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**DINÂMICA DE COMUNIDADES ARBÓREAS EM ÁREAS EXPOSTAS AO**  
**DESMATAMENTO NA AMAZÔNIA BRASILEIRA**

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Manaus, Amazonas

Novembro, 2023

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DESMATAMENTO NA AMAZÔNIA BRASILEIRA**

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**Sinopse:**

Estudamos a proporção da ocorrência de espécies hiper dominantes no ciclo do Carbono suprimidas por desmatamento ou em área de borda florestal no Bioma Amazônico brasileiro. Também avaliamos como o efeito de borda pode afetar a mortalidade e o acúmulo de Carbono pelas espécies hiper dominantes localizadas na ARIE PDBFF, Amazônia Central.

**Palavras-chave:** Dinâmica florestal, registro de ocorrência, árvores amazônicas, ciclo do C, desmatamento

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## RESUMO

A Bacia Amazônica exerce uma forte influência no ciclo global do Carbono, armazenando milhões de toneladas desse elemento em um número relativamente pequeno de espécies arbóreas consideradas "hiperdominantes". Contudo, este estoque de Carbono está sob ameaça devido ao desmatamento e fragmentação das florestas. Árvores em bordas florestais enfrentam diversos efeitos físicos e bióticos que podem impactar a sobrevivência e o crescimento das plantas. Este estudo investigou a ocorrência de espécies arbóreas hiperdominantes na Amazônia brasileira entre 1988 e 2021, visando avaliar como os efeitos de borda afetaram a mortalidade das árvores e o armazenamento de Carbono. Utilizando dados de parcelas permanentes e herbários, identificamos que 22% das árvores foram registradas em áreas que hoje já foram desmatadas, 35% a uma distância de até 1 km da borda da floresta e 43% em áreas de floresta contínua. Na escala local, na região central da Amazônia, o monitoramento ao longo de 30 anos revelou que os fragmentos florestais apresentaram o dobro da taxa de mortalidade em comparação com as florestas contínuas, devido aos efeitos de borda. Embora as árvores nos fragmentos florestais tenham inicialmente demonstrado um crescimento mais rápido, esse padrão diminuiu ao longo do tempo, resultando em uma perda significativa de Carbono, principalmente devido à mortalidade das árvores. Os efeitos de borda conduziram a declínios anuais na biomassa dos remanescentes florestais, indicando que mesmo as espécies hiperdominantes são suscetíveis a perturbações que levam à degradação e perda de áreas florestais. Portanto, a conservação das florestas amazônicas demanda uma abordagem que considere os efeitos das perturbações locais nos estoques de Carbono da região.



## INTRODUÇÃO GERAL

A Amazônia desempenha um papel crucial na regulação do ciclo global de Carbono (C) devido à capacidade das florestas de fixar e armazenar esse elemento em forma de biomassa (Dixon et al. 1994; Pan et al. 2011). Estimativas indicam que a Amazônia estoca aproximadamente 150-200 Pg de C (pentagramas de C) em biomassa acima do solo (Brienen et al. 2015), e que a maior parcela dessa biomassa está no componente arbóreo (Pan et al. 2011). Notavelmente, apenas 1% das espécies de árvores concentram cerca da metade do C estocado. A hiper dominância de espécies arbóreas no bioma e o maior porte das árvores são os principais fatores por haver esse acúmulo desproporcional de biomassa (Fauset et al. 2015; ter Steege et al. 2013).

A floresta Amazônica já foi um importante sumidouro de C, contudo, o aumento de CO<sub>2</sub> na atmosfera, as altas taxas de desmatamento e as oscilações climáticas, têm enfraquecido este padrão (Schimel et al. 2015; Hubau et al. 2020; Zhao & Running, 2010). Nos últimos anos, mais de 70% das emissões de gases de efeito estufa do Brasil tiveram origem nas mudanças no uso da terra. Como resultado, particularmente entre 2010 e 2020, o Sudeste amazônico tornou-se uma fonte de C, em decorrência do intenso desmatamento da área, e conseqüentemente um aumento na frequência de incêndios florestais (Gatti et al. 2021; Rosan et al. 2023; SEEG, 2021). Sinergicamente, a remoção da vegetação eleva a probabilidade de ocorrência de incêndios em áreas que raramente queimam naturalmente, propiciando a liberação do C armazenado na biomassa e necromassa (Nepstad et al. 1999; Pan et al. 2011).

O balanço de C é regulado pelas taxas de crescimento e mortalidade das árvores, que determinam a quantidade de C na biomassa e posteriormente sua transferência para a necromassa. O processo de transferência é fortemente influenciado pelas flutuações na disponibilidade de água e variações de temperatura (Roebroek et al. 2020). A elevada

heterogeneidade na dinâmica florestal do bioma resulta em uma variação regional no acúmulo de C, fortemente relacionada com a concentração de nutrientes no solo e regime climático, além das características espécie-específicas, como a densidade da madeira (ter Steege et al. 2000; Baker et al. 2004; Saatchi et al. 2007). Locais com maiores períodos de seca apresentam menor área basal; maior concentração de nutrientes no solo desencadeia crescimento acelerado nas árvores, que se relaciona com uma menor densidade da madeira e maior taxa de reposição de árvores (*turnover*). Regiões que somam essas características, como a Amazônia Ocidental, tendem a acumular menos C acima do solo. Já nas regiões Oriental e Central da Amazônia, ocorre o inverso, com períodos de seca mais curtos, solos com menor disponibilidade de nutrientes e, portanto, registram um crescimento mais lento, menor *turnover*, e assim, um maior acúmulo de C na vegetação (Malhi et al. 2006).

Do total da cobertura florestal original da Amazônia, 18% já foram desmatados (INPE, 2023). A prática agropecuária, que engloba em seu processo de desmatamento desde a mudança de uso da terra a extensas monoculturas, à exploração madeireira e indução de fogo para conversão de florestas em pastagem, além de frequentemente associada a práticas ilegais, é a principal causa dessas mudanças (Margulis, 2003). Aqui nos referimos a “desmatamento” como a completa remoção da cobertura florestal em um curto intervalo de tempo e subsequente substituição por outras coberturas de usos, enquanto “degradação florestal” faz referência às perturbações antrópicas que impactam consideravelmente a estrutura florestal, como exploração madeireira, incêndios e a fragmentação florestal (INPE, 2008; Putz & Redford, 2010; Berenguer, 2021). Tais perturbações chegam inclusive a superar os níveis de C emitidos por desmatamento em anos de secas extremas, uma vez que a extensão das áreas degradadas no período seco é maior (Qin et al. 2021; Matricardi et al. 2020; Lapola et al. 2023). Neste estudo vamos focar na degradação decorrente da fragmentação florestal associada ao desmatamento.

O desmatamento do bioma apresenta variações significativas no espaço e no tempo. A partir da década de 1950 e principalmente de 1970, incentivada por benefícios fiscais federais advindos de uma política de desenvolvimento econômico, houve uma migração intensa de colonos de outras áreas do Brasil para a Amazônia Oriental, agravando o desmatamento na região (Becker, 2010). Com o estabelecimento de latifundiários, a expansão procedeu para a Amazônia Central e as taxas de desmatamento seguiram o ritmo da migração e expropriação de terras, resultando em degradação e na maior taxa de crescimento urbano do país (Porto-Gonçalves, 2015; Becker, 2004). A ampliação da frente de desmatamento foi viabilizada, em grande parte, pelo aumento da acessibilidade, com a construção oficial de rodovias e significativa abertura ilegal de uma malha de estradas vicinais (Mahar, 1989; Brandão Jr. & Souza Jr., 2006; Perz et al., 2006; Haddad et al., 2015).

Porém, antes do ano 1990, apenas cerca de 5% da floresta havia sido desmatada, proporção que dobrou nos 10 anos subsequentes e continuou a aumentar (Berenguer, 2021). Após o segundo maior pico de desmatamento anual já registrado, em 2004, houve redução das taxas, devido a um conjunto de políticas de monitoramento, controle e fiscalização, aumento da demarcação de áreas protegidas e implementação das moratórias da soja e da carne (Macedo et al. 2012). Essas ações contribuíram para uma redução de 82% do desmatamento em 10 anos, atingindo uma das taxas mais baixas registradas em 2012 (4.6 mil km<sup>2</sup>) (INPE, 2023; Arima et al., 2014, West & Fearnside, 2021; Nepstad et al. 2009).

Nos anos seguintes, o aumento progressivo nas taxas de desmatamento foi impulsionado pelo enfraquecimento das políticas ambientais, desmonte governamental das medidas de controle e fiscalização, cortes orçamentários de órgãos ambientais, e condições econômicas favoráveis à expansão agrícola. É importante salientar que a partir de 2018, as alterações drásticas nas políticas ambientais brasileiras, levaram a um aumento abrupto nas

taxas de desmatamento, chegando a um acréscimo de 22% entre os anos 2020 e 2021 (Berenguer, 2021, INPE, 2022).

Atualmente, vem ocorrendo uma redução do desmatamento, com a diminuição de metade de área desmatada no primeiro semestre de 2023 em comparação ao mesmo período de 2022 (INPE, 2023), resultado da transição para um governo que vem dando ênfase à diminuição do desmatamento com um compromisso político de alcançar a erradicação do desmatamento até 2030 (MMA, 2023). Ao longo da história do desmatamento da Amazônia, é evidente o papel determinante de fatores econômicos e políticos como propulsores das oscilações nas taxas registradas (Messias, 2021; Margulis, 2003; Börner et al., 2014; Gibbs et al., 2015; Nepstad et al. 2014; Assunção et al. 2015; Gatti et al. 2023).

A conversão de florestas contínuas em áreas de agricultura e pecuária ocasiona a perda do habitat original e a fragmentação florestal, reconfigurando espacialmente a paisagem e aumentando a extensão de perímetro e áreas de bordas de floresta (Bennett et al. 2006; Laurance & Bierregaard, 1997; Berenguer, 2021). A vegetação que compõe as bordas de um fragmento florestal é duramente afetada, exposta a uma maior radiação solar, intensidade de ventos e conseqüente dessecação, prejudicando a sobrevivência e o crescimento de árvores. Essas novas condições ambientais são geralmente reunidas sob o termo efeito de borda e interferem na dinâmica da vegetação de forma mais pronunciada em até 300 metros para dentro de um fragmento florestal (Laurance, 2004).

