Increased carbon sequestration of actively restored tropical forests in Sabah, Malaysia

A comparison of natural regeneration and active restoration

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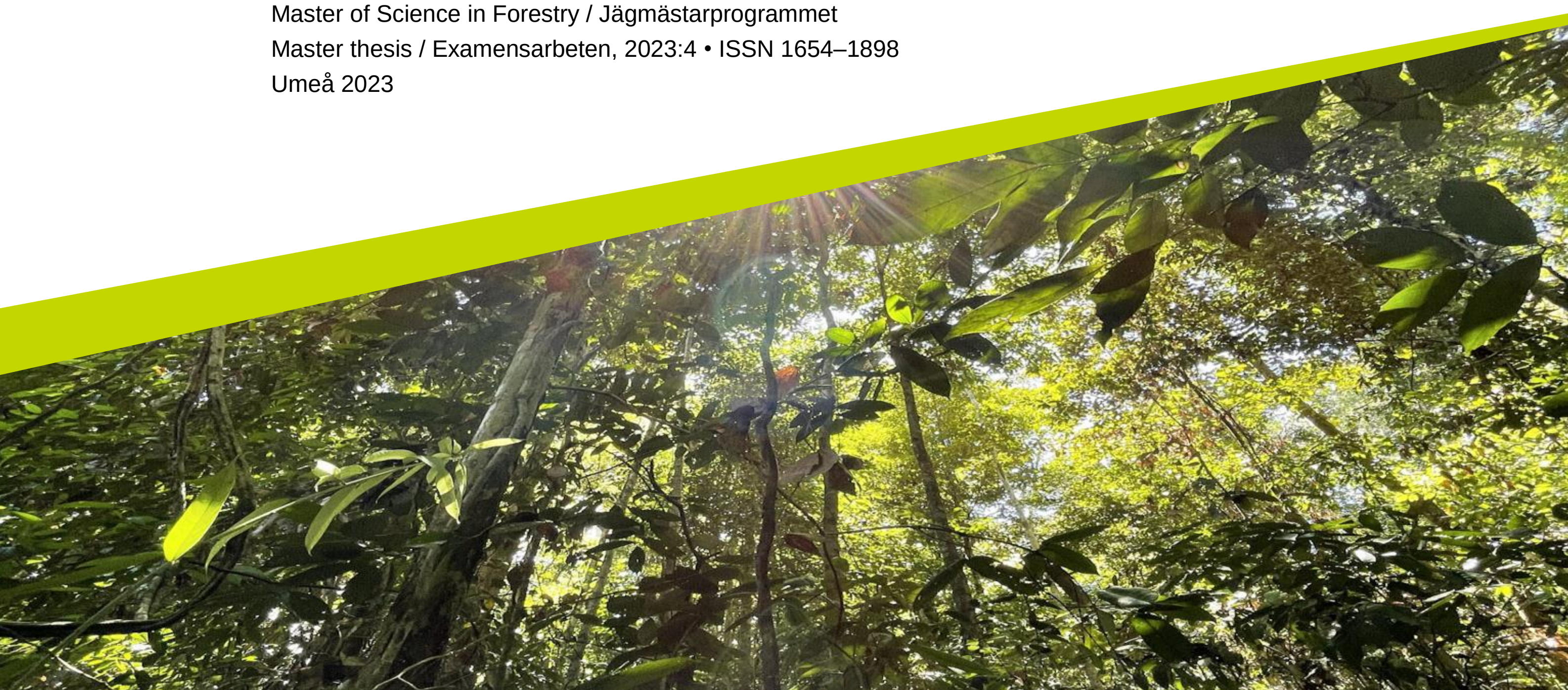
Master's thesis • 30 credits
Swedish University of Agricultural Sciences, SLU
Department of Forest Ecology and Management
Master of Science in Forestry / Jägmästarprogrammet
Master thesis / Examensarbeten, 2023:4 • ISSN 1654–1898
Umeå 2023



Increased carbon sequestration of actively restored tropical forests in Sabah, Malaysia.

A comparison of natural regeneration and active restoration

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Credits:	30 credits
Level:	Advanced level A2E
Course title:	Masterarbete i Skogsvetenskap, A2E - Skogens ekologi och skötsel
Course code:	EX0958
Programme/education:	Jägmästarprogrammet / Master of Science in Forestry
Course coordinating dept:	Department of Forest Ecology and Management
Place of publication:	Umeå
Year of publication:	2023
Cover picture:	Johan Engström
Copyright:	All featured images are used with permission from the copyright owner.
Title of series:	Master theses / Examensarbeten
Part number:	2023:04
ISSN:	1654–1898
Keywords:	Degraded tropical forest, active restoration, carbon sequestration, natural regeneration, enrichment planting, biomass accumulation

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Abstract

As restoration with assisted natural regeneration and enrichment planting becomes a more important and popular method to restore ecosystem services and mitigate negative climate effects, these treatments and their effectiveness is heavily debated. There are three main groups of opinions regarding restoration, one is that humans actively need to help nature recover and the second is that nature itself does a better job without our help and the third is somewhere in between. In my study, I compared active and passive restoration in degraded tropical forests within the INIKEA project in Sabah, Malaysia. Along with previously collected data from 2017, I used my own data collected in 2022 to investigate the effects of active restoration with focus on enrichment line-planting followed by continuous maintenance for 10 years. My result showed that some control plots were still almost completely bare of trees, even 40 years after the last severe disturbance. This indicates that heavy degradation sometimes hinders natural regeneration for decades, whereas no such delayed regeneration was found in actively restored forests. In contrast, after 23 years actively restored forests had 41.8 Mg C ha^{-1} higher carbon storage ($p=0.015$) compared to naturally regenerated control plots. However, the increase in carbon sequestration cannot be attributed to planted trees since only 30 trees per hectare out of 300 planted had grown into the measurable diameter size (dbh ≥ 10). The average diameter of these planted trees was still low, 17.4 cm, which contributes 4.4 % or 1.84 Mg C ha^{-1} of the added carbon sequestration as a result from active restoration. I conclude that most of the additional carbon sequestration from active restoration came from the continuous maintenance where lianas, climbers and other competing vegetation is removed which led to an increased growth of the already established and naturally regenerated trees. Consequently, knowledge like this could be useful in decision-making concerning restoration methods to maximize benefits in a cost-effective way.

Keywords*: Degraded tropical forest, active restoration, carbon sequestration, natural regeneration, enrichment planting, biomass accumulation*

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1. Introduction

The world's forests serve as a rich carbon storage and sequester more carbon than any other terrestrial ecosystem (Gibbs et al. 2007). This makes forests an indispensable tool in mitigating climate change. Global forests carbon stock is estimated to 861 ± 66 Pg C and 55 % of this carbon is stored in tropical forest, 32 % in boreal forests and 14 % in temperate forests (Pan et al. 2011). Tropical forest, as opposed to boreal forests, store most of its carbon in aboveground biomass and less in the soil (Ibid), resulting in faster and greater carbon releases when subjected to deforestation and land use change. It is estimated that during the 1990s, tropical deforestation and land use change released 15-25 % of the annual global greenhouse gas emissions (Malhi & Grace 2000; Gibbs et al. 2007; Pan et al. 2011).

Pristine tropical forests are currently disappearing at a rate of 1-2 % per year (Ibid). Decades of logging and land use conversion of tropical forests has resulted in large areas of degraded landscapes and secondary forests (Chokkalingam & De Jong 2001; van der Werf et al. 2009). Secondary forests are a successional forest state developed after severe natural or anthropogenic disturbance of the original forests (Chokkalingam & De Jong 2001). Examples of this could be forest regrowth after fire, logging or agriculture (Zhu 2002). Degraded tropical forests cover approximately 550 million hectares, an area equal to half of Europe (Pan et al., 2011). The main causes of degradation and deforestation in the tropics are commercial logging and land use transformation to large-scale commercial agriculture (FAO 2020).

During the current decade, several countries, and global organizations have come together in large scale movements like the Bonn Challenge to restore 350 million hectares of deforested lands and ecosystems by 2030 (UN 2021; IUCN 2021). Even though active restoration in many cases are considered to be successful with increased recovery rates for aboveground biomass (AGB) (Wheeler et al. 2016; Philipson et al. 2020), questions about its effectiveness still needs to be evaluated (Crouzeilles et al. 2017). In large scale restoration projects, economy and timeframe are important factors as active restoration can be a slow, lengthy, and expensive process. The mean cost for tropical forest restoration in developing countries is roughly \$1600 ha^{-1} and for Australia between \$6000 to \$15,000 ha^{-1} (Philipson et al. 2020). A more cost-effective alternative to active restoration could be passive regeneration (Crouzeilles et al. 2020). However, natural aboveground carbon (AGC) recovery for degraded topical forests show a wide variation of accumulation rates ranging from 0.3 Mg C ha^{-1} to 4.3 Mg C ha^{-1} and depends on many factors (Galante et al. 2018). Therefore, more knowledge is needed to evaluate the success and cost effectiveness of active restoration, compared to more passive forms of restoration (Crouzeilles et al. 2017).

The island of Borneo has some of the world's most biodiverse and carbon-dense tropical forests (Asner et al. 2018) known for its abundant wildlife and unique species compositions. It is estimated that Borneo has up to 15,000 flowering plant species, comparable to the entire continent of Africa, which is about 40 times larger in size (MacKinnon et al. 1996). In total, about 3000 tree species grows on Borneo and 267 of them belong to the *Dipterocarpaceae* family (dipterocarps) (Appanah & Turnbull 1998). Borneo's forests are the world's richest in dipterocarp species and about half of them are endemic to the island (Ashton 1982). From an ecological standpoint, dipterocarp trees are a key component of the mature tropical forest, shade tolerant in early development and canopy forming in later stages (Ashton 1988). Furthermore, dipterocarps generally have high wood density (WD) and are long lived (ibid) compared to pioneer species and therefore contribute to the long-term carbon storage of the tropical forest. However, many dipterocarps do not have a dormant seed stage and germinate soon after falling which sometimes complicates natural establishment after disturbance. In logging operations, Dipterocarps is regarded as the most important group of trees because of its high commercially valued timber (MacKinnon et al., 1996). Between 1973 and 2015 a new industrial era emerged on Borneo where selective logging and plantations industries grew in a rapid pace (Gaveau et al. 2016). Intensified selective logging operations during this time period resulted in a loss of 18.7 million ha of Borneo's old growth forests (Gaveau et al. 2016). The most common land use change resulting in loss of forest in Southeast Asia in recent decades is conversion of forest into oil palm and pulp wood plantations (Gaveau et al. 2016). Although, it is not clear if this land-use conversion is the reason for deforestation or merely follows it (Fitzherbert et al. 2008).

