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Biomass and Carbon Loss from Selective Logging and Associated Collateral Damage in Eastern Amazonia, Brazil

Malin S. Aannestad
Master of Science in Ecology

Preface

This thesis is submitted as the final part of my master's degree in Ecology at the Norwegian University of Life Sciences.

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Thank you all,
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Abstract

Tropical forests are pivotal in global climate regulation and biodiversity conservation. Even though the value of tropical forests has been shown multiple times, they are still experiencing multiple pressures, threatening their existence. Timber industry is one of them, with over half of the world's tropical forests having already been logged. Production for timber has increased in recent years, and in 2017 and 2018, 29 million m^3 of roundwood was extracted from the Brazil alone. There have been many studies on the effect of selective logging, but few on collateral damage and the associated biomass and carbon loss. I sought to close this knowledge gap by investigating the direct and collateral biomass and carbon loss within a logging concession in Caxiuanã National Forest, Brazil.

I found that each extracted tree in the concession damaged on average 18.5 trees in the residual stand. The damaged trees were mostly of smaller DBH (diameter at breast height) classes. Loss of more than 2/3 of the crown was found for 33.5% of the residual trees, however most of the damage to the crown and bole was found to be bark scrapes without any damage to the cambial tissue, while the roots of the tree were most often uninjured. From the extracted trees and broken parts of residual trees I calculated a mean biomass and carbon loss per extracted tree of 9.11 ± 0.20 Mg and 4.29 ± 0.88 Mg, respectively. To see the impact of this result, I scaled this up for the 4 552 trees to be extracted within the concession. This led to an estimate of 41 468.7 Mg biomass and 19 531 ± 8542 carbon lost during operations. My results show that even best-case scenarios of logging with the Brazilian Amazon are highly damaging, and further research needs to focus on methods to reduce collateral damage and associated biomass and carbon loss.

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Introduction

Tropical forests play a pivotal role in global climate regulation and biodiversity conservation (Foley et al., 2007; Gibson et al., 2011). They store more carbon as woody biomass per unit area than any other vegetation type (Pan et al., 2011). Through the absorption of atmospheric carbon, tropical forests help mitigate climate change and capture as much as one third of the annual global carbon emissions (Lewis, 2006; Malhi & Grace, 2000). They also contain most of the global biodiversity. For example, although the Amazon rainforest comprises the largest tropical forest in the world, it only accounts for approximately 3.6% of the terrestrial global surface but harbours an estimated 10% of the world's known species (Maretti, 2014). Yet, tropical forests experience multiple pressures threatening their very existence and the ecosystem services they provide.

Timber is a big global industry and logging for timber is an extensive form of land-use change in tropical forest regions. About half of the world's tropical forests has already been logged (Asner et al., 2009), and selective logging is expanding in the remaining global tropical forest as market demand for roundwood has increased in recent years (FAO, 2020). For example, in 2009 more than 14 million m^3 of roundwood was extracted from the legal Amazon (Pereira et al., 2010), but the production of total roundwood had doubled to approximately 29 million m^3 in 2017 and 2018 (ITTO, 2019). Human population growth and accompanying demand makes it unlikely that logging activities will be curbed in the near future.

In the Brazilian Amazon, selective logging has been most intense in the states of Mato Grosso and Pará (Asner et al., 2005), where the total area affected by logging is equal to the amount of deforested area. These states are part of the deforestation arc where cattle ranching and soya bean drive the deforestation frontier and cause habitat fragmentation and additional associated forest degradation (Asner et al., 2006; Pearson et al., 2014), such as wildfires (Cochrane, 2001). However, logging activities generally happen first and often paves the way for total deforestation as logged forests are considered to have little or no economic value or value for biodiversity conservation (Asner et al., 2006; Dunn, 2004). Similar trends are found in other tropical areas, such as south-east Asia (Edwards et al., 2014).

Human disturbance, such as infrastructure development, hunting and logging, in primary forests is a driver of biodiversity loss and disturbed forests are not able to support the same level of species richness as primary forests (Gibson et al., 2011; Rozendaal et al., 2019). However, many studies show that logged forest retain most biodiversity also present in unlogged forest, including large vertebrates (Carvalho et al., 2020) , plants and invertebrates, (Putz et al., 2012), and many red-listed species (Edwards et al., 2011; Edwards et al., 2014). Given the vast areas of tropical forests affected by logging, these may therefore be important in the global conservation agenda.

In addition, logging practices have been improved in recent years, moving on from very damaging conventional logging towards reduced impact logging (RIL) techniques. This is significant, as above-ground biomass and carbon losses from selective logging is a direct effect of the harvest intensity and the level of care at which the harvest is performed. In turn, this affects the level of collateral damage to the residual stand (Piponiot et al., 2016). RIL has been found to damage fewer trees and reduces the amount of severely damaged trees compared to conventional logging practises, thereby retaining over 20% more biomass and reducing the amount of wood waste left behind (Pinard & Putz, 1996; Putz et al., 2012; Sasaki et al., 2016). This is an extremely important development in the global effort to preserve carbon stocks in tropical forests to combat climate change.

However, although many studies have investigated the effects of logging on forest structure (De Carvalho et al., 2017; Jackson et al., 2002; Rutishauser et al., 2016), forest gaps and edges (Asner et al., 2004; Rangel Pinagé et al., 2019; Ruslandi et al., 2012), fire susceptibility (Cochrane, 2001; Cochrane & Laurance, 2008), and biodiversity recovery following logging (De Carvalho et al., 2017; Gaui et al., 2019; Richardson & Peres, 2016), few studies have investigated the direct and indirect (collateral) loss of above-ground biomass and carbon stocks caused by logging damage (but see Jackson et al., 2002; Mazzei et al., 2010; West et al., 2014). As selective logging is a major player in land-use change throughout the tropics, the effect that this has on carbon retention and release needs to be better understood. In this thesis, I try to redress this knowledge gap by investigating the direct and collateral biomass and carbon loss from a logging operation in the eastern Amazon. More specifically, I use species-specific wood density estimates to calculate the direct loss of carbon stocks from felled trees. I also document the type of damage inflicted on residual trees from selective tree felling and quantify the additional biomass and carbon loss associated with this collateral damage. I discuss my findings in the context of existing information on logging and the connection collateral damage has on carbon emissions in the tropics.