Os impactos dos efeitos de borda entre as espécies que constituem a comunidade arbórea variam. Árvores grandes são mais suscetíveis a complicações fisiológicas causadas por dessecação, principalmente pelo porte que possuem e pela baixa flexibilidade. Além disso, a taxa de mortalidade é maior para árvores emergentes (>60 cm de DAP) em comparação com árvores menores submetidas às mesmas perturbações, reduzindo

significativamente o estoque de C local (Laurence et al, 2000; Albiero-Junior, 2021). Há uma perda considerável de biomassa, sobretudo nos primeiros 100 metros próximos à borda, intensificada pelo estabelecimento de espécies pioneiras de madeira leve, típicas de estágios iniciais de sucessão ecológica, que substituem as árvores grandes ao longo dos anos pós perturbação, causando um colapso de C (Laurance et al., 1997; Gora, 2021). Assim, o processo leva à mudança na composição e na estrutura da comunidade arbórea (Laurance et al., 1998a,b, 2000; Haddad et al. 2015).

Uma vez que grandes árvores fixam grandes quantidades de C em comparação com árvores menores, é importante ponderar a dinâmica da taxa de crescimento dessas espécies e a possível alteração no potencial de armazenamento de C (Stephenson et al., 2014). Para uma espécie arbórea comum de grande porte como *Scleronema micranthum*, por exemplo, houve redução da taxa de crescimento por 10 anos após o processo de fragmentação florestal em indivíduos localizados próximos às bordas (Albiero-Junior et al. 2019). Esse fenômeno, com ocorrência em escala ampla como a do bioma, pode impactar nos estoques de C ao longo de bordas florestais (Albiero-Junior et al., 2019, 2021).

Com base na relevância de espécies arbóreas na regulação do ciclo do C e na magnitude do desmatamento da Amazônia, este estudo propõe investigar o impacto da degradação ambiental por efeito de borda na dinâmica do estoque de C das espécies arbóreas hiper dominantes. Para isso, endereçamos as seguintes perguntas: (1) O quanto da ocorrência conhecida de espécies hiper dominantes foi perdida no Bioma Amazônico brasileiro entre 1988-2021? Para os registros remanescentes, (2) qual é a proporção das ocorrências localizadas em áreas de borda de floresta? E, (3) como o efeito de borda pode afetar a mortalidade e o acúmulo de C pelas espécies-alvo? Levantamos a hipótese de que a exposição ao efeito de borda devido ao desmatamento altera a dinâmica e o estoque de

carbono nas populações das espécies hiper dominantes. Assim, esperamos um aumento na taxa de mortalidade dentre as populações arbóreas investigadas neste estudo, e alteração da taxa de crescimento dos indivíduos sobreviventes, respondendo negativamente aos efeitos de borda com a diminuição na taxa ou, ao contrário, apresentar aumento na taxa de crescimento, favorecidos por um ambiente com maior oferta de luz. O que se pretende estimar com o estudo é o balanço de C por inferência indireta a partir das perdas e ganhos de C por biomassa, representados por mortalidade e crescimento, respectivamente.

Em capítulo único apresentado em inglês que compõe o corpo principal da dissertação, buscamos responder às indagações levantadas neste estudo. O capítulo se encontra formatado segundo as regras da revista *Forest Ecology and Management*.

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## Capítulo Único

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Giancola, DT et al. 2023. **Degradation Exposure Scenario in the Brazilian Amazon: Edge Effect on Hyperdominant C-cycle Tree Species.**

Manuscrito formatado para *Forest Ecology and Management*

# Degradation Exposure Scenario in the Brazilian Amazon: Edge Effect on Hyperdominant C-cycle Tree Species

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## Abstract

The Amazon basin strongly influences the global carbon cycle, storing billions of tonnes of carbon in a relatively small number of ‘hyperdominant’ tree species. However, the Amazon carbon stock is threatened. Forests are being cleared and fragmented, with trees near forest margins subjected to various physical and biotic edge effects that can alter plant survival and growth. We investigated the occurrence of hyperdominant tree species in the Brazilian Amazon between 1988 and 2021, to assess how edge effects influence tree mortality and carbon storage. Using plot and herbaria data, we found 22% of tree occurrence records in deforested areas, 35% within 1 km of the forest edge, and 43% in continuous forest. At the local scale in central Amazonia, tree monitoring data over 30 years also showed that forest fragments had twice the mortality rate of continuous forests due to edge effects. Although trees in fragments had higher initial growth, this pattern declined over the years and eventually resulted in significant carbon loss, mainly from tree mortality. Edge effects have led to annual declines in the biomass of forest remnants, suggesting that hyperdominant species are also susceptible to disturbances that lead to degradation and forest losses. Conservation of the Amazon forests requires an approach that considers the effects of local disturbances on carbon stocks in the region.

**Keywords:** Amazon basin, carbon storage, edge effects, forest dynamics, hyperdominant trees, occurrence register.

## 1. Introduction

The Amazon rainforest plays a crucial role in regulating the global Carbon (C) cycle due to its capacity for C fixation and biomass storage (Melillo et al. 1993; Dixon et al. 1994; Pan et al. 2011). Most of this C is stored above ground, notably concentrated within a small

fraction of tree species; just 1% of tree species in Amazonia store half of the C (Fauset et al. 2015). The disproportionate accumulation is primarily driven by the hyperdominance of certain species in the biome and their large sizes (ter Steege et al. 2013). The amount of C stored as biomass or transferred to necromass is governed by the rates of tree growth and mortality, which, in turn, are influenced by fluctuations in water availability and temperature variations, shaping the forest dynamic (Roebroek et al. 2020).

However, the rising levels of atmospheric CO<sub>2</sub>, high deforestation rates, and increasingly frequent climate fluctuations have undermined the notion of the Amazon as a significant C sink (Schimel et al. 2015; Hubau et al. 2020; Zhao & Running, 2010). Land-use changes have accounted for over 70% of Brazil's greenhouse gas emissions recently (SEEG, 2021). Currently, the Southeastern region of the biome has transitioned from a C sink to a source due to significant deforestation and the subsequent increase in forest fires (Gatti et al. 2021; Rosan et al. 2023). Furthermore, INPE (2023) reports that approximately 18% of the Brazilian Amazon's original forest cover has been lost in forested areas where all vegetation has been removed. Vegetation removal raises the risk of wildfires in regions where they would not naturally occur due to humidity, leading to the emission of stored carbon from both living and dead biomass. (Nepstad et al. 1999; Pan et al. 2011).

We refer to 'deforestation' as the complete removal of forest cover within a short period, followed by its subsequent replacement with other land uses. On the other hand, 'forest degradation' refers to anthropogenic disturbances significantly impacting forest structure, such as logging, fires, and forest fragmentation (INPE, 2008; Matricardi et al., 2020; Berenguer, 2021). This study focuses on degradation caused by fragmentation related to deforestation.

Deforestation in the Amazon rainforest exhibits significant spatial and temporal variation, mainly driven by agricultural expansion. The process of agricultural and livestock expansion stems from a history of public policies and is often associated with illegal practices (Margulis, 2003; Becker, 2005). In the 1970s, a government project in the eastern Amazon implemented a series of measures, including building infrastructure such as roads, hydroelectric dams, and communication networks. Settlement projects were also initiated, providing tax incentives to attract migrants from other regions of Brazil, who would subsequently arrive and occupy the area (Becker, 2009). Large and small landowners settled in the area, and the expansion extended into the Central Amazon through the highways established for agribusiness expansion, a significant factor in the region's devastation (Becker, 2004). The expansion of the agricultural frontier was mainly enabled by the greater accessibility resulting from the official construction of highways and the substantial illegal opening of secondary roads (Mahar, 1989; Brandão Jr. & Souza Jr., 2006; Perz et al., 2006; Haddad et al., 2015).

Deforestation decreased between 2022 and 2023 due to a political commitment to eliminate deforestation in the Amazon by 2030 (INPE, 2023; MMA, 2023). Several studies have examined the economic and political drivers of deforestation patterns throughout Amazonian history. (Messias, 2021; Margulis, 2003; Arima et al., 2014; Börner et al., 2014; Gibbs et al., 2015; Nepstad et al., 2014; Assunção et al., 2015; Gatti et al., 2023).

Converting continuous forests into agricultural and livestock areas leads to losing the initial forest habitat and forest fragmentation. Consequently, the landscape is spatially reconfigured, and the exposure extent of forest edges to non-forest matrices is increased (Bennett et al. 2006; Laurance & Bierregaard, 1997; Berenguer, 2021). Forests close to the edge of forest fragments experience adverse environmental conditions, including exposure to

higher solar radiation, increased wind intensity, and subsequent desiccation, which affect tree growth and survival. These environmental conditions are known as edge effects and significantly impact vegetation dynamics up to 300 meters within a forest fragment or any forested edge (Laurance, 2004).

The reverberation of edge effects varies among the tree species that comprise the arboreal community. Big trees are more susceptible to physiological complications caused by desiccation, which increases their likelihood of dying (Laurance et al. 2000). The death of large trees (>60 cm diameter at breast height) significantly reduces the local C stock, leading to its collapse, especially within the first 100 meters from the edge (Laurance et al. 1997; Gora, 2021). The fall of large trees leads to subsequent canopy opening, favoring the growth and establishment of pioneer trees that sequester less C. This significantly modifies the potential storage of C in the edge arboreal community (Laurance et al., 1998a,b, 2000; Stephenson et al., 2014). Albiero-Junior et al. (2019) observed a reduction in the growth rate of *Scleronema micranthum*, a common Amazonian large-sized species, in individuals located on the edges over ten years after forest fragmentation. This large-scale phenomenon, occurring within the biome, can affect C stocks at forest edges (Albiero-Junior et al., 2019, 2021).