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Borneo's northern part, the state Sabah, Malaysia has a forest area of 7.25 million hectares and during the 20th century it has been subjected to widespread exploitation via selective timber logging and land-use conversion (Pinard et al. 1996; Jomo et al. 2004). A more recent development of exploitation has led to large clear-cutting of secondary forests which is then converted into oil palm or timber plantations (Bryan et al. 2013). It is estimated that about 3.7 million hectares, excluding mangroves, of natural forests remains in Sabah (Asner et al 2018). Of these 3.7 million hectares, more than 40% of forests with the highest carbon stock are located outside regions that are designated for maximum protection. This suggests that there are still large amounts of carbon storage that potentially could be lost since conversion from secondary forest into oil palm plantation regularly occur (Asner et al. 2018). However, there exists great potential of increasing carbon sequestration in Sabah by natural regeneration and letting previously logged forests recover. Based on current data, an additional 362.5 Tg (million metric tons) of carbon, a doubling of current stocks, could be gained by providing logged forests with sufficient time to recover (Asner et al. 2018). Many characteristics of post logging sites such as compacted soils, decreased water infiltration rates and extensive growth of weedy plants may limit natural re-establishment of trees (Malmer & Grip 1990; Pinard et al. 1996). Furthermore, selective logging is known to increase the risk of forest fire, especially during dry periods which was evident during the severe drought event of El Niño 1983 (Siegert et al. 2001). Once a forest has burned, it has increased risk of entering a cycle of repeated fires which hinders natural recovery and increase degradation (Cochrane 2003; Hoscilo et al. 2011)

The main objective of this master thesis was to increase the knowledge concerning restoration of degraded tropical forests and investigate the differences of natural and assisted regeneration. Potentially, increased aboveground biomass (AGB) recovery of actively restored forests occurs for two main reasons. The first reason is that active removal of climbers, lianas and vegetation as well as later shade adjustment through girdling of pioneer tree species reduces competition and can promote growth in already naturally established secondary trees. The second reason is that planted trees of mainly late successional species with high wood density (WD) start to reach the canopy and experience increased growth. Wheeler et al. (2018) showed slow initial AGB accumulation of restored tropical forest in Uganda which accelerated 10 years after initial restoration treatment. Furthermore, a study by Clark & Clark et al. (1990) revealed that tall growing, late-successional tree species in tropical forests of Costa Rica had an exponential growth pattern which start out slow and increased with increased basal area. I therefore hypothesis that while the naturally established trees will benefit from restoration treatments, the planting of dipterocarp species will have a low initial effect on AGB, since any significant effect would require longer than 20 years to be expressed.

To investigate these questions and hypotheses I inventoried a tropical rainforest in the INIKEA project area in Sabah, Malaysia that has been naturally recovering for 40 years since last severe disturbance. These data were then compared to previously collected measurements from similarly degraded forest within the INIKEA project where enrichment line planting and 10-year maintenance had been applied.

I anticipate that the outcome of this study can contribute to a better understanding of when and where to use active restoration and when natural regeneration might be sufficient, as of today there are divided opinions in this matter. I will use two research question to guide me through analyses and discussion:

i) What is the carbon storage of a secondary tropical forest within the INIKEA-project Malaysia Borneo 40 years after disturbance without active restoration?

ii) Has active restoration efforts in a similarly degraded tropical forests within the INIKEA-project had any additional effect on carbon storage compared to the forest without active restoration?

2. Material and method

2.1 Study site

This study was conducted within the Innoprise-IKEA (INIKEA) Forest Rehabilitation Project in Kalabakan Forest Reserve located in the southern part of Yayasan Sabah’s Forest Management Area in Sabah, Malaysia, Borneo (Figure 1). The INIKEA project, also called the Sow-A-Seed project, was launched in 1998 as a collaboration between the Malaysian forestry organisation Yayasan Sabah, the Swedish furniture company IKEA together with the Swedish University of Agricultural Sciences (SLU). The goal was to restore 18,500 hectares of degraded forest landscape through different methods of active restorations (Laneng et al. 2021). During the 1970s, the study area was subjected to selective conventional logging. Selective logging is the most common form of harvest in the tropics where a limited number of marketable tree species of a certain dbh size (≥ 60 cm) are harvested (Asner et al. 2005). In Borneo, dipterocarp tree species are the main target for extraction due to their high value as commercial timber (MacKinnon et al., 1996). During the drought of El Niño in 1983-1984 the study area was subjected to forest fires which hindered natural recovery and increased degradation. These events lead to a forest landscape in different states of degradation composed of mainly secondary forest dominated by pioneer species such as *Macaranga* spp. Since the beginning in 1998 till today, about 14 000 hectares has undergone active restoration. The most pervasive restoration method has been enrichment planting where 80 different native species been planted, followed by repeated maintenance to remove climbers and other competing vegetation. Some areas have varied extent of old growth trees that were untouched during logging events while others were dominated by pioneer species, lianas and climbers. The geographical landscape has a heterogeneous topography with steep slopes, hills, rivers and ravines. The soil is characterized by older clay-rich subsoils which are acidic in nature with a yellow-red colour (Dahlgren et al. 2008). The annual mean precipitation of 2517 ± 760 mm (Gustafsson et al. 2016) together with a stable temperature varying between 22-32.7 degrees Celsius (Romell 2007) creates a warm and humid climate all year round. Roughly, once every ten years, severe droughts linked to El Niño events affect the area (Newbery et al. 1999; Malhi et al. 2004)

Figure 1. Map displaying Borneo's location in Southeast Asia in the top left quadrant and a colourful figure representing Borneo in the centre. The INIKEA-project is marked by a red dot. The state of Sabah and Sarawak, Malaysia is represented by dark grey colour. Brunei is shown in blue colour and the Indonesian section of Borneo is represented by green colour. Published with permission from Daniel Lussetti

2.2 INIKEA-project

The INIKEA Sow-A-Seed project was divided into four different phases where every phase consists of smaller blocks averaging from 20-200 hectares (Figure 2). During a five years period, forests within a phase were inventoried and if needed active restoration was performed. After restoration was finished in one phase, the next phase started. After five initial years of enrichment planting, five to ten years of maintenance followed, including removal of climbers and competing vegetation. Phase 1 began in 1998 and continued until 2003, phase 2 took place during 2003-2008, phase 3 between 2008-2013 and lastly, phase 4 from 2015 till 2020. Since the start up in 1998, over 5 million seedlings have been planted over an area of

11 000 hectares through enrichment planting with the goal to restore biodiversity and ecosystem services.

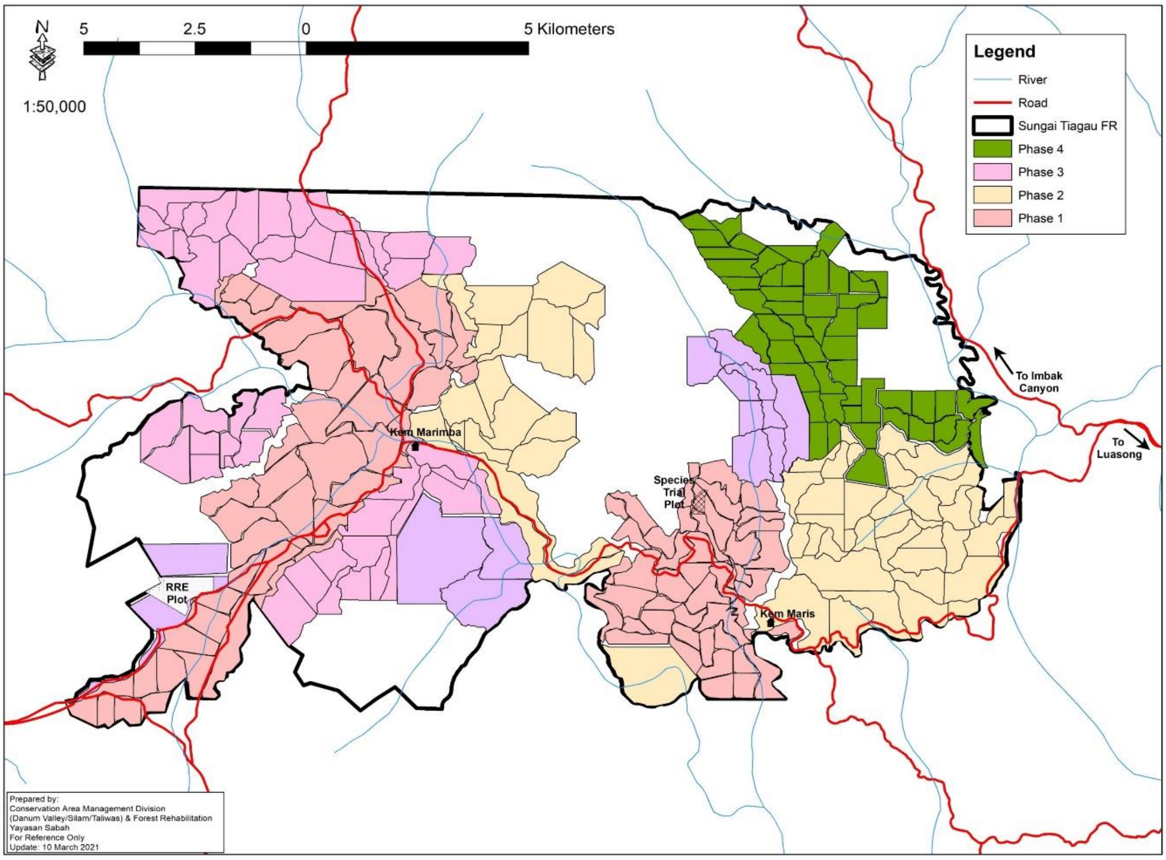

Figure 2. A schematic image showing the different phases, rivers, roads, camps, and outlines of the INIKEA-project area in Sabah, Malaysia, Borneo. Phase 1 is represented by orange colour, phase 2 by khaki colour, phase 3 by pink colour, phase 4 by green colour, experimental plots by purple colour. The border of the INIKEA-project is represented by a thick black line surrounding the project-area and thin black lines denotes the block boarders. Permanent camps inside the project area are represented by black squares. Roads are shown as think red lines. Rivers are shown as pale blue lines. The scale is 1:50000

Three main methods of active restoration have been used. The methods used were enrichment planting through gap-cluster or line planting and liberation. Line planting was performed in the most degraded areas where the canopy was severely fragmented. Seedlings are then planted in parallel lines with ten meters spacing along a trajectory. The lines are two meter wide, cleared from vegetation and seedlings are planted every three meters (Figure 3B). Gap-cluster planting was generally performed in areas that are a bit less degraded, these forests usually have a partial canopy with scattered gaps. An area of 400 m^2 was divided into ten meters squares that were examined and if gaps were present within these squares, vegetation was cleared, and seedlings planted (Figure 3A). However, if more than three naturally regenerated tree species existed in the 10 x 10 square, no additional seedlings were planted. The tree species were randomly selected from 80 native species with pre-decided portions of 70% dipterocarps, 25% non-dipterocarps and 5% fruit trees. Liberation was mainly used in less degraded areas with an existing forest canopy to facilitate natural regeneration and tree growth of existing trees. The liberation technique has been applied on 2 600 hectares and include a one-time removal treatment of climbers and lianas, in some cases shade adjustment is preformed where pioneer species like *Macaranga* spp are girdled. During phase 4, about 500 hectares of untreated forests were set aside for research purposes. All blocks in phase 4 were randomly assigned a treatment which included liberation, gap cluster planting, line planting and control. Control plots which had been left untreated as well as treated plots had two Permanent Sample Plots (PSP) with the purpose to establish a baseline for degraded forests prior to active restoration. Although randomly assigned treatments are good for scientific reference, it can become a problem when a control plot, that was supposed to act as a baseline for a heavy degraded forest harbours aboveground biomass equivalent to a pristine and untouched forest. During the earlier phases of 1, 2 and 3, blocks were not randomly assigned a treatment but inventoried and assessed before deciding what treatment would be best suited. If the block was largely intact and already had functioning natural regeneration, liberation would be applied. If the block consisted of mostly heavily degraded open areas, line planting was performed. When a block was dominated by pioneer species and climbers with smaller openings, gap-cluster planting would be applied.

