes (n = 8 plots per logging class). Participatory mapping was used to verify these historical logging activities, but the logging intensity according to the Landsat analysis was used in the analyses. This approach was described in detail in Chapter 2.2.3.

4.2.3.2. Estimation of vital rates

Vital rates, i.e. dbh growth, recruitment and mortality rates, were calculated for stems ≥10 cm dbh. The yearly dbh growth rate per tree was calculated by dividing the change in dbh between the censuses by the years of the census interval. We defined the range of acceptable dbh changes of trees ≥ -0.2 cm yr^{-1} and $\leq +4$ cm yr^{-1} (Sheil, 1995). Trees with dbh growth rates outside this range, i.e. likely due to measurement errors, or a missing dbh in the second census were excluded from the dbh growth rate analysis. Yearly dbh growth rates were averaged by plot to derive mean plot-level dbh growth rates. A field study showed that six threatened long-lived forest tree species in four Vietnamese protected areas started, in general, to be reproductive at 10–20 cm dbh (Chien et al., 2008). Those six study species were selected using the following criteria: conservation status (i.e. included in Vietnam's Red List), preferably endemic to the region, taxonomically well described, known locations of populations in conservation areas, sufficiently large populations and from a variety of forest types (i.e. altitudes and locations).

Based on this, we assume that all trees ≥10 cm dbh in our study are vegetatively and/or sexually reproductive. The annual recruitment rate per plot was calculated by dividing the yearly number of recruits (i.e. number of recruits observed in the second census divided by the census interval) by the total number of living stems in the first census and multiplying by 100% (Flores et al., 2014). Equally, the annual mortality rate for each plot was derived by dividing the yearly number of dead stems (i.e. number of dead stems observed in the second census divided by two years) by the initial number of living stems and multiplying by 100% (Flores et al., 2014; Wright et al., 2010). Other studies used different formulas to estimate recruitment and mortality rates in order to correct for biases such as differences in census intervals and unseen recruitment and mortality events during the census interval (e.g. Kohyama et al., 2018; Lewis et al., 2004; Sheil et al., 1995; Sheil and May, 1996). In my study, census intervals of all plots are equal in length and short, i.e. two years, therefore it is reasonable to expect that these potential biases do not confound our results. I use basal area as a measure for stand density and explore relationships between the vital rates and initial plot basal area.

4.2.3.3. Estimation of AGC stocks and AGC fluxes

From the trees that had a dbh change rate within the acceptable range, we calculated the mean dbh growth rate in the following dbh classes (all plots grouped): 5-10, 10-20, 20-30, 30-40, 40-50 and ≥50 cm dbh. For trees with a dbh growth rate outside the acceptable dbh change range, we assigned this mean dbh growth rate according to its dbh class and corrected the dbh in the first census accordingly. If a tree showed a substantial growth or decline in the second census, it was carefully checked in the field if the dbh was measured correctly. Therefore, we assume that it is more likely that a measurement error was made in the first census and we corrected the dbh in the first census. We also assigned the mean dbh growth rate to stems from which the dbh was not recorded in the second census. All analyses (except the dbh growth rate calculation, in which trees with dbh growth rates outside the acceptable dbh change range or a missing dbh in the second census were excluded) were performed with these corrected dbh values.

The BIOMASS package in R (Rejou-Mechain et al., 2017) was used to develop a local HD-model, extract wood density (wood specific gravity; wsg) values and calculate plot-level living aboveground biomass (AGB) of stems ≥5 cm dbh. From the subset of intact trees with height measurements in the first census, a local HD-model (n = 1065, Weibull function) was developed, using the corrected dbh values in the initial census. The linear relationship between logarithmic height and logarithmic diameter did not differ significantly with logging class, therefore the intact trees from all plots were grouped to develop one HD-model. The HD-model was used to estimate heights of all intact trees, i.e. also intact trees with height measurements, in both censuses. Heights of stems with a broken top in the second census were assigned from the individual height measurements during the second census. The other deviating stems in the second census (not with broken top, but other reasons for deviating stems) and all deviating stems in the first census were assigned the individual height measurements taken during the initial census. We thus assumed that these deviating stems did not grow in tree height during the census interval. Wood densities were assigned using the global wood density database (Chave et al., 2009; Zanne et al., 2009). Tree AGB was computed using the generic allometric equation 4 of Chave et al. (2014), including dbh, height and wsg as input parameters (Rejou-Mechain et al., 2017), and summed for all living stems per plot and census to retrieve plot-level AGB for both censuses. AGB stocks were converted into AGC stocks using a default value of 47% C content (IPCC, 2006). Further details on assigning wood densities are described in Chapter 2.2.4.1.

AGC gains during the census interval were retrieved by summing the AGC dbh growth of surviving trees and the AGC of recruits. AGC losses represent the AGC in trees that died during the census interval. The AGC net change was calculated by subtracting the AGC losses from the AGC gains (equal to AGC growth + AGC recruitment – AGC mortality).

The AGC stocks in the first census are calculated slightly different compared to the AGC stocks in Chapter 2. The initial AGC stocks in this chapter are calculated with corrected dbh values (for trees with dbh growth rates outside the acceptable dbh change range) and a new corresponding HD-model was constructed. Further, we assigned tree heights differently: here, all intact trees were assigned the height derived from the HD-model, while in Chapter 2 heights from the HD-model were only assigned to intact trees without height measurements.

4.2.3.4. Testing differences between logging classes

Differences in vital rates, AGC stocks and AGC fluxes were tested amongst logging classes using ANOVA or, when distributions within groups were not normally distributed, the Kruskal-Wallis rank sum test. For parameters that showed differences among logging classes, a Tukey test was performed to identify which logging classes differ. All analyses were carried out using R (version 3.5.3; R Core Team, 2019).

4.3. Results

4.3.1. Vital rates

At tree-level, measured dbh growth rates of stems ≥5 cm dbh ranged from -5 cm yr^{-1} to +8.65 cm yr^{-1}. Forty-two stems had a dbh growth rate outside the limits of acceptable dbh change (88% of these were negative values, 12% positive values) and three stems had a missing dbh in the second census; these trees were excluded from the dbh growth rate analysis. The dbh growth rates did not show a clear pattern with initial dbh (Fig. 4.1).

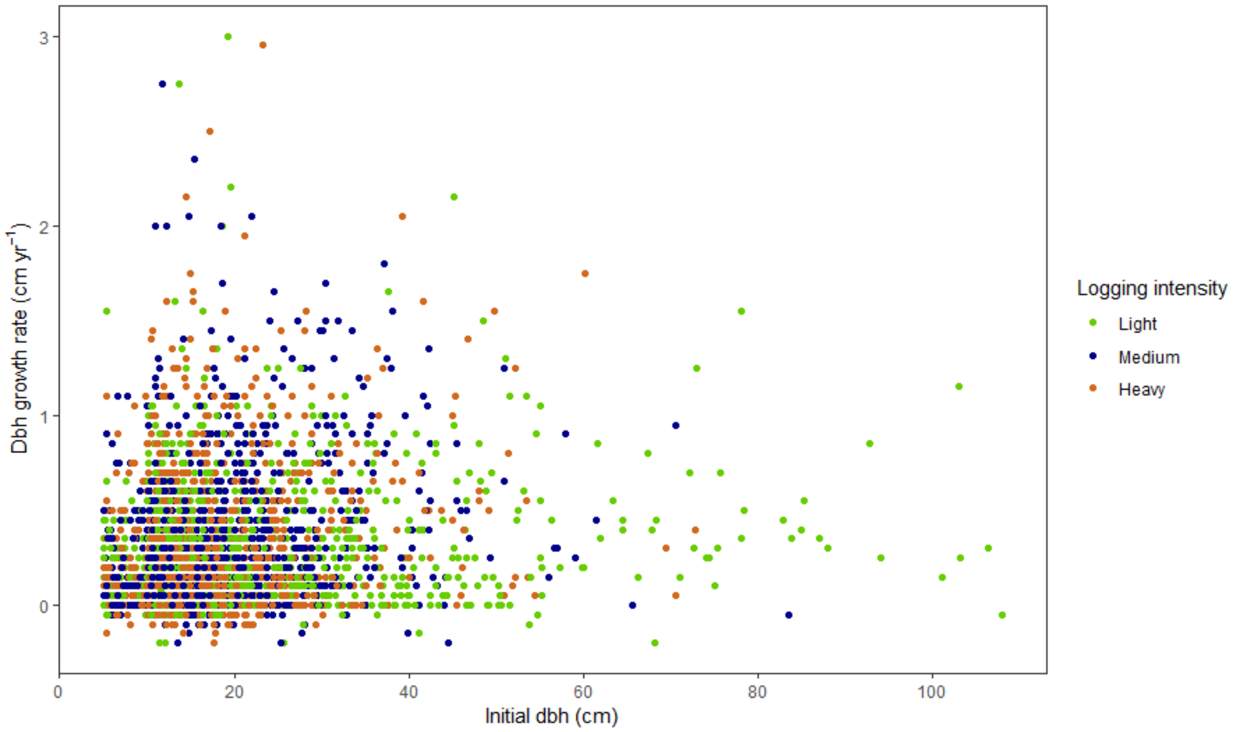


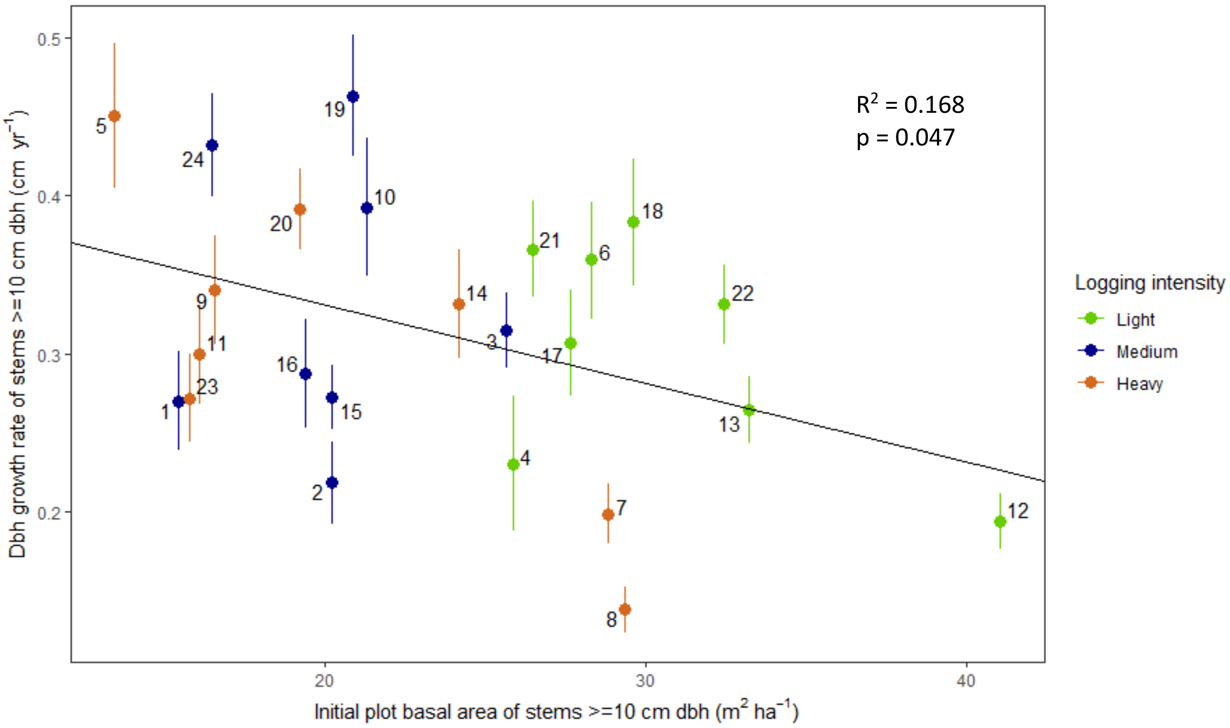
Figure 4.1. The yearly dbh growth rate versus the initial dbh for each individual stem ≥5 cm dbh.

Each dot represents a stem and has been coloured by logging class.

Plot-level dbh growth rates decreased significantly with plot basal area (Fig. 4.2a). Plots with lower basal area have a less dense forest stand and likely experience less competition for light and nutrients, which can enable higher dbh growth rates. Heavily and medium logged plots had generally a lower basal area than lightly logged plots (Fig. 4.2a), but still no significant differences in dbh growth rates were observed with logging class (Table 4.1; Appendix 4.A shows vital rates per plot). Plots 7 and 8 had a remarkable high basal area for heavily logged plots, due to the very high number of stems in these plots (Appendix 4.B). In contrary, no relationship was observed between plot-level recruitment rates and plot basal area (Fig. 4.2b).