Given the relevance of tree species in regulating the C cycle and the magnitude of deforestation in the Amazon, we investigated the effects of environmental degradation through edge effects on the C stock dynamics of hyperdominant tree species. In doing so, we ask: (1) How much of the known occurrence of hyperdominant species was lost in the Brazilian Amazon biome between 1988 and 2021<sup>1</sup>? Of the remaining tree species records, (2) what proportion are located in forest edges? (3) How does the edge effect affect mortality and C accumulation in the target species? We hypothesize that exposure to the edge effect due to

deforestation alters the dynamics and C stock in populations of hyperdominant species. Thus, we expect an increase in the mortality rate of the tree populations investigated, along with changes in the growth rate of surviving individuals. These changes may be manifested as a decrease in growth rate in negative response to edge effects or, conversely, an increase in growth rate favored by an environment with greater light availability. The study aims to estimate the C balance using indirect inference based on losses and gains represented by mortality and growth, thus providing valuable insights for conservation strategies amidst the ongoing challenges posed by Amazon deforestation.

<sup>1</sup>The period defined for this study corresponds to the Brazilian official monitoring data on deforestation in the Brazilian Amazon.

## **2. Material and Methods**

### **2.1 Study area**

In this study, we analyzed databases covering two spatial scales. To address questions 1 and 2, we used a regional scale corresponding to the Brazilian Amazon biome with deforestation data generated by the Satellite Deforestation Monitoring Project in Legal Amazon – PRODES (INPE, 2022) (Fig. 1).

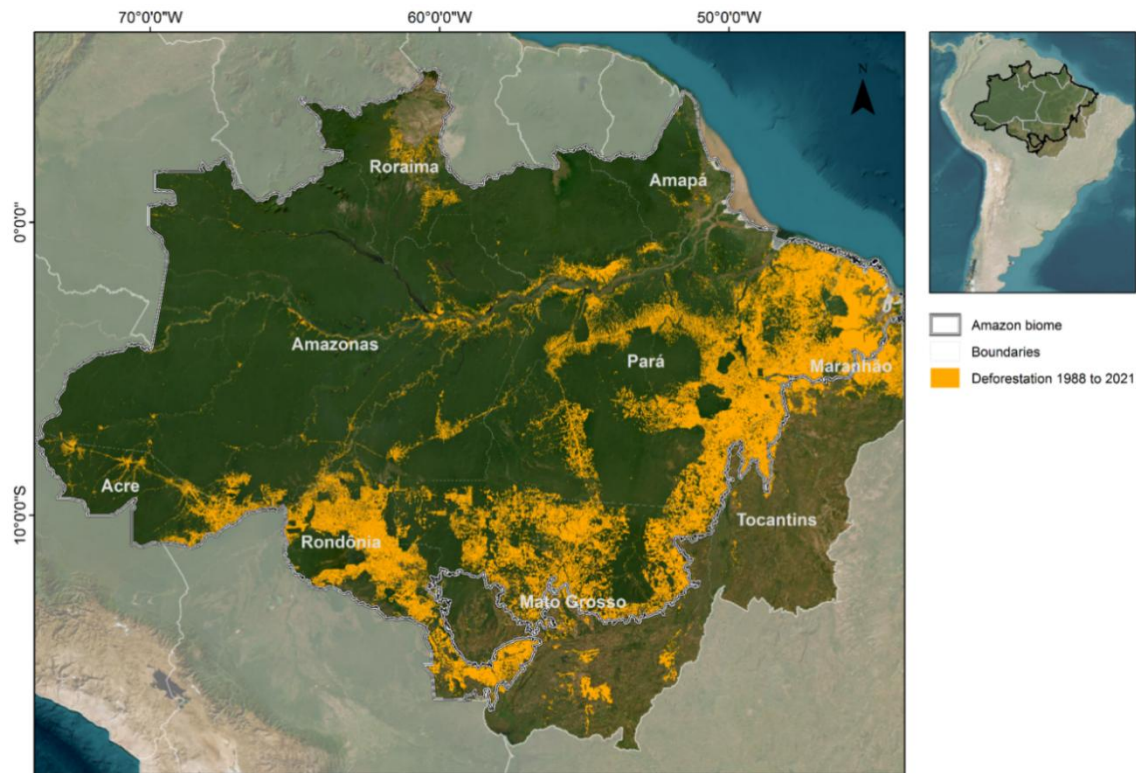


Figure 1. Amazon biome within Brazilian territory, predominantly covered by tropical forest (in green) and cumulative deforestation between 1988-2021 (in yellow) based on the historical records produced by PRODES (2021).

To answer the third question raised, we focused on a local study area using tree monitoring data from the Biological Dynamics of Forest Fragments Project (BDFFP), located approximately 80 km north of Manaus in the Central Amazon (see Supplementary Material, Figure A4). The BDFFP is the longest-running project monitoring biodiversity and assessing the consequences of forest fragmentation in tropical forests (Laurance et al. 2018). The experimental area established in 1979 covers 1.000 km<sup>2</sup>, with forest areas of different sizes within continuous mature forests beyond 100 m from the forest edge and remaining forest fragments represented by samples of permanent forest plots of 1 ha (100 x 100 m) located within 100 m. Isolated plots (forest fragments) were initially surrounded by a matrix of pasture created by clear-cutting techniques and, subsequently, after being abandoned, a matrix of secondary forest (Lovejoy et al. 1986; Laurance et al. 2002). Every five years, a strip of 100 m surrounding the forest fragments was cut to keep them isolated. With data

from these two different environments, we could comparatively assess the consequences of forest edge effects on mortality rates and C accumulation in selected tree species populations.

For the Biome scale study, we used a subdivision of the Brazilian Amazon proposed by Becker (2004), which divides it into three macro-regions. This regionalization considers the different historical patterns of local settlements, aiding in discussing the Amazon spatial and temporal heterogeneity related to human occupation and forest loss and its effects in the species occurrence patterns within the biome. The Densely Populated Arc region consists mainly of Maranhão, Tocantins, Mato Grosso, and Rondônia states. The Central Amazon corresponds to most of Pará, Amapá, and the Eastern part of Amazonas state. Finally, the western region includes the states of Roraima, the remaining part of Amazonas, and part of Acre (Fig. 2A).

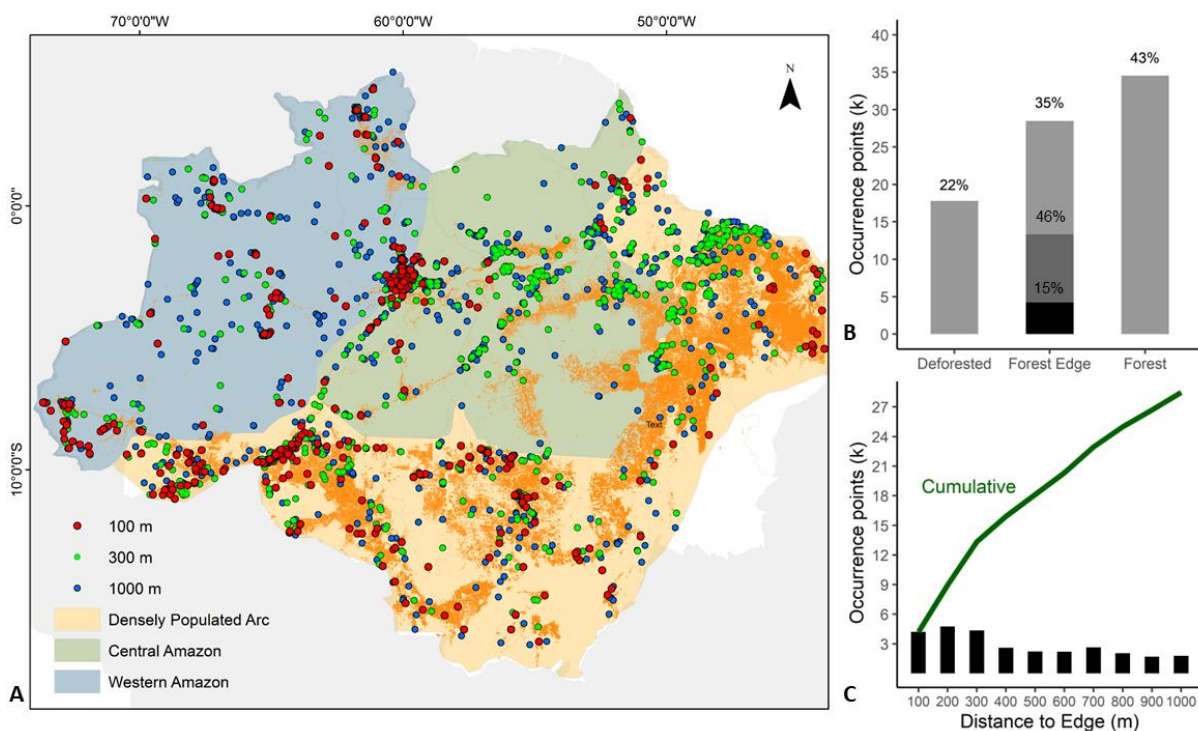


Figure 2. Tree species occurrence in the Brazilian Amazon. A. Distribution map of occurrence records of hyperdominant tree species on 100 m, 300 m and 1 000 m from the edge and territory subdivision proposed by Becker et al. (2004). The orange layer corresponds to accumulated deforestation from 1988 to 2021 (PRODES, 2021) B. Proportion



of occurrence records (in thousands) in deforested areas, forest edge areas, and continuous forest. The dark gray portion represents the proportion of occurrence points up to 300 m from the edge, and the black portion represents occurrences up to 100 m from the forest edge. C. Occurrence points (in thousands) at each distance from the edge and the increasing accumulation in distance categories (green line).

## **2.2 Database and sampling design**

We focused on the list of hyperdominant species as presented by Fauset and colleagues (2015), which included a dataset from 530 permanent plots distributed throughout the Amazon biome, maintained and organized by the RAINFOR and ForestPlots.net projects (Lopez-Gonzalez, 2011). From this list of approximately 3,000 species, we selected the 182 tree species responsible for storing 50% of the forest biomass in the monitored plots. We then excluded from the list palm species and tree species occurring exclusively outside the Brazilian Amazon, resulting in a final selection of 146 tree species.

