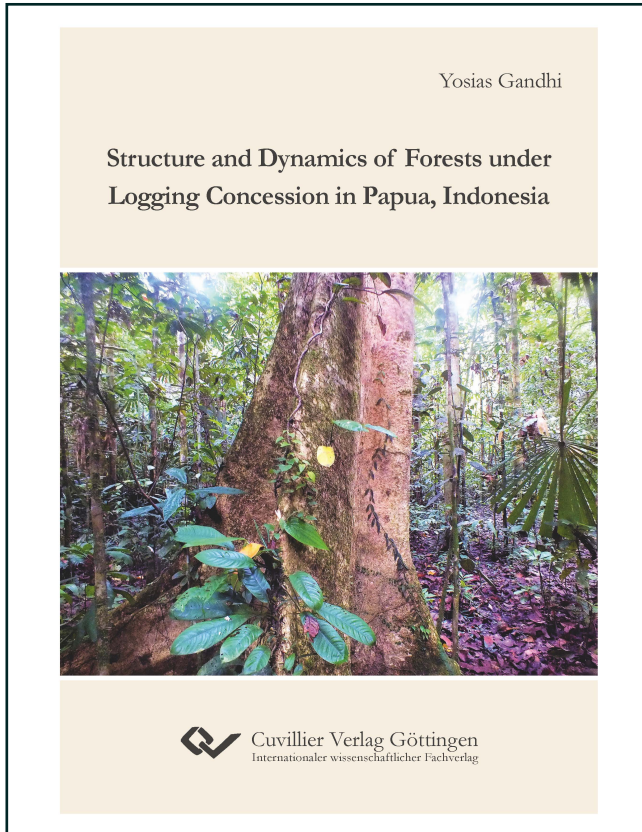




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**Structure and Dynamics of Forests under Logging
Concession in Papua, Indonesia**



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1. Introduction

1.1. The threat to tropical forests of insular South East Asia

Southeast (SE) Asia and the Pacific Archipelago (New Guinea and the Solomon Islands) harbour about 15% of the world's tropical forests (FAO, 1995). The region varies in forest type, including mixed deciduous forests, highly productive dipterocarp forests, and tropical moist forests (Richards, 1996; Thaman *et al.*, 2006). This tropical region is also home to carbon-rich ecosystems such as mangrove and peat swamp forests that are still located throughout coastal zones (Donato *et al.*, 2011; Page *et al.*, 2011). Moreover, the region's tropical forests play an essential role in environmental protection and biodiversity, socio-economy, and the living conditions of forest-dependent populations (Lee, 2009). Furthermore, these forests are of importance in the context of global carbon balance (Fox *et al.*, 2010; Page *et al.*, 2011).

A significant challenge to sustaining the tropical forests of SE Asia and the Pacific is rampant deforestation and forest degradation, driven by growing demands for food, timber, and other natural resources (DeFries *et al.*, 2010; Gibbs *et al.*, 2010; Foley *et al.*, 2011; Wilcove *et al.*, 2013). Forest cover in these regions decreased significantly between 1990 and 2010, with the vast majority of loss (about 2/3 of the loss experienced between 2000 and 2010) occurring in insular SE Asia, driven by logging and the use of peatlands for tree crops (in particular oil palm) (Houghton & Hackler, 1999; Miettinen *et al.*, 2011; Stibig *et al.*, 2014; Miettinen *et al.*, 2017). These pressures on forests are predicted to account for 13–85% of species losses in the region by 2100 (Sodhi *et al.*, 2010). Besides threatening existing endemic species in the region, reductions in forest cover will further accelerate CO₂ emissions from opened peatlands, impacting climate change (Miettinen *et al.*, 2011; Miettinen *et al.*, 2017).

The insular sub-region of Brunei, East Timor, Indonesia, Malaysia, the Philippines, Papua New Guinea (PNG) and the Solomon Islands, which together