Recruitment rates of stems in the ≥10 cm diameter class were low in most plots (Appendix 4.A) and there was no significant difference with logging intensity (Table 4.1). No relationship between plot-level mortality rates and plot basal area (Fig. 4.2c) suggests that there are no direct signs of competition-driven mortality in denser forest stands and that other disturbances might explain the mortality events at our sites. Mortality rates did not differ between logging classes (Table 4.1).

a)



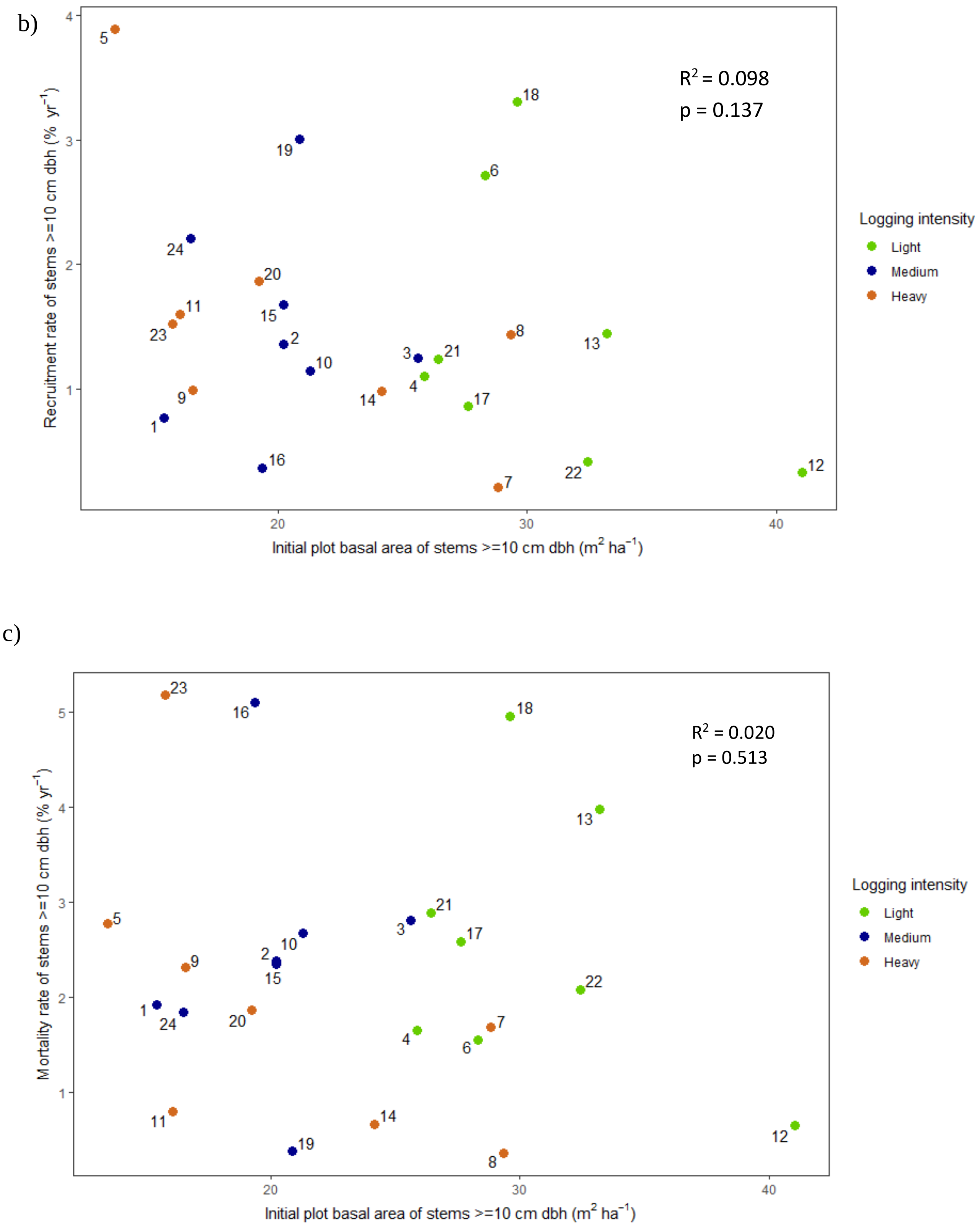


Figure 4.2. The relationship between plot-level vital rates, i.e. the mean dbh growth rate (± standard error) (a), total recruitment rate (b) and total mortality rate (c), and the initial plot basal area.

Each dot represents a plot, marked with the corresponding plot number, and has been coloured by logging class.

Table 4.1. Mean ± standard deviation for vital rates for stems ≥10 cm dbh for each logging class.

Vital rates per plot are shown in Appendix 4.A.

Vital rates for stems ≥10 cm dbh	Logging intensity			p	Test
	Light	Medium	Heavy		
Dbh growth rate (cm yr^{-1})	0.3 ± 0.1^a	0.3 ± 0.1^a	0.3 ± 0.1^a	0.762	ANOVA
Recruitment rate (% yr^{-1})	1.4 ± 1.1^a	1.5 ± 0.8^a	1.6 ± 1.1^a	0.962	ANOVA
Mortality rate (% yr^{-1})	2.5 ± 1.4^a	2.4 ± 1.3^a	2.0 ± 1.6^a	0.685	ANOVA

4.3.2. Aboveground carbon dynamics

Thirty-eight percent of the plots had a negative annual AGC net change (Fig. 4.3; Appendix 4.B). Plots with a negative AGC net change have greater AGC losses due to mortality and negative dbh growth than AGC gains by dbh growth and recruitment. AGC net changes were predominantly driven by mortality events, with exceptional high mortality in plot 18. Plots 1 and 13 showed negative AGC dbh growth values, even though we corrected the dbh of trees that showed an unacceptable dbh change. Plots 1 and 13 had many surviving trees with a broken top (reduced tree height), in the second census and this resulted in almost all cases in negative AGC dbh growth of those trees, which led to negative AGC dbh growth values for those plots. In all plots, recruits contributed a neglectable small fraction to AGC gains (Fig. 4.3).

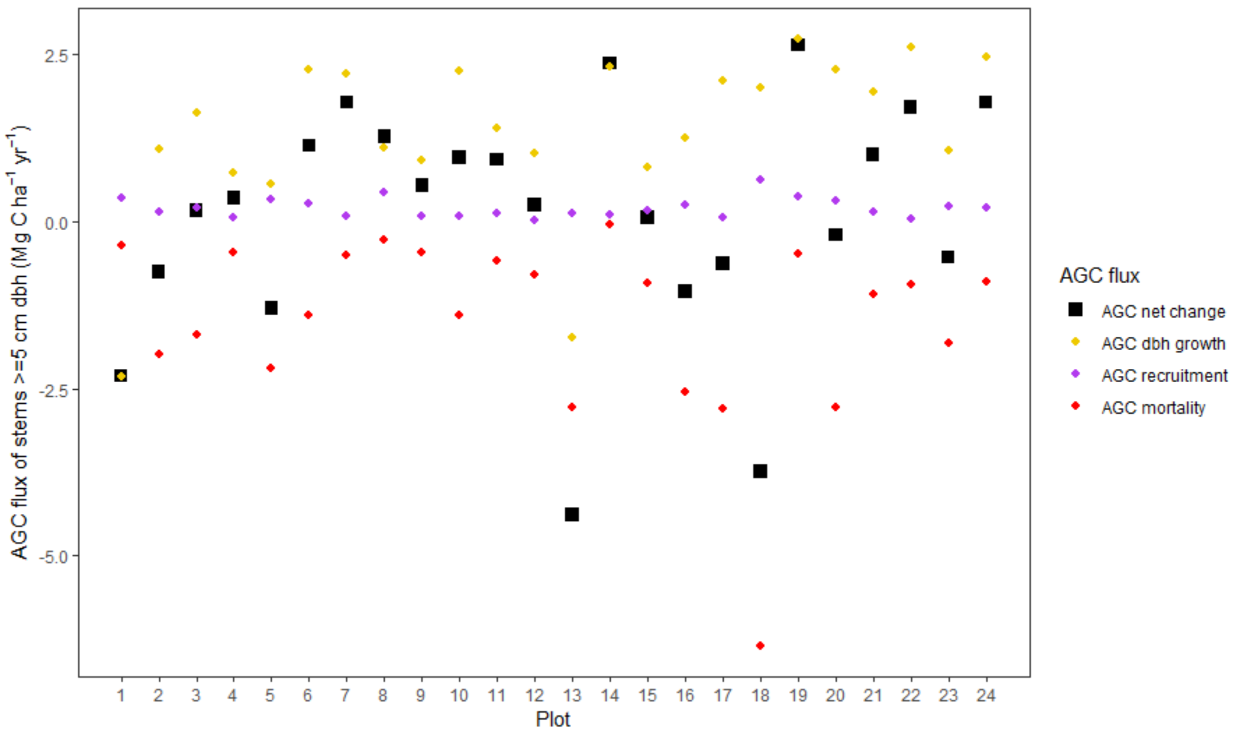


Figure 4.3. Yearly AGC fluxes of AGC net change, AGC dbh growth, AGC recruitment and AGC mortality of stems ≥5 cm dbh for each plot.

Positive values indicate carbon accumulations, negative values indicate carbon emissions. AGC net change was calculated by subtracting the AGC losses (AGC mortality) from the AGC gains (AGC dbh growth and AGC recruitment).

Large AGC losses were never caused by the mortality of a single big tree (Fig. 4.4). Plot 18 had a large number of dead stems ≥10 cm dbh, including big stems that died (i.e. half of the dead stems were ≥30 cm dbh), leading to the low AGC mortality value. Dead stems of 5-10 cm dbh contributed little to AGC mortality: 11 plots did not have dead stems of 5-10 cm dbh, while another plot had a maximum of 2 dead stems of 5-10 cm dbh yr^{-1}. Almost all mortality, i.e. 98% of all dead stems ≥10 cm dbh in the plots grouped, was natural and the remaining 2% was logged.

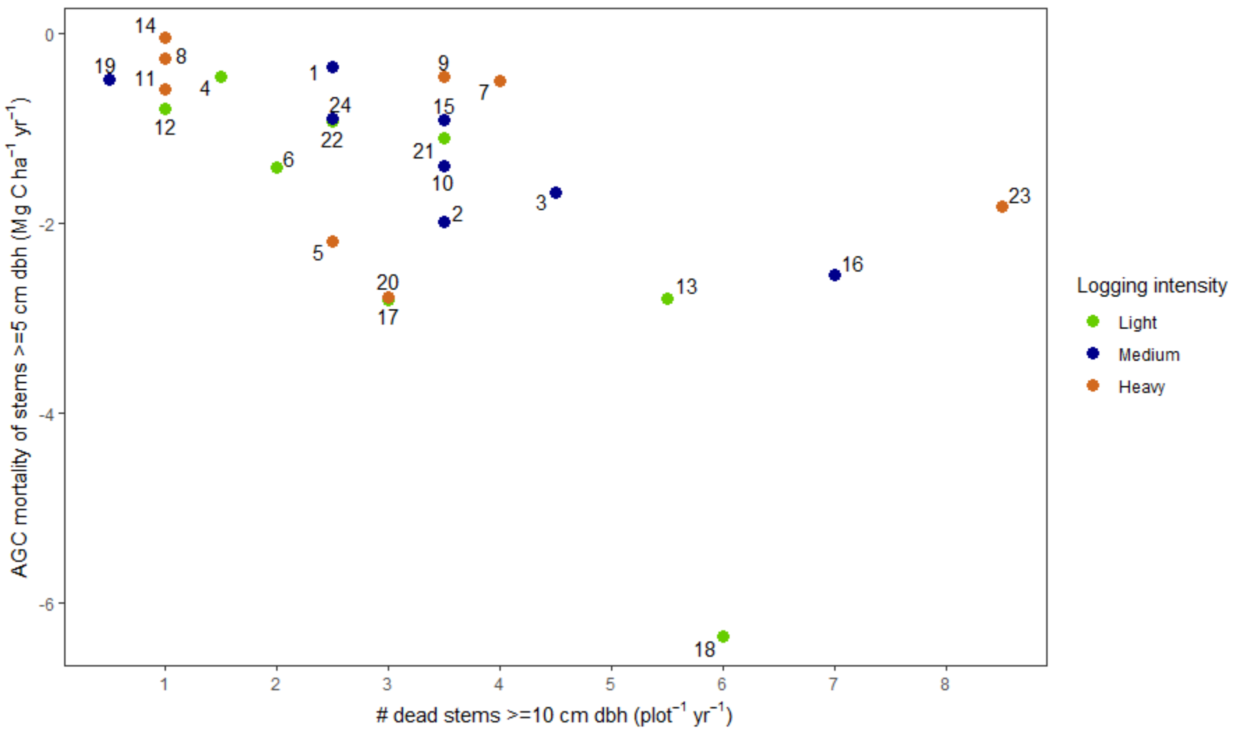


Figure 4.4. The AGC mortality of stems ≥5 cm dbh versus the number of dead stems ≥10 cm dbh for each plot.