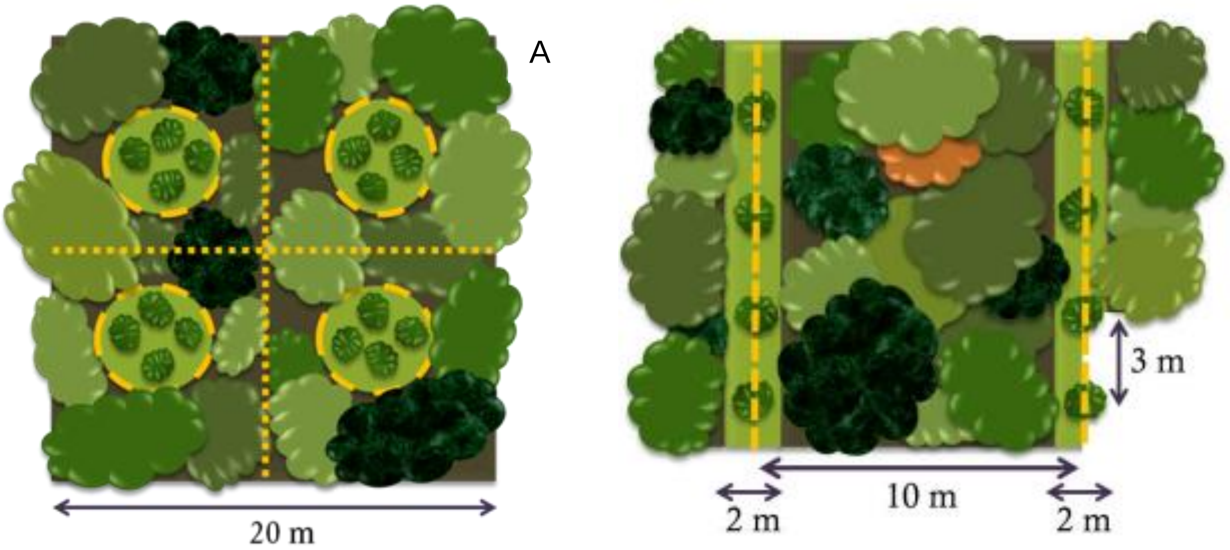

Figure 3. Schematic illustrations of A) gap-cluster planting and B) line planting as implemented in the INIKEA Project Area located in the Yayasan Sabah’s Forest Management Area in Sabah, Malaysia Borneo. For gap-cluster planting (A) squares of 20 x 20 m were divided into smaller sub-plots of 10 x 10 m. If gaps were found in a sub-plot they were planted with seedling. However, if natural regeneration of more than three species was found in the sub-plot gap, no seedlings were planted. When line planting was preformed (B), 2 m wide transects were cleared from competing vegetation with a spacing of 10 m. In each transect, a seedling was planted every three meters. Area cleared from competing vegetation is shown as light green rows and circles.

2.3 Data collection for control plots

Field work and data collection were conducted during September and October of 2022 as 16 plots, 40 x 40 meters were established on previously established Permanent Sample Plot (PSP). Coordinates were gathered in corner No.1 of the PSP and the plot was then laid out with the sides extending to the north (0 or 360 degrees) and east (90 degrees) from this corner (Figure 4). Within every sampling plot, smaller sub-plots with a 10 x 10-meter grid were established to locate the position of trees more precisely. Every sub-plot was given a code representing its position in the main plot. All trees within the plots with a diameter ≥ 10 cm was identified, tagged, and measured for diameter at breast height (dbh = 1.3 m). Dead trees and lianas with a dbh ≥ 10 cm were also measured. Trees with buttresses were measured 30 cm above the highest buttress.

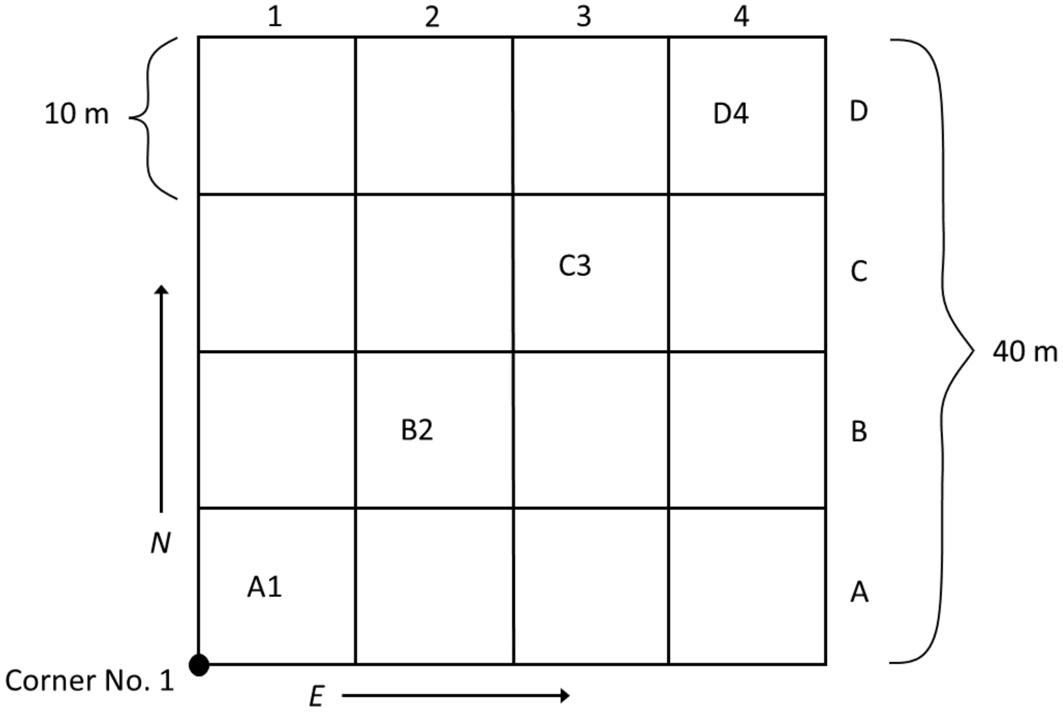

Figure 4. Plot design used during inventory of fall 2022 in the INIKEA Project Area located in the Yayasan Sabah's Forest Management Area in Sabah, Malaysia Borneo. The main plot was established by a 40x40 m square containing 16 smaller sub-plots of 10x10 m. Every sup-plot was given a unique code starting with a letter and ending with a number depending on its position in the main plot. Plots started from the corner of A1 and sides were laid out in north and east directions.

During phase 1, 2 and 3, blocks were given treatments depending on their level of degradation, if an area was heavily degraded, it was assigned with enrichment planting, and if less degraded assigned liberation treatment or left without treatment. In contrast to phase 4 where all plots were randomly assigned a treatment or control. This leads to data bias as all restored plots within early phases were heavily degraded and some control plots from phase 4 was almost untouched by logging and fire due to randomised selection.

Due to this bias, levels of degradation from 1 to 4 was implemented for all control plots within phase 4. The aim of this classification was to determine what treatment, if any a forest plots would have received if they were to be restored during 1998 at the start of the project. In this way, some plots could be excluded from the comparison of restored plots which all were heavily degraded in 1998. The most degraded plots were assigned degradation level 4 and had AGC values ≤ 40 Mg C ha^{-1}, level 3 had AGC values between 40 Mg C ha^{-1} and ≤ 110 Mg C ha^{-1}, level 2 had AGC values between 110 Mg C ha^{-1} and ≤ 170 Mg C ha^{-1} while the least degraded plots were assigned level 1 with AGC values > 170 Mg C ha^{-1}. It is likely that control plots with degradation level 3 and 4 would have received active restoration treatment of enrichments planting back in 1998 based on rangers' experience. However, plots with degradation level 2 would probably receive liberation and plots assigned level 1 all had AGC values higher than 170 Mg C ha^{-1} and plenty of large trees present which could not have been regenerated after 1998. These forests would not have been considered for active restoration according to the rangers. Based on discussions with local field ranges and previous studies about carbon storage in degraded tropical forest (Asner et al. 2018; Philipson et al. 2020) I then excluded control plots within degradation level 1 (all with an AGC above 170 Mg C ha^{-1}) from the comparison with actively restored plots.

When referring to aboveground carbon (AGC) in this study, I include only live, standing trees with a dbh ≥ 10 cm; lianas, dead wood and other vegetations biomasses were excluded. Calculations were based on three datasets containing measurements from forests located inside the 18,500 ha INIKEA Project Area. The most recent data was collected by me as described in the beginning of section 2.3. My dataset consisted of 16 randomised control plots with no active restoration. The second dataset was collected during 2017 and consists of 12 plots of 60 x 60 meter (Jensen 2020). Three control plots located in phase 4 and three restored plots in each phase of 1, 2 and 4. The third and last dataset, was collected by researchers at Universiti Malaysia Sabah (UMS) in collaboration with the restoration project during 2017. This data was composed of 31 plots that have undergone line planting treatment about 20 years ago and was located in phase 1.

To address research question *i)* data from 19 control plots will be used which 16 were inventoried during 2022 and three were inventoried during 2017.

To address research question *ii)* 14 control plots (five plots within degradation level 1 excluded) will be compared to 40 restored plots that was actively restored with enrichment planting 5-23 years ago and inventoried 2017.

2.4 Data analyses

To analyse the different datasets, an accumulation of annual growth between the years of 2017 till 2022 were calculated and added to all plots inventoried during 2017. In a study by Philipson et al (2020), annual growth per hectare was calculated for degraded tropical forests in Sabah during the first 30 to 35 years after logging. Measurements were based on 257 forest plots near Danum Valley, approximately 50 km from the INIKEA project, treated with active restoration or natural regeneration. They found that natural regenerated forests annually accumulated an average AGC of 2.9 Mg C ha^{-1}, while actively restored forests accumulated an average AGC of 4.4 Mg C ha^{-1} year^{-1}. Based on these carbon accumulation rates I added 3.65 Mg C ha^{-1} for every year between inventory 2017 till 2022, a total of 18.25 Mg C ha^{-1}.

Calculations of aboveground biomass was performed using an equation described by Chave et al. (2014):

$$\mathrm{AGB} = \exp[-1.803 - 0.976E + 0.976 \ln(\rho) + 2.673 \ln(D) - 0.0299[\ln(D)]^2] \qquad (1)$$

Where AGB represent the estimated aboveground biomass, E denotes an environmental variable specific for the area, ρ is the specific wood density of the tree species, genus or family and D is the diameter of the tree trunk at breast height (1.3 m).