To compile the known occurrence points of the selected species, we check the information from the permanent plots database of ATDN (Amazon Tree Diversity Network, 2022), which includes the data from ForestPlot.net, as well as in online herbarium platforms: Botanical Information and Ecological Network (BIEN, Enquist et al., 2009), Global Biodiversity Information Facility (GBIF), and SpeciesLink. Also, and quite notably, we included herbarium data, which provided a historical perspective on species occurrence records at sampling points before these areas underwent significant land use conversion, typically through forest suppression. To remove duplicate data, correct and update species names, the data has been thoroughly curated.

## **2.3 Data preparation and statistical analysis**

To estimate how much of the species' original known range has been lost, we overlaid the tree species occurrence points with information derived from the PRODES cumulative deforestation maps for the period of 1988 to 2021, freely available at

(<http://terrabrasilis.dpi.inpe.br/downloads/>). The proportion of each species' occurrence within areas classified as deforested at the regional scale was then estimated.

To assess how much of the selected species' occurrences are located at forest edges, we defined distances from PRODES cumulative deforestation perimeters toward the forest core. Distances of 100 m, incremented by 100 meters up to 1 km, were then created from the perimeter of each detected forest fragment across the biome, creating an area considered to be progressively affected by forest edge effects. We considered areas most affected within the first 100 to 300 meters from the edge and areas susceptible to progressive deforestation and subsequent edge effects from 300 to 1.000 meters (Laurance, 2018). This procedure allowed us to determine the proportion of occurrences present in forested areas but exposed to edge effects. Species whose populations are located at forest edges may be more susceptible to changes in forest dynamics and biomass accumulation processes.

To assess changes in forest dynamics under edge effects at the local scale, we focused on relative growth (RGR), mortality, and turnover rates of individuals of hyperdominant species in the BDFFP. We calculated the community mortality rate for each time interval using the formula:  $[\log(N1)-\log(N2)]/[T2-T1]$ , where N1 is the number of individuals in the first census, N2 is the number of individuals in the last census excluding recruits, that is, individuals who were included in the census for reaching the minimum DBH (10 cm) in that year, T1 is the year of the first census, and T2 is the year of the last census (Laurance et al. 2009). The equation used to calculate the recruitment rate was  $[\log(N2)-\log(N1)]/[T2-T1]$ , and by averaging the estimated mortality and recruitment rates, we calculated the turnover rate. Because the plots differed in the interval between censuses, a correction factor was applied to both mortality and recruitment rates, as well as the turnover rate, to minimize bias caused by differences in time intervals:  $\lambda_{\text{corr}} = \lambda t^{0.08}$ , where  $\lambda$  is the rate and t is the interval

between censuses in years (Lewis et al. 2004). Annual relative growth was calculated using the equation  $[\log(\text{DAP2}) - \log(\text{DAP1})] / [T2 - T1]$ , where DAP1 represents diameter at breast height (cm) in the first census and DAP2 in the last. The non-parametric Mann-Whitney test (Wilcoxon rank sum test) was used to compare rates between continuous and fragmented forests for each time interval.

We assessed the edge effect on community-wide growth and at the species level. For community analyses, we selected data from species with more than five individuals in each area (93 spp.), covering approximately 30 years of census data. We compared data from 36 of 1ha permanent plots (100 x 100 m) in forest fragments (FF), and data from 29 similarly sized permanent plots in continuous forests (FC). 14,083 trees were analyzed, 8,115 located in forest fragments, and 5,967 in continuous forest, representing 21.5% of the total trees monitored in the BDFFP. For species-level analysis of RGR, we selected only species with at least five individuals in all time intervals in each area (30 spp.).

To assess the difference in dynamics between edge and continuous forests over time, we fitted mixed-effects linear models using the *lmer* function from the *lme4* package (Bates et al. 2015). For RGR, we fitted the model (rate ~ year \* location + (1 | plot/individual (ID))) at the community level (93 spp.) and for each of the selected species (30 spp.). Location refers to fragmented and continuous forests. For the analysis of mortality and turnover, we fitted the model (rate ~ year \* location + (1 | plot)) at the plot level (n = 65). Both the year of establishment of forest fragments and the frequency of demographic censuses were different among plots. Considering the timing differences, the timeline was divided into six periods ranging from 5 to 30 years, considering the start year of the forest fragmentation process for each plot. We applied a logarithmic transformation to the community rates to ensure a normal distribution of the residuals. Marginal and conditional R-squared values (proportion of

variance explained by fixed effects -  $R^2m$ , and random and fixed effects -  $R^2c$ ) were calculated following Nakagawa & Schielzeth (2013). All analyses were performed in the R programming environment.

### **3. Results**

#### **3.1 Exposure to deforestation and edge effect - Regional scale**

Of the 80 thousand occurrence points of the selected hyperdominant species in the Brazilian Amazon, 57% are located in areas that have already been deforested or are influenced by forest edges (up to 1.000 m). Specifically, 22% of the records were found in areas deforested between 1988 and 2021, 35% located within 1 000 m of the edge, and 43% in continuous forest areas. Of the occurrences in edge areas, 15% were located within the first 100 m, and about half (46%) of the records were within the first 300 m of the edge (Figure 2B). Thus, there was a concentration of occurrence records within the first 300 m from the edge, with a decrease in the records between 400 and 700 m from the edge (Figure 2C).

The highest concentration of occurrences was recorded in the Central Amazon (45.5%), followed by the Densely Populated Arc region (33.5%), while the lowest concentration was recorded in the Western Amazon (21%) It is necessary to take into account the differences in territorial size in order to make a comparison between the subregions. The Western and Central Amazon are similar in size, and the Arco is about 35% larger than each of the other areas. The distribution pattern changes when only points in deforested areas are considered, with a higher concentration in the Arc (48%), followed by the Central Amazon (36%), and again a lower concentration in the Western Amazon (16%). At the forest edge, points are mainly located in the Central Amazon (38%) and the Arc (38%). This pattern continues for points 300 m from the forest edge. For points within the forest, there is a predominant concentration in the Central Amazon (58.5%), followed by a balanced distribution between

the Arc (21%) and the Western Amazon (20.5%). It is important to consider that the Western and Central sub-regions have a similar area size, while the Arc is approximately 35% larger than the other areas.

For about 4% of the species, most occurrences (>50%) overlap with deforested areas. Another 30% of species occur predominantly at forest edges, and of these, 80% showed a higher concentration of records within the first 300 m of the edge (Table 1). Among the species with the majority of records in already deforested areas or at the edge, predominantly within the first 300 m, three main patterns of occurrence were observed based on the registers: species restricted to one region of the Brazilian Amazon, generally with little information on their distribution; species with few records in the Brazilian Amazon, but widespread in the rest of South America; and species with widely distributed records throughout the biome (Table 1).

Table 1. Proportional distribution pattern of occurrence of tree species in deforested areas, forests located at 300 and 1.000 meters far from the edge, and in continuous forests: Part A. With higher occurrence in deforested areas, and Part B. With higher occurrence in edge areas. With a predominance of occurrence at the edge, mainly in the first 300 meters. All points within 1.000 meters are also within 300 meters.

#### A. Predominance in deforested area

Occurrence pattern	Species	Deforested (%)	Forest Edge - Distance range		Forest (%)
			300 m	1000 m	
Wide distribution in South America	<i>Anadenanthera colubrina</i>	54.1	5.4	29.7	16.2
	<i>Dipteryx micrantha</i>	42.3	11.5	32.7	25
Wide distribution in the biome	<i>Hymenaea courbaril</i>	38.6	18.5	35.6	25.8
	<i>Jupunba trapezifolia</i>	36.7	16.5	33.3	30
Restricted to Brazilian Amazon	<i>Vouacapoua americana</i>	40.4	24.3	32.1	27.5
	<i>Lecythis idatimon</i>	39.9	15.2	30.2	29.9

#### B. Predominance in forest edge area

Occurrence pattern	Species	Deforested (%)	Forest Edge - Distance range		Forest (%)
			300 m	1000 m	

			300 m	1000 m	
Wide distribution in South America	<i>Parkia pendula</i>	23.4	20.7	50.8	25.9
	<i>Handroanthus serratifolius</i>	22.4	22	48.2	29.4
	<i>Ficus gomelleira</i>	29.2	21.3	46.6	24.2
	<i>Inga alba</i>	25	22.8	43.8	31.2
	<i>Jacaranda copaia</i>	25.7	20.5	42.8	31.5
	<i>Lecythis pisonis</i>	33.6	20.8	39	27.4
	<i>Sloanea guianensis</i>	18.9	22.3	44.5	36.6
Wide distribution in the biome	<i>Pseudolmedia macrophylla</i>	26.7	20.6	47.1	26.2
	<i>Chrysophyllum lucentifolium</i>	27.5	19.8	40.7	31.9
	<i>Licania canescens</i>	25.9	19.6	40.4	33.7
	<i>Astronium lecointei</i>	22.8	16.1	39.5	37.6
	<i>Sterculia pruriens</i>	24.7	19.8	39.5	35.8
	<i>Bertholletia excelsa</i>	33.1	16.3	38.2	28.7
Restricted to Brazilian Amazon	<i>Tachigali paraensis</i>	35.8	22.6	45.3	19
	<i>Monopteryx uaucu</i>	22.6	22.6	45.2	32.3
	<i>Tachigali melinonii</i>	17.7	25.5	43.1	39.2
	<i>Protium altissimum</i>	30.8	17.1	41.5	27.7
	<i>Aspidosperma carapanauba</i>	19.1	18.2	39.14	38.6
	<i>Pouteria oppositifolia</i>	34	20.2	36.2	29.8

### 3.2 Edge Effect on Tree Mortality and Carbon Accumulation - Local scale

Considering the forest dynamics among the hyperdominant species occurring in the BDFFP, in a comparative scenario between fragmented forest areas (FF) and continuous forest areas (CF), we found that for the selected tree community (93 spp.), the average mortality rate was more than twice as high in FF compared to CF (FF =  $1.79\% \pm 0.24\% \text{ yr}^{-1}$  vs. CF =  $0.75\% \pm 0.18\% \text{ yr}^{-1}$ ,  $W = 188$ ,  $p < 0.05$ ) during the first five years after the area isolation, maintaining this pattern in FF up to 15 years after fragmentation ( $W = 771$ ,  $p < 0.05$ ). After this period, there was a decrease in the mortality rate in FF, while simultaneously, there was an increase in CF (Figure 3A). Following the mortality pattern, the turnover rate was higher in the early years of FF and decreased over time, while there was an increase in CF (Figure 3B).