Each dot represents a plot, marked with the corresponding plot number, and has been coloured by logging class.

The relationship (linear model) between AGC net change and logging intensity was not significant ($p=0.134$, $R^2=0.099$) (Fig. 4.5).

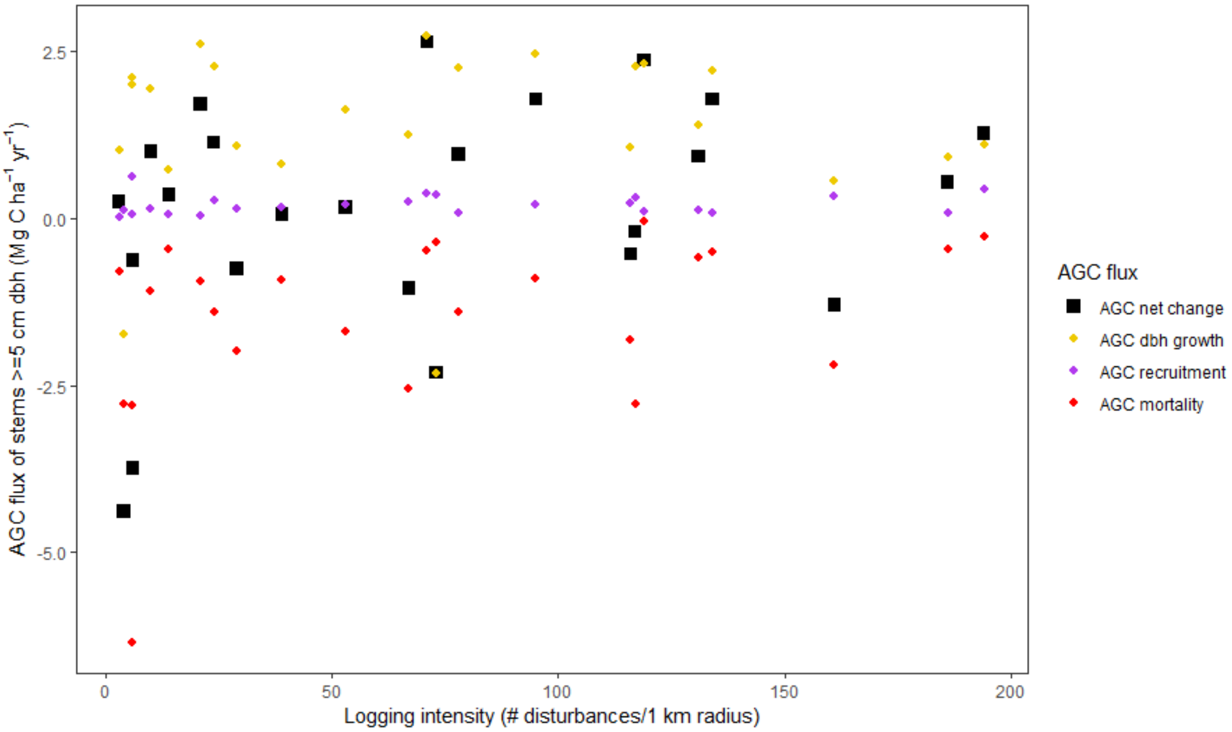


Figure 4.5. The AGC fluxes of stems ≥5 cm dbh versus the logging intensity.

The AGC net change has been shown, together with the separate AGC fluxes from which it was calculated (AGC dbh growth, AGC recruitment and AGC mortality).

For each census, AGC stocks decreased significantly with logging intensity (Table 4.2). All AGC fluxes, i.e. AGC dbh growth, AGC recruitment, AGC mortality and the AGC net change, did not differ with logging intensity (Table 4.2).

Table 4.2. AGC stocks and fluxes for stems ≥5 cm dbh for each logging class.

Values represent the mean ± standard deviation. Significant p-values are annotated with *, different letters within a given row indicate significant differences between logging classes. KW = Kruskal-Wallis rank sum test.

Stocks & fluxes	Logging intensity			p	Test
	Light	Medium	Heavy		
AGC stock census 1 (Mg C ha^{-1})	109 ± 23^{a}	60 ± 10^{b}	57 ± 19^{b}	<0.001*	ANOVA
AGC stock census 2 (Mg C ha^{-1})	107 ± 23^{a}	60 ± 11^{b}	59 ± 21^{b}	<0.001*	KW
AGC net change (Mg C ha^{-1} yr^{-1})	-0.5 ± 2.3^{a}	0.2 ± 1.6^{a}	0.6 ± 1.2^{a}	0.434	ANOVA
AGC dbh growth (Mg C ha^{-1} yr^{-1})	1.4 ± 1.4^{a}	1.2 ± 1.6^{a}	1.5 ± 0.7^{a}	0.968	KW
AGC recruitment (Mg C ha^{-1} yr^{-1})	0.2 ± 0.2^{a}	0.2 ± 0.1^{a}	0.2 ± 0.1^{a}	0.265	KW
AGC mortality (Mg C ha^{-1} yr^{-1})	-2.1 ± 1.9^{a}	-1.3 ± 0.8^{a}	-1.1 ± 1.0^{a}	0.330	KW

To assess whether plots with a low initial AGC, i.e. medium or heavy logged forests in my study, recover with the same rate as plots with a high initial AGC, i.e. lightly logged forests, we assessed the distribution of AGC net change versus initial AGC. No clear relationship was observed between the AGC net change and the initial AGC (Fig. 4.6). Plots 1, 13 and 18, the plots with highest AGC losses due to reduced tree heights (broken tops) and high mortality events form outliers in this graph.

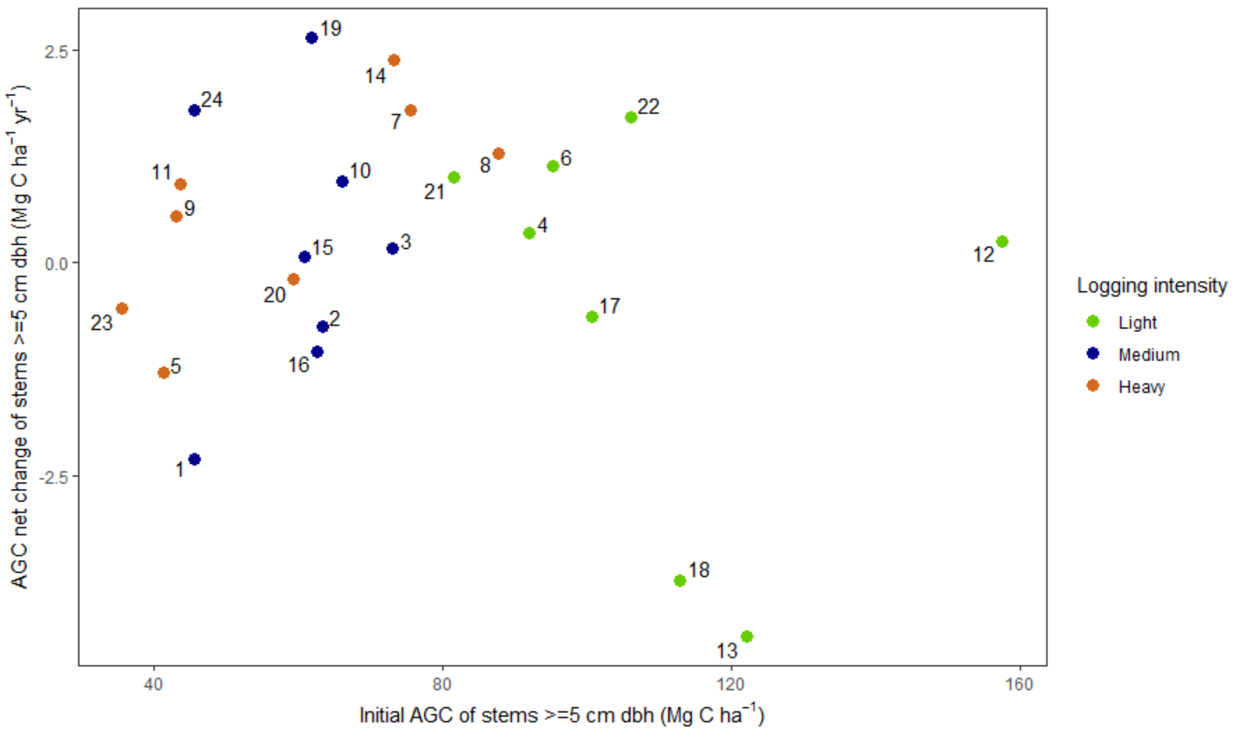


Figure 4.6. The annual AGC net change of stems ≥5 cm versus the AGC stock of stems ≥5 cm dbh in the first census.

Each dot represents a sample plot, marked with the corresponding plot number, and has been coloured by logging class.

4.4. Discussion

We estimated vital rates and AGC dynamics over a two-year period for forests nine years post-logging. Vital rates, AGC fluxes and AGC net changes did not differ with logging intensity. Thirty-eight percent of the plots showed a negative annual AGC net change, predominantly driven by mortality. Almost all mortality, i.e. 98%, was due to natural causes.

4.4.1. Vital rates

In this study, for all plots combined, dbh growth rates averaged 0.3 cm yr^{-1}, recruitment rates averaged 1.5% yr^{-1} and mortality rates averaged 2.3% yr^{-1}, and all rates did not differ with logging intensity. A similar mean dbh growth rate (0.4 cm yr^{-1}) was observed 30 years-post logging elsewhere in Vietnam, while the recruitment rate (2.5% yr^{-1}) in that study was higher and the mortality rate (1.4% yr^{-1}) lower (Nam et al., 2018). Other logged forests in Vietnam reported lower dbh growth rates (0.1 cm yr^{-1}), higher recruitment rates (2.2% yr^{-1}) and lower mortality rates (0.9% yr^{-1}) than our study (Do et al., 2018).

Elsewhere in the tropics, logged forests have commonly faster dbh growth rates than undisturbed forests in the first four years after logging (Delcamp et al., 2008; Hérault et al., 2010; Silva et al., 1995; Sist and Nguyen-The, 2002), although logging still stimulated dbh growth rates of trees even 50 years after logging in a Malaysian forest (Yamada et al., 2013). This is likely due to more open canopies and longer sunfleck time in logged forests than in primary forests (Yamada et al., 2014). Recruitment rates were generally higher in logged forests compared to undisturbed forests in the first four years after logging (Sist and Nguyen-The, 2002) and 4-15 years after logging (Delcamp et al., 2008). On the other hand, the recruitment rate in a 50-year post-logging forest was significantly lower than in the primary forest, possibly due to the extraction of adult trees and soil compaction by logging and a higher density of wild pigs (Yamada et al., 2013). Although logging gaps are often rapidly colonized, the establishment and growth of timber species is often strongly limited by the heavy regeneration of competing plants such as pioneer species and lianas (Fredericksen and Mostacedo, 2000; Sist and Nguyen-The, 2002). Higher mortality rates in logged forests than in unlogged forests were observed the first two (Sist and Nguyen-The, 2002) and eight years (Blanc et al., 2009) after logging, likely due to a higher mortality of injured trees during logging events (Sist and Nguyen-The, 2002). Thereafter, mortality rates in the logged and unlogged forests were similar (Blanc et al., 2009; Sist and Nguyen-The, 2002). In the long-term, i.e. 50 years after logging, no significant differences in mortality rates were observed between logged and primary forests in Malaysia (Yamada et al., 2013).

Contrary to our findings, other studies that analysed the impacts of logging intensity on vital rates directly (2-4 years) and longer (18 years) after logging in forests in Borneo found that growth rates of surviving trees, recruitment rates and mortality rates all increased with logging intensity, likely due to increased resource availability and more injured trees at higher logging intensities (Lussetti et al., 2016; Sist and Nguyen-The, 2002). We studied the impacts of logging on vital rates in forests that experienced the majority of logging nine years earlier for a two-year census interval. The effects of logging on vital rates can differ between studies, but it is evident that my study has exceptionally high mortality rates compared to other logged forests in the region (Do et al., 2018; Nam et al., 2018). High mortality rates at my site are likely due to the impacts of a heavy typhoon during the monitoring period and has probably changed forest dynamics in KNT, which is discussed in detail in Chapter 4.4.2. Further, it could be that nine years after logging any enhanced effects of logging on vital rates at our site have already declined and no significant differences can be detected anymore between logging classes. Additionally, differences in census intervals (Lewis et al., 2004) and the time lag between logging events and the study period complicate comparisons of vital rates among studies, which likely contributes to the differences in post-logging period in which vital rates are affected by logging.