Value for tree species wood densities were obtained from the global wood density database (Chave et al. 2009; Zanne et al. 2009). If no value was found for a specific species, an average of values for genus or family was used instead. AGB was then converted into Kg carbon by multiplying it with a value of 0.475 (Eggleston et al. 2006) and further converted to Mg C ha^{-1}.

All statistical analyses were made using Jamovi version 2.3.18 with a significant level of 0.05. Two kinds of statistical tests were used while analysing the data, Man Whitney U-test and a general linear model (GLM). I used a one-sided, non-parametric Man Whitney U-test when comparing control and restored plots. Man Whitney U-test was used since the data was not normally distributed. Two tests were performed, one with AGC for all trees and one with AGC for only dipterocarp species. When analysing the AGC in relation to years passed since last restoration, GLM was utilized with the same dataset as before. All control plots were assigned year zero while restored plots varied from 5 to 23 years since restoration. The general linear model used was:

$$y_i = \beta_0 + \beta_1 x_i + \varepsilon_i$$

Where y is the response variable, β coefficient estimates, x represents an independent variable and ε random error.

3. Results

i) What is the carbon storage of a secondary tropical forest within the INIKEA-project Malaysia Borneo 40 years after disturbance without active restoration?

The carbon storage of 19 secondary tropical forest plots is on average 135.1 Mg C ha^{-1} with wide-ranging variation from 19.5 Mg C ha^{-1} to 400.7 Mg C ha^{-1} (Table 1). Trees per hectare show large difference in tree structure within control plots varying from 44 trees to 706. Mean trees ha^{-1} was 460. Plots with a low level of degradation were also overrepresented in phase 4 with five plots in degradation level one, and four plots in degradation level two (Table 1). The highest level of degradation was only observed in two plots. Mean dbh for all control plots was 22.9 cm.

Table 1. Data from 19 control plots within phase 4, INIKEA Project Area, Yayasan Sabah's Forest Management Area in Sabah, Malaysia Borneo. Starting from the left, Block number, Plot number, aboveground carbon (Mg C ha^{-1}), level of degradation, Trees per hectare (dbh ≥ 10 cm), Mean dbh (cm) for trees within each plot. The level of degradation goes from 1 till 4 where 4 represent the most degraded. Only plots inventoried during 2022 have a level of degradation, this includes 16 plots.

Block	Plot	Mg C ha^{-1}	Level of deg	Trees ha^{-1}	Mean dbh (cm)
136	3	19.5	4	44	30.8
140	5	21.2	4	206	18.6
138	11	45.4	3	363	18.3
154	39	69.7	3	394	21.2
140	8	71.3	3	425	21.9
136	-	86.5	-	578	16.65
136	2	103.2	3	406	22.4
138	12	106.5	3	375	23.2
140	-	121.7	-	439	22.6
165	57	127.5	2	631	20.7
147	24	134.6	2	400	25.8
138	-	137.5	-	522	22.9
152	36	150.7	2	531	23.0
147	23	151.1	2	419	23.9
156	54	173.4	1	494	26.6
156	53	179.7	1	550	24.5
154	40	217.7	1	650	23.7
152	35	249.1	1	606	23.5
165	58	400.7	1	706	26.6

Even though 85 % of all trees for degradation level 1 (n=601) and 2 (n=495) are represented by trees in diameter class 10-39.9 cm (Figure 4), most of the carbon stored was found in diameter classes over 39.9 cm (Figure 5). For degradation level 1, more than half of all AGC ha^{-1} was stored in trees with dimeter ≥ 80 cm (n=20). Degradation level 4 had the highest proportion of small trees (dbh = 10-19.9 cm) with the lowest mean tree dbh of 20.7 ± 1.93 cm (n=2). The pattern persists for all levels as mean tree size increased at lower degradation levels (Figure 4). Mean dbh for degradation levels 3, 2 and 1 are 21.5 ± 0.86 cm (n=5), 23.0 ± 0.99 cm (n=4) and 25.8 ± 1.78 cm (n=5).

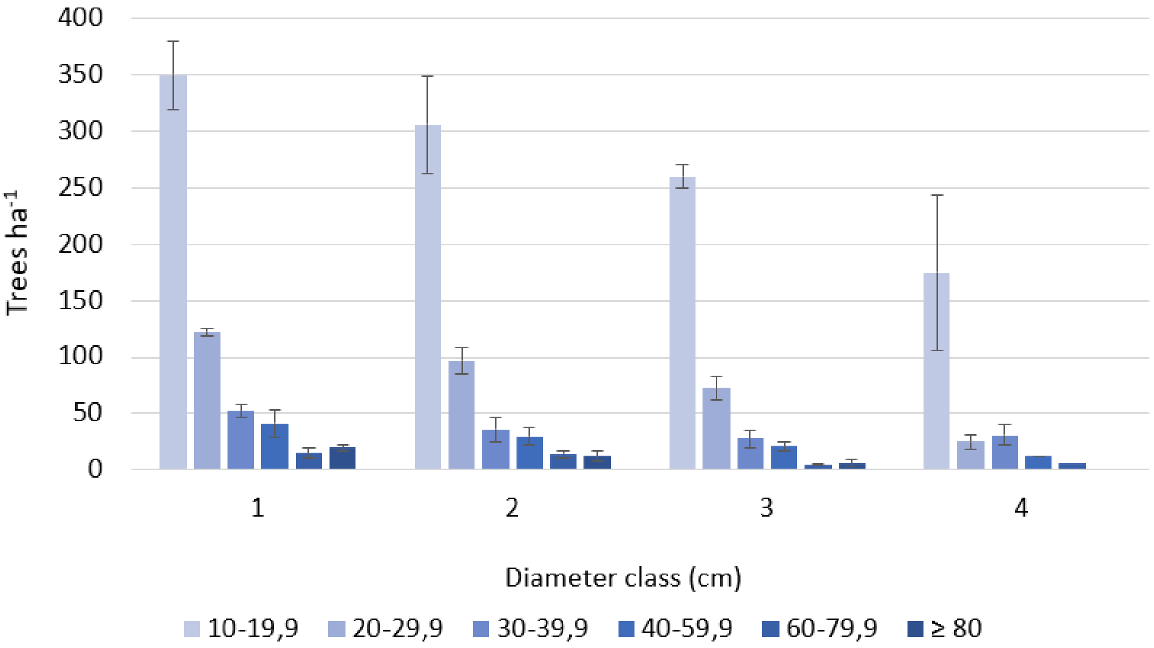

Figure 4. Diagram showing average trees per hectare per diameter class for four levels of degradation, level 1 is least degraded and level 4 highly degraded. Trees are divided into six diameter classes; 10-19.9, 20-39.9, 40-59.9, 60-79.9, ≥ 80 and represented by different shades of blue. Data is derived from 16 control plots inventoried during 2022 in phase 4, INIKEA Project Area, Yayasan Sabah's Forest Management Area in Sabah, Malaysia Borneo. Only live trees with dbh ≥ 10 are included in the data.

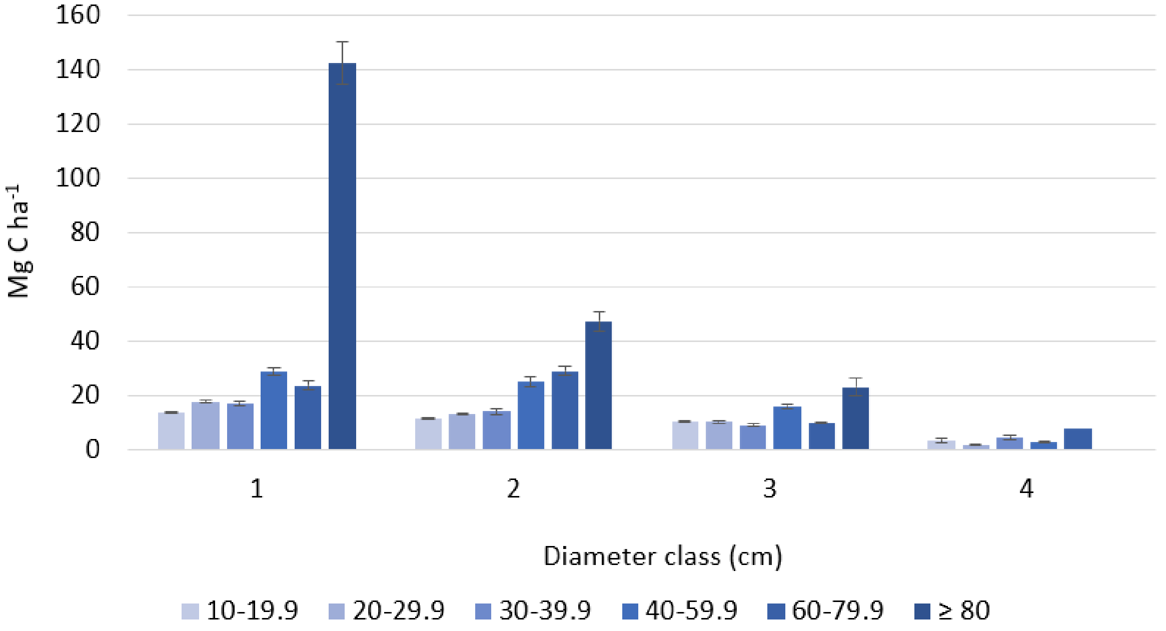

Figure 5. Diagram showing the average AGC for each diameter class divided into four levels of degradation where level 1 is least degraded and 4 highly degraded. Trees are divided into six diameter classes; 10-19.9, 20-39.9, 40-59.9, 60-79.9, ≥ 80 and represented by different shades of blue. Data was derived from 16 control plots inventoried during 2022 in phase 4, INIKEA Project Area, Yayasan Sabah's Forest Management Area in Sabah, Malaysia Borneo. Only live trees with dbh ≥ 10 are included in the data.

ii) Has active restoration efforts in a similarly degraded tropical forests within the INIKEA-project had any additional effect on carbon storage compared to the forest without active restoration?

Aboveground carbon in relation to years since restoration was analysed for 54 plots divided into two datasets dependent on tree species. Active restoration by enrichment line-planting resulted in an average annual increase of 1.91 Mg C ha^{-1} yr^{-1} (p=0.005; Fig 6A). When only dipterocarp tree species were included, active restoration by enrichment line-planting show an average annual increase of carbon with 0.385 Mg C ha^{-1} yr^{-1} (p=0.473; Fig 6B). The number of trees planted through enrichment line planting and measured with a dbh ≥ 10 cm in actively restored plots were 208 (30.2 trees ha^{-1}) with an average dbh of 17.4 ± 0.55 cm out of total 4296 trees (523 trees ha^{-1}). This number of planted trees corresponds to 4.4 % of the increased carbon sequestration achieved by active restoration in restored plots or an average of 1.84 Mg C ha^{-1}.