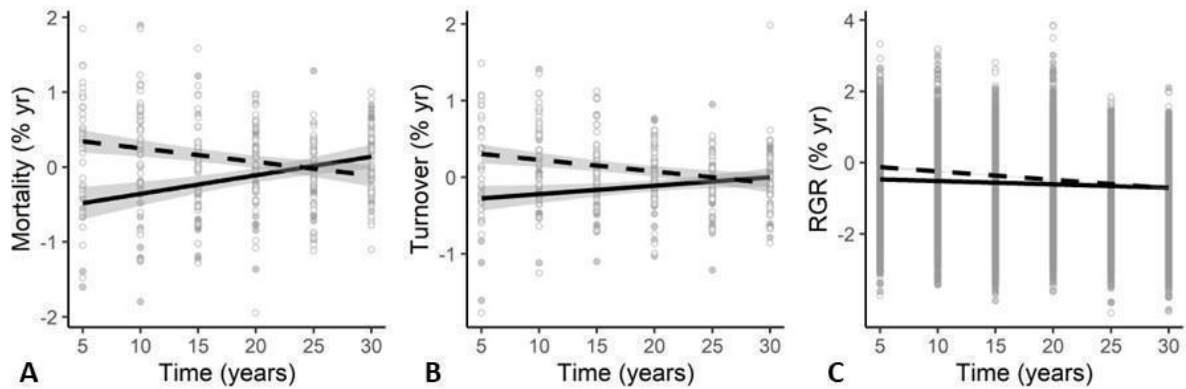


Figure 3. Community dynamics of hyperdominant tree species over 30 years in fragmented forest areas (FF, dashed line) and adjacent continuous forest areas (CF, solid line) from BDFFP, Central Amazon, based on the mixed-effects linear model. A. Mortality rate (% yr<sup>-1</sup>); B. Turnover rate (% yr<sup>-1</sup>) and C. Relative growth rate (% yr<sup>-1</sup>). All values are in logarithmic scale.

During the first 15 years after fragment isolation, RGR was higher in FF than in CF (W5 = 3268243; W10 = 6871246; W15 = 9473088,  $p < 0.05$ ). In addition, the decline in RGR was more significant in FF, converging after 30 years to annual rate values similar to CF, indicating a reduction in the intensity of the consequences of the fragmentation process over time (Figure 3C). In the RGR species analyses (30 spp.), 18 showed an interaction between time and location influencing the variation in growth rate ( $p < 0.05$ ). There was a predominant trend of convergence in RGR, except for *S. reticulata*, which exhibited higher growth in CF than in FF after 30 years (Figure 4 and see Supplementary Material, Table A2).

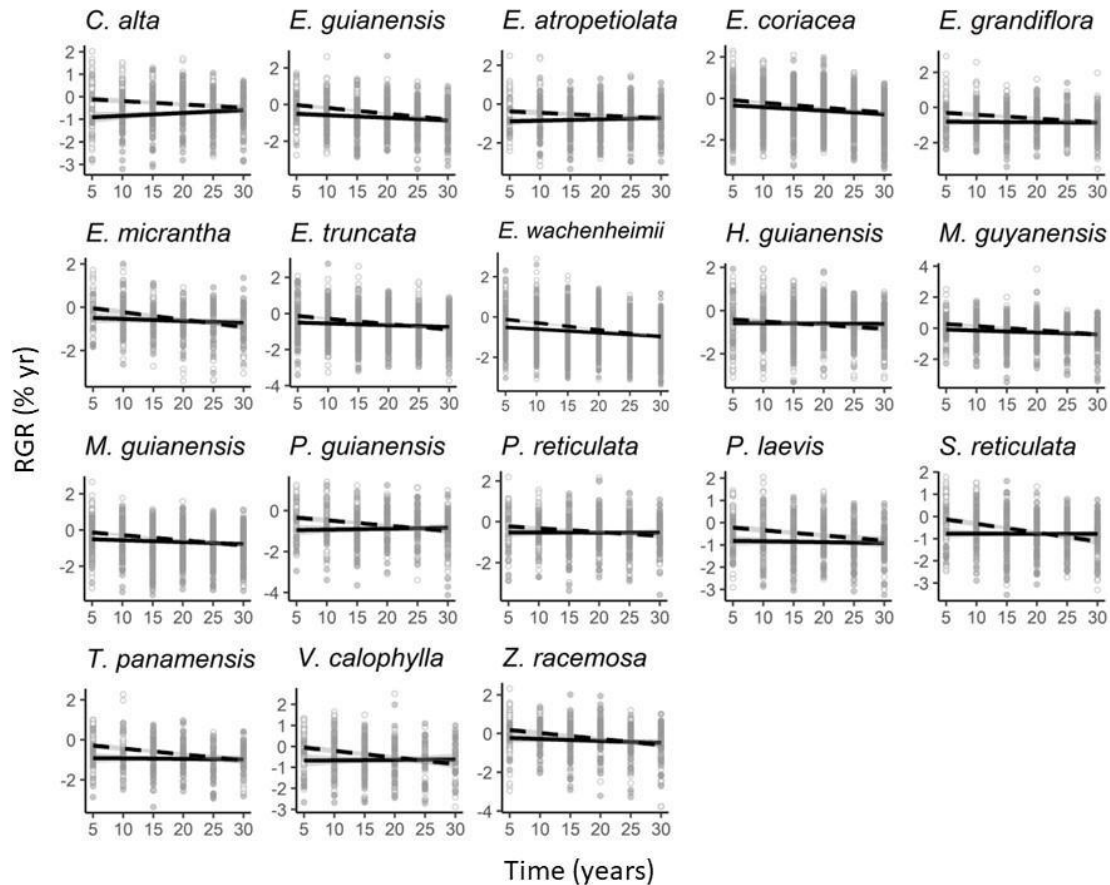


Figure 4. Relative Growth Rate ( $\% \text{ yr}^{-1}$ ) of hyperdominant tree species over 30 years in fragments (FF, dashed lines) and continuous forests (CF, solid lines) from BDFFP, Central Amazon, based on the mixed-effects linear model.

The regional-scale results provide an overview of species vulnerability, while the local-scale results provide insight into how species dynamics have changed over time in a given region. These results combined suggest that the identified trends represent significant regional dynamics in the Amazon basin. A similar pattern could occur in at least 15% of the sites located within the first 100 m of the forest edge.

#### 4. Discussion

We found that about 20% of the occurrence records have already been lost due to deforestation. These points were prevalent in the Densely Populated Arc deforested areas, which can be attributed to the ongoing deforestation in the region. As suggested by Becker (2004), this area is characterized by consolidated settlements in a historical process of



occupation, where deforestation was considered an improvement that added value to the land (Gomez et al. 2015; Aguiar et al. 2007). The presented data is consistent with a recently published study that demonstrates a one-third reduction in the recorded occurrence area of tree species due to deforestation, mainly in the Arc region. (Stropp et al., 2020).

One of the critical global consequences of deforestation is the release of stored C into the atmosphere, resulting from the loss of forest biomass and necromass (Loarie et al. 2009; Galford, 2011). Areas previously covered by forest and converted to pasture, agricultural land, or burned lose their accumulated C stocks significantly above ground (Nunes et al. 2022; Berenguer et al., 2014; MapBiomas, 2023), compromising the functional capacity of the forest to act as a sink (Gatti et al., 2022). Thus, we highlight the contribution of the 20% occurrence records of the most C-storing tree species to the conversion of forests, once sinks, into net C sources due to land use conversion.

In addition to C emissions from deforestation, which remains the primary source of C loss, it is imperative to consider emissions associated with edge effects and forest degradation due to deforestation (Pearson et al. 2017). Edge effects alone can account for up to half of the carbon lost from deforestation in the Amazon (Silva Júnior et al., 2020). As a result, approximately one-third of species within 1 km of the edge are vulnerable to C loss due to the adverse effects of forest fragmentation. Even more alarming, approximately half of these occurrences may already be experiencing these effects within 300 m of the forest edge, where the effects are significantly more pronounced (Laurence et al. 1997, 1998a). However, this pattern could be influenced by biases resulting from data collection strategies and the establishment of forest inventory plots, which are often located in easily accessible areas near roads and rivers and, thus, in general, close to forest edges (Oliveira et al. 2016). Therefore, it is imperative to conduct more comprehensive analyses to investigate this bias and its implications for studies related to edge-induced forest degradation.

Another critical consideration regarding the concentration of over 30% of the species analyzed in the edges is the fact that these areas are also vulnerable to deforestation. Given the ongoing deforestation trends of expressive annual rates, these populations are at imminent risk of decline, in line with projections that deforestation will advance up to 50% of the forest by 2050 if we continue with current trends in agricultural expansion (Leite-Filho et al. 2021).

We also demonstrated a greater loss of C in FF, primarily represented by continued high mortality rates, especially within the first 15 years after the generation of forest edges through the fragmentation process. In addition, dead trees no longer sequester or accumulate C, reducing the overall potential for its accumulation. There is also C loss through branch breakage, one of the main forms of biomass loss in areas under the edge effect, which poses a risk of damage to smaller nearby trees, also reducing their C contribution (Esquivel-Muelbert et al. 2020; Nunes et al. 2023). The increased mortality and turnover is a result of the new microclimatic conditions established at the edge. High temperatures and strong winds lead to canopy desiccation and physical damage (Camargo & Kapos, 1995; Laurance et al. 1998a, 2000, 2004; Ferreira & Laurence, 1997).

Although turnover is more pronounced in FF, studies show that recruitment consists primarily of smaller pioneer tree species, shrubs, and lianas (Laurance et al., 1997; Mesquita et al. 2001). This process potentially leads to C accumulation below pre-fragmentation levels (Silva Junior et al. 2020), implying a reduction of C storage in FF and a continuous loss of C accumulation potential over time (Ferreira & Laurence, 1997; Laurence et al. 1997). Therefore, there is an acceleration of the C cycle in FF, characterized by a shorter residence time of C in the biomass along the successional gradient compared to large trees in continuous areas (Nascimento & Laurance, 2004; Reis et al., 2022).