4.4.2. Aboveground carbon dynamics

Plot-level AGC net changes in our study varied from -4.4 to 2.6 Mg C ha^{-1} yr^{-1}, with a mean of 0.1 Mg C ha^{-1} yr^{-1}. Averaged by plot, mean AGC dbh growth was equal to 1.4 Mg C ha^{-1} yr^{-1}, mean AGC recruitment equal to 0.2 Mg C ha^{-1} yr^{-1} and mean AGC mortality equal to 1.5 Mg

C ha^{-1} yr^{-1}. Nam et al. (2018) estimated mean net AGC net change rates at 3.0 Mg C ha^{-1} yr^{-1} for forests 22-30 years after logging in Vietnam, which is remarkably higher than the mean in my study and comparable to values of my plots that experienced the highest AGC increments. They found higher mean AGC dbh growth values (3.9 Mg C ha^{-1} yr^{-1}) and similar values for AGC recruitment (0.3 Mg C ha^{-1} yr^{-1}) and AGC mortality (1.2 Mg C ha^{-1} yr^{-1}). Further, considerable higher AGC net change rates were observed in logged forests elsewhere in Vietnam (5.4 Mg C ha^{-1} yr^{-1}; Do et al., 2018), Borneo (1.4 Mg C ha^{-1} yr^{-1}; Berry et al., 2010) and the Amazon (ranging from 0.04–2.96 Mg C ha^{-1} yr^{-1}, mean of 1.33 Mg C ha^{-1} yr^{-1}; Rutishauser et al., 2015). My study is the first work that reports mean AGC recovery rates close to zero in logged forests.

Generally, AGC net change rates in logged forests were higher than in unlogged forests (Berry et al., 2010; Roopsind et al., 2017). In our study, annual AGC net changes did not differ with logging intensity. Previous work showed that AGC stock losses increased with logging intensity immediately after logging, because more trees were killed by logging (Roopsind et al., 2018). After this initial period, AGC recovery rates increased with logging intensity (Gourlet-Fleury et al., 2013; Roopsind et al., 2018), likely because remaining trees benefit from reduced competition (Villegas et al., 2009). The gain in AGC decreased with time after logging, because competition increases as stands accumulate AGC (Gourlet-Fleury et al., 2013). However, at high logging intensities enhanced residual tree growth was insufficient to compensate for post-logging mortality (Roopsind et al., 2018). Recovery times depended mainly on the presence of remaining old growth trees and therefore the time to recover pre-logging AGC stocks was fastest in lightly logged forests (Roopsind et al., 2018). Consistently, Amazonia-wide results that show that the time needed to recover initial AGC stocks increases with AGC losses due to logging (Rutishauser et al., 2015).

It is likely that other disturbances at our sites interfere with any noticeable impacts of logging intensity on AGC net changes. In our study, plot-level AGC dynamics after logging were driven by mortality events, almost all due to natural causes. In 38% of the plots, high mortality and/or a large amount of surviving trees with reduced heights resulted in negative AGC net changes and these sites were thus not recovering. Although negative AGC net changes have been commonly observed in post-logging forests, due to high tree mortality in the first years after logging (Blanc et al., 2009; Mazzei et al., 2010; Sist et al., 2014), I expect that mortality events at my site are also impacted by heavy storms. KNT is located in Quang Binh province and experiences yearly storms from June to September (pers. comm. Trai Trong Le, 2019). In September 2017, i.e. during the census interval of my study, typhoon Doksuri severely impacted extensive areas in northern Vietnam, including the forests of KNT. From all typhoon landfalls in Vietnam between 1977 and 2017, typhoon Doksuri had the strongest typhoon intensity (Takagi, 2019). A study in the Philippines showed the plot-level tree mortality rate was significantly higher during a census interval with a strong typhoon (i.e. 2.27% yr^{-1}) compared to a non-typhoon interval and resulted in a small decrease in AGC stock after the typhoon (Yap et al., 2016). As a considerable amount of trees was uprooted or had broken tops during the re-measurements, I expect that besides post-logging mortality, the mortality in our site is largely caused by storm impacts and is higher than in years with less severe typhoons.

I did not observe a relationship between the AGC net change and the initial AGC of my plots. Contrary, standing AGC was positively related to net AGC increment, both at the individual and population level in logged Vietnamese forests (Nam et al., 2018). Similarly, AGC recovery rates were positively correlated with initial AGC stocks in logged Amazonian forests, i.e. forests with larger AGC stocks recovered faster (Rutishauser et al., 2015). The likely impact of storms on AGC net change rates could have disturbed such a relationship at my site.

4.4.3. Implications for forest recovery

The recovery to pre-logging AGC stocks after logging can take several decades or even more than a century (Blanc et al., 2009; Pinard and Cropper, 2000; Rutishauser et al., 2015). Improved logging practices (West et al., 2014) and silvicultural treatments (Peña-Claros et al., 2008) can accelerate forest recovery. Historical logging reduced AGC stocks in KNT (Chapter 2), but recent tree mortality, probably partly related to the impacts of a heavy typhoon, slows current forest recovery. Therefore, it is important to understand better the impacts of typhoons on tree damage and mortality in KNT and whether the historical logging intensity has changed the susceptibility of these forests to tree damages by typhoons. This will greatly improve predictions on the recovery capacity of these forests.

4.5. Conclusions

I quantified vital rates and AGC dynamics during two years in forests in Vietnam nine years after logging. I evaluated whether these vital rates and AGC fluxes differed with logging intensity. Although other studies usually observe increased dbh growth, recruitment and mortality rates with logging intensity, I did not find an effect of logging intensity on any of the vital rates. Plot-level AGC net changes varied from -4.4 to 2.6 Mg C ha^{-1} yr^{-1}, with a mean of 0.1 Mg C ha^{-1} yr^{-1}, and did not differ with logging intensity. AGC recovery was predominantly driven by mortality, which resulted in negative AGC net changes in 38% of the plots. The forests suffer from typhoon-induced tree damages and mortality, which currently retards AGC recovery in these forests. KNT is one of the last remaining lowland forests in Vietnam with significant conservation values. Logged forests have often a large recovery potential, but further degradation should be prevented. It remains important to continue monitoring whether these forests are recovering from historical logging events, ongoing illegal logging and impacts of typhoons.

Chapter 5. Synthesis and conclusions

Logging is the prevalent driver of forest degradation in Southeast Asia (Hosonuma et al., 2012). In Vietnam, extensive areas have been degraded by wood extraction and illegal logging in natural forests remains a serious concern (McElwee, 2004; Sikor and To, 2011). To prevent further degradation and allow recovery of logged forests, it is of critical importance to understand the impacts of logging on forest ecology in Vietnam. In this thesis, I assessed the impacts of logging intensity on carbon (C) stocks, tree diversity and forest recovery in Khe Nuoc Trong forest (KNT) in north-central Vietnam, nine years after the majority of logging had ceased.

5.1. Main findings

In Chapter 2, I presented an approach, combining Landsat analysis and participatory mapping, to estimate historical logging intensities. This method can be applied for sites that lack accurate records on logging history, which is the case for most forests in the tropics especially where illegal logging occurs. I quantified forest C stocks, i.e. carbon in aboveground carbon (AGC), dead wood, belowground carbon (BGC) and soil (SOC), in sites that vary in logging intensity. Next, I explored correlations between logging intensity, soil, topography and AGC stocks. Heavily logged forests stored only half the amount of AGC of stems ≥10 cm dbh as lightly logged forests, mainly due to a reduction in the number of trees ≥60 cm dbh. Carbon in AGC of small trees (5–10 cm dbh), dead wood and BGC stored small fractions of total C stocks, while SOC in the topsoil of 0–30 cm depth stored ~50% of total C stocks. Combining logging intensities with soil and topographic data showed that the variability in AGC was largely determined by the logging intensity. This chapter showed large reductions in AGC in medium and heavily logged forests, the critical role of big trees to maintain high forest C stocks and the importance to include SOC in total C stock estimates.

In Chapter 3, I analysed the impacts of logging intensity on tree diversity, composition, community wood density and availability of timber species in KNT. At my site, no significant relationships were observed between tree diversity and logging intensity and more plots in heavily logged forests would be needed to assess whether patterns follow predictions of the intermediate disturbance hypothesis. Some compositional differences were observed with logging intensity, but the challenge of sampling at the species level sufficiently in high diversity tropical forests complicates interpretation. Community wood density differed with logging intensity, shifting towards lighter wood density species in heavily logged forests. Timber species were not very abundant in our plots and especially the heavily logged forests showed reduced numbers. Most timber trees from harvestable size were present in the lightly logged plots. In this chapter, I showed the importance of reducing logging intensities to maintain high community wood density and to prevent the dominance of light-demanding pioneer species and the depletion of timber species after logging.

In Chapter 4, I assessed the impacts of logging intensity on vital rates, i.e. diameter growth, recruitment and mortality rates, and associated AGC dynamics over a two-year period. None of the vital rates and their equivalent AGC fluxes differed with logging intensity. Meanwhile, plot-level AGC net changes ranged from -4.4 to 2.6 Mg C ha^{-1} yr^{-1} and averaged 0.1 Mg C ha^{-1} yr^{-1}, and did not differ with logging intensity. Nearly 40% of plots showed a negative annual AGC net change during the monitoring period, predominantly driven by the natural mortality of trees and breakage of stems. KNT suffers from typhoon-induced tree damages and mortality, and this is likely to be retarding post-logging AGC recovery of these forests.

5.2. Connections between carbon, tree diversity and forest recovery in logged forests

This thesis shows that logging intensity has multiple impacts on forest ecology, including changes in forest carbon stocks, community wood density and the availability of timber species. Tree AGC is calculated with diameter and height data, to estimate the tree volume, and wood density of the species (Chave et al., 2014). Selective logging involves the direct extraction of predominantly big trees with high wood density (Macpherson et al., 2012), but it can also alter further forest stand dynamics. Although the height-diameter allometry in this study did not differ with logging intensity (Chapter 2.2.4.1), total and merchantable bole height (i.e. the lowest branch at crown base) of tropical canopy trees were reduced after selective logging in a forest in French Guiana (Rutishauser et al., 2016). Logging intensity affected tree heights negatively, which resulted in 2–13% decreases in tree AGC (Rutishauser et al., 2016). Logging thus had adverse effects on C sequestration and timber production.

Growth strategies of tree species are often reflected in the wood density of a species: fast-growing pioneer species usually invest in rapid growth, reflected in less dense wood, while slower-growing species invest in durable wood with high wood density (King et al., 2005; Poorter et al., 2010). Disturbance, including logging, can alter the species composition in forests and usually stimulates early successional, light-demanding and fast-growing species, while it reduces the richness of late successional and slow-growing species (Bongers et al., 2009; Carreño-Rocabado et al., 2012; Clark and Covey, 2012; Hoang et al., 2011). Logging-induced changes in species composition can thus lead to changes in community wood density (Verburg and Van Eijk-Bos, 2003). Indeed, in this study community wood density decreased with logging intensity (Chapter 3), which can result in a further decrease in AGC stocks in heavily logged forests. Amazonian studies in mature tropical forests showed that variation in tree species composition and wood density regulated AGC storage (Baker et al., 2004; Phillips et al., 2019). Species-related variation in wood density impacted AGC storage at all scales, i.e. from the individual tree up to the biome (Phillips et al., 2019). Stand-level wood density explained up to 45% of the total variation in AGC across Amazonia (Baker et al., 2004). In logged forests in Vietnam, species with high wood density contributed more to AGC and net AGC recovery rates than low wood density species (Nam et al., 2018). Additionally, net AGC recovery rates were positively related to AGC stocks in logged forests, meaning that forests with larger AGC stocks recovered faster, both in Vietnam (Nam et al., 2018) and across the Amazon Basin (Rutishauser et al., 2015). In KNT, such a trend was not detected, likely caused by sporadic tree mortality and retarded AGC recovery due to the impacts of a recent heavy typhoon (Chapter 4). Logging can thus alter all parameters used to estimate AGC, i.e. tree diameter distribution, height and wood density, resulting in changes in AGC stocks and AGC dynamics.