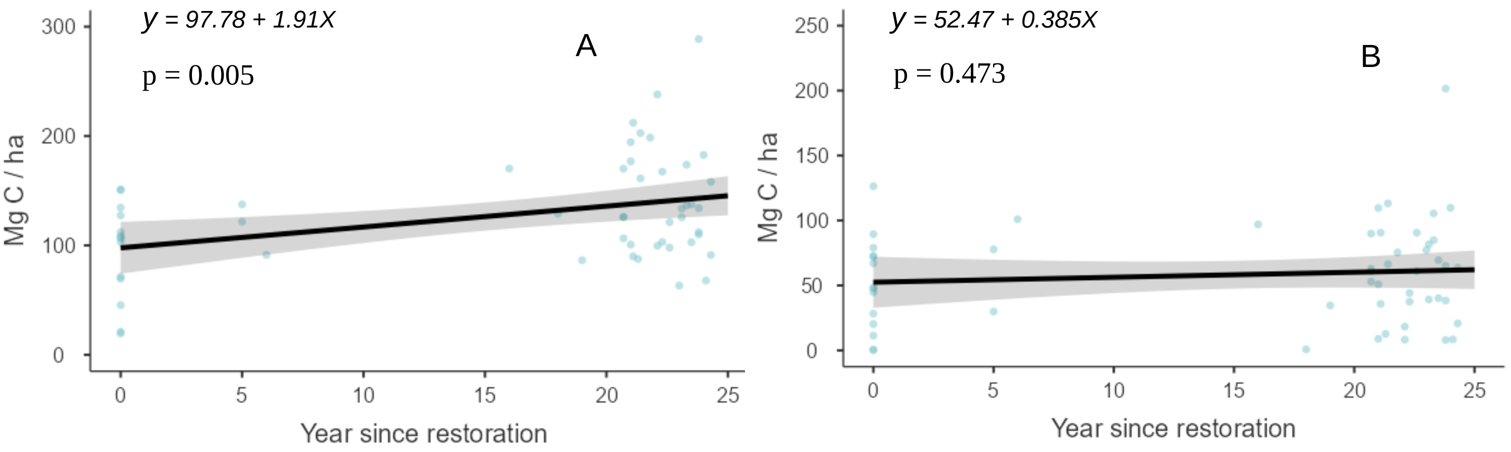

Figure 6. Carbon stock on the y-axis (Mg C ha^{-1}) and years since restoration on the x-axis. Diagram A is represented by all trees regardless of species and diagram B by only dipterocarp species. Control plots were assigned year 0 and restored plots spanned from 5 till 23.8 years since restoration. In total 54 plots where 14 are control and 40 restored plots that is represented by blue dots. These plots were inventoried within the INIKEA Project Area, Yayasan Sabah's Forest Management Area in Sabah, Malaysia Borneo.

Restored plots (n=40) had a higher average ACG of 138 ± 7.6 Mg C ha^{-1} than control plots (n=14), which had 96.2 ± 12.1 Mg C ha^{-1} ($p = 0.015$; Fig 7A). When only dipterocarp species was included in the analysis, mean aboveground carbon for restored plots were slightly higher, 61,1.3 ± 6.29 Mg C ha^{-1} than for control plots, 50.6 ± 9.76 Mg C ha^{-1} ($p = 0.212$; Fig 7B).

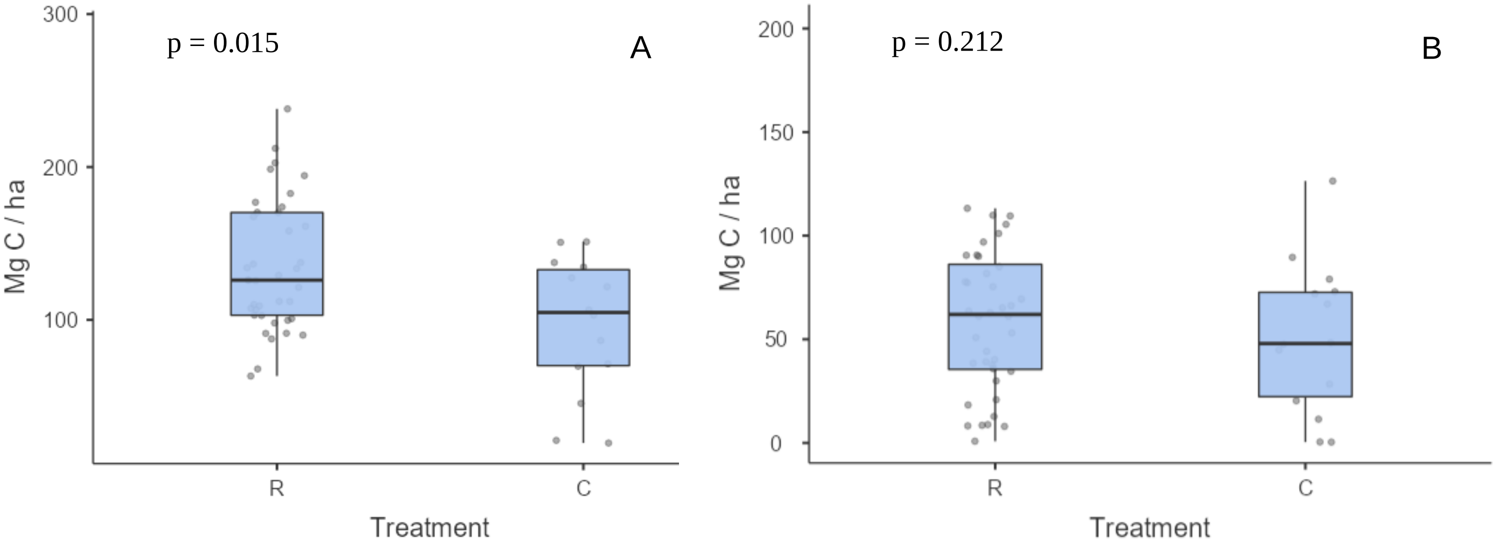

Figure 7. Boxplots presenting aboveground carbon in plots which are either control (C) or restored (R). Restored AGC is represented by 40 plots and AGC for control plots by 14 plots. AGC values for each plot is represented by grey dots. Boxplot A represents all trees independent of species family. Boxplot B represents only dipterocarp trees species. Data presented in the diagram were collected from total 54 plots inventoried in INIKEA Project Area, Yayasan Sabah's Forest Management Area in Sabah, Malaysia Borneo.

The proportion of trees divided into diameter classes showed that control plots have a higher share of total trees (66 %) within the lowest diameter class of 10-19.9 cm compared to restored plots (58 %) (Figure 8). While restored plots have higher proportion of total trees (31 %) compared to control plots (22 %) in the second diameter class of 20-39.9 cm. Mean number of trees per hectare for restored plots were 523 ± 22.4, mean trees per hectare for control plots were 424 ± 45.1. The mean dbh for trees, independent of family were 23.1 ± 0.24 cm for restored plots and 21.8 ± 0.44 cm for control plots. The proportion of dipterocarp trees for diameter classes 10-19.9 cm and 20-39.9 cm for both restored and control plots follow the same pattern as for total trees. The mean dbh for dipterocarps are in general higher than for all trees, regardless of treatment. Mean dbh for control plots were 24.3 ± 0.91 cm and for restored plots 26.4 ± 0.51 cm. Restored plot had an average of 137 ± 14.2 dipterocarp trees ha^{-1} and control plots had 134 ± 20.3 dipterocarps ha^{-1}.

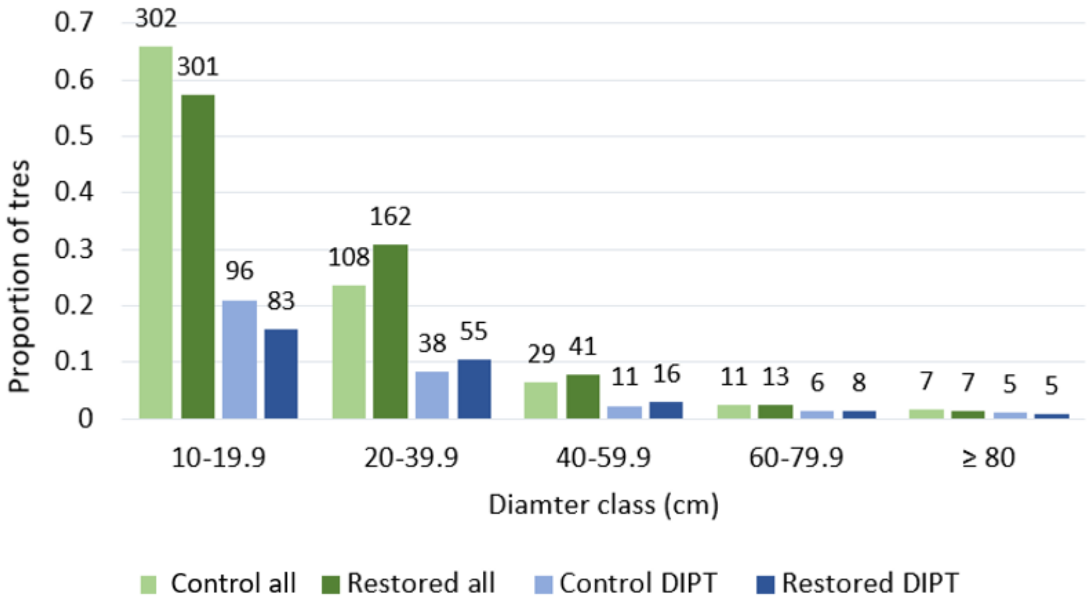

Figure 8. Proportions of trees (y-axis) within five diameter classes; 10-19.9, 20-39.9, 40-59.9, 60-79.9, ≥ 80 (x-axis) from total 54 plots inventoried in INIKEA Project Area, Yayasan Sabah's Forest Management Area in Sabah, Malaysia Borneo. Every diameter class has four coloured bars, control plots with all trees included (light green), restored plots with all trees (dark green), control with only dipterocarp tree species (light blue) and restored plots with only dipterocarp species (dark blue). Average number of trees ha^{-1} for treatments and diameter class is displayed above each bar. Control plots are represented by 14 plots and restored by 40.

Dipterocarp trees represent 34.2 % of the total number of trees for control and 31.6 % for restored plots. However, dipterocarps were overrepresented in the larger dbh-classes and represent more than 56 % of all trees in dbh-class 60-79.9 for both restore and control plots. In the largest dbh-class, 70 % of trees were dipterocarps in control plots and 63 % in restored (Figure 9).