In addition, FF individuals exhibited higher RGR immediately after edge establishment, which decreased steadily over a 15-year period until it equaled the relative growth observed in CF. The initial higher growth in FF could be attributed to increased light availability due to increased mortality and consequent clearcutting in the early years following forest fragmentation (Oliver & Larson, 1996). However, this pattern did not hold for the species studied, all of which are slow-growing. Thus, while edge formation may favor initial growth, it does not translate into a consistent contribution on C stock over time.

The reduction in growth over 30 years, although more pronounced in FF, has been observed in both areas. Additionally, increased mortality has been observed in FC. These changes may be related to the increased frequency of extreme climatic events. This is evidenced by the decrease in tree growth during intense El Niño years due to the increase in the intensity of droughts (Stahle et al., 2020; Aleixo et al., 2019).

#### **4.1 Conservation relevance and management of protected areas**

Growth analysis allows us to infer the dynamics of populations in fragmented areas within the biome. Significantly, *M. guianensis* and *E. truncata*, two species affected by forest fragmentation, predominantly inhabit edge areas. Also, other similarly affected species have a significant proportion of their occurrence in these areas. At the species level, *L. idatimon* and *V. americana* also show a worrying conservation status, with a higher concentration in deforested areas and a restricted distribution in Brazil, mainly in the Northern regions of the Amazon. More alarmingly, *V. americana* is listed as "Critically Endangered" in the International Union for Conservation of Nature's (IUCN) Red List of Threatened Species (IUCN, 2022). We also highlight species such as *P. altissimum*, *P. oppositifolia*, and *T. paraensis*, which are not only restricted to forest edges, but also have the second-highest

concentration in deforested areas. Such situations highlight the need for conservation measures, such as deforestation control, to prevent the suppression of vegetation in new areas.

Species occurrence data patterns in the Central Amazon result from a complex interplay of urbanization, territorial policies and land use dynamics driven by integration into national economic cycles (Becker, 2004; Fearnside, 2001; Aguiar et al. 2002, 2007). Conversely, the Western region is characterized by a low concentration of records, likely due to information gaps caused by local inaccessibility and significant distances from major research centers, making data collection and material gathering challenging (Aguiar et al. 2007; Carvalho et al., 2023). These regions contain the most significant areas of Conservation Units (UCs) and Indigenous Territories (TIs), which are essential for the legal protection of vegetation. However, they are vulnerable to future degradation due to the expansion of deforestation for agricultural activities, the exploitation of natural resources such as minerals and timber, and especially the establishment along rivers and the invasion of TIs and UCs (Oliveira Souza, 2020; Schielein & Börner, 2018, Silva et al. 2021).

For species that occur mainly in contiguous areas, 40% of the points located in forest areas are within UCs or TIs, highlighting the critical role these areas play in environmental conservation. Such fact emphasizes the need to allocate areas for conservation, combat deforestation, and consequently reduce C emissions (Walker et al. 2009, 2020; Blackman & Veit 2018; Sousa et al. 2023; Londono et al. 2016; Soares-Filho et al. 2023).

These findings have important implications for conservation and management in the Amazon, emphasizing the need to protect degraded areas for regeneration into secondary forests, an essential strategy for C accumulation and the maintenance of local biodiversity and ecosystem functions (Lewis et al. 2019). However, studies highlight that more than 70 years of regeneration are required for forests to reach C stocks similar to pre-disturbance

levels, indicating a slow and gradual process that should be aligned with policies to combat deforestation in new areas (Poorter et al. 2016; Lennox et al. 2018). Successional pathways vary according to the intensity, extent, frequency, and history of land-use (Jakovac et al. 2015; Villa et al. 2018), affecting the speed of regeneration and the recomposition of species (Mesquita et al. 2001; Jakovac et al. 2016). For instance, slower regeneration was observed in abandoned pastures compared to clear-cut areas (Zarin et al., 2005; Mesquita et al., 2015). The development of strategies that are sensitive to local specificities while considering overarching patterns at larger scales is imperative. While this study provides valuable information into these patterns, in-depth investigations are essential for a comprehensive understanding of tree species dynamics in the Amazon. Therefore, we recommend using community forest C measurement methods in future studies involving fragmented areas, such as LiDAR, which has shown robust results by capturing not only mortality-related changes but also recurrent tree breakage in edge areas (Nunes et al. 2023). In addition, robust models that include specific environmental variables of edge areas and tree age are needed to better understand the effect of forest fragmentation on tree growth patterns. Furthermore, it is crucial to consider the age of forest edges in future studies, as naturally regenerating vegetation tends to mitigate the influence of edges over the years.

## **5. Conclusion**

Deforestation led to a 22% reduction in the existing knowledge about the occurrence of hyperdominant tree species, resulting in the release of carbon into the atmosphere and consequent negative impacts due to tree removal. In addition, there has been a considerable decrease in the scientific information accessible on these species. In addition, about 35% of the registers are located at the edge of the forest, where they are vulnerable to both ongoing deforestation and edge effects.

Hyperdominant tree communities in the Central Amazon show twice the mortality of trees exposed to forest edges, accompanied by initial growth that is not sustained over time. This scenario suggests a significant carbon loss in these areas, mainly due to unbalanced tree mortality. Furthermore, it is possible that the populations of several hyperdominant species residing along the edges of various parts of the Amazon exhibit similar responses to edge-induced degradation, leading to a reduction in biomass that in general is not accounted for. Thus, this consequence underscores the importance of these dynamic trends in Amazonian carbon estimates and balance. It highlights the need for conservation and management strategies that consider the responses of high-carbon stock trees to environmental change and local disturbance.

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## **Declarations**

**Competing interests** The authors have no competing interests to declare that are relevant to the content of this article

**Ethics approval** Ethics approval was not required for this study according to local legislation

**Consent to participate** Not applicable

**Consent for publication** Not applicable

**Availability of data and material** The datasets used and analyzed during the current study are available from the corresponding author on reasonable request.

**Code availability** Study is available from the corresponding author on reasonable request.

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## CONCLUSÃO

O desmatamento levou à redução de 22% dos registros existentes de ocorrência das espécies arbóreas hiper dominantes, resultando na liberação de C na atmosfera e consequentes impactos negativos devido à remoção de árvores. Além disso, há dessa forma uma diminuição considerável da informação científica disponível sobre estas espécies. Em relação aos registros encontrados em borda de floresta, do total, cerca de 35% encontram-se nessas áreas, vulneráveis tanto ao avanço do desmatamento, quanto aos efeitos de borda.

Referente à dinâmica do estoque de C nas árvores hiper dominantes na Amazônia Central, observamos taxas de mortalidade mais elevadas em áreas de borda florestal e uma tendência de aumento inicial no crescimento após perturbações, mas que não se sustentam ao longo do tempo. Isso indica uma perda no saldo de C estocado nessas áreas, principalmente devido à morte de indivíduos.

Considerando que árvores dessas mesmas espécies enfrentam condições semelhantes de exposição a perturbações em diferentes partes da Amazônia, é plausível supor que várias dessas populações estejam respondendo de maneira semelhante, ocasionando declínio de biomassa nos remanescentes de floresta ou ao longo de qualquer outra borda florestal. Esse fenômeno e a expressiva perda de registros por desmatamento de espécies hiper dominantes no ciclo do C levantam a relevância do impacto dessas tendências nas concentrações de C na Amazônia e destaca a necessidade de estratégias de conservação e manejo que levem em consideração as respostas das árvores de alto estoque de carbono às mudanças ambientais e perturbações locais.

## SUPPLEMENTARY MATERIAL

Table A1. Proportional occurrence of hyper dominant tree species in forest edges, deforested areas and forest areas in the Brazilian Amazon biome.

<b>Species</b>	<b>Deforested (%)</b>	<b>Forest Edge (%)</b>	<b>Forest (%)</b>
<i>Anadenanthera colubrina</i>	54.1	29.7	16.2
<i>Apuleia leiocarpa</i>	34.4	45.5	20.1
<i>Aspidosperma carapanauba</i>	19.1	42.3	38.6
<i>Aspidosperma excelsum</i>	17.6	39.1	43.3
<i>Aspidosperma spruceanum</i>	14.0	40.8	45.2
<i>Astronium lecointei</i>	22.8	39.5	37.6
<i>Balizia pedicellaris</i>	28.1	37.3	34.7
<i>Bertholletia excelsa</i>	33.1	38.2	28.7
<i>Brosimum guianense</i>	15.6	28.4	56.1
<i>Brosimum lactescens</i>	17.9	37.3	44.8
<i>Brosimum rubescens</i>	12.2	26.6	61.2
<i>Brosimum utile</i>	16.7	22.0	61.3
<i>Calycophyllum spruceanum</i>	21.1	39.5	39.5
<i>Carapa guianensis</i>	27.2	34.4	38.4
<i>Carapa surinamensis</i>	15.8	36.8	47.4
<i>Cariniana micrantha</i>	18.3	39.4	42.3
<i>Caryocar glabrum</i>	24.1	35.4	40.4
<i>Caryocar pallidum</i>	19.7	41.6	38.7
<i>Caryocar villosum</i>	29.3	34.7	36.0
<i>Casearia bicolor</i>	25.9	35.8	38.3
<i>Cedrelinga cateniformis</i>	28.3	39.8	31.9
<i>Ceiba pentandra</i>	26.5	38.6	34.9
<i>Ceiba samauma</i>	15.8	51.8	32.5
<i>Chaetocarpus schomburgkianus</i>	21.7	36.8	41.5
<i>Chimarrhis turbinata</i>	22.9	36.9	40.2
<i>Chrysophyllum lucentifolium</i>	27.5	40.7	31.9
<i>Chrysophyllum prieurii</i>	17.4	40.9	41.7
<i>Chrysophyllum sanguinolentum</i>	14.6	25.8	59.6
<i>Clarisia racemosa</i>	18.8	37.3	44.0
<i>Corythophora alta</i>	15.6	35.0	49.5
<i>Couratari guianensis</i>	23.9	36.5	39.7
<i>Couratari stellata</i>	12.1	37.8	50.2
<i>Dialium guianense</i>	26.6	37.4	36.1
<i>Dinizia excelsa</i>	21.9	38.2	40.0
<i>Diploptropis purpurea</i>	26.0	41.0	33.0
<i>Dipteryx magnifica</i>	22.6	36.1	41.3
<i>Dipteryx micrantha</i>	42.3	32.7	25.0
<i>Dipteryx odorata</i>	31.0	33.6	35.5
<i>Duckeodendron cestroides</i>	19.6	33.6	46.7
<i>Ecclinusa guianensis</i>	12.9	29.9	57.2