5.3. Recovery of logged forests

I found that recent AGC recovery after logging was retarded in KNT and did not differ with logging intensity (Chapter 4). Previous work found that recovery of pre-logging AGC stocks can take decades to over a century in logged Southeast Asian and Amazonian forests (Blanc et al., 2009; Pinard and Cropper, 2000; Rutishauser et al., 2015). The recovery time depended on logging intensity, with longer times required for forests logged with higher intensity (Roopsind et al., 2018, 2017; Rutishauser et al., 2015). In this study, nine years after logging, tree diversity did not differ significantly with logging intensity, while the species composition seemed to differ slightly with logging intensity (Chapter 3). The time needed to recover tree diversity in other studies differs: while the plant diversity of logged mixed Dipterocarp forest in Southern Vietnam was relatively well recovered after an average of 27 years (H.T.T. Do et al., 2019), a forest 50-years post-logging in Malaysia was still in secondary succession (Yamada et al., 2013). Timber species were only occurring in small numbers in our plots, especially in the heavily logged forests (Chapter 3). Timber stocks do not always recover at the same speed as AGC stocks. In forests in the Central African Republic, the AGC recovered very quickly after logging and thinning, at a rate proportional to the disturbance intensity (average recovery of 144% after 24 years), while timber stocks recovered much slower (mean recovery of 41% after 24 years) (Gourlet-Fleury et al., 2013). A pantropical meta-analysis showed that timber yields decreased with approximately 46% after the first logging cycle, but were subsequently sustained at that level (Putz et al., 2012). Timber stocks did not regain primary forest levels and logging cycles would need to be lengthened for further recovery of timber stocks (Putz et al., 2012). Sustainable cutting cycles in forests in Borneo that were logged with low, medium and high intensity significantly differed between groups and equalled 27, 41 and 89 years, respectively (Sist et al., 2003a). For KNT, it remains important to monitor C stocks, illegal logging events and impacts of typhoons, to assess whether KNT's forests are recovering from historical logging.

Logging gaps tend to be rapidly colonized by competing plants such as pioneer species and lianas, which suppress the establishment and growth of timber species in logged forests (Fredericksen and Mostacedo, 2000; Sist and Nguyen-The, 2002). Silvicultural practices can be implemented to stimulate the growth and recruitment of timber species, such as the liberation of valuable trees from lianas and neighbouring competing trees, soil scarification and stand refinements (Peña-Claros et al., 2008), but outcomes depended on the intensity of the silvicultural treatment (de Avila et al., 2017). Reducing logging intensities, elongating logging cycles and implementing silvicultural treatments can accelerate recovery of AGC stocks, species and timber yields after logging.

5.4. Implications for forest conservation

KNT is one of the last remaining extensive evergreen lowland forests in Vietnam with significant conservation value. Communities surrounding KNT derive an essential part, i.e. 28%, of their total household income from natural forests (Ngo et al., 2020). From this, roughly half was obtained through illegally extracted timber and the remaining half from the sale of other forest products, such as honey, mushrooms and hunted animals (Ngo et al., 2020). Heavily logged forests stored only half the amount of AGC as lightly logged forests (Chapter 2), highlighting the damage caused by unsustainable extraction, the potential in AGC storage if these forests are protected and enabled to recover and the vital contribution that these logged forests can make to Vietnam's efforts to reduce climate change. As 88% of the plots had stumps, indicating recent logging (Chapter 2), it is essential to reduce the amount of illegal logging. For this, it is crucial to work in consultation with communities on providing alternative livelihood strategies, to provide an income to communities and to reduce the pressure on KNT's natural forests. Viet Nature Conservation Centre, our partner organization, supports and empowers communities surrounding KNT, such as increasing income from alternative livelihood activities, providing school materials and environmental awareness raising. Research in this area showed that plantation forests contributed 15% of total household income for communities living around KNT (Ngo et al., 2020). Improved plantation management can likely increase income from existing plantations. Viet Nature Conservation Centre is currently supporting smallholders with *Acacia* plantations surrounding KNT to certify their wood according to the Forest Stewardship Council (FSC) standard, which provides evidence for sustainable forest management and enables access to certain timber markets.

Another possibility is to derive profits from conservation through reducing emissions from deforestation and forest degradation (REDD+) projects. Based on the results of this thesis and additional research, the project document for Climate, Community and Biodiversity Alliance (CCBA) certification is being prepared for KNT (CCBA, 2013). CCBA certification ensures that the project stores carbon, supports local communities and protects biodiversity. Money generated through this certification scheme can further support alternative livelihood incomes and enhanced conservation. An analysis for KNT suggests that activities to reduce forest degradation are likely financially viable through Vietnam's REDD+ program (Ngo et al., 2020). As KNT is located in Vietnam's REDD+ focus region, i.e. the north-central coast, and logging is a key driver of forest degradation in Vietnam, our project and analyses are excellent examples for REDD+ projects in this region aimed at avoiding forest degradation through reduced logging. To further enhance protection of KNT's valuable forests, a proposal is currently being reviewed to designate KNT as a nature reserve, the highest government protection status in Vietnam. Such a protection status would give access to a higher level of government funding for conservation.

Spending extensive periods in KNT also taught me that forest patrols supporting law enforcement are necessary to reduce the amount of illegal logging in the area. When we camped in KNT and rangers from checkpoints left the forest for a few days, we immediately heard the sound of chainsaws, stopping again when the rangers came back to their checkpoints. Full-time occupation of checkpoints on crucial timber extraction routes seems needed to reduce illegal timber extraction activities in KNT. An analysis on the political economy of illegal logging in Vietnam suggests that a singular focus on law enforcement likely neither benefits local livelihoods nor forest conservation in areas with smallholder extraction (Sikor and To, 2011). Law enforcement activities should therefore always be accompanied with the provision of alternative livelihood activities, which are developed together with forest-dependent communities.

In KNT, lightly logged forests contained the highest AGC (Chapter 2), while tree diversity did not differ significantly with logging intensity (Chapter 3). At the pantropical scale, no relationship was detected between tree diversity and AGC storage in intact old-growth tropical forests, but a weak relationship was observed within Asia (Sullivan et al., 2017). Absence of a clear diversity-carbon relationship means that conservation strategies solely focussing on C stocks will miss important high diversity areas (Sullivan et al., 2017). To avoid trade-offs, both carbon and biodiversity require explicit consideration in conservation planning.

5.5. Directions for future research

Several AGC estimates have been made in Vietnamese forests, mainly in old-growth forests (Con et al., 2013; T.V. Do et al., 2019; Hai et al., 2015; Luong et al., 2015; Nam et al., 2018; Stas et al., 2020). While extensive areas of Vietnam have been degraded by logging, there is limited knowledge on AGC and its variations in logged forests at the national-scale. A permanent plot network has been established under Vietnam's national forest inventory project during 2000-2015 and represents various forest types throughout Vietnam, with measurements conducted every five years (Nam, 2017). This network provides unique opportunities to refine AGC estimates and dynamics in Vietnamese forests. However, limitations with data sharing prevent current access to these data. Several pantropical AGC maps exist that combined field measurements with remote sensing (Avitabile et al., 2016; Baccini et al., 2012; Saatchi et al., 2011), but regional AGC estimates from ground plots and satellites appeared to differ significantly (Mitchard et al., 2014). That said, significant relationships were found between field-measured forest structure and high-spatial resolution satellite sensor data across a degradation landscape, including logged forests, in Malaysia, which could be used to map AGC and canopy structure (Pfeifer et al., 2016). An AGC map for Vietnam that is derived from detailed national field measurements does not yet exist. Such a national forest C map, constructed by combining field data and remote sensing, is highly valuable for REDD+ projects in Vietnam, especially for baseline measurements and large-scale monitoring of AGC stocks.

Observed forest structures and plant diversity are a result of past disturbances and subsequent forest recovery. Besides logging intensity, time since logging is an important parameter to interpret observed AGC stocks, tree diversity and forest dynamics. While chronosequence studies are common to study recovery in secondary forests (e.g., Do et al., 2010; Poorter et al., 2016), only a few studies used such an approach in logged tropical forests (de Carvalho et al., 2017; Lasco et al., 2006; Lussetti et al., 2016; Pinard et al., 2000). Studies considering variations in both logging intensity and time since logging, such as done in Borneo (Lussetti et al., 2016), would greatly improve our understanding of recovery in logged forests.

The geographical spread of logging studies is not uniformly across the tropics. The majority of the work on the impacts of logging on forest ecology has been performed in South America. Studies that analysed the responses of Southeast Asian forests to logging were predominantly performed in Dipterocarp forests in Borneo. As logging is the dominant driver of forest degradation in Southeast Asia (Hosonuma et al., 2012), also beyond Borneo and in other forest types, more research on this in mainland Southeast Asia and in non-Dipterocarp forests is recommended. Overall, studies on logging differ largely in logging intensity, recovery time since the last logging operation, number of times logged, time interval between logging cycles and applied harvest practices (e.g., reduced impact logging or conventional logging). Individual studies assessing the impacts of logging on forest C stocks and tree diversity can therefore differ significantly. As already a large fraction of the world’s tropical forests has been logged and the area of logged forests is still increasing in the tropics (Asner et al., 2009), it is highly important to understand broad-scale and long-term responses of tropical forests to logging. Pantropical networks that include permanent sample plots in logged forests, such as the Tropical managed Forests Observatory (TmFO; Sist et al., 2015) and ForestPlots.net (Lopez-Gonzalez et al., 2011), are crucial for such large-scale analyses and thereby predicting the future of logged tropical forests.

5.6. Final remarks

Selectively logged tropical forests can retain substantial C and timber stocks, biodiversity and other important ecosystem functions. This thesis contributes to the understanding of the effects of long-term logging activities on forest C stocks, tree diversity and forest recovery in seasonal evergreen broadleaf forests in Southeast Asia. These data are crucial for the development and implementation of REDD+ approaches, aimed at preventing further degradation and allowing eventual recovery of logged forests in Vietnam. A large part of the tropical forest biome has already been logged, yet logged forests have often a large recovery potential. Researchers, conservationists and policy makers should recognize the value of logged forests for conservation and their crucial role in mitigating global climate change, which will ultimately determine the fate of logged tropical forests.

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Appendices

Appendices Chapter 2

Appendix 2.A

The decay classes used in our study (Baker and Chao, 2011) and the reduction in wsg for each decay class relative to the wsg in living trees, as estimated by Chao et al. (2017) in tropical lowland forests in Taiwan (2.A1). For each plot, the wsg per decay class was estimated (2.A2) by multiplying the mean plot-level wsg of living trees ≥10 cm dbh with the proportional reduction in wsg according to Chao et al. (2017).

1)

Decay class	Description	Relative reduction wsg (%)
1	Solid wood, recently fallen, with intact bark and fine branches (<10 cm) still attached	32
1.5	Solid wood, but with no fine branches, and bark starting to fall off	42
2	Non-solid wood, in poorer condition, but where it is still difficult to push a nail into the wood by hand	51
2.5	Soft, rotten wood, where a nail can be pushed into the wood easily	54
3	Soft, rotten wood, which collapses easily when stepped on	61

2)

Plot	Mean plot-level wsg living trees ≥10 cm dbh (g cm^{-3})	wsg class 1 (g cm^{-3})	wsg class 1.5 (g cm^{-3})	wsg class 2 (g cm^{-3})	wsg class 2.5 (g cm^{-3})	wsg class 3 (g cm^{-3})
1	0.558	0.378	0.322	0.274	0.255	0.218
2	0.583	0.395	0.336	0.286	0.267	0.227
3	0.509	0.345	0.293	0.250	0.233	0.198
4	0.561	0.381	0.323	0.276	0.257	0.219
5	0.505	0.342	0.291	0.248	0.231	0.197
6	0.537	0.364	0.310	0.264	0.246	0.209
7	0.525	0.356	0.303	0.258	0.240	0.205
8	0.533	0.361	0.307	0.262	0.244	0.208
9	0.508	0.344	0.293	0.250	0.232	0.198
10	0.565	0.383	0.326	0.278	0.259	0.220
11	0.532	0.361	0.307	0.262	0.244	0.207
12	0.597	0.404	0.344	0.293	0.273	0.233
13	0.588	0.399	0.339	0.289	0.269	0.229
14	0.534	0.362	0.308	0.262	0.244	0.208
15	0.545	0.369	0.314	0.268	0.249	0.212
16	0.580	0.393	0.334	0.285	0.265	0.226
17	0.570	0.386	0.328	0.280	0.261	0.222
18	0.574	0.389	0.331	0.282	0.263	0.224
19	0.571	0.387	0.329	0.281	0.261	0.223
20	0.604	0.410	0.348	0.297	0.277	0.236
21	0.526	0.356	0.303	0.258	0.241	0.205
22	0.571	0.387	0.329	0.281	0.262	0.223
23	0.549	0.372	0.317	0.270	0.251	0.214
24	0.537	0.364	0.310	0.264	0.246	0.209

Appendix 2.B

Methods of the soil laboratory analyses

All analyses were performed in the soil laboratory of the Department of Soil Science, Faculty of Agronomy at Hue University of Agriculture and Forestry in Hue, Vietnam. For the chemical analyses, samples were prepared by air-drying and removal of coarse materials such as roots and stones. Subsequently, the samples were crushed and sieved at <2 mm to retrieve the fine earth fraction. This sample material was stored in sealed plastic bags to prevent moisture uptake. To improve homogeneity and digestion of the samples for the “total analysis” (total organic C and total N, P, K), fine earth material was downsized to <0.25 mm. Results are expressed on oven-dry basis.