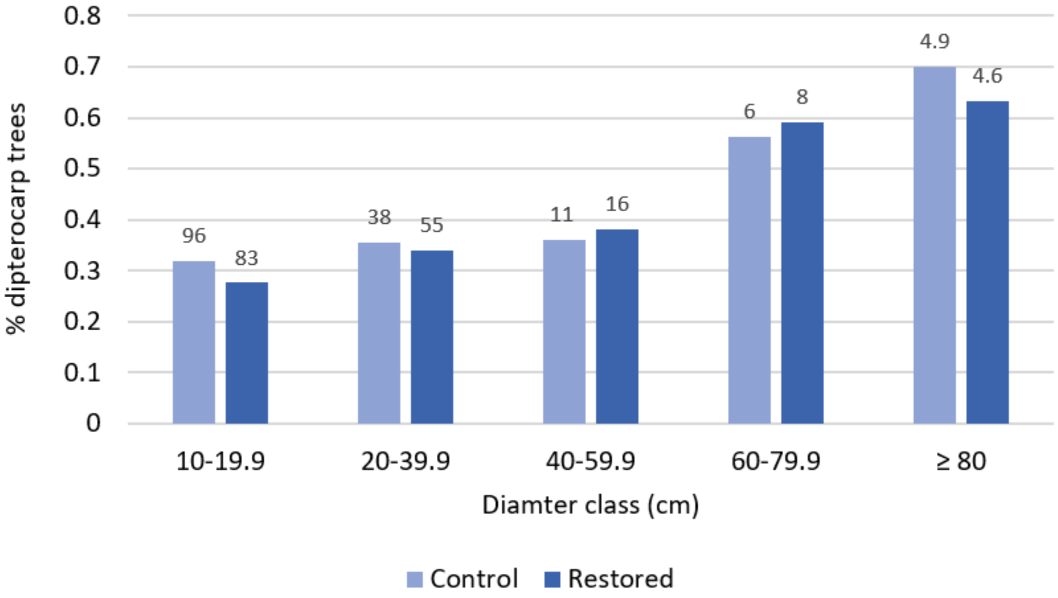

Figure 9. Proportions of dipterocarp trees of all trees (y-axis) within a specific diameter class for control and restored plots. On the x-axis five diameter classes: 10-19.9, 20-39.9, 40-59.9, 60-79.9, ≥ 80 are presented. In total 54 plots where 14 are control and 40 restored plots, inventoried within the INIKEA Project Area, Yayasan Sabah's Forest Management Area in Sabah, Malaysia Borneo. Above every bar is the number of total dipterocarp trees represented within the corresponding diameter class and treatment. Restored plots are represented by dark blue bars and control plots by light blue.

4. Discussion

In my study, I explored how degraded tropical forests in Sabah, Malaysia recovered over time with or without active help such as planting and removal of competing vegetation. I found that degraded tropical forests plots that has been actively restored had in average 41.8 Mg C ha^{-1} more AGC compared to naturally regenerated control plots (p=0.015; Fig 7 A & B). Both mean diameter of trees and trees per hectare was higher for restored plots compared to the control plots (Fig 8). No increased effect on AGC for restored plots for late successional dipterocarp species was found for any of the tests (Fig 7B & 6B). The data also showed that actively restored plots had fewer cases of plots with very low AGC compared to control plots (Fig 6A). The analysis of AGC in relation to year since restoration, with control plots assigned year 0 since restoration, also show a positive increasing trend of 1.91 Mg C ha^{-1} (p=0.005; Fig 6 A & B). These results support previous findings from Philipson et al. (2020) which showed that degraded tropical forests that had undergone active restoration treatments recover faster (4.4 Mg C ha^{-1}) than natural regenerating forests (2.9 Mg C ha^{-1}). Wheeler et al. (2016) stated similar results but also concluded that initial recovery of AGC after active restoration might be slow but increases over time. Furthermore, Schwarts et al. (2013) showed that tending natural established seedlings with liberation as well as enrichment planting with maintenance resulted in higher initial growth rates and survival. However, a meta-study by Coruzerilles et al. (2017) challenged these results and claimed that natural regeneration is more successful in achieving both biodiversity and biomass goals.

The lowest AGC for restored plots (n=40) were 63.3 Mg C ha^{-1} while three control plots (n=19) had AGC below 50 Mg C ha^{-1}, despite being represented by half as many plots. This indicates that some natural regenerating forests have had a hard time recovering on their own, even 40 years after last severe disturbance, control plots with as low AGC as 19.5 Mg C ha^{-1} was observed (Table 1). This could be due to the extensive ground vegetation shading young seedlings while also preventing seeds to reach the forest floor in combination with heavily disturbed and compacted soils. At control sites with very low AGC the local rangers and I experienced a dense wall of fast-growing climbers and weedy vegetation, sometimes three meters high, leaving the soil surface in complete darkness.

Extensive growth of climbers, lianas and other vegetation has been shown in previous studies to hinder regeneration of trees (Pinard et al. 1996; Schnitzer et al. 2000). Evaluation of the datasets showed that planted seedlings in actively restored plots had a minor impact on AGC accumulation, contributing only 1.5 % (1.84 Mg C ha^{-1}) of current AGC. In the restored plots, out of total 4296 measured trees (523 trees ha^{-1}), only 208 were planted (30.2 trees ha^{-1}). The planted trees had a mean dbh of 17.4 ± 0.55 cm and would not have contributed much to the increased AGC that was observed between control and restored plots. Instead, it is more likely that the maintenance practices preformed regularly during a ten-year period after planting in phase 1-3 resulted in increased AGC in already established trees (Schnitzer et al. 2000; Schwartz et al. 2013). My data presents a picture that fit well with the hypothesis, that severely degraded forests can have difficulties to naturally regenerate. However, in several cases it seems that natural regeneration can be sufficient for forests to start recovering and accumulating biomass since the majority of control plots had an AGC of 100 Mg C ha^{-1} or above. The slow ingrowth of planted trees is likely the explanation why I did not detect any difference in AGC for dipterocarp species between restored and control plots. The forests at INIKEA project area are still in early succession of regeneration and only a small percentage of the planted trees, mainly represented by dipterocarps, have grown into measurement size (≥10cm dbh). Dipterocarps species build the canopy in late successional rainforests and have higher WD than pioneer species (Ashton 1982, 1988; Appanah & Turnbull 1998) which are dominant in the current stage at INIKEA. Furthermore, dipterocarps generally grow slower in the beginning and has an exponential increase in growth after 20-30 cm in dbh when they reach the canopy (Clark & Clark 1999). Therefore, it is likely that many of the positive effects on biodiversity and carbon sequestration from enrichment planting is yet to come.

The results in my study could have a bias if the two study areas were not similarly degraded at the start in 1998. According to David Alloysius, the manager at INIKEA and rangers who was present during the startup of the INIKEA project, it is revealed that areas most in need of restoration were prioritised first. The most degraded areas in 1998 were phases 1 and 2 due to their accessibility by roads and a more favourable topography for logging operations. Phase 4 however, were more heterogeneous in both topography and level of degradation and was farther from the main road. Therefore, it is possible that in the beginning of 1998, phases 1 and 2 were more or equally degraded as phase 4. Some control plots located in phase 4 had carbon stocks that were high above the average AGC for degraded forests, reaching values greater than are common for pristine, unlogged forests. Five control plots with degradation level 1 had an AGC of 170 Mg C ha^{-1} or more and one plot reaching 400.7 Mg C ha^{-1}. Such forests would not have been restored with enrichment planting but left for natural regeneration with or without a onetime liberation treatment including climber-cutting and selective girdling of pioneer species. This type of treatment was uncommon in phase 1 but increased towards phase 3 that had more areas with less degradation. Exceptionally high values of AGC in some control plots, combined with randomized selection, as opposed to phase 1 which was originally restored due to its severe degradation made it necessary to exclude some control plots. Control plots that would not have been considered for enrichment planting (i.e. plots that I classified with degradation level of 1; Table 1) with an AGC over 170 Mg C ha^{-1} and an average of 601 trees ha^{-1} (dbh ≥ 10) (Fig. 4 & 5) were therefore excluded. As a comparison, intact and protected forests in Maliau Basin and Danum Valley has a mean AGC of 220 ± 69 Mg C ha^{-1} and 207 ± 71 Mg C ha^{-1} (± SE) (Asner et al. 2018). Asner et al. (2020) estimated average AGC for recovering tropical forests on Borneo, initially logged during 1973 to be 80 Mg C ha^{-1} and a general AGC for logged forest of Sabah varying between 60-140 Mg C ha^{-1}.

If we are to reach ambitious restoration commitments like The Bonn Challenge, which pledge to restore 350 million hectares of deforested and degraded lands by 2030 (IUCN 2021), it is vital to know how and when to use the different tools of restoration at our disposal. Here, both active restoration and natural regeneration have their role. The ability to correctly asses sites were natural regeneration would be successful or where active restoration is needed is of critical importance for future restoration efforts (Chazdon & Guariguata 2016; Crouzeilles et al. 2017, 2020). Natural regeneration is an indispensable tool and has the potential to be applied cost-effectively at large scale globally (Poorter et al. 2016; Lamb 2017; Nunes et al. 2017; Crouzeilles et al. 2020). However, there are many factors regulating the success of restoration by natural regeneration, both ecological and socio-economic factors (Chazdon & Guariguata 2016). Natural regeneration, which typically is a slow process, can be interpreted as ineffective and inappropriate land-use by local communities and stakeholders which might lead to discontent and premature termination of a project (Zahawi et al. 2014). There are also ecological circumstances that obstructs natural regeneration, mainly degraded soils, frequent fires, competing climbers and lianas as well as seed dispersal opportunities (Pinard et al. 1996; Cochrane 2003; Tymen et al. 2016; Crouzeilles et al. 2020). This is also something that I observed during my field inventory. I noted that control plots with least amount of AGC were also the most open, covered in ground vegetation and climbers, lacking canopy and larger trees in the nearby vicinity. In comparison, I noted that naturally regenerated control plots with an existing canopy and a few large trees often had and abundant number of trees in the lower diameter classes (10-19.9 cm) growing in the understory (Figure 4). A partial canopy cover provides a more stable microclimate with lower and more stable temperature, higher humidity and less intensive solar radiation (Holl 1999) which also might help seedling to compete with other fast growing herbaceous vegetation. Pinard et al. (1996) stated that heavily degraded sites, denuded of vegetation after logging operations needed pioneer trees to establish quickly before other vegetation became too extensive. The canopy cover of pioneer species will provide environmental conditions needed for shade tolerant late successional tree species, such as dipterocarps to establish. However, dipterocarps have large gyration-dispersed seeds with a short range of dispersal where 90% of seed land within 10 meters from the parent tree (Smith et al. 2015). This complicates natural establishment in degraded forests since mature dipterocarp species were targeted during logging operations and might be absent in large areas. With a natural dispersal range of 10-100 meter, although slope and topography might increase this distance, it can take decades or centuries for some species to spread over a landscape scale. Active restoration could therefore be a helpful tool to bring back both biodiversity and higher, long-term carbon storage more rapidly via planting of long-lived climax species were seed trees are absent.

The question of ecological restoration will become increasingly more important in coming decades and there is a large need for further studies in this area (Chazdon & Guariguata 2016). Restoration projects will always be limited by time and economy so continuous assessments of active restorations success and cost effectiveness is needed. However, most studies examining cost effectiveness and carbon sequestration are, in a forest development perspective very short-term and mostly capture an early timeframe in development. More studies are needed to investigate how active and passive restoration attempt develop in the long term. The forest at INIKEA, which were initially restored for more than 20 years ago, are still young and there are many unanswered questions how they will develop in the future. In this study I only consider the carbon sequestration, but equally important is the biodiversity of the ecosystem. In this context time is even more relevant since it takes even longer for biodiversity to reach climax levels than it does for carbon stocks (Wheeler et al. 2016).