<i>Endopleura uchi</i>	20.0	32.7	47.4
<i>Eperua leucantha</i>	26.8	31.0	42.3
<i>Eperua purpurea</i>	22.9	41.4	35.7
<i>Erisma uncinatum</i>	27.7	38.4	34.0
<i>Eschweilera atropetiolata</i>	19.1	33.4	47.6
<i>Eschweilera coriacea</i>	21.3	35.5	43.2
<i>Eschweilera cyathiformis</i>	14.2	38.3	47.5
<i>Eschweilera grandiflora</i>	21.8	34.8	43.4
<i>Eschweilera micrantha</i>	24.6	34.1	41.3
<i>Eschweilera parviflora</i>	15.1	34.3	50.7
<i>Eschweilera pedicellata</i>	13.7	39.1	47.3
<i>Eschweilera sagotiana</i>	20.3	30.5	49.2
<i>Eschweilera truncata</i>	22.0	41.3	36.7
<i>Eschweilera wachenheimii</i>	10.3	28.3	61.4
<i>Ficus gomelleira</i>	29.2	46.6	24.2
<i>Geissospermum argenteum</i>	15.6	34.4	50.0
<i>Geissospermum sericeum</i>	31.9	37.0	31.0
<i>Goupia glabra</i>	22.6	35.1	42.2
<i>Guarea kunthiana</i>	32.3	36.8	31.0
<i>Gustavia hexapetala</i>	19.6	35.5	44.9
<i>Handroanthus serratifolius</i>	22.4	48.3	29.4
<i>Helicostylis tomentosa</i>	19.0	37.8	43.2
<i>Hevea guianensis</i>	27.4	35.0	37.7
<i>Hura crepitans</i>	20.8	36.9	42.3
<i>Hymenaea courbaril</i>	38.6	35.6	25.8
<i>Hymenopus heteromorphus</i>	18.1	33.0	49.0
<i>Inga alba</i>	25.0	43.8	31.2
<i>Iriartea deltoidea</i>	21.5	44.9	33.6
<i>Iryanthera juruensis</i>	23.9	33.8	42.4
<i>Iryanthera sagotiana</i>	18.6	35.8	45.6
<i>Jacaranda copaia</i>	25.7	42.8	31.5
<i>Jupunba trapezifolia</i>	36.7	33.3	30.0
<i>Lecythis chartacea</i>	26.3	34.4	39.3
<i>Lecythis confertiflora</i>	18.0	24.4	57.7
<i>Lecythis corrugata</i>	18.0	34.3	47.7
<i>Lecythis idatimon</i>	39.9	30.2	29.9
<i>Lecythis pisonis</i>	33.6	39.0	27.4
<i>Lecythis poiteaui</i>	14.8	31.5	53.7
<i>Lecythis prancei</i>	17.0	31.1	51.9
<i>Lecythis zabucajo</i>	13.8	37.1	49.1
<i>Leonia glycyarpa</i>	27.4	39.2	33.4
<i>Leptobalanus apetalus</i>	22.0	34.2	43.8
<i>Leptobalanus octandrus</i>	29.0	31.6	39.4
<i>Licania alba</i>	27.3	23.4	49.4
<i>Licania canescens</i>	25.9	40.5	33.7
<i>Licania membranacea</i>	25.6	36.6	37.9

<i>Licaria cannella</i>	24.8	27.0	48.1
<i>Luehea cymulosa</i>	27.3	37.6	35.1
<i>Manilkara bidentata</i>	15.8	26.7	57.5
<i>Manilkara elata</i>	22.2	29.7	48.1
<i>Micropholis guyanensis</i>	16.9	30.4	52.7
<i>Micropholis venulosa</i>	19.8	36.8	43.4
<i>Minuartia guianensis</i>	16.8	43.0	40.2
<i>Monopteryx uauacu</i>	22.6	45.2	32.3
<i>Osteophloeum platyspermum</i>	15.5	31.0	53.6
<i>Parkia nitida</i>	29.1	41.2	29.7
<i>Parkia pendula</i>	23.4	50.8	25.9
<i>Poraqueiba guianensis</i>	12.5	21.4	66.0
<i>Pourouma minor</i>	19.6	39.1	41.3
<i>Pouteria anomala</i>	17.2	25.3	57.5
<i>Pouteria caimito</i>	26.7	27.8	45.6
<i>Pouteria eugeniifolia</i>	23.5	29.8	46.7
<i>Pouteria guianensis</i>	14.5	24.9	60.7
<i>Pouteria oppositifolia</i>	34.0	36.2	29.8
<i>Pouteria reticulata</i>	18.8	25.8	55.5
<i>Pouteria torta</i>	27.6	30.3	42.2
<i>Pradosia cochlearia</i>	23.2	27.6	49.2
<i>Protium altissimum</i>	30.8	41.5	27.7
<i>Protium altsonii</i>	22.5	39.4	38.1
<i>Protium stevensonii</i>	20.8	34.6	44.5
<i>Pseudolmedia laevigata</i>	13.2	32.3	54.5
<i>Pseudolmedia laevis</i>	16.3	40.6	43.2
<i>Pseudolmedia macrophylla</i>	26.7	47.1	26.2
<i>Pseudopiptadenia suaveolens</i>	24.7	36.6	38.8
<i>Qualea paraensis</i>	18.6	38.3	43.1
<i>Rinorea guianensis</i>	20.3	37.7	42.1
<i>Robrichia schomburgkii</i>	26.9	39.3	33.8
<i>Ruizterania albiflora</i>	25.7	25.1	49.2
<i>Sacoglottis guianensis</i>	17.8	30.8	51.4
<i>Sapium marmieri</i>	30.8	40.0	29.2
<i>Scleronema micranthum</i>	12.4	30.6	57.0
<i>Sextonia rubra</i>	18.9	32.8	48.3
<i>Sloanea guianensis</i>	18.9	44.5	36.6
<i>Spondias mombin</i>	35.2	32.7	32.0
<i>Sterculia pruriens</i>	24.7	39.5	35.8
<i>Swartzia benthamiana</i>	18.2	34.6	47.3
<i>Swartzia polyphylla</i>	19.7	33.2	47.1
<i>Swartzia reticulata</i>	11.9	32.6	55.6
<i>Symphonia globulifera</i>	22.3	35.3	42.4
<i>Tachigali melinonii</i>	17.7	43.1	39.2
<i>Tachigali paraensis</i>	35.8	45.3	19.0
<i>Tachigali poeppigiana</i>	28.3	39.1	32.6



<i>Tapirira guianensis</i>	29.3	35.5	35.2
<i>Terminalia amazonia</i>	29.4	41.2	29.4
<i>Terminalia grandis</i>	16.2	30.2	53.6
<i>Terminalia guyanensis</i>	32.3	3.2	64.5
<i>Terminalia oblonga</i>	33.0	62.6	4.4
<i>Trattinnickia burserifolia</i>	25.2	34.6	40.1
<i>Trattinnickia glaziovii</i>	21.2	35.4	43.4
<i>Vantanea parviflora</i>	15.7	30.5	53.8
<i>Virola calophylla</i>	22.9	37.2	39.9
<i>Virola michelii</i>	24.2	38.2	37.6
<i>Virola pavonis</i>	18.8	33.3	47.9
<i>Vochysia guianensis</i>	32.3	33.9	33.9
<i>Vouacapoua americana</i>	40.4	32.1	27.5
<i>Zygia racemosa</i>	19.4	33.0	47.6

Table A2. Time effect on RGR of tree species located in Forest Fragments (FF) and Continuous Forest (FC) in Central Amazon. The list showed only the tree species with a significant interaction result ( $p < 0.05$ ).

<b>Species</b>	<b>Slope FF</b>	<b>Slope FC</b>	<b>R<sup>2</sup>m</b>	<b>R<sup>2</sup>c</b>
<i>Corythophora alta</i>	-0.0204	-0.0150	0.07	0.54
<i>Ecclinusa guianensis</i>	-0.0199	-0.0316	0.08	0.5
<i>Eschweilera atropetiolata</i>	-0.0158	-0.0143	0.02	0.44
<i>Eschweilera coriacea</i>	-0.0077	-0.0238	0.05	0.41
<i>Eschweilera grandiflora</i>	-0.0178	-0.0216	0.04	0.34
<i>Eschweilera micrantha</i>	-0.0281	-0.0344	0.07	0.47
<i>Eschweilera truncata</i>	-0.0207	-0.0301	0.06	0.45
<i>Eschweilera wachenheimii</i>	-0.0163	-0.0348	0.08	0.39
<i>Hevea guianensis</i>	-0.0184	-0.0170	0.01	0.56
<i>Micropholis guyanensis</i>	-0.0147	-0.0267	0.06	0.48
<i>Minquartia guianensis</i>	-0.0207	-0.0287	0.06	0.47
<i>Pouteria guianensis</i>	-0.0257	-0.0259	0.04	0.42
<i>Pouteria reticulata</i>	-0.0172	-0.0185	0.02	0.48
<i>Pseudolmedia laevis</i>	-0.0238	-0.0233	0.07	0.57
<i>Swartzia reticulata</i>	-0.0379	-0.0393	0.08	0.43
<i>Protium stevensonii</i>	-0.0208	-0.0295	0.08	0.5
<i>Virola calophylla</i>	-0.0196	-0.0316	0.08	0.44
<i>Zygia racemosa</i>	-0.0211	-0.0306	0.05	0.39