constitute approximately 70% of SE Asia's tropical forests, experienced rapid deforestation between 1990 and 2010 (Stibig *et al.*, 2014). The expansion of oil palm plantations during the 1990s and 2000s has been the greatest driver of deforestation in eastern Sumatra, coastal Sarawak, central and northeast Borneo, and southeast Papua (Miettinen *et al.*, 2017). There are also indications of the establishment of new forest plantations in Papua New Guinea, Peninsular Malaysia, Sumatra, and Sarawak (Stibig *et al.*, 2014). Indonesia has accounted for approximately 80% of forest conversion over the two decades (Stibig *et al.*, 2014). Sumatera and Borneo (Indonesia) accounted for approximately 10 Mha of the total 14.7 Mha forest cover losses in SE Asia between 2000 and 2010, with fibre (*Acacia mangium*) plantation and oil palm plantation concessions contributing the most (1.1 Mha and 1.2 Mha, respectively) (Abood *et al.*, 2015). Even though logging concessions have made a relatively small contribution to total deforestation, there might have been an occurrence of deforestation during logging operations (inside logging concessions) before their conversion to the tree and oil palm plantations (Casson, 2000; Kartodihardjo & Supriono, 2000). Before 2013, logging accounted for only marginal forest losses in Papua Indonesia, where deforestation was less than half that experienced by Sumatera and Borneo. Today, logging has become one of the leading drivers of industrial deforestation, and is responsible for increasing forest conversion on the island (Stibig *et al.*, 2014; Abood *et al.*, 2015; Austin *et al.*, 2017; Miettinen *et al.*, 2017).

Although the industrial sector might be expected to be the primary culprit, it was responsible for less than 50% (6.6 Mha) of Indonesia's deforestation (Stibig *et al.*, 2014). Some enterprise-based plantation activities have been shown to occur outside of concession boundaries (Gaveau & Salim, 2013). Nevertheless, small- and medium-scale illegal logging, land clearing, and forest fires, which also drive forest losses, should not be neglected (Curran *et al.*, 2004; Langner *et al.*, 2007; Ekadinata *et al.*, 2013). In some instances, protected forest areas are becoming isolated and even deforested, while buffer zones are getting degraded (Curran *et al.*, 2004) as the result of expanding populations and rapidly growing economies (Sodhi *et al.*, 2010). The factors described here are inter-correlated. Unmanaged

(and/or illegal) logging methods may leave a forest in such poor condition that it becomes fragmented. Fragmentation effects and degradation caused by logging may leave residuals that are vulnerable to intentional and accidental fire and further encroachment, which in turn may lead to deforestation and conversion.

1.2. Regeneration of natural tropical primary forest

The main challenge of achieving sustainable forest management goals is the successful mimicry of pristine forest conditions through adjustment of harvesting procedures in conformance with tropical forests' natural regeneration processes. It is crucial to consider that virgin tropical forests vary in terms of tree species composition, diversity, and structure due to variations in the environmental setting. Topography and biogeography firmly control pollination and seed dispersal agents of tree species (Walker & Hope, 1982; Petocz, 1989; Sist *et al.*, 2003b). Differences in rainfall, humidity, and temperature (Richards, 1996) combine to contribute to the formation of varying soil types, soil textures, and nutrients, which are essential for the establishment and development of particular species (Gentry, 1988; Ashton & Hall, 1992; Tuomisto *et al.*, 2003). By comprehending the physical and biological factors that shape a forest community, one can more easily understand the historical, functional, and successional behaviours of forest ecosystems (Oliver & Larson, 1990; Spies, 1998) and better understand how to reproduce natural processes in an anthropogenically altered or created forest. This understanding can serve as an important reference in the process of planning forest management techniques such as harvesting and thinning (Spies, 1998). It can also inform optimal silvicultural practices in forest restoration and rehabilitation, as well as the identification of relevant data to support conservation programs (Teketay, 1995; Teketay & Bekele, 1995; Pyke *et al.*, 2001).

The tropical forest harbours hundreds of tree species. The tropics' tree communities are typically composed of a few species with many individuals (common species) and many species with one or two individuals (rare species). This composition allows for high species diversity in natural forests. Because

many rare species not only occur locally (e.g., at the stand level) but also span an extensive biogeographical range, they may represent a vast absolute number of individuals at the landscape level (Primack & Hall, 1992; Poorter *et al.*, 1996; Pitman *et al.*, 1999). The mechanisms that allow many tree species to coexist in the same assemblage, in which the common species do not outcompete the many rare species, are vital to the natural regeneration of a tropical forest. In other words, the possibility of reproducing any individual tree depends on substantial factors of randomness in regeneration processes.

The number of tropical tree species is positively correlated with the number of pollinators and seed-dispersal animals: Insects, birds, bats, and various other mammals are vital to the successful reproduction of tropical trees (Howe & Miriti, 2000; Sist *et al.*, 2003b; Anitha *et al.*, 2010). Dispersion is essential to improving the probability of seed germination through the movement of seeds to more favourable sites that are far from the mother tree, such as tree-fall gaps or areas of nutrient-rich soil (Howe & Miriti, 2000; Babweteera & Brown, 2010).