Moisture content

Moisture content of fine earth samples was determined by overnight heating at 100-105 °C (Dewis and Freitas, 1970).

pH

Both pH H_2O and pH KCl were determined in a 1:5 (w/v) extract of H_2O and KCl 1 M (Dewis and Freitas, 1970), respectively, after shaking for 10 minutes.

Total Organic Carbon (C)

The Walkley and Black (1934) method was followed, compensating for incomplete oxidation (75%) for conversion to total organic C (see also Page et al., 1982, p. 539).*Total Nitrogen (N)*

Total nitrogen was determined using Kjeldahl method. Organic matter was digested without catalyst in concentrated H_2SO_4 heated at ca. 300 °C. Digestion was completed by dropwise additions of $HClO_4$ (Batey et al., 1974). From this digest, N was quantified by liberating NH_3 by steam distillation, trapping into H_3BO_3 and subsequent titration.

Total Phosphorus (P)

Sample material was digested with $HClO_4$ (60%) and HNO_3 (Sommers and Nelson, 1972). P was determined by the 'Molybdenum blue method' (Murphy and Riley, 1962), using a mixture of $SnCl_2$ and $C_6H_8O_6$ as reductant and measuring absorbance at 882 nm.

Available Phosphorus (P)

Available P was extracted according to the Bray ll method, using a mixture of NH_4F (0.03 M) and HCl (0.1 M) (Bray and Kurtz, 1945). P was quantified by the 'Molybdenum blue method' (Murphy and Riley, 1962) with a mixture of $SnCl_2$ and $C_6H_8O_6$ for colour development. Absorbance was measured at 720 nm.

Total Potassium (K)

Sample material was digested with boiling concentrated H_2SO_4. Digestion was completed with repeated treatments of $HClO_4$ (70%) until the residue turned white. The digest was diluted and the residual mineral matter was sedimented. K was determined in the supernatant by flame photometer and expressed as K_2O.

Exchangeable bases and Cation Exchange Capacity (CEC)

Exchangeable bases and CEC were determined in the same procedure. The exchangeable bases (K^+, Na^+, Ca^{2+}, Mg^{2+}) were leached from the sample by subsequent additions of NH_4OAc 1M at pH 7 (Page et al., 1982; p. 160). The exchangeable bases K^+, Na^+ and Ca^{2+} were determined by flamephotometer and Mg^{2+} by atomic absorption spectrometer. Excess NH_4^+ was washed with C_2H_5OH (96%). Absorbed NH_4^+ was displaced by leaching with unbuffered KCl 1 M. NH_4^+ in the leachate was determined by micro steam distillation and used for the calculation of the CEC (Dewis and Freitas, 1970).

Exchangeable hydrogen and aluminium (exchangeable acidity) (H^+ and Al^{3+})

The sample was leached with stepwise additions of 1 M KCl solution. The exchanged acidity was determined by titration with NaOH. Subsequently, NaF was added to the leachate and Al^{3+}

determined by titration with HCl. Exchangeable H^+ was calculated by subtraction of exchangeable Al^{3+} from exchangeable acidity (Dewis and Freitas, 1970).

Particle size distribution

Particles were measured in the following size fractions: sand: 0.05-2 mm, silt: 0.002-0.05 mm and clay: <0.002 mm. The fine earth sample was treated with H_2O_2 to remove organic matter. The remaining primary particles were dispersed with $(NaPO_3)_6$ and Na_2CO_3 ("Calgon type solution") and sieved over a 0.050 mm sieve to retrieve the sand fraction. This fraction was dried at 100-105 °C and weighed. With sieving, the clay and silt fractions were washed into a sedimentation cylinder. The clay fraction was estimated by pipetting an aliquot at a depth calculated by Stokes law (van Reeuwijk, 2002). The percentage silt was calculated by subtracting the weight of sand and clay fractions from the initial sample weight.

Bulk density

In the field, the bulk density samples were taken out of the cores and stored in plastic bags. In the laboratory, the sample material was dried at 105 °C to constant weight. The bulk density was calculated by dividing the oven dry weight by the core volume (Dewis and Freitas, 1970).

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Appendix 2.C

Parameters describing disturbances, site information and topography (2.C1) and forest structure (2.C2) per plot, ordered by the logging intensity derived from the Landsat analysis. Total C includes AGC ≥5 cm dbh, C in dead wood, BGC and SOC 0-30 cm depth (2.C2).

1)

	Disturbances				Site & topography		
	Landsat analysis		Participatory mapping	Field			
Plot	Logging intensity	# disturbances (km^{-1})	Logging intensity	B.A. logged stumps (m^2ha^{-1})	Site	Elevation (m)	Slope (°)
4	Light	14	Light	0.0	2	379	26
6	Light	24	Medium	3.4	3	241	24
12	Light	3	Light	0.1	6	440	19
13	Light	4	Light	3.3	6	428	19
17	Light	6	Light	0.0	6	381	25
18	Light	6	Light	0.8	6	410	30
21	Light	10	Light	0.0	2	358	21
22	Light	21	Light	0.7	2	319	22
1	Medium	73	Heavy	4.3	1	146	12
2	Medium	29	Medium	8.1	1	186	21
3	Medium	53	Light	0.3	2	644	23
10	Medium	78	Heavy	3.3	5	114	19
15	Medium	39	Heavy	1.8	7	185	12
16	Medium	67	Heavy	5.3	7	210	18
19	Medium	71	Medium	5.1	8	277	16
24	Medium	95	Heavy	0.7	9	105	10
5	Heavy	161	Medium	0.8	3	236	24
7	Heavy	134	Medium	0.3	4	618	12
8	Heavy	194	Medium	0.6	4	547	18
9	Heavy	186	Heavy	1.1	5	83	7
11	Heavy	131	Heavy	3.0	5	112	21
14	Heavy	119	Heavy	2.5	7	138	13
20	Heavy	117	Medium	3.7	8	204	16
23	Heavy	116	Heavy	4.1	9	93	13

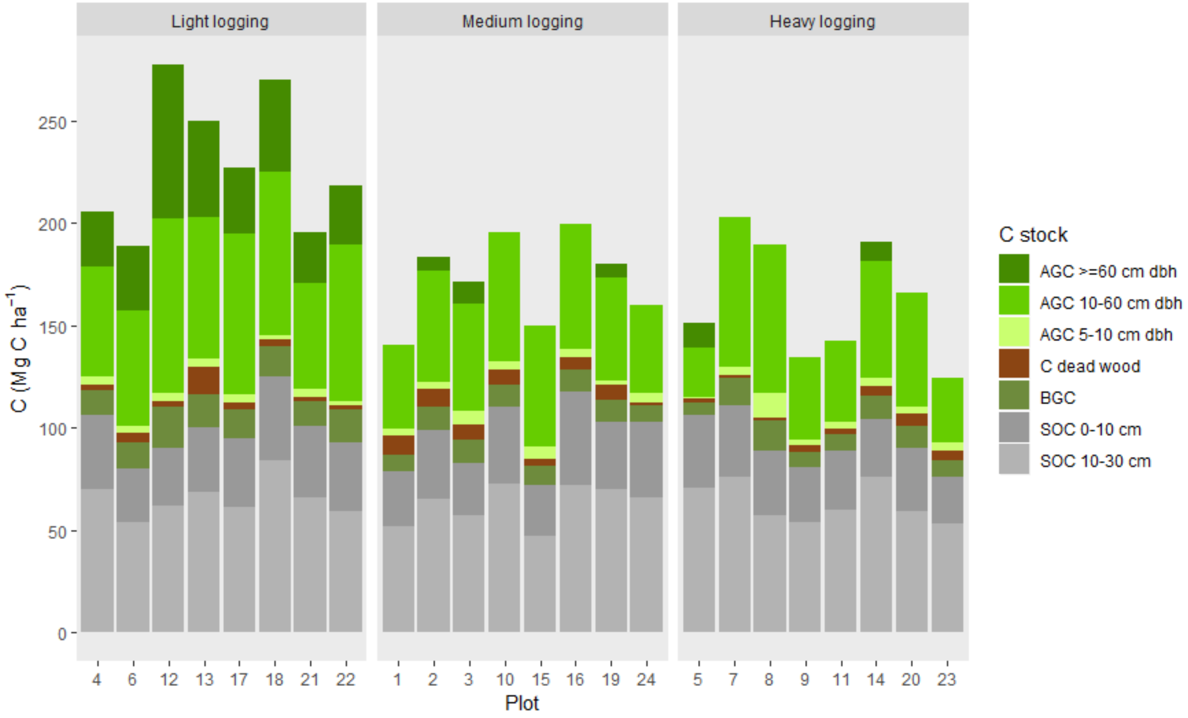
2)

		Living stems 5-10 cm dbh		Living stems ≥10 cm dbh							
Plot	Max. dbh (cm)	# stems (ha^{-1})	AGC (Mg C ha^{-1})	# stems (ha^{-1})	B.A. (m^{2}ha^{-1})	AGC (Mg C ha^{-1})	C dead wood (Mg C ha^{-1})	BGC (Mg C ha^{-1})	SOC 0-10 cm (Mg C ha^{-1})	SOC 10-30 cm (Mg C ha^{-1})	Total C (Mg C ha^{-1})
4	101.0	900	4	364	26	80	2	13	36	70	206
6	107.9	800	4	516	28	88	4	13	26	54	189
12	106.3	750	4	612	41	160	3	20	28	62	277
13	87.9	600	4	552	33	116	13	16	31	69	250
17	78.1	300	4	480	29	111	3	14	34	61	227
18	103.1	450	2	484	30	125	3	15	41	84	270
21	94.0	800	4	484	26	77	2	12	35	66	195
22	78.4	300	2	480	32	105	2	16	34	59	218
1	59.0	750	4	520	15	41	9	8	27	52	141
2	70.5	800	4	604	21	61	8	11	34	65	183
3	83.5	1400	7	644	26	63	8	11	26	57	171
10	46.5	750	4	524	21	63	8	11	37	73	196
15	50.9	1000	6	596	20	59	3	10	25	47	150
16	58.0	500	4	548	20	62	6	11	46	72	200
19	65.6	200	2	532	21	57	8	11	33	70	180
24	41.2	1150	5	548	17	42	1	8	37	66	160
5	72.8	250	1	360	14	36	2	6	35	71	151
7	53.6	700	4	948	29	73	1	14	35	76	203
8	53.4	1600	12	1116	29	73	2	14	32	57	189
9	48.0	500	3	616	17	40	3	7	27	54	134
11	52.2	750	4	500	16	40	3	8	29	60	143
14	69.4	750	4	608	24	66	5	11	28	76	191
20	45.5	450	3	644	19	56	7	11	31	59	166
23	35.0	800	4	656	16	31	5	8	23	53	124

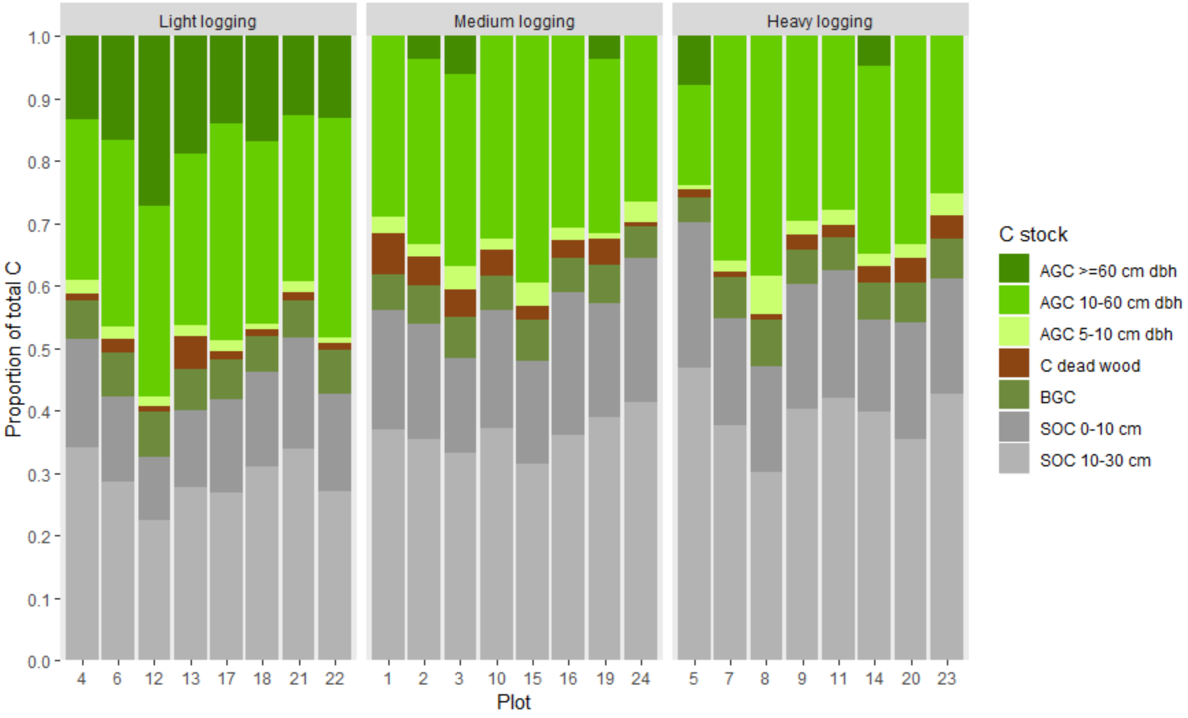
Appendix 2.D

The absolute (2.D1) and proportional (2.D2) total C stock for each plot, divided into the various C pools.