Methods

Study site

The work was carried out from September to October 2019 in the Benevides Madeiras logging concession located within the Caxiuanã National Forest in Pará state, eastern Amazonia, Brazil (Figure 1). The area is characterized as lowland *terra firme* forest. The forest has a mean annual rainfall of 2272 (± 193) mm, with a clear seasonality in precipitation (Fisher et al., 2006). Most of the rains occur in the wet season during December-May (~76% of rainfall), with June-November being relatively dry (~24% of rainfall; (Muniz, 2017; Oliveira et al., 2008). Logging operations are active during the dry season but are suspended during the wet season.

The national forest was recently opened for sustainable use of natural resources, leading to the establishment of the forest concession. The concession consists of three forest management units (UMFs), and I worked with Benevides in one of these, UMF II (Figure 3). Each UMF is divided into annual production units (UPAs), which are further divided into working units (UTs) of 1000 m * 1000 m (100 ha). The concession is led by the principles of Projeta Aflora, a forest concession system initiated by Benevides to show their commitment to ensure both sustainable use of the forest resources and to integrating the local communities in their activities.

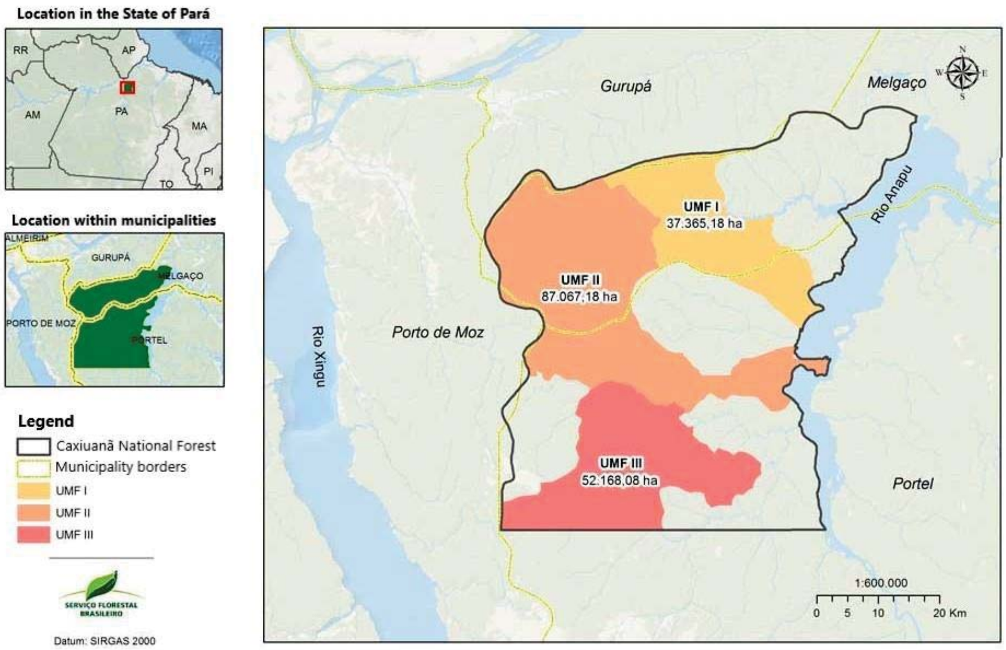

Figure 1. Map showing Caxiuanã National Forest in the state of Pará (PA; two smaller maps left side) and the location of the logging concession within the National Forest with the three production units (UMF I-III). From the Brazilian Forest Service (Serviço Florestal Brasileiro).

Roads are constructed to gain access to the working units (UTs) within the UMFs. The road system consists of a main (primary) road going vertically all the way through the UMF, with connected secondary roads constructed horizontally through the UMF. Stockyards (log storage sites) are constructed onto the secondary roads to store wood from the exploration in the nearby area (Figure 2). Logs from the stockyards are transported using trucks to a central yard at the entrance of the concession (Figure 3), where they will be further loaded onto timber cargo ships and transported to a sawmill.

A total of 3482 trees from 19 different species will be harvested in UMF II. Species names and number of individuals to be harvested is presented in Appendix 1.

Figure 3. Stockyard where logs are stored within the working units (UTs).

Figure 3. Central yard at the edge of the concession where timber is stored before further transportation to the industry.

Data collection

Felled trees

To estimate the direct loss of above-ground biomass and carbon due to logging, I accompanied loggers to their daily areas pre-selected by the company. The first five trees they harvested every day were included in the study. Therefore, I had no prior knowledge of what tree or species was selected.

All trees were checked to see if they were hollow prior to felling, as that makes them unfit for timber use. Any hollow trees were excluded from the harvesting operations. After a tree had been checked and cleared for felling, I estimated the relative tree density in the area around the target tree. This was done holding an AA battery at an arm's length and counting all trees wider than the battery around the tree to be felled. This method filters out trees far away from the base of the tree, as they will seem smaller than the battery with increasing distance.

After felling, the tree was left until the next day as a safety precaution. If big branches, lianas, or other trees were still suspended above the impact area upon return the next day, the area was deemed unsafe and no data collection activities were performed. If the impact area was deemed safe, the bole length of the tree was measured from stump to crown with a 50 m measuring tape. Species name and DBH (diameter at breast height = 130 cm) for all felled trees were supplied by the logging company.

Collateral damage

To estimate collateral above-ground biomass and carbon loss from logging, all trees with DBH >10 cm either in the impact site (see Figure 4) of the felled tree or that had in other ways been directly or indirectly (e.g. by pulls from lianas or damaged by trees that were uprooted/broken) affected by the felling (hereafter called residual trees) were measured and damage severity noted. DBH was measured with a measuring tape and height estimation was done visually by standing at a point where both the bottom and the top of the tree was visible. A 1 m section was marked on the tree trunk and by using a pencil as a proxy for the 1 m section I counted how many meters tall the tree was. The damage to different parts of each tree was visually determined after the damage classification in Krueger (2004). See Table 1 for details. If there was no visible damage, this was noted.

Table 1. Damage classification for residual trees, modified from Krueger (2004)

DAMAGE DEGREE	BOLE	ROOT	CROWN
SEVERE	Snapped at base, bent, or severely leaning	Uprooted	Loss of entire crown, loss of less than entire crown but more than 2/3 of crown
MODERATE	Exposed and damaged cambial tissue	Exposed and damaged cambial tissue	Loss of less than 2/3 but more than 1/3 of crown
MINOR	Exposed cambial tissue but no damage, bark scrape	Exposed cambial tissue but no damage, root scrape	Loss of less than 1/3 of crown

In addition, larger broken pieces (diameter >10 cm) from residual trees that could be identified to which tree it had broken off from had their diameter and length measured to estimate the additional biomass and carbon loss.