The probability of tropical forests' successful regeneration mainly depends on small-scale natural disturbances that cause disordered canopy gaps (Denslow, 1987). Small openings in a canopy resulting from full or partial tree falls allow for the transmission of sunlight to seedlings and juvenile trees. Mid- and upper-canopy tree species mainly rely on this gap-light for their regeneration (Denslow, 1987; Kuusipalo *et al.*, 1997; Struhsaker, 1997). Tree species are grouped along a gradient of light dependence, from the light-demanding pioneers that depend on light throughout their lives to shade-tolerant types that may survive for many years in the full shade beneath the canopy (Sist *et al.*, 2003a; Vieira *et al.*, 2005; Herault *et al.*, 2010). Therefore, the species composition of forest recovering after disturbance varies with canopy-opening size and the species of seedlings or seeds that have been carried into the gap (Denslow, 1987; Howe & Miriti, 2000).

Primary tropical forest hardwoods, including most high-value timber species, are characterised by long lifespans and slow growth rates (Richards, 1996). These

species mainly regenerate in a gap and adapt to shade over the course of their lives (Vieira *et al.*, 2005). Seedlings of the commercially high-value mahogany require sunlight during their early years but can tolerate shade after becoming established (Sist *et al.*, 2003b; Hall, 2008). In fact, many dipterocarps of Southeast Asia tolerate shade throughout most of their lives (Kuusipalo *et al.*, 1997; Sist *et al.*, 2003b). Tropical timber trees are often hundreds of years old at harvest; some species with high wood density reach as much as 1000 years (Kurokawa *et al.*, 2003). Tree species vary markedly in diameter increment between and within tree species, depending on various factors such as age, edaphic and climatic conditions, and associated lianas (Peña-Claros *et al.*, 2008; Toledo *et al.*, 2011). Although most tropical timber species have innately low growth rates (Dauber *et al.*, 2005; Vieira *et al.*, 2005; Valle *et al.*, 2007), they are capable of responding swiftly to light provided by canopy gaps and increasing their diameter increment (Herauld *et al.*, 2010).

The huge number of tropical tree species is reflected in the population structure of the forest assemblage. The long tail part of the inverted J-shape curve of size class distribution represents larger stems with few seedlings, saplings, and poles. For instance, 46 out of 49 valuable timber species from the Bolivian forest are large-sized trees with low numbers of individuals and low regeneration rates; these are mostly pan-neotropical species, such as *Swietenia macrophylla* and *Cedrela fissilis* (Fredericksen, 1999). Valuable timber species with low regeneration rates are common throughout the Amazon, as well as in African and Southeast Asian forests (Poorter *et al.*, 1996; Kammesheidt *et al.*, 2001; Sist *et al.*, 2003b; Hall, 2008; Schulze *et al.*, 2008; Anitha *et al.*, 2010; Babweteera & Brown, 2010). Their low population density is mainly associated with their status as nonpioneer light-demanding species whose seedlings cannot tolerate shade, causing high seed and seedling mortality rates in natural forests (Hall, 2008). The sparse regeneration implies a scant probability of tree recruitment in a high-diversity assemblage. However, their dispersal ability allows them to scatter across the natural forest such that at a landscape level, they can persist as a viable population.

1.3. Logging impacts on tropical forests

Forest destruction caused by logging in tropical forests varies depending on the harvest intensity level, indicated by the number or volume of timber harvested per ha. When logs are felled and dragged ('skidded') from their stump to the log landing, they create tree gaps and skid trails. Tree felling can destroy surrounding trees through collateral canopy damage and skidding processes, which clear seedlings and saplings and injures future crop trees, often leading to mortality (Hawthorne *et al.*, 2011). Thus, the magnitude of forest damage increases with the number of felled trees to a point at which where primary forest, non-pioneer trees, cannot naturally grow (Struhsaker, 1997; Sist & Nguyen-Thé, 2002; van Gardingen *et al.*, 2006). Under high logging intensity conditions, in which more than eight trees are typically extracted per ha (Sist *et al.*, 2003a), the pressure on the natural regeneration of primary forest substantially kills most of the future crops and seed trees (Verissimo *et al.*, 1995; Sist *et al.*, 1998). Felling creates gaps that strongly favour fast-growing pioneer vegetation that easily outcompetes slow-growth species (Kuusipalo *et al.*, 1997; Struhsaker, 1997; Sist & Nguyen-Thé, 2002; Sist *et al.*, 2003b; Park *et al.*, 2005). This may also have a detrimental influence on populations of animal pollinators and seed dispersers (Struhsaker, 1997; Babweteera & Brown, 2010).