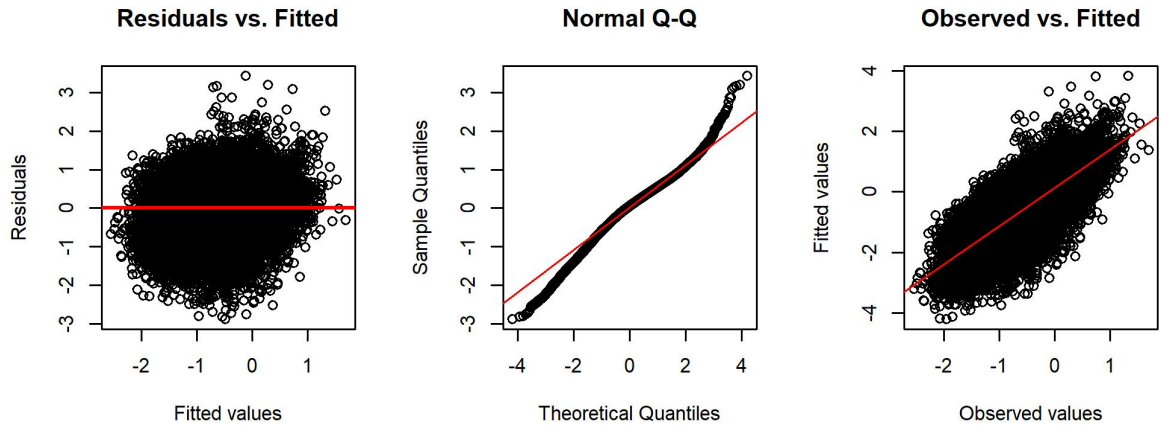


Figure A1. Relationship between residuals and adjusted values, Q-Q normal plot, and the relationship between observed and adjusted data of RGR (% yr<sup>-1</sup>) ~ time \* area + (1 | plot/individual identification (ID) for the community (93 spp.).

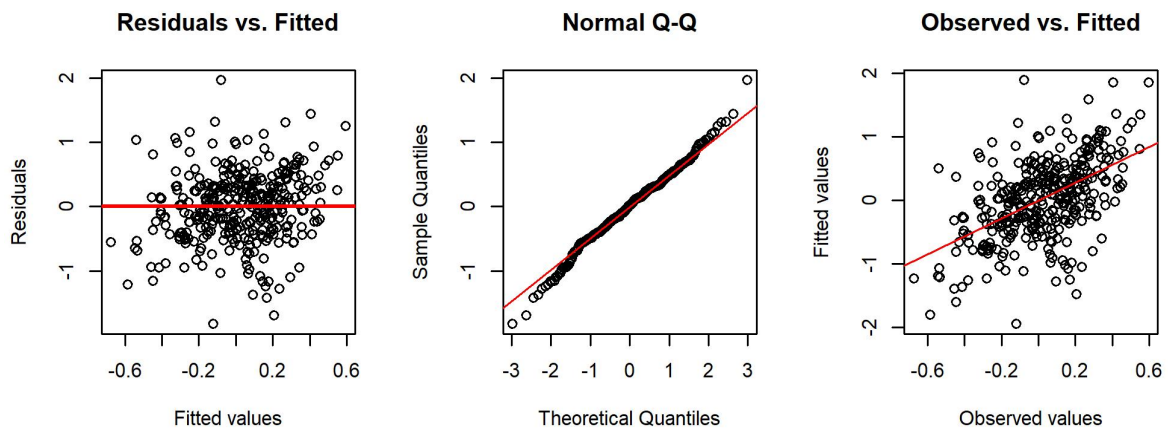


Figure A2. Relationship between residuals and adjusted values, Q-Q normal plot, and the relationship between observed and adjusted data of Mortality (% yr<sup>-1</sup>) ~ time \* area + (1 | plot/individual identification (ID) for the community (93 spp.).

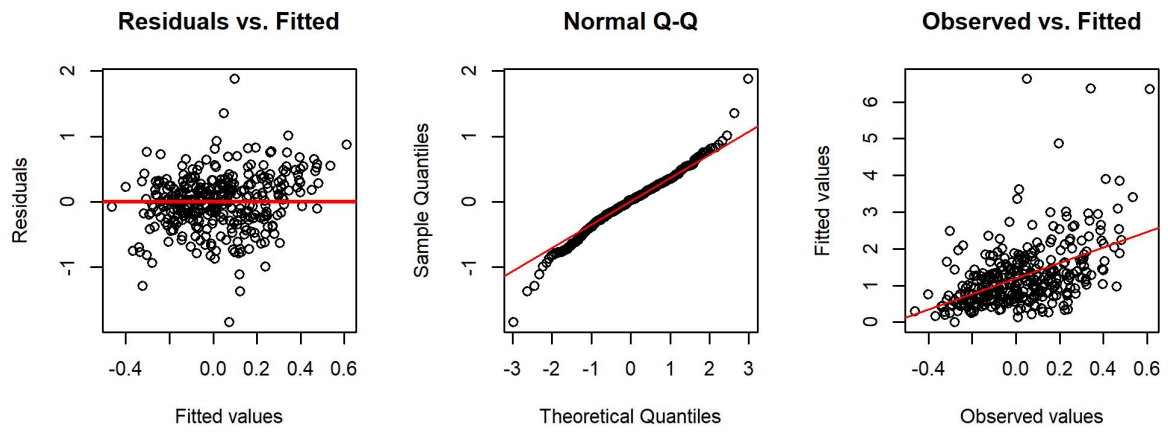
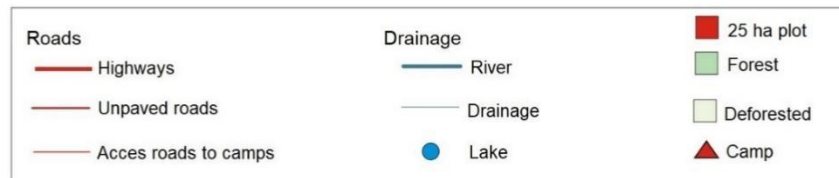
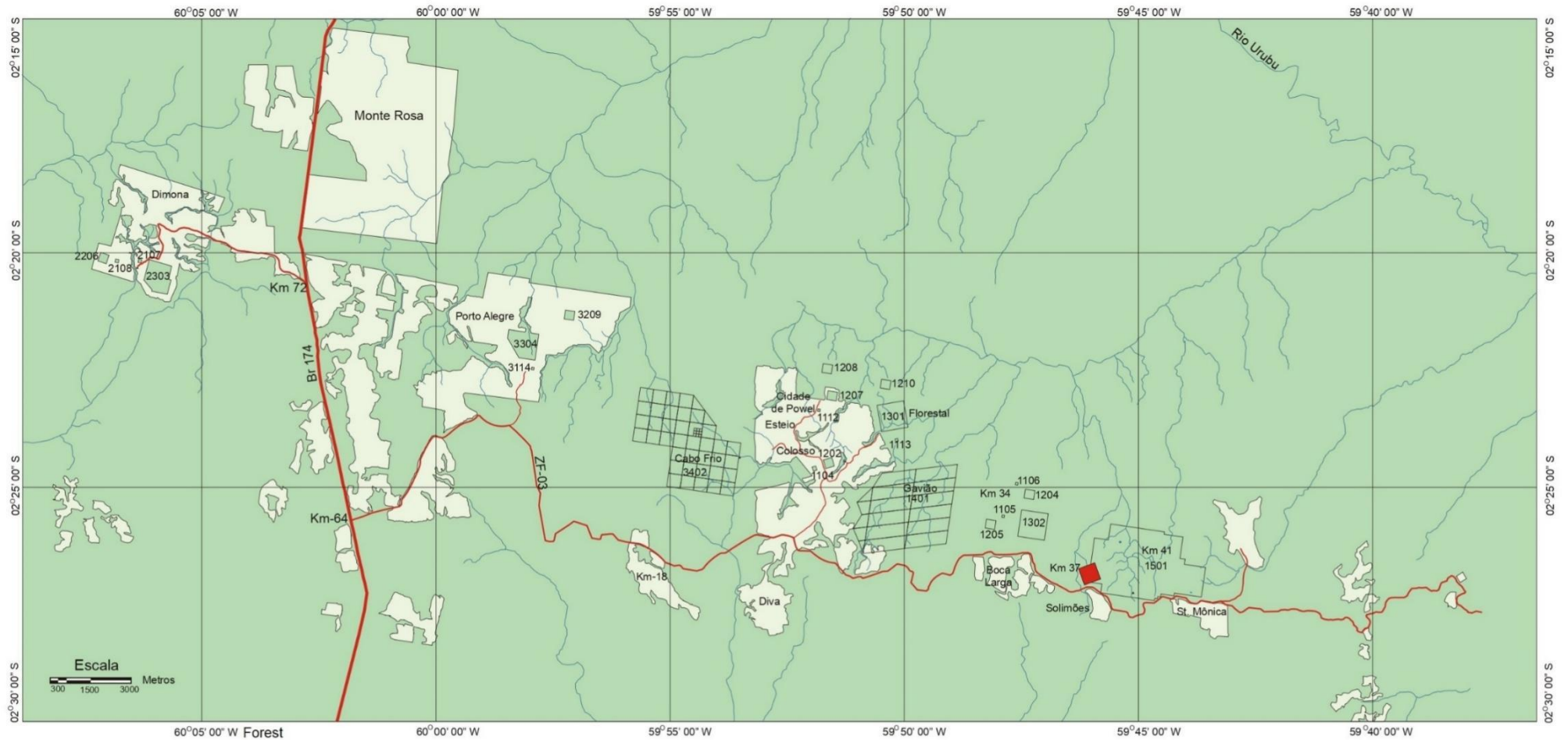


Figure A3. Relationship between residuals and adjusted values, Q-Q normal plot, and the relationship between observed and adjusted data of Turnover (% yr<sup>-1</sup>) ~ time \* area + (1 | plot/individual identification (ID) for the community (93 spp.).



INPE Landsat TM 5,4,3 - RGB, 1995. June, 1998  
 by Venticinque, E.M and Fernandes, T. L. N.

Figure A4. ARIE – PDBFF, located ca. 80 km North of Manaus, Central Amazon, Brazil. Figure adapted from LabTrop USP.