Figure 4. Two examples of a typical impact site after felling operations. Note the damage to the bark of closest trees and trees toppled by the felled tree.

Statistical analysis

Above-ground biomass (AGB) was calculated using the R package BIOMASS (Réjou-Méchain et al., 2017) in R version 3.6.3 (R Core Team, 2020). Species-specific wood density of each harvested species was obtained from the global wood density (GWD) database (Zanne et al., 2009). Due to lack of information about species in the residual stand, I used the mean wood density value for trees in tropical South America (0.632 g/cm^3, n = 4192), calculated using the GWD database (Zanne et al., 2009). A biomass to carbon ratio of 0.471 ± 0.206 (Réjou-Méchain et al., 2017) was used to calculate carbon loss.

Linear regression models were constructed to test the relationship between felled tree height, DBH, relative tree density, and the number of residual trees. I also constructed an interaction model to see if the height and the DBH of the felled tree both influenced the number of residual trees. All analyses were performed in R version 3.6.3 (R Core Team, 2020) using a significance level of 0.05.

Results

Felled trees

In total, 54 felled trees from 12 different species were included in the study, and *Erisma uncinatum* (n = 20) and *Manilkara huberi* (n = 15) were most abundant (Table 2). The felled trees were relatively evenly distributed in each DBH class (Figure 2), ranging from 55-130.5

cm. Tree height varied from 11.5-42.2 m, with a mean height of 21.3 ± 0.8 m. For a full list of data for each felled tree, see Appendix 2.

The height and DBH of the felled trees were tested to see if they were correlated, but this was not significant (adjusted $R^2 = -0.00836$, $p = 0.457$, Appendix 3). DBH and height were therefore both used in further analyses.

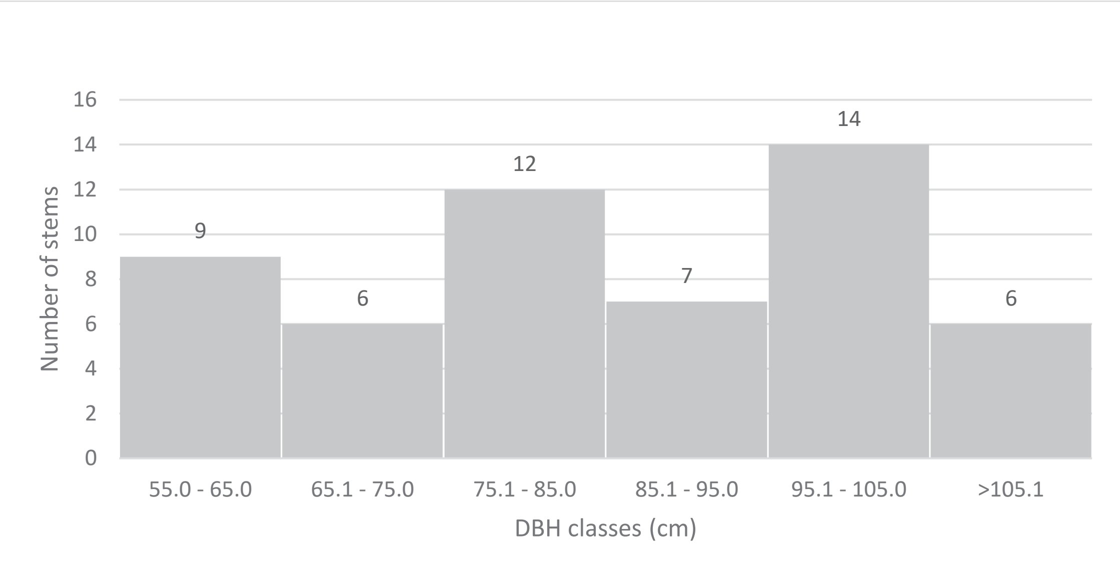

Figure 5. The distribution of stem DBH for the 54 felled trees. All felled trees must have a minimum DBH of 50 cm to be eligible for felling.

Table 2 Overview of data from the felled trees included in the study. The table includes scientific as well as common names for all species studied, number of trees sampled from each species, mean values for wood density, DBH, and height, and mean number of residual trees damaged per felled tree from each species.

Scientific name	**Common name**	**Number of trees sampled**	**Mean wood density (g/cm³)**	**Mean DBH (m)**	**Mean height (m)**	**Mean number of residual trees**
Astronium lecointei	Muiracatiara	3	0.790	0.902 ± 0.053	32.8 ± 7.6	20.3 ± 5.2
Bagassa guianensis	Tatajuba	1	0.706	0.824	20.4	29.0
Caryocar gracile	Pequiá	2	0.690	0.977 ± 0.051	17.4 ± 0.13	16.0 ± 2.0
Chrysophyllum spp.	Guajará-bolacha	2	0.665	0.796 ± 0.13	22.3 ± 1.1	15.0 ± 2.0
Couratari guianensis	Tauari	4	0.507	0.938 ± 0.030	26.5 ± 2.7	22.5 ± 4.5
Dipteryx odorata	Cumaru-amarelo	1	0.914	0.764	25.6	32.0
Erisma uncinatum	Quarubarana	20	0.523	0.923 ± 0.047	19.9 ± 0.97	17.8 ± 1.4
Goupia glabra	Cupiúba	1	0.727	0.859	15.1	8.00
Hymenaea courbaril	Jatobá	1	0.792	0.955	30.6	15.0
Machaerium macrophyllum	Timborana	1	0.733	0.697	15.0	24.0
Manilkara huberi	Maçaranduba	15	0.921	0.768 ± 0.055	21.1 ± 0.79	18.3 ± 1.2
Manilkara paraenesis	Maparajuba	3	0.860	0.849 ± 0.11	15.3 ± 3.1	13.3 ± 5.6

Collateral damage

There were a total of 1000 residual trees registered, giving a mean number of 18.5 ± 0.92 residual trees per felled tree. Most of the residual trees were small, with a DBH ranging between 10-20 (54%) and 20-30 cm DBH (24%; Figure 4). Very few trees (107 trees, 11%) were larger than 40 cm DBH. There were 135 broken pieces with diameter >10 cm counted from the residual trees included in my study.