Most of the tropical commercial timber species are non-pioneer, light-demanding species. During the recruitment stage (up to a DBH of 10 cm), trees' growth rates are optimal when canopy openings are few and relatively little soil disruption occurs (Kuusipalo *et al.*, 1997; Hall, 2008). Therefore, the gap sizes created during felling and skidding are vital to a balance between sustaining valuable tree species and restricting pioneer vegetation growth in logged forests. In a less than 500 m² felling area, the promotion of non-pioneer, shade-intolerant juveniles were found to be ideal, with a maximum of 10% stand canopy opening and a minimum of 85% maintained stand basal area (Struhsaker, 1997; Sist *et al.*, 2003a). In other words, the growth of valuable timber species is optimized by low-intensity logging. The practice of cutting less than five stems per ha has been suggested to minimize adverse impacts on future trees and drive stand recovery toward

initial composition and structure (Struhsaker, 1997; Sist & Nguyen-Thé, 2002; ter Steege *et al.*, 2002; Hall *et al.*, 2003).

However, the current Indonesian silvicultural system has less control over cutting-intensity if an intact forest with large high-value timbers is the targeted logging area (Sist *et al.*, 1998). The Bornean dipterocarp forest, where commercially valuable trees are in abundance (about 200 species), is mainly harvested at high intensities of more than ten trees per ha, which results in 50% canopy gaps and severe damage on the residual stand (Sist & Nguyen-Thé, 2002; Sist *et al.*, 2003a; Forshed *et al.*, 2008). Most of these logged forests remain low in timber value; many are eventually termed "degraded forests" and reclassified for other land uses (Kartodihardjo & Supriono, 2000; Gaveau *et al.*, 2013).

Even though patchily distributed and represented by only three genera (*Anisoptera*, *Vatica*, and *Hopea*) and about five species, the Dipterocarpaceae family dominates the forest canopy in South-West New Guinea (Paijmans, 1976). In these forests, approximately ten out of every fifteen commercial trees cut per ha are dipterocarps, which has led to the domination of 8-year logged stands by the pioneer, light-demanding family *Moraceae*, followed by the dipterocarp *Vatica rasak* (Gandhi & Mitlöhner, 2014). Similarly, the heavily exploited Merbau tree (*Intsia bijuga* O.K.) in the northern part of the island (Thaman *et al.*, 2006; Sadono, 2014) has been harvested at high intensities, constituting approximately five of every thirteen harvestable trees (personal interview). This high-value species has also attracted destructive, illegal logging in the Papuan forest (Newman, 2008). Currently, the disappearing Merbau stock is triggering a demand to classify the species as endangered in APPENDIX III CITES. Should the current Indonesian logging practices, which fail to meet the sustainability qualifications, continue, subsequent harvests of less-valuable species within these forests raise the threat of reclassification to "degraded forest", as has occurred in other parts of Indonesia. In short, a trade-off between the number of each tree species cut above which the species locally vanishes is urgently necessary.

1.4. Sustaining tropical logged forests

Sustainable Forest Management (SFM) has been broadly defined. It ranges from the simple-narrowed that is 'management capable of maintaining timber productivity in a management cycle' to the modern-broadly as 'management preserving productivity, forest structure, diversity, and the fundamental ecological process of populations, communities, and the ecosystem' (FAO, 2014). SFM has been of global relevance in the effort to combat large-scale, industrially driven deforestation in the tropics, and SFM is one of the United Nations' instruments for addressing sustainable development and biodiversity conservation. The UN Framework Convention on Climate Change (UNFCCC) provides incentives for the implementation of SFM in tropical forests through its Reducing Emissions from Deforestation and Forest Degradation (REDD+) policy (UNFCCC, 2008). Several tropical countries, mainly members of the International Tropical Timber Organization (ITTO), have demonstrated a serious commitment to SFM as a means of preserving tropical forests (Blaser *et al.*, 2011). For example, Indonesia has established its Criteria and Indicators, developed national standards for forest certification, and introduced the mandatory verification of SFM into its management system (ITTO, 2006). The country's revised Criteria and Indicators for SFM consist of four criteria (enabling conditions, production, ecology, and social aspects) and 24 indicators, endorsed by the national government through Ministerial Forestry Regulation No. 4795/Kpts-II/2002 issued on 3 June 2002.