The INIKEA project has so far restored approximately 11,000 hectares of degraded forest with enrichment planting followed by maintenance. My results show that actively restored plots have significantly higher AGC than naturally restored plots (41.8 Mg C ha^{-1} ;p=0.015). Via conversation with David Alloysius, the manager of INIKEA, I estimated a cost ha^{-1} for enrichment planting around \$3150 (USD), converting carbon into carbon dioxide gives a price of \$20.5 per ton CO_2 additionally sequestered by active restoration after 23 years. This price is quite high compared to the average carbon credit price of \$4.1 per ton CO_2, although the price range vastly from \$0.45 to \$45 (Hamilton et al. 2009, n.d.). However, Philipson et al. (2020) argues that higher carbon prices of \$40 to \$80 per ton CO_2 would be needed to fulfil the 2016 Paris climate agreement and that prices today are too low. Furthermore, this is the state 23 after active restoration and this number will probably continue to increase more rapidly in the future when planted climax tree species with higher WD experience an increased growth as they reach the canopy.

It is however likely that initial increased carbon accumulation of forest within the INIKEA project could have been reached for a lower cost. About 5 million seedlings have been planted but many of these have not yet grown into meaningful size to impact carbon sequestration, implying maintenance to be the biggest contributor of carbon sequestration at current state. This suggests that a continuous liberation treatment over a few years by removal of weeds, lianas and climbers might have been sufficient in several areas since large seed trees of dipterocarps are present and growing (Figure 4 & 5). If enrichment planting and maintenance was only utilized on the most degraded plots, with conditions as described before, a restoration with lower cost per hectare could be achieved. This would maximize the initial carbon sequestration for a lower cost but would probably in result in lower long-term carbon sequestration and biodiversity. Future studies are needed to investigate the trade-off between cost effectiveness and carbon sequestration for passive and active restoration in similar degradation conditions to ensure that desired goals are met.

My result show that active restoration increased carbon sequestration with about 2 ton annually during 23 years after enrichment line-planting with 10 years maintenance. However, the positive effect of active restoration seen so far mainly comes from the continuous maintenance which greatly influence the growth of already well-established trees. The planted trees of largely late successional dipterocarp species have yet to make an impact on the carbon sequestration. These species have slower initial growth, higher WD and are a major contribution to the carbon storage in later stages of forest development (Ashton 1988; Clark & Clark 1999). In comparison, planted trees contribute an average of 0.09 ton carbon annually the first 23 years after restoration. Although, for every year that passes, this number will continue to increase as more and more planted trees reach the threshold dbh of ≥ 10 cm. As planted trees continue to grow and reach the canopy, an even greater growth is expected. This will in turn affect the balance of cost-effectiveness and carbon sequestration of active restorations. It is therefore of great importance that projects like INIKEA remain active so continuous measurements and studies can be conducted to increase our understanding of long-term development of restored forests that is lacking today.

References

Appanah, S. & Turnbull, J.M. (1998). *A Review of Dipterocarps: Taxonomy, Ecology, and Silviculture*. CIFOR.

Ashton, P.S. (1982). Dipterocarpaceae. *Dipterocarpaceae.*, 9 (2), 237–552

Ashton, P.S. (1988). Dipterocarp Biology as a Window to the Understanding of Tropical Forest Structure. *Annual Review of Ecology and Systematics*, 19, 347–370

Asner, G.P., Brodrick, P.G., Philipson, C., Vaughn, N.R., Martin, R.E., Knapp, D.E., Heckler, J., Evans, L.J., Jucker, T., Goossens, B., Stark, D.J., Reynolds, G., Ong, R., Renneboog, N., Kugan, F. & Coomes, D.A. (2018). Mapped aboveground carbon stocks to advance forest conservation and recovery in Malaysian Borneo. *Biological Conservation*, 217, 289–310. https://doi.org/10.1016/j.biocon.2017.10.020

Asner, G.P., Knapp, D.E., Broadbent, E.N., Oliveira, P.J.C., Keller, M. & Silva, J.N. (2005). Selective Logging in the Brazilian Amazon. *Science*, 310 (5747), 480–482. https://doi.org/10.1126/science.1118051

Bryan, J.E., Shearman, P.L., Asner, G.P., Knapp, D.E., Aoro, G. & Lokes, B. (2013). Extreme Differences in Forest Degradation in Borneo: Comparing Practices in Sarawak, Sabah, and Brunei. *PLOS ONE*, 8 (7), e69679. https://doi.org/10.1371/journal.pone.0069679

Chave, J., Coomes, D., Jansen, S., Lewis, S.L., Swenson, N.G. & Zanne, A.E. (2009). Towards a worldwide wood economics spectrum. *Ecology Letters*, 12 (4), 351–366. https://doi.org/10.1111/j.1461-0248.2009.01285.x

Chazdon, R.L. & Guariguata, M.R. (2016). Natural regeneration as a tool for large-scale forest restoration in the tropics: prospects and challenges. *Biotropica*, 48 (6), 716–730. https://doi.org/10.1111/btp.12381

Chokkalingam, U. & De Jong, W. (2001). Secondary forest: a working definition and typology. *The International Forestry Review*, 3 (1), 19–26

Clark, D.A. & Clark, D.B. (1999). Assessing the Growth of Tropical Rain Forest Trees: Issues for Forest Modeling and Management. *Ecological Applications*, 9 (3), 981–997. https://doi.org/10.1890/1051-0761(1999)009[0981:ATGOTR]2.0.CO;2

Cochrane, M.A. (2003). Fire science for rainforests. *Nature*, 421 (6926), 913–919. https://doi.org/10.1038/nature01437

Crouzeilles, R., Beyer, H.L., Monteiro, L.M., Feltran-Barbieri, R., Pessôa, A.C.M., Barros, F.S.M., Lindenmayer, D.B., Lino, E.D.S.M., Grelle, C.E.V., Chazdon, R.L., Matsumoto, M., Rosa, M., Latawiec, A.E. & Strassburg, B.B.N. (2020). Achieving cost-effective landscape-scale forest restoration through targeted natural regeneration. *Conservation Letters*, 13 (3), e12709. https://doi.org/10.1111/conl.12709

Crouzeilles, R., Ferreira, M.S., Chazdon, R.L., Lindenmayer, D.B., Sansevero, J.B.B., Monteiro, L., Iribarrem, A., Latawiec, A.E. & Strassburg, B.B.N. (2017). Ecological restoration success is higher for natural regeneration than for active restoration in tropical forests. *Science Advances*, 3 (11), e1701345. https://doi.org/10.1126/sciadv.1701345

Dahlgren, R.A., Macías, F., Arbestain, M.C., Chesworth, W., Robarge, W.P. & Macías, F. (2008). Acrisols. In: Chesworth, W. (ed.) *Encyclopedia of Soil Science*. Dordrecht: Springer Netherlands. 22–24. https://doi.org/10.1007/978-1-4020-3995-9_11

Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T. & Tanabe, K. (2006). 2006 IPCC Guidelines for National Greenhouse Gas Inventories. https://www.osti.gov/etdeweb/biblio/20880391 [2023-01-22]

FAO and UNEP. (2020). *The State of the World's Forests 2020. Forests, biodiversity and people.* Rome. https://doi.org/10.4060/ca8642en

Fitzherbert, E.B., Struebig, M.J., Morel, A., Danielsen, F., Brühl, C.A., Donald, P.F. & Phalan, B. (2008). How will oil palm expansion affect biodiversity? *Trends in Ecology & Evolution*, 23 (10), 538–545. https://doi.org/10.1016/j.tree.2008.06.012

Galante, M.V., Pinard, M.A. & Mencuccini, M. (2018). Estimating carbon avoided from the implementation of reduced-impact logging in Sabah, Malaysia. *International Forestry Review*, 20 (1), 58–78. https://doi.org/10.1505/146554818822824192

Gaveau, D.L.A., Sheil, D., Husnayaen, Salim, M.A., Arjasakusuma, S., Ancrenaz, M., Pacheco, P. & Meijaard, E. (2016). Rapid conversions and avoided deforestation: examining four decades of industrial plantation expansion in Borneo. *Scientific Reports*, 6 (1), 32017. https://doi.org/10.1038/srep32017

Gibbs, H.K., Brown, S., Niles, J.O. & Foley, J.A. (2007). Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Research Letters*, 2 (4), 045023. https://doi.org/10.1088/1748-9326/2/4/045023

Gustafsson, M., Gustafsson, L., Alloysius, D., Falck, J., Yap, S., Karlsson, A. & Ilstedt, U. (2016). Life history traits predict the response to increased light among 33 tropical rainforest tree species. *Forest Ecology and Management*, 362, 20–28. https://doi.org/10.1016/j.foreco.2015.11.017

Hamilton, K., Bayon, R., Turner, G. & Higgins, D. (n.d.). State of the Voluntary Carbon Market 2007 Picking Up Steam 17th July 2007.

Hamilton, K., Sjardin, M., Shapiro, A. & Marcello, T. (2009). Fortifying the foundation: state of the voluntary carbon markets 2009. *Fortifying the foundation: state of the voluntary carbon markets 2009.*,. https://www.cabdirect.org/cabdirect/abstract/20093167352 [2023-03-01]

Holl, K.D. (1999). Factors Limiting Tropical Rain Forest Regeneration in Abandoned Pasture: Seed Rain, Seed Germination, Microclimate, and Soil1. *Biotropica*, 31 (2), 229–242. https://doi.org/10.1111/j.1744-7429.1999.tb00135.x

Hoscilo, A., Page, S.E., Tansey, K.J., Rieley, J.O., Hoscilo, A., Page, S.E., Tansey, K.J. & Rieley, J.O. (2011). Effect of repeated fires on land-cover change on peatland in southern Central Kalimantan, Indonesia, from 1973 to 2005. *International Journal of Wildland Fire*, 20 (4), 578–588. https://doi.org/10.1071/WF10029

IUCN (2021). The Bonn Challenge https://www.iucn.org/theme/forests/our-work/forest-landscape-restoration/bonn-challenge / [2021-11-20]

Jensen, J. (2020). *Above- and belowground carbon stocks and effects of enrichment planting in a tropical secondary lowland dipterocarp rainforest*. [Avancerad nivå, A2E]. https://stud.epsilon.slu.se/15287/ [2022-11-22]

Jomo, K.S., Chang, Y.T. & Khoo, K.J. (2004). Deforesting Malaysia: the political economy and social ecology of agricultural expansion and commercial logging. *Deforesting Malaysia: the political economy and social ecology of agricultural expansion and commercial logging.*,. https://www.cabdirect.org/cabdirect/abstract/20056702120 [2022-12-19]

Lamb, D. (2017). *Large-scale Forest Restoration*. London: Routledge. https://doi.org/10.4324/9780203071649

Laneng, L.A., Nakamura, F., Tachiki, Y. & Vairappan, C.S. (2021). Camera-trapping assessment of terrestrial mammals and birds in rehabilitated forest in INIKEA

Project Area, Sabah, Malaysian Borneo. *Landscape and Ecological Engineering*, 17 (2), 135–146. https://doi.org/10.1007/s11355-020-00442-7

MacKinnon, K., Hatta, G., Mangalik, A. & Halim, H. (1996). *The Ecology of Kalimantan*. Periplus Editions.