Severe crown damage with loss of over 2/3 of the crown was recorded for 335 of the residual trees (Figure 5). However, most trees showed minor damage to both the bole and crown (489 and 404 trees, respectively; Figure 5). Roots were less likely than bole and crown to be damaged, with 603 trees not having any root damage (Figure 5).

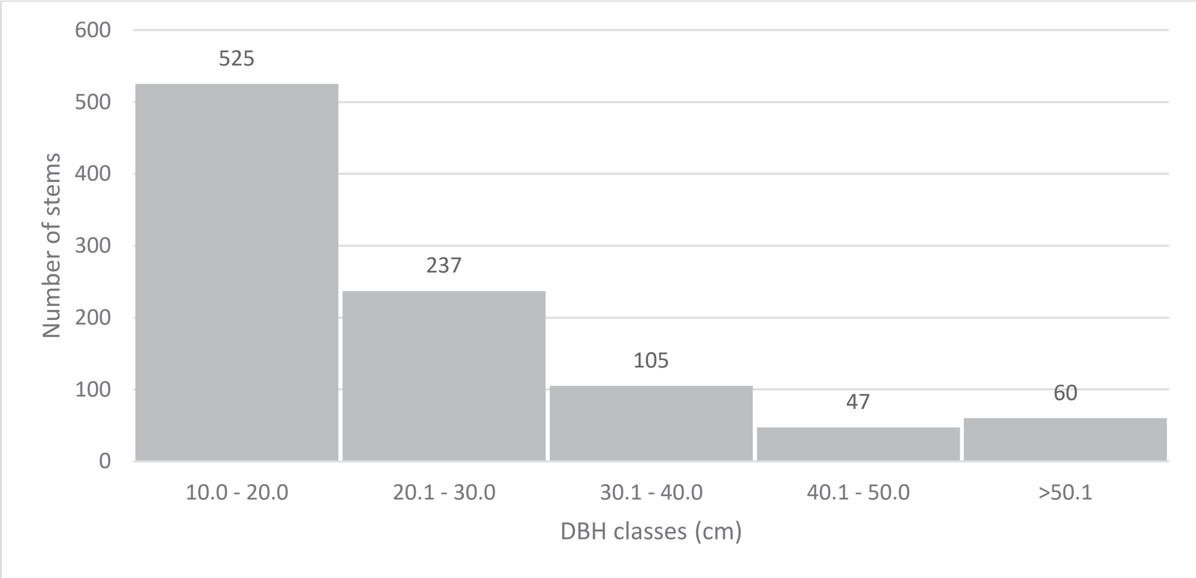

Figure 6. Distribution of stem DBH measured for the residual trees. Most of the residual trees were in the smallest DBH class, while there was a decreasing number of trees for the larger DBH classes. For 26 of the damaged trees it was not possible to measure DBH, and they are therefore not included in this figure.

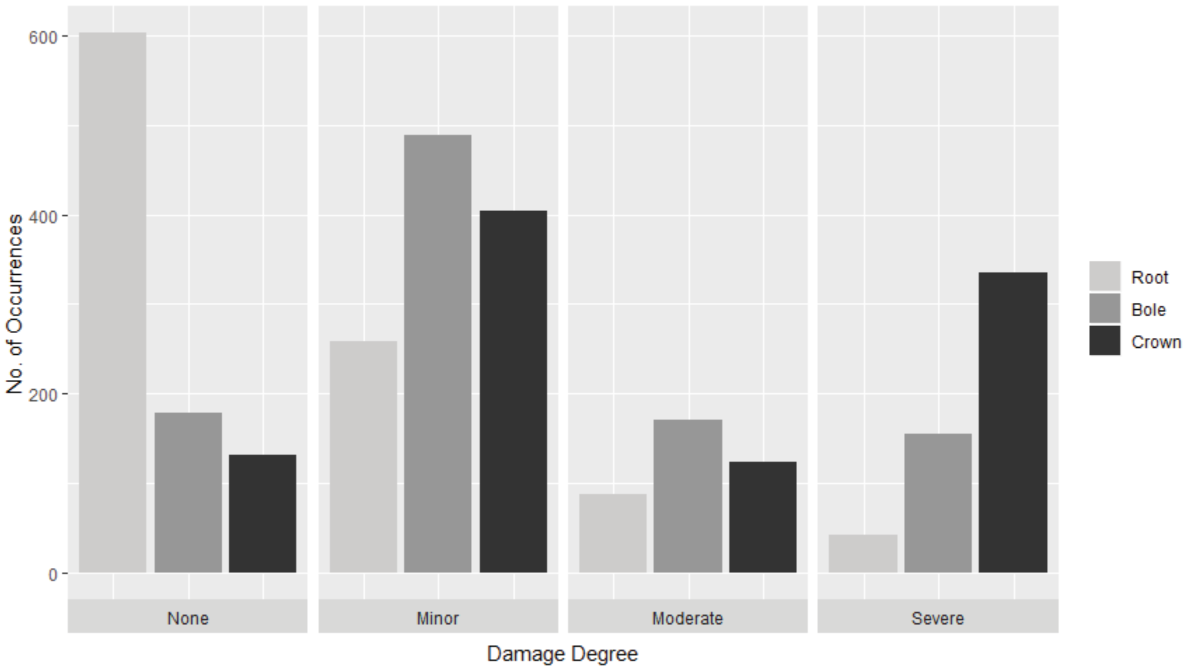

Figure 7. Distribution of damage to the residual stand, classified using Krueger (2004). Roots were most often not damaged, while the bole and crown mostly suffered minor damage. However, there was also a large number of trees with severe crown damage.

The number of residual trees increased significantly with increasing height of felled trees (adjusted $R^2 = 0.0655$, $p = 0.0345$; Figure 7), whereas DBH had no significant effect ($p = 0.321$; Figure 9). The number of trees surrounding each felled tree (relative tree density) did not have a significant effect on the number of residual trees ($p = 0.475$; Figure 10).