It is debatable whether the practice of harvesting high-value timber for more than a single cutting cycle can assure the sustainability of the natural forest ecosystem, as stated by SFM goals (Nasi & Frost, 2009; Zimmerman & Kormos, 2012). As previously discussed, many tropical trees of intact forests, including high-value timber species, have life spans and growth rates such that they take years to become harvestable. These species' seeds and seedlings also experience high mortality rates. At last, these timber species rely on a diversity of animal dispersers for reproduction. All of these traits imply that vast, continuous areas of forests are important in maintaining viable population sizes and enabling sustainable timber harvesting in tropical forests (Pitman *et al.*, 1999). In general,

however, most of the timber concession areas are limited to one cycle period of 30-35 years, which compels timber companies to reenter the same places for subsequent cutting cycles (Zimmerman & Kormos, 2012). It is a considerable challenge for profit-oriented companies with limited knowledge of the tropical trees' life history to maintain the productivity of high-value timber using current cycles (Zimmerman & Kormos, 2012). Therefore, without implementing financial incentives and new forest regulations, sustainability goals are unlikely to be achievable (Schulze, 2008). The problem is exacerbated if little is known about the interactions of individual trees with their environment, e.g., species-specific growth rate, germination and growth requirements, animal pollinator and disperser species, and edaphic preferences (de Freitas & Pinard, 2008; Grogan *et al.*, 2008).

Although most countries in the tropics have a standard protocol for timber logging that includes anticipation of drivers and impacts (ITTO, 2006), the life histories of timber species are universally poorly understood (Zimmerman & Kormos, 2012). The high international demand for tropical timber has led to increasingly profit-oriented motivations and the harvesting of valuable species first. When these species are less available for exploitation in the second cycle, the next lower-grade timbers are targeted, until the commercial species are threatened by disappearance altogether, and the land becomes more financially viable for uses other than forestry (Casson, 2000; Asner *et al.*, 2006). The voracious consumption of commercial trees in the export market has forced the expansion of logging areas far into the pristine forests of the Amazon, Central Africa, and Borneo (Kammesheidt *et al.*, 2001; Laporte *et al.*, 2007; Hall, 2008; Bryan *et al.*, 2010). To compound the problem in some instances, abundant timber supply from illegal and conventionally unmanaged logging diminishes the incentive for management or conservation (Nawir & Rumboko, 2007). Conventional logging has reduced many Southeast Asian primary forestlands to degraded secondary forests, some of which have been cleared for agriculture (Casson, 2000; Kartodihardjo & Supriono, 2000; Stibig *et al.*, 2014; Abood *et al.*, 2015). This

relentless process is also well underway in the wild forests of the last frontiers of Indonesia, including the Papuan forest (Austin *et al.*, 2017; Austin *et al.*, 2019).

SFM protocols that rely on government-mandated silviculture systems have been practised in many tropical timber-producing countries. Blaser *et al.* (2011) reported that many ITTO countries practice cutting cycles (30-35 years), minimum felling diameters (40-60 cm DBH for all species), per-unit-area harvest intensities, and seed tree retention rates applied in combination with proven techniques for reducing damage to the residual stand during logging operations (reduced-impact logging, RIL). The report also indicates that silvicultural techniques that can enhance the abundance and diameter increment of timber species' seedlings and saplings for recruitment are not yet clearly stated in SFM procedures.

RIL is a set of timber-harvesting techniques for reducing the mortality rates of residual trees caused by logging and preserving the ecological integrity of the logged stand. These techniques are useful for increasing the survival of seedlings, saplings, and future trees of commercial species, and have proven effective in reducing collateral damage to the residual stand by 20%–50% (Putz *et al.*, 2008). However, if logging at higher intensities (>five trees ha⁻¹) and employing diameter-limit cutting RIL alone fail to sustain production and take no marked effect in future harvests (Kammesheidt *et al.*, 2001; Sist *et al.*, 2003b; van Gardingen *et al.*, 2006; Peña-Claros *et al.*, 2008; Kukkonen & Hohnwald, 2009).

Furthermore, some studies have indicated that logging operated under current SFM protocols is inconsistent with sustainability in terms of production and ecological integrity (Poorter *et al.*, 1996; Sist *et al.*, 2003b; Dauber *et al.*, 2005; de Freitas & Pinard, 2008; Schulze *et al.*, 2008; Herault *et al.*, 2010). These researchers show that minimum rotation cycles (typically 25-35 years) are too short to maintain the quality and quantity of the first yield in the third cycle, minimum felling diameters (50 cm DBH) are too small to preserve adequate populations of reproductive adults and control harvest intensity, harvest intensities (around eight trees per ha) are too high and leave residual stands with