LEVERHULME
Centre for Wildfires, Environment and Society

Imperial College London

Beyond Controlling Deforestation: Addressing Emerging Wildfire Risk in Rapidly Changing Landscapes to Protect Ecosystems and Communities in the Brazilian Amazon

Submitted to Imperial College London for the degree of Doctor of Philosophy

Michel Valette

June 2025

Centre for Environmental Policy, Faculty of Natural Sciences



Statement of Originality

I declare that this thesis is the result of my own work, conducted under the oversight of my main supervisor, Dr Morena Mills, and my second supervisor, Prof. Jayalaxshmi Mistry. When building on the work of others, this is appropriately referenced or acknowledged. The list of co-authors for publication prepared from the work of this thesis is detailed in the thesis outline (section 1.5). The illustration on the cover page, in Chapter 4 and the design of the poster in supplementary appendices from Chapter 4 was done by Manini Bansal.

Copyright declaration

The copyright of this thesis rests with the author. Unless otherwise indicated, its contents are licensed under a Creative Commons Attribution-Non-Commercial 4.0 International Licence (CC BY-NC). Under this licence, you may copy and redistribute the material in any medium or format. You may also create and distribute modified versions of the work. This is on the condition that: you credit the author and do not use it, or any derivative works, for a commercial purpose. When reusing or sharing this work, ensure you make the licence terms clear to others by naming the licence and linking to the licence text. Where a work has been adapted, you should indicate that the work has been changed and describe those changes. Please seek permission from the copyright holder for uses of this work that are not included in this licence or permitted under UK Copyright Law.

Acknowledgement

I would like to take this unique opportunity to thank all of the people who have contributed to my personal and professional journey in environmental sciences.

First of all, I am extremely grateful to Dr Morena Mills for your supervision over the past years, the time you dedicated to giving me feedback, the opportunity to undertake fieldwork with you and your support whenever it was needed. I am also deeply indebted to Prof. Jayalaxshmi Mistry, not only for trying to bring the best out of every early career researcher at the centre, but also for the time devoted to discussing our research project and my general approach to research and fieldwork. I am also thankful to Dr Adriana Ford, who has been an amazing and supportive manager for the Leverhulme Wildfires Centre, helping the cohort of PhD students as much as she can. I would like to extend my sincere thanks to Prof. Jeremy Woods for your ability to make me see the bigger picture of my thesis, as well as Dr Kountouris, Dr Freni-Sterrantino and Prof. James Millington for your valuable feedback during the early stages of my thesis.

I am also deeply indebted to Roiti Metuktire, Kokokrotire Metuktire, Yarika Juruna and Juliana Bonanomi for your warm welcome in Brazil and in Capoto/Jarina, and your continuous support throughout this project. You have always been patient with my silly questions and given so much of your energy to make all of this possible. I also could not have made it without your experience in organising the logistics for fieldwork and your expertise in the ecosystems of the Xingu region. I should extent my gratitude to all the participants of Piaraçu, Capoto and Mupa for their warm welcome inside their communities and the time devoted to participating in this research project.

I would like to express my deepest appreciation to my former supervisor and mentor from Bioversity International, Dr Riina Jalonen. Your mentorship has been precious to me, and I am not sure I would have persevered in conservation without the long discussions we had during our drives between UPM and Bukit Bintang or the amazing research opportunities you provided me with prior to my PhD. I am also thankful to Dr Barbara Vinceti for the wonderful opportunity you offered me to work in Burkina Faso, giving me a chance to learn about the local ecological knowledge of farmers and nursery owners in the Plateau Central.

I am also extremely grateful to all of my colleagues who became close friends along the way: Abha, Amira, Ana, Alvaro, Aurore, Clémentine, Hermann, Jean-François, Juliana, Max, Sarah, Tania, Thibaut and Vincent. I feel extremely lucky to be around people with whom I can discuss ways to make this world a fairer and greener place. I am sure we will have many other opportunities to meet, and hopefully collaborate, during our future careers.

I would also like to extend my thanks to my colleagues from the Leverhulme Wildfires Centre, David, Cathy, Farrer, Haleema, Henri, Jose, Kapil, Kayla, Matt, Ol, Olivia, Sophie and Will, for the discussions we had about my project around a cup of coffee. This has truly helped me to dive into the world of fire sciences and explore its multidisciplinary aspect.

I would be remiss not to mention my former biology teacher, Mr Delangle, for your dedication to providing high-quality education to people from my hometown in France and for having pushed me to go to university, and my former fly-fishing coach, Mr Vincent, for your time teaching teenagers about the wonderful freshwater ecology of French rivers. I should also extend my thanks to my dear friends Louise, Nicolas, William, and Selene for reminding me that there is more to life than our work and always be there for me.

Words cannot express the gratitude that I have for my family, who have helped me to become the person I am today. I have two generations to look up to, who provided me with a constant example of dedication through volunteering and who taught me to be kind, try to help others, and always reach out to people who need it most. The time we spent together going hiking, mushroom picking, or tending the garden and orchards contributed a lot to my career choices, and your constant support encouraged me to pursue my studies as much as I could. I should especially thank my parents, who have supported me through my studies and provided me with an amazing work environment during the initial months of my thesis, amidst the COVID lockdown, as well as during the final months of writing.

Finally, words cannot express my gratitude to Charmaine, my beloved wife. Even before the beginning of this doctoral journey, you believed in me even when I didn't and pushed me to be the best version of myself. I still am not sure how you put up with me through these years and the many late-night sessions tapping on my keyboard in our small room in London, but I feel like the luckiest person on earth to have shared these years with you.

"There is in all things a pattern that is part of our universe. It has symmetry, elegance, and grace [...] Yet, it is possible to see peril in the finding of ultimate perfection. It is clear that the ultimate pattern contains its own fixity. In such perfection, all things move toward death." Frank Herbert, Dune

Abstract

The Amazon rainforest is threatened simultaneously by climate change, deforestation and forest degradation. While deforestation rates have decreased over the last two decades, wildfires remain a major source of environmental degradation, jeopardizing biodiversity, carbon stock and the well-being of local communities. While historically fires were mostly linked to deforestation, they are increasingly driven by climatic and ecological factors altering wildfire risk. In this thesis, I aim to understand