Malhi, Y. & Grace, J. (2000). Tropical forests and atmospheric carbon dioxide. *Trends in Ecology & Evolution*, 15 (8), 332–337. https://doi.org/10.1016/S0169-5347(00)01906-6

Malhi, Y., Phillips, O.L., Malhi, Y. & Wright, J. (2004). Spatial patterns and recent trends in the climate of tropical rainforest regions. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*, 359 (1443), 311–329. https://doi.org/10.1098/rstb.2003.1433

Malmer, A. & Grip, H. (1990). Soil disturbance and loss of infiltrability caused by mechanized and manual extraction of tropical rainforest in Sabah, Malaysia. *Forest Ecology and Management*, 38 (1), 1–12. https://doi.org/10.1016/0378-1127(90)90081-L

Newbery, D.M., Clutton–Brock, T.H., Prance, G.T., Walsh, R.P.D. & Newbery, D.M. (1999). The ecoclimatology of Danum, Sabah, in the context of the world's rainforest regions, with particular reference to dry periods and their impact. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*, 354 (1391), 1869–1883. https://doi.org/10.1098/rstb.1999.0528

Nunes, F.S.M., Soares-Filho, B.S., Rajão, R. & Merry, F. (2017). Enabling large-scale forest restoration in Minas Gerais state, Brazil. *Environmental Research Letters*, 12 (4), 044022. https://doi.org/10.1088/1748-9326/aa6658

Pan, Y., Birdsey, R.A., Fang, J., Houghton, R., Kauppi, P.E., Kurz, W.A., Phillips, O.L., Shvidenko, A., Lewis, S.L., Canadell, J.G., Ciais, P., Jackson, R.B., Pacala, S.W., McGuire, A.D., Piao, S., Rautiainen, A., Sitch, S. & Hayes, D. (2011). A Large and Persistent Carbon Sink in the World's Forests. *Science*, 333 (6045), 988–993. https://doi.org/10.1126/science.1201609

Philipson, C.D., Cutler, M.E.J., Brodrick, P.G., Asner, G.P., Boyd, D.S., Moura Costa, P., Fiddes, J., Foody, G.M., van der Heijden, G.M.F., Ledo, A., Lincoln, P.R., Margrove, J.A., Martin, R.E., Milne, S., Pinard, M.A., Reynolds, G., Snoep, M., Tangki, H., Sau Wai, Y., Wheeler, C.E. & Burslem, D.F.R.P. (2020). Active restoration accelerates the carbon recovery of human-modified tropical forests. *Science*, 369 (6505), 838–841. https://doi.org/10.1126/science.aay4490

Pinard, M., Howlett, B. & Davidson, D. (1996). Site Conditions Limit Pioneer Tree Recruitment After Logging of Dipterocarp Forests in Sabah, Malaysia. *Biotropica*, 28 (1), 2–12. https://doi.org/10.2307/2388766

Poorter, L., Bongers, F., Aide, T.M., Almeyda Zambrano, A.M., Balvanera, P., Becknell, J.M., Boukili, V., Brancalion, P.H.S., Broadbent, E.N., Chazdon, R.L., Craven, D., de Almeida-Cortez, J.S., Cabral, G.A.L., de Jong, B.H.J., Denslow, J.S., Dent, D.H., DeWalt, S.J., Dupuy, J.M., Durán, S.M., Espírito-Santo, M.M., Fandino, M.C., César, R.G., Hall, J.S., Hernandez-Stefanoni, J.L., Jakovac, C.C., Junqueira, A.B., Kennard, D., Letcher, S.G., Licona, J.-C., Lohbeck, M., Marín-Spiotta, E., Martínez-Ramos, M., Massoca, P., Meave, J.A., Mesquita, R., Mora, F., Muñoz, R., Muscarella, R., Nunes, Y.R.F., Ochoa-Gaona, S., de Oliveira, A.A., Orihuela-Belmonte, E., Peña-Claros, M., Pérez-García, E.A., Piotto, D., Powers, J.S., Rodríguez-Velázquez, J., Romero-Pérez, I.E., Ruíz, J., Saldarriaga, J.G., Sanchez-Azofeifa, A., Schwartz, N.B., Steininger, M.K., Swenson, N.G., Toledo, M., Uriarte, M., van Breugel, M., van der Wal, H., Veloso, M.D.M., Vester, H.F.M., Vicentini, A., Vieira, I.C.G., Bentos, T.V., Williamson, G.B. & Rozendaal, D.M.A. (2016). Biomass resilience of Neotropical secondary forests. *Nature*, 530 (7589), 211–214. https://doi.org/10.1038/nature16512

Romell, E. (2007). *Artificial canopy gaps and the establishment of planted dipterocarp seedlings in Macaranga spp. dominated secondary tropical rain forests of Sabah, Borneo*. [Licentiatavhandling]. https://pub.epsilon.slu.se/1323/ [2023-01-05]

Schnitzer, S.A., Dalling, J.W. & Carson, W.P. (2000). The impact of lianas on tree regeneration in tropical forest canopy gaps: evidence for an alternative pathway of gap-phase regeneration. *Journal of Ecology*, 88 (4), 655–666. https://doi.org/10.1046/j.1365-2745.2000.00489.x

Schwartz, G., Lopes, J.C.A., Mohren, G.M.J. & Peña-Claros, M. (2013). Post-harvesting silvicultural treatments in logging gaps: A comparison between enrichment planting and tending of natural regeneration. *Forest Ecology and Management*, 293, 57–64. https://doi.org/10.1016/j.foreco.2012.12.040

Siegert, F., Ruecker, G., Hinrichs, A. & Hoffmann, A.A. (2001). Increased damage from fires in logged forests during droughts caused by El Niño. *Nature*, 414 (6862), 437–440. https://doi.org/10.1038/35106547

Smith, J.R., Bagchi, R., Ellens, J., Kettle, C.J., Burslem, D.F.R.P., Maycock, C.R., Khoo, E. & Ghazoul, J. (2015). Predicting dispersal of auto-gyrating fruit in tropical trees: a case study from the Dipterocarpaceae. *Ecology and Evolution*, 5 (9), 1794–1801. https://doi.org/10.1002/ece3.1469

The jamovi project (2022). *jamovi* (Version 2.3) [Computer Software]. Retrieved from https://www.jamovi.org

Tymen, B., Réjou-Méchain, M., Dalling, J.W., Fauset, S., Feldpausch, T.R., Norden, N., Phillips, O.L., Turner, B.L., Viers, J. & Chave, J. (2016). Evidence for arrested succession in a liana-infested Amazonian forest. *Journal of Ecology*, 104 (1), 149–159. https://doi.org/10.1111/1365-2745.12504

UN (2021). Forest Restoration: A Path to Recovery and Well-Being. https://www.un.org/en/un-chronicle/forest-restoration-path-recovery-and-wellbeing-0 / [2021-03-21]

USDA (2014). Keys to soil taxonomy. Soil Survey Staff. van der Werf, G.R., Morton, D.C., DeFries, R.S., Olivier, J.G.J., Kasibhatla, P.S., Jackson, R.B., Collatz, G.J. & Randerson, J.T. (2009). CO2 emissions from forest loss. Nature Geoscience, 2, p. 737.

van der Werf, G.R., Morton, D.C., DeFries, R.S., Olivier, J.G.J., Kasibhatla, P.S., Jackson, R.B., Collatz, G.J. & Randerson, J.T. (2009). CO2 emissions from forest loss. *Nature Geoscience*, 2 (11), 737–738. https://doi.org/10.1038/ngeo671

Wheeler, C.E., Omeja, P.A., Chapman, C.A., Glipin, M., Tumwesigye, C. & Lewis, S.L. (2016). Carbon sequestration and biodiversity following 18years of active tropical forest restoration. *Forest Ecology and Management*, 373, 44–55. https://doi.org/10.1016/j.foreco.2016.04.025

WWF (2020). Fakta om tropikskogar. https://www.wwf.se/skog/regnskog/fakta-om tropikskogar / [2021-11-16]

Zahawi, R.A., Reid, J.L. & Holl, K.D. (2014). Hidden Costs of Passive Restoration. *Restoration Ecology*, 22 (3), 284–287. https://doi.org/10.1111/rec.12098

Zanne AE, Lopez-Gonzalez G, Coomes DA, Ilic J, Jansen S, Lewis SL, Miller RB, Swenson NG, Wiemann MC, Chave J (2009) Data from: Towards a worldwide wood economics spectrum. Dryad Digital Repository. doi:10.5061/dryad.234

Zhu, J. (2002). [A review on fundamental studies of secondary forest management]. *Ying yong sheng tai xue bao = The journal of applied ecology*, 13 (12), 1689–1694

Popular science summary

The need and commitment for restoring large areas of degraded ecosystem are becoming greater for every year as humans continue to harvest natural resources in unsustainable ways and release greenhouse gases into the atmosphere. Restoration of degraded forests can contribute to many positive effects of ecosystem services such as increased carbon sequestration and biodiversity. There are two main opinions on how we should approach the question of restoration. One idea is that actively restoration, where humans interfere and help nature to recover faster is the best alternative. The other opinion is that humans should take a more passive role and that nature itself does a better job if left to its own devices and allowed to naturally regenerate. In my study, I compared active and passive restoration in degraded tropical forests within the INIKEA project in Sabah, Malaysia. I used previously collected data from 2017 together with data collected by myself during the fall of 2022. The forest areas inventoried have been treated by either enrichment line planting followed by 10 years of maintenance or nothing, and these untreated plots will act as a control plots and baseline for natural regeneration.

My results show that actively restored forests after 23 years had an increased carbon sequestration of 41.8 Mg C ha^{-1} ($p=0.015$) compared to naturally regenerated forests. This is an annual increase of almost two ton carbon per hectare for line planting treatment with maintenance. However, after closer examination of the data it was discovered that planted trees contributed only about 1.5 % of carbon or 1.84 Mg C ha^{-1} in total. My conclusion is that continuous maintenance where lianas, climbers and other competing vegetation is removed during active restoration had led to an increased growth of already established and naturally regenerated trees. Furthermore, a few control plots that have been naturally regeneration for about 50 years since last severe disturbance still is almost completely bare from trees with AGC lower than 50 Mg C ha^{-1}. This indicates that sometimes, severe degradation can hinder natural regeneration for decades. A similar delay in regeneration could however not be found for any of the actively restored plots.

During enrichment line planting, mainly late successional dipterocarp tree species were used, which are adapted to shaded condition as seedlings and form the tall canopy in mature tropical forests of Borneo. This initially slow growth pattern is probably one of the reasons we do not see more carbon addition in planted trees. My believe is that the planted trees, of which about 30 trees per hectare have grown into diameter size dbh $\geq$ 10 cm, will contribute more to total carbon sequestration in coming years as they reach the canopy. We know too little about the long-term development of actively restored forest and need to continue projects like the INIKEA and conduct more studies going forward.

Acknowledgements

Firstly, I want to thank my supervisors Ulrik Ilstedt and Daniel Lussetti who has supported me during all parts of this thesis; field work, statistical analyses and report writing. As well as the people from SLU that accompanied me on my first two weeks in Sabah, Malaysia and helped me get started with the field work as well as introduced me to the rest of the team in Luasong. Lastly, I want to thank David Alloysius, Vita Juin and all the rangers and workers that aided me during my time in Luasong, this thesis would not have been possible without your knowledge, hard work, and support, thank you!

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