1)

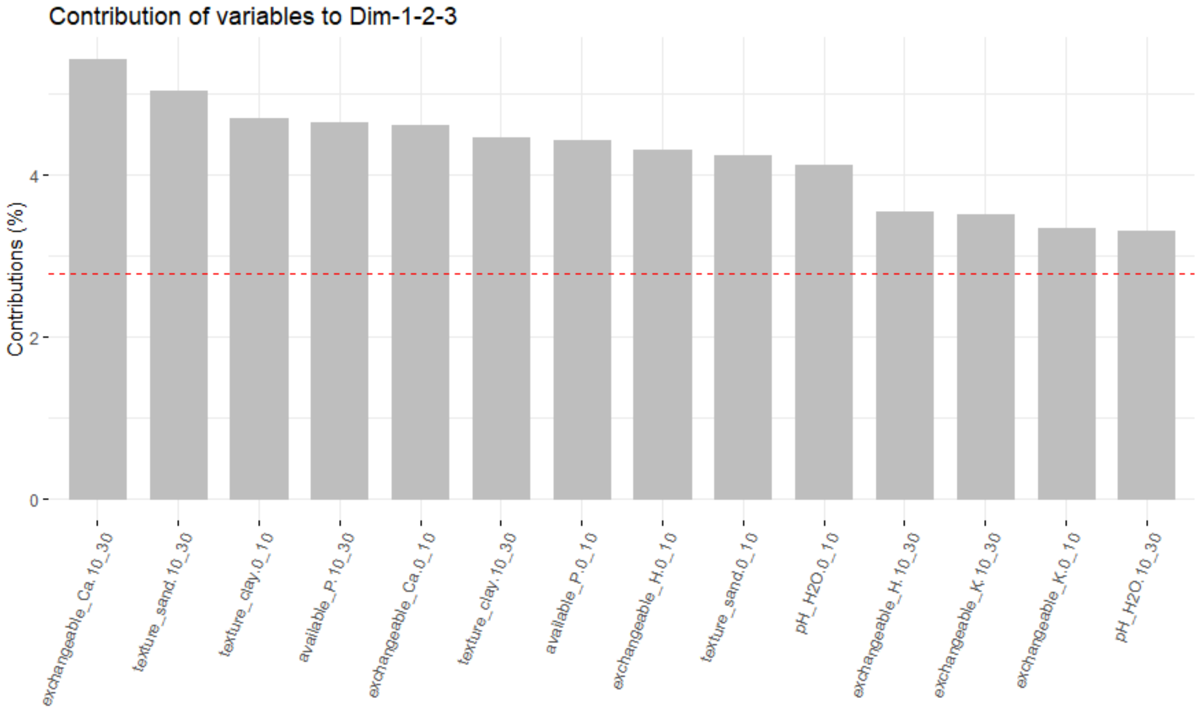


2)



Appendix 2.E

The contributions of the soil parameters that explain most of the variances in the soil dataset, combining both the 0-10 cm and 10-30 cm depth layer. Nineteen percent of the variation is explained by the first eigenvalue, 11% by the second eigenvalue and 11% by the third eigenvalue. Forty-one percent of the variation is explained by the first three eigenvalues together.



Appendix 2.F

Soil parameters for each plot.

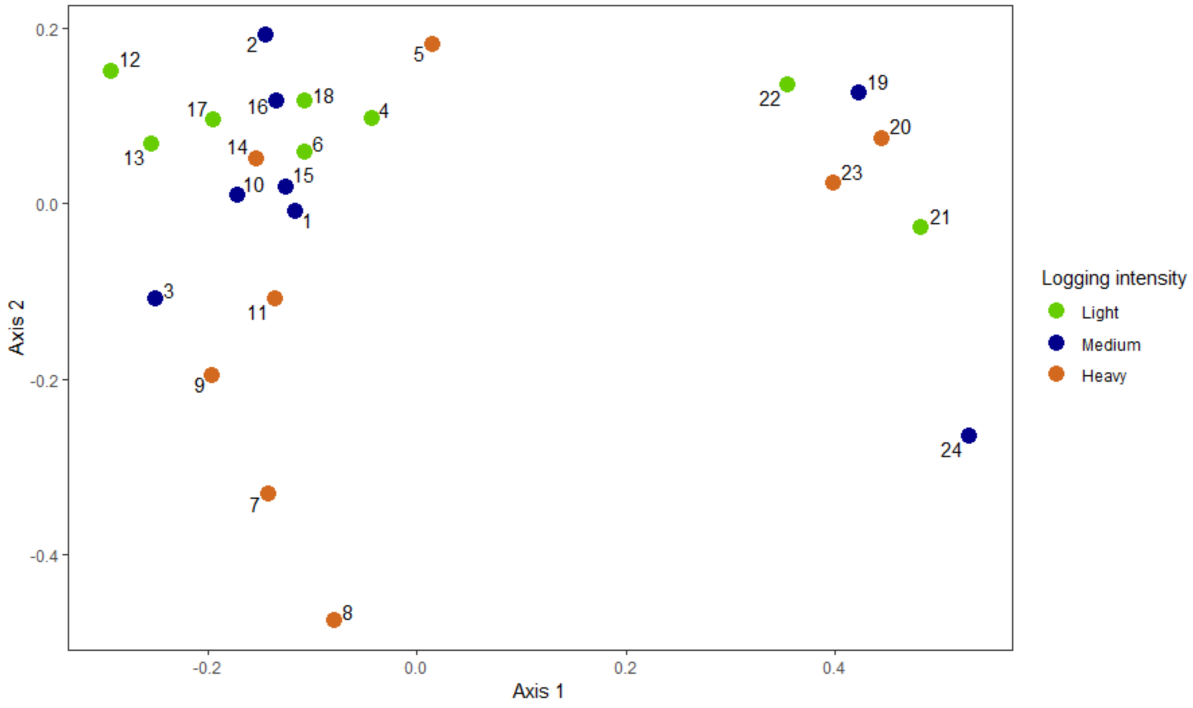
Parameter	Depth	Plot																							
	(cm)	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24
pH H_2O	0-10	4.69	5.24	4.79	5.29	5.08	5.46	4.98	4.45	5.20	4.78	5.36	4.80	5.05	4.45	5.35	4.39	5.01	5.05	5.75	4.54	6.29	4.81	4.86	5.54
	10-30	5.23	5.42	5.10	5.44	5.65	5.27	5.13	4.35	5.37	5.08	5.06	4.92	5.23	4.76	5.55	4.56	5.12	5.10	5.82	4.72	5.89	5.10	5.18	5.67
pH KCl	0-10	3.45	4.51	3.81	3.87	3.97	4.03	4.33	3.84	4.01	3.67	4.22	3.76	3.86	3.67	4.02	3.54	3.81	3.86	3.69	4.51	3.78	3.97	4.10	4.20
	10-30	3.83	4.65	4.03	3.90	4.21	4.20	4.23	3.89	4.23	3.84	4.35	3.89	4.02	3.79	4.31	3.61	4.02	4.05	3.99	4.63	4.04	4.08	4.09	4.23
SOC	0-10	2.34	3.09	2.37	3.35	2.97	2.53	2.63	2.67	2.29	3.26	2.55	2.57	2.53	2.45	2.19	3.45	2.65	3.11	3.12	2.67	3.02	3.17	2.42	3.12
(%)	10-30	2.18	2.67	2.16	2.76	2.70	2.16	3.04	2.32	2.09	3.03	2.53	2.27	2.34	3.21	1.89	2.90	2.18	2.92	3.02	2.21	2.35	2.65	2.28	3.09
Total N	0-10	0.15	0.22	0.22	0.22	0.15	0.17	0.24	0.18	0.14	0.16	0.10	0.14	0.14	0.11	0.17	0.12	0.19	0.19	0.13	0.14	0.18	0.19	0.16	0.21
(%)	10-30	0.14	0.13	0.15	0.19	0.14	0.13	0.17	0.15	0.12	0.12	0.07	0.09	0.09	0.10	0.13	0.09	0.15	0.16	0.11	0.07	0.12	0.13	0.14	0.12
Total P	0-10	0.09	0.12	0.09	0.12	0.16	0.14	0.13	0.11	0.13	0.11	0.10	0.11	0.11	0.14	0.10	0.11	0.13	0.13	0.13	0.08	0.14	0.08	0.13	0.10
(%)	10-30	0.07	0.09	0.09	0.11	0.12	0.10	0.10	0.08	0.11	0.09	0.07	0.08	0.09	0.11	0.09	0.10	0.10	0.09	0.07	0.08	0.09	0.07	0.08	0.08
Available P	0-10	9.00	9.50	12.50	14.00	15.50	14.50	10.50	11.50	10.50	12.00	12.00	16.50	16.00	12.50	16.00	16.00	14.50	20.50	12.50	16.50	16.00	15.00	20.50	16.50
(mg/100 gram soil)	10-30	9.00	9.50	11.50	14.00	14.50	14.00	9.00	10.50	10.00	12.00	11.00	12.50	14.50	10.50	12.50	15.00	12.50	20.50	12.50	12.50	14.25	14.50	16.50	12.50
Total K_2O	0-10	0.45	0.58	0.60	0.75	0.60	0.62	0.70	0.65	0.58	0.55	0.63	0.70	0.60	0.72	0.63	0.58	0.58	0.60	0.55	0.67	0.75	0.58	0.63	0.53
(%)	10-30	0.40	0.53	0.55	0.67	0.57	0.58	0.65	0.62	0.50	0.55	0.59	0.60	0.51	0.65	0.60	0.55	0.55	0.55	0.55	0.63	0.65	0.52	0.58	0.49
Exchangeable K^+	0-10	0.05	0.05	0.05	0.05	0.06	0.05	0.06	0.05	0.07	0.05	0.06	0.06	0.06	0.06	0.08	0.06	0.08	0.06	0.05	0.07	0.05	0.06	0.07	0.08
(cmol$_c$/kg)	10-30	0.03	0.05	0.04	0.03	0.05	0.05	0.05	0.05	0.04	0.04	0.05	0.05	0.05	0.05	0.06	0.05	0.06	0.05	0.05	0.05	0.05	0.04	0.05	0.06
Exchangeable Na^+	0-10	0.03	0.04	0.04	0.03	0.05	0.03	0.04	0.03	0.04	0.03	0.05	0.05	0.03	0.03	0.04	0.03	0.04	0.03	0.03	0.03	0.03	0.03	0.02	0.03
(cmol$_c$/kg soil)	10-30	0.03	0.02	0.03	0.03	0.03	0.03	0.03	0.03	0.02	0.03	0.03	0.04	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.02	0.03	0.02	0.03	0.03
Exchangeable Ca^{2+}	0-10	1.50	2.50	3.12	3.20	2.75	2.80	3.10	2.10	2.50	3.10	2.50	3.08	2.80	2.50	1.80	2.30	2.84	3.40	2.80	2.49	3.78	2.80	3.60	3.20
(cmol$_c$/kg soil)	10-30	1.30	1.89	2.35	3.11	2.58	2.83	2.20	1.90	2.30	2.67	2.30	2.87	2.72	2.20	1.54	1.92	2.75	3.10	2.50	1.90	3.50	2.70	3.30	2.50
Exchangeable Mg^{2+}	0-10	0.34	0.53	0.50	0.37	0.60	1.25	0.65	0.43	0.64	0.44	0.60	0.55	0.80	0.75	0.55	1.20	1.12	0.55	0.65	0.75	0.69	0.43	1.15	1.10
(cmol$_c$/kg soil)	10-30	0.45	0.41	0.71	0.32	0.40	0.92	0.60	0.55	0.55	0.38	0.50	0.43	0.72	0.58	0.45	0.90	0.81	0.50	0.60	0.70	0.62	0.35	0.90	0.90
CEC	0-10	10.55	9.50	10.78	10.10	10.21	11.21	10.50	7.50	11.50	9.50	10.00	10.13	11.76	7.50	8.00	12.50	10.23	12.32	12.00	9.00	13.42	9.80	11.01	13.00
(cmol$_c$/kg)	10-30	11.00	8.00	12.06	8.21	8.12	8.45	9.00	7.00	10.00	9.00	9.50	8.21	9.21	6.50	9.50	10.00	8.21	11.78	10.50	7.50	8.21	8.21	9.12	10.50