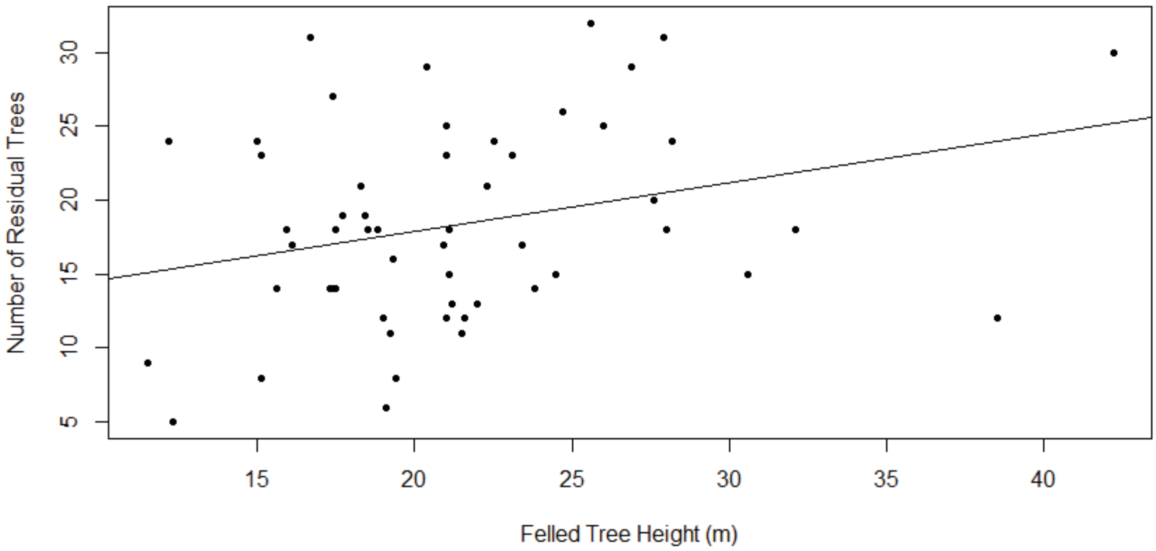

Figure 8. Number of residual trees plotted against the height of the felled tree. Adjusted $R^2 = 0.0655$, $p = 0.0345$.

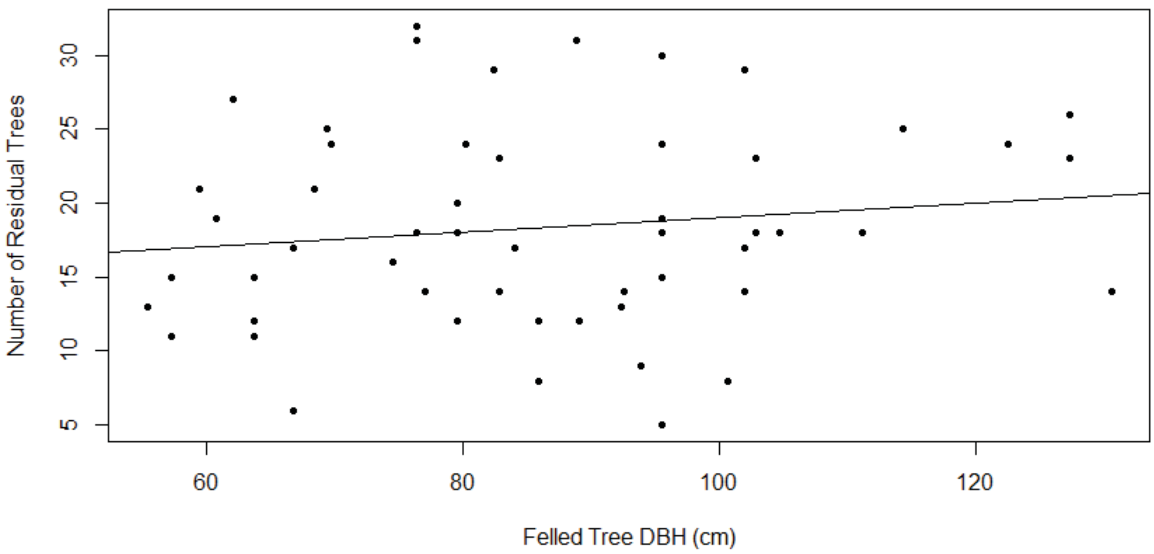

Figure 9. Number of residual trees plotted against the DBH of the felled tree. Adjusted $R^2 = 8.137 \times 10^{-5}$, $p = 0.321$.

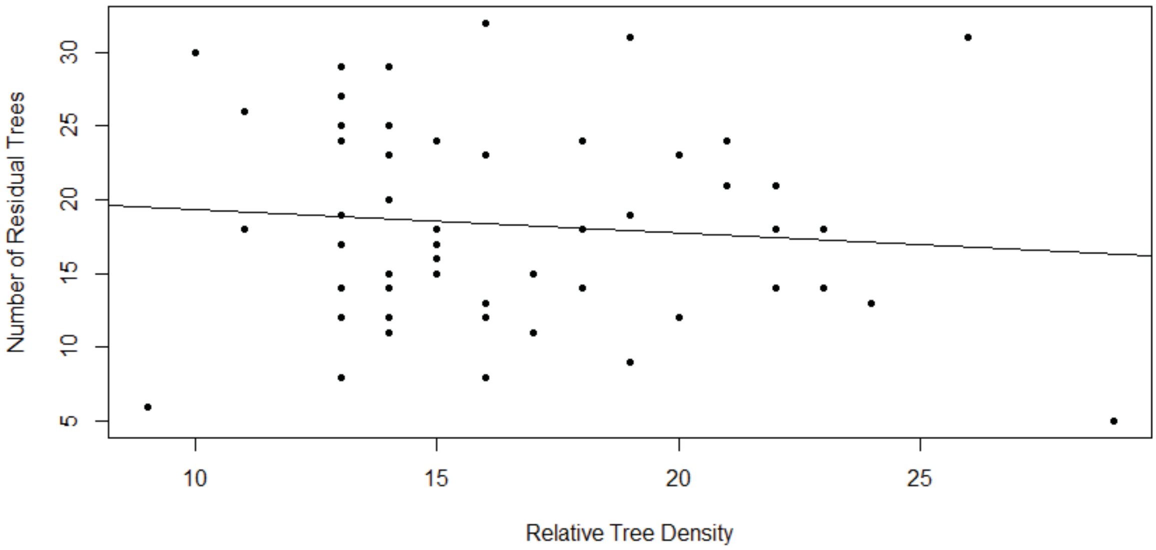

Figure 10. Number of damaged trees plotted against the relative tree density around the felled tree. Adjusted R^2 = -0.00919, p = 0.475.

The interaction between height and DBH of the felled tree plotted against the number of damaged trees was significant (p = 0.03), showing that the number of damaged trees increases as the height and DBH of the felled tree increases (Figure 11).