Exchangeable H^+	0-10	0.95	1.00	0.75	1.10	1.40	1.05	0.75	1.03	1.20	0.60	1.34	1.15	1.22	0.93	1.25	1.05	0.90	0.78	1.15	0.95	1.15	0.70	1.35	1.00
(cmol$_c$/kg)	10-30	1.20	1.06	0.80	1.10	1.50	1.00	0.80	1.05	1.25	0.68	1.40	1.25	1.28	0.95	1.30	1.15	1.00	1.15	1.25	1.09	1.23	0.84	1.38	1.05
Exchangeable Al^{3+}	0-10	0.46	0.70	0.65	0.70	0.63	0.55	1.41	1.48	0.87	1.55	0.62	1.44	0.49	0.17	0.45	1.20	0.35	1.48	0.55	1.05	0.05	1.48	1.30	0.13
(cmol$_c$/kg)	10-30	0.39	0.46	0.80	1.03	0.67	0.95	0.30	1.61	1.08	0.93	0.85	1.35	0.65	0.39	0.60	1.25	0.75	1.25	0.76	0.91	0.00	1.36	1.48	0.20
Texture - % sand	0-10	33.48	43.76	37.89	43.34	45.76	35.89	36.97	44.24	38.28	34.78	43.76	39.21	46.32	44.56	42.65	41.68	46.87	48.21	44.98	36.87	42.45	49.34	46.23	50.34
Texture - % silt	0-10	20.98	23.21	22.17	25.15	20.65	30.21	21.26	19.21	19.78	20.13	22.87	22.87	22.32	22.67	21.65	22.65	23.12	18.65	24.54	22.67	25.87	23.13	24.43	17.32
Texture - % clay	0-10	45.54	33.03	39.94	31.51	33.59	33.90	41.77	36.55	41.94	45.09	33.37	37.92	31.36	32.77	35.70	35.67	30.01	33.14	30.48	40.46	31.68	27.53	29.34	32.34
Texture - % sand	10-30	31.76	41.23	30.19	35.45	37.12	32.34	33.89	38.76	35.87	29.95	39.45	36.78	36.21	45.54	40.21	44.89	38.21	45.23	43.89	38.21	38.89	45.34	44.78	44.78
Texture - % silt	10-30	23.47	21.21	20.21	21.34	27.08	24.21	20.12	22.13	18.67	21.56	23.21	20.18	23.45	21.65	19.78	18.78	20.23	20.23	22.12	18.94	27.34	20.78	23.43	18.49
Texture - % clay	10-30	44.77	37.56	49.60	43.21	35.80	43.45	45.99	39.11	45.46	48.49	37.34	43.04	40.34	32.81	40.01	36.33	41.56	34.54	33.99	42.85	33.77	33.88	31.79	36.73
Bulk density	0-10	1.17	1.11	1.09	1.07	1.18	1.03	1.33	1.19	1.17	1.13	1.12	1.08	1.23	1.15	1.16	1.32	1.29	1.31	1.07	1.18	1.16	1.09	0.95	1.17
(g/cm^3)	10-30	1.18	1.21	1.33	1.26	1.31	1.25	1.25	1.22	1.30	1.21	1.19	1.36	1.47	1.19	1.23	1.24	1.40	1.44	1.17	1.32	1.40	1.12	1.16	1.07

Appendices Chapter 3

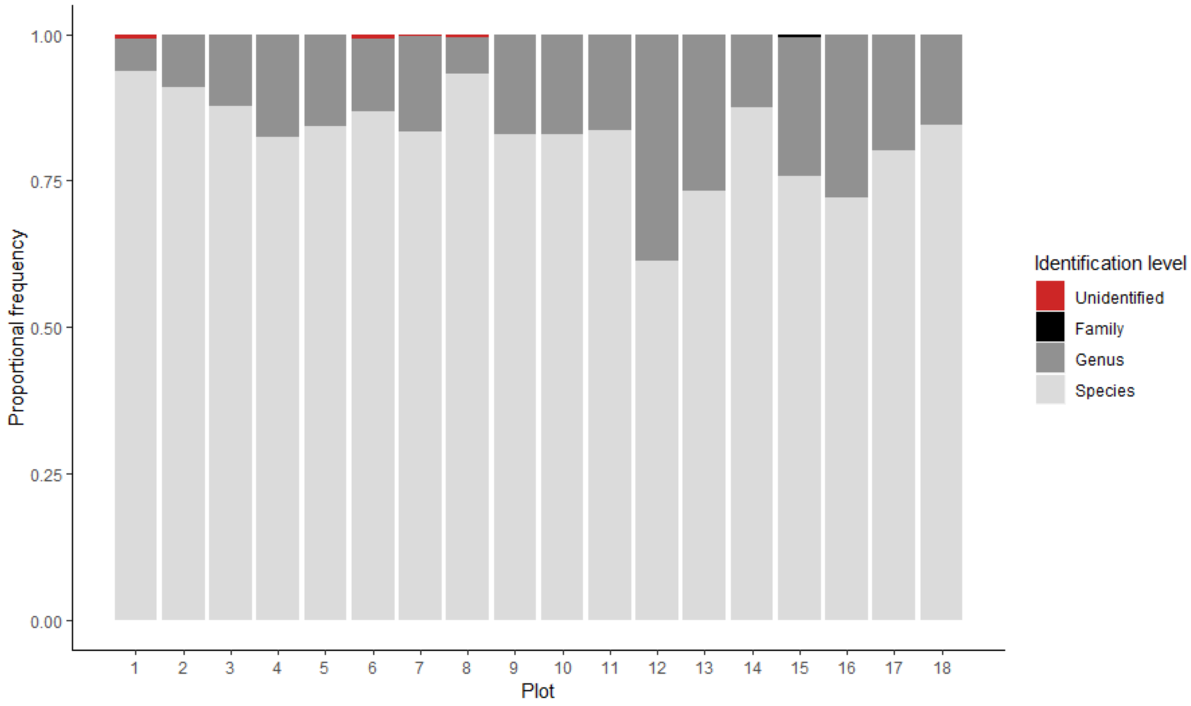
Appendix 3.A

NMDS plot showing the dissimilarity in tree species composition of stems ≥5 cm dbh between the 24 plots. Each dot represents a sample plot, marked with the corresponding plot number, and has been coloured by logging class.



Appendix 3.B

Proportional frequencies of the levels of tree identifications for stems ≥5 cm dbh for each plot.



Appendix 3.C

Diversity metrics for stems ≥5 cm dbh for each plot. Metrics are standardized by the lowest number of stems in the plots and permutated 1000 times to derive a mean value. Data have been ordered by logging intensity.

Plot	Logging intensity	Shannon	Simpson	Fisher's α	Species richness	Genus richness	Family richness	Evenness
4	Light	3.27	0.95	17.53	33	27	20	0.94
6	Light	3.22	0.95	19.06	34	26	20	0.91
12	Light	2.63	0.91	7.56	20	18	15	0.88
13	Light	3.24	0.95	17.44	32	27	21	0.93
17	Light	3.11	0.94	14.57	29	24	18	0.92
18	Light	2.98	0.93	13.84	28	26	19	0.89
1	Medium	3.27	0.95	20.00	35	29	21	0.92
2	Medium	3.05	0.94	14.82	30	26	20	0.90
3	Medium	3.10	0.94	17.11	32	26	18	0.90
10	Medium	3.16	0.94	17.53	32	27	20	0.91
15	Medium	2.81	0.92	11.74	26	23	19	0.87
16	Medium	3.01	0.93	14.85	30	26	19	0.89
5	Heavy	2.30	0.80	8.76	22	20	17	0.75
7	Heavy	2.91	0.92	14.24	29	25	17	0.86
8	Heavy	1.94	0.69	8.49	21	19	15	0.64
9	Heavy	3.18	0.94	18.81	34	26	21	0.90
11	Heavy	3.40	0.96	24.41	39	31	22	0.93
14	Heavy	3.27	0.95	19.37	34	30	23	0.93

Appendices Chapter 4

Appendix 4.A

Vital rates of stems ≥10 cm dbh per plot. Mean dbh growth rates, total recruitment rates and total mortality rates are shown. Data have been ordered by logging intensity.

Plot	Logging intensity	Dbh growth rate (cm yr^{-1})	Recruitment rate (% yr^{-1})	Mortality rate (% yr^{-1})
4	Light	0.2	1.1	1.6
6	Light	0.4	2.7	1.6
12	Light	0.2	0.3	0.7
13	Light	0.3	1.4	4.0
17	Light	0.3	0.9	2.6
18	Light	0.4	3.3	5.0
21	Light	0.4	1.2	2.9
22	Light	0.3	0.4	2.1
1	Medium	0.3	0.8	1.9
2	Medium	0.2	1.4	2.4
3	Medium	0.3	1.3	2.8
10	Medium	0.4	1.1	2.7
15	Medium	0.3	1.7	2.3
16	Medium	0.3	0.4	5.1
19	Medium	0.5	3.0	0.4
24	Medium	0.4	2.2	1.8
5	Heavy	0.5	3.9	2.8
7	Heavy	0.2	0.2	1.7
8	Heavy	0.1	1.4	0.4
9	Heavy	0.3	1.0	2.3
11	Heavy	0.3	1.6	0.8
14	Heavy	0.3	1.0	0.7
20	Heavy	0.4	1.9	1.9
23	Heavy	0.3	1.5	5.2

Appendix 4.B

Number of living stems, AGC stocks and AGC fluxes of stems ≥5 cm dbh per plot. Positive fluxes indicate carbon accumulations, negative fluxes indicate carbon emissions. Data have been ordered by logging intensity.

Plot	Logging intensity	# stems (ha^{-1})		AGC stocks ($Mg\ C\ ha^{-1}$)		AGC fluxes ($Mg\ C\ ha^{-1}\ yr^{-1}$)			
		Census 1	Census 2	Census 1	Census 2	AGC net change	AGC dbh growth	AGC recruitment	AGC mortality
4	Light	1264	1214	92	93	0.4	0.7	0.1	-0.5
6	Light	1316	1378	95	98	1.1	2.3	0.3	-1.4
12	Light	1362	1258	158	158	0.3	1.0	0.0	-0.8
13	Light	1152	932	122	113	-4.4	-1.7	0.1	-2.8
17	Light	764	652	101	99	-0.6	2.1	0.1	-2.8
18	Light	934	918	113	105	-3.7	2.0	0.6	-6.4
21	Light	1284	1172	82	84	1.0	1.9	0.2	-1.1
22	Light	780	664	106	109	1.7	2.6	0.0	-0.9
1	Medium	1270	1358	45	41	-2.3	-2.3	0.4	-0.4
2	Medium	1388	1472	63	62	-0.8	1.1	0.2	-2.0
3	Medium	2090	1978	73	73	0.2	1.6	0.2	-1.7
10	Medium	1274	1258	66	68	1.0	2.3	0.1	-1.4
15	Medium	1596	1446	61	61	0.1	0.8	0.2	-0.9
16	Medium	1048	1000	63	60	-1.0	1.3	0.2	-2.5
19	Medium	732	814	62	67	2.6	2.7	0.4	-0.5
24	Medium	1694	1498	45	49	1.8	2.5	0.2	-0.9
5	Heavy	610	626	41	39	-1.3	0.6	0.3	-2.2
7	Heavy	1648	1670	75	79	1.8	2.2	0.1	-0.5
8	Heavy	2716	2748	88	90	1.3	1.1	0.4	-0.3
9	Heavy	1254	1100	43	44	0.5	0.9	0.1	-0.5
11	Heavy	1250	1112	44	45	0.9	1.4	0.1	-0.6
14	Heavy	1358	1362	73	78	2.4	2.3	0.1	0.0
20	Heavy	1094	1048	59	59	-0.2	2.3	0.3	-2.8
23	Heavy	1456	1362	36	34	-0.5	1.1	0.2	-1.8