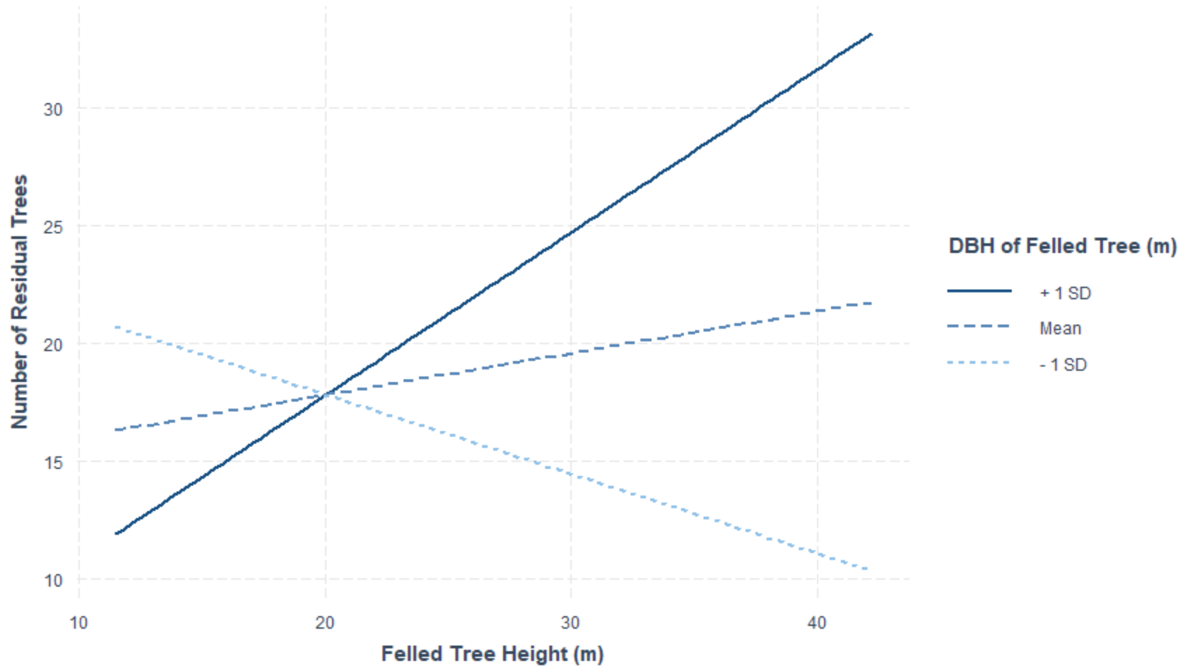

Figure 11. Interaction model showing how the number of residual trees changes depending on the height and DBH of the felled tree. Model fit: adjusted R^2= 0.13, p= 0.02. P-value for the interaction model DBH:Height = 0.03.

Biomass and carbon loss

Biomass loss from the 52 felled trees included in this study was 313.2 Mg, while the biomass loss for broken parts from the residual stand totalled to 178.8 Mg. Carbon losses were estimated to be 147.5 ± 57.9 Mg and 84.2 ± 36.8 Mg for the felled trees and the residual stand, respectively. These results mean that each felled tree will lead to an average of 9.11 ± 0.20 Mg biomass lost and 4.29 ± 0.88 Mg carbon lost when including losses from the residual stand.

Discussion

Felled trees

I sampled 12 of the 19 species targeted for felling in UMF II. The trees sampled in this study are therefore reasonably representative for all trees to be harvested in this production unit. Most notably, I included 20 *Erisma uncinatum* and 15 *Manilkara huberi*, the two species that will be harvested most intensely (Appendix 1). The trees included in the study also covered the entire size spectrum of trees to be harvested. The legal minimum DBH for trees to be eligible for logging is 50 cm, which is why no trees were below this size.

The species harvested at Caxiuana are common timber species throughout the eastern Amazon and beyond (Global Forest Atlas). Most of them are hardwood species and very valuable in the timber industry. There has been some concern expressed about the sustainability of logging for a few of these species. Research has shown that the current legal cutting cycle of minimum 30 years is too short to ensure sustainable timber harvests during future cycles for *M. huberi*, *B. guianensis*, *A. lecointei*, and *H. courbaril* (Schulze et al., 2008; Sebbenn et al., 2008). Vast, continuous areas of forest are also needed to maintain an effective population of *M. huberi* and *D. excelsa* (Azevedo et al., 2007; Dick et al., 2003), as there is a concern about inbreeding due to poor pollen dispersal. *M. huberi* is the second most exploited species in UMF II, and the continued high harvest is therefore cause for concern about the sustainability of this timber species.

Collateral damage

From the 52 felled trees that I studied, I found a positive relationship between the height of the felled tree and the number of trees damaged by the harvest, and an interaction between the height and DBH of the felled tree and number of residual trees. However, I could not find any such correlation with DBH or relative tree density. A possible reason for this is that I did not sample enough felled trees. The battery method used to measure relative tree density can be inaccurate since it only “sees” trees close to the felled tree. The felled trees are quite tall and may therefore affect residual trees quite a distance from its base when felled. Also, since I visually determined what trees were inside the impact zone and what trees were damaged, I could have mistakenly omitted trees that were in fact affected by the felling, thereby underestimating the number of residual trees.

I found that each felled tree affected approx. 18 residual trees, where they mostly suffered minor damage to the bole and crown, while roots were most often without any visible damage. This contrasts with findings by Jackson et al. (2002) who reported that in a selective logging concession in Bolivia the most common types of damage included uprooted stems (severe damage), damage going through the cambial tissue (severe damage), and bark scrapes (minor damage). Yet, over 400 of the residual trees recorded had a loss of over 2/3 of their crown or had lost the crown completely. This is a significant number, as trees that have suffered severe crown damage have a very high mortality rate (Arellano et al., 2019; Pinard & Putz, 1996). Similarly, almost 30% of trees recorded as leaning after logging will have an increased risk of dying (Pearson et al., 2014).

Trees that were broken due to collateral damage may be able to resprout. However, if the damage to the tree is too high for regeneration, the tree may die due to the damages. Therefore, trees that are considered to be living shortly after logging may add to the dead biomass due to delayed mortality. Damaged and exposed cambial tissue is also a potential infection site for various pathogens or insect attacks, which may also lead to mortality (Gilbert & Hubbell, 1996). The collateral damage and wood waste left after felling will function as fuel and therefore increase fire susceptibility in the forest (Holdsworth & Uhl, 1997; Matricardi et al., 2010). The increased risk of fire is both due to the increase in available fuel, but also due to the formation of logging gaps increasing light penetration into the forest. The combined effect of logging and fire releases more carbon stocks out into the atmosphere, especially during periods of drought (Nepstad et al., 1999). This represents a big risk if the occurrence and length of drought periods increase with climate change (Malhi et al., 2008).

I do not know the identity of affected trees (vouchers were collected in the field, but the identification of these were unfortunately not completed in time). Yet, their identity has significance as the area is supposed to be explored again in a second rotation 30 years from now. If many of the trees collaterally damaged during the first rotation are young individuals of timber species, this could affect the economic viability and sustainability of that second rotation. In fact, avoiding damage to future crop trees (FCTs) would help ensuring more sustainable harvests and larger future yields. Krueger (2004) found that flagging FCTs at a cost of US$0.38/ha could reduce damage to the residual stand by 20 and 10% in felling gaps and skidding trails, respectively.

Additional biomass and carbon loss

Assuming a biomass loss of 9.11 Mg per tree, this means a biomass loss of 31 721 Mg for the 3482 trees to be harvested in UMF II. This translates to a carbon loss of 14 940.6 ± 3077.8 Mg. Including the 1070 trees to be harvested in UMF I, the total estimated loss from these two management units will be 41 468.7 Mg biomass and 19 531 ± 8542 Mg carbon. Considering that rainforest in the eastern Amazon contains on average 197 000-256 000 tons biomass per hectare (Mello et al., 2016), this amounts to the complete deforestation of 0.16-0.21 ha of rainforest from these two management units alone.

However, biomass and carbon loss estimates provided here are conservative for several reasons. First, this logging concession is a best-case scenario in the Brazilian Amazon. With up to 80-95% of the timber extraction in the Brazilian Amazon being illegal (Hirschberger, 2008; Smith, 2004), such activities are likely to carry a greater impact on forest structure and thus a higher collateral damage. The number of trees removed per defined area will also likely be much higher. Second, only bole height was measured for the felled trees. This means that biomass and associated carbon loss from the stump of the tree and the whole of the crown is not included in my estimates. As most crowns were large (pers. obs.), this will considerately underestimate losses from each tree. Third, I did not collect individual-specific wood densities, but used data from the global wood density database. Previous research suggest that this can significantly impact above-ground biomass and carbon estimates (Y. Bredin pers. comm.; Fearnside, 1997).

Fourth, this estimate does not include losses from road construction, forest cleared for stockyards and camps, or from skidders extracting timber from within the forest. Pearson et al. (2014) found that emissions from extracted logs amount to only a small proportion (15-25%) of total emissions from selective logging, while emissions from collateral damage account for 38-51% of the total. The amount of carbon emissions from this concession could therefore be between 30-50% higher than what is estimated here.

My estimates here assume that all carbon emissions are emitted at the time of felling and does not consider the fact that timber from the felled trees can be stored in the long term in products. Construction products are one such example, while timber used for paper, for example, is a very short-lived product. As most of the tree species considered for felling at this concession are hardwoods (see Appendix 1), the probability that they will be made into building material and other long-lived products is high.

Conclusion

Even though this concession uses RIL principles, the amount of collateral damage observed is still significant. Each extracted tree in the concession leads to an estimated 9.11 Mg carbon loss, which results in close to 41 500 Mg biomass lost and around 20 000 Mg carbon released. However, this estimate is likely to be very conservative as they do not include losses from infrastructure construction inside of the concession or species-specific wood density values for losses from the residual stand.

Considering how important logged forests are for biodiversity conservation, and that tropical forests are essential in the mitigation of climate change effects through carbon sequestration, more research is needed to understand the processes that alter and degrade these environments. There is very little research on collateral damage during selective logging and how unnecessary damage can be avoided. Future studies should focus on methods for reducing collateral damage during selective logging and how to preserve carbon stocks in logging concessions. However, improving legal practices will likely make a small difference due to the amount of illegal logging present in the Brazilian Amazon. The pressing issue of preserving ecosystem services and carbon in future Amazonia will therefore likely not be solved unless steps are taken to reduce the amount of illegal logging.

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Appendix 1

Table A1. Species harvested at the concession in Caxiuanã, number of individuals harvested and mean DBH.

Scientific name	**Common name**	**Individuals to be harvested**	**Mean DBH (m)**
Astronium lecointei	Muiracatiara	221	0.763 ± 0.010
Bagassa guianensis	Tatajuba	43	0.879 ± 0.014
Caryocar gracile	Pequiá	37	1.00 ± 0.021
Chrysophyllum spp.	Guarajá-bolacha	90	0.976 ± 0.066
Copaifera multijuga	Copaiba	35	0.890 ± 0.026
Cordia goeldiana	Freijó	10	0.750 ± 0.037
Couratari guianensis	Tauari	219	0.930 ± 0.0020
Dinizia excelsa	Angelim-vermelho	84	1.21 ± 0.026
Diplotropis racemosa	Sucupira	13	0.789 ± 0.050
Dipteryx odorata	Cumaru-amarelo	106	0.772 ± 0.015
Endopleura uchi	Uxi	42	0.767 ± 0.016
Erisma uncinatum	Quarubarana	1003	0.846 ± 0.0040
Goupia glabra	Cupiúba	136	0.859 ± 0.012
Hymenaea courbaril	Jatobá	159	0.897 ± 0.013
Hymenolobium excelsum	Angelim-pedra	50	1.05 ± 0.025
Licaria cannella	Louro-canela	35	0.677 ± 0.019
Machaerium macrophyllum	Timborana	315	0.764 ± 0.0070
Manilkara huberi	Maçaranduba	819	0.775 ± 0.0060
Manilkara paraenesis	Maparajuba	65	0.866 ± 0.010

Appendix 2

Table A2. Overview of the felled trees included in my study and collected data associated with them.

UT	Felled tree ID	Scientific name	Common name	Relative tree density	DBH (m)	Bole height (m)
14	22729	Astronium lecointei	Muiracatiara	14	0.796	38.5
4	6451	Astronium lecointei	Muiracatiara	19	0.955	17.7
13	22122	Astronium lecointei	Muiracatiara	10	0.955	42.2
10	16856	Bagassa guianensis	Tatajuba	13	0.824	20.4
14	23393	Caryocar gracile	Pequiá	22	1.028	17.5
2	2393	Caryocar gracile	Pequiá	13	0.926	17.3
14	23257	Chrysophyllum spp.	Guajará-bolacha	24	0.923	21.2
14	23076	Chrysophyllum spp.	Guajará-bolacha	15	0.668	23.4
7	10196	Couratari guianensis	Tauari	16	0.891	19.0
14	23315	Couratari guianensis	Tauari	26	0.888	27.9
10	16973	Couratari guianensis	Tauari	23	0.955	32.1
8	15261	Couratari guianensis	Tauari	14	1.019	26.9
2	2821	Dipteryx odorata	Cumaru-amarelo	16	0.764	25.6
7	10133	Erisma uncinatum	Quarubarana	15	0.764	18.8
7	10495	Erisma uncinatum	Quarubarana	15	0.745	19.3
7	10672	Erisma uncinatum	Quarubarana	14	1.143	26.0
7	10439	Erisma uncinatum	Quarubarana	14	1.019	17.5
14	23677	Erisma uncinatum	Quarubarana	23	0.955	18.5
14	22922	Erisma uncinatum	Quarubarana	16	1.006	19.4
14	23074	Erisma uncinatum	Quarubarana	20	1.273	23.1
14	23069	Erisma uncinatum	Quarubarana	19	0.939	11.5
14	23079	Erisma uncinatum	Quarubarana	23	1.305	17.4
14	22735	Erisma uncinatum	Quarubarana	15	0.84	20.9
10	16809	Erisma uncinatum	Quarubarana	13	0.694	21.0
10	16981	Erisma uncinatum	Quarubarana	19	0.764	16.7
2	2387	Erisma uncinatum	Quarubarana	18	0.77	15.6
2	2315	Erisma uncinatum	Quarubarana	9	0.668	19.1
2	2690	Erisma uncinatum	Quarubarana	16	1.028	15.1
10	17138	Erisma uncinatum	Quarubarana	13	0.637	21.6
10	16931	Erisma uncinatum	Quarubarana	14	0.796	27.6
6	8953	Erisma uncinatum	Quarubarana	15	0.796	15.9
6	8955	Erisma uncinatum	Quarubarana	11	1.047	28.0
13	22246	Erisma uncinatum	Quarubarana	11	1.273	24.7

UT	Felled tree ID	Scientific name	Common name	Relative tree density	DBH (m)	Bole height (m)
2	2323	Goupia glabra	Cupiúba	13	0.859	15.1
14	22854	Hymenaea courbaril	Jatobá	17	0.955	30.6
9	15409	Machaerium macrophyllum	Timborana	15	0.697	15.0
7	10395	Manilkara huberi	Maçaranduba	18	0.802	22.5
7	10254	Manilkara huberi	Maçaranduba	21	1.225	28.2
7	10144	Manilkara huberi	Maçaranduba	14	0.573	19.2
2	2909	Manilkara huberi	Maçaranduba	22	0.828	23.8
6	9381	Manilkara huberi	Maçaranduba	14	0.828	21.0
6	9450	Manilkara huberi	Maçaranduba	13	1.019	16.1
6	9452	Manilkara huberi	Maçaranduba	18	1.111	21.1
4	6503	Manilkara huberi	Maçaranduba	22	0.684	22.3
4	6505	Manilkara huberi	Maçaranduba	20	0.859	21.0
3	4604	Manilkara huberi	Maçaranduba	15	0.637	21.1
3	4674	Manilkara huberi	Maçaranduba	21	0.595	18.3
6	8145	Manilkara huberi	Maçaranduba	14	0.573	24.5
6	8146	Manilkara huberi	Maçaranduba	13	0.621	17.4
13	22355	Manilkara huberi	Maçaranduba	13	0.608	18.4
8	15258	Manilkara huberi	Maçaranduba	16	0.554	22.0
14	23391	Manilkara paraensis	Maparajuba	29	0.955	12.3
3	4457	Manilkara paraensis	Maparajuba	17	0.637	21.5
3	4606	Manilkara paraensis	Maparajuba	13	0.955	12.2

Appendix 3

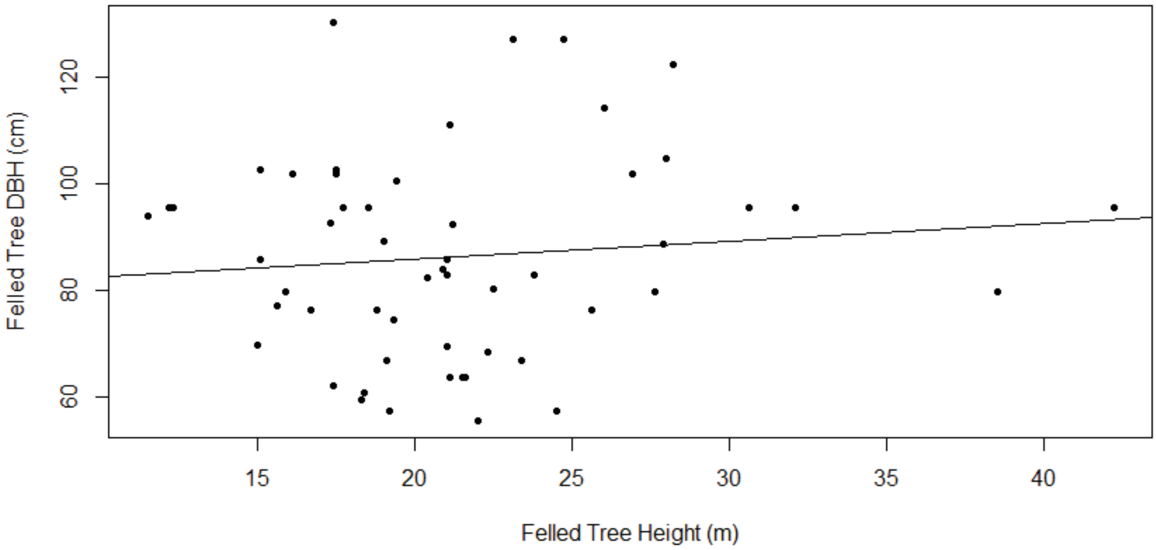

Figure A 1. The relationship between felled tree height (m) and felled tree DBH (cm). Adjusted $R^2 = -0.00836$ *and* $p = 0.457$.

Norges miljø- og biovitenskapelige universitet
Noregs miljø- og biovitskapelege universitet
Norwegian University of Life Sciences

Postboks 5003
NO-1432 Ås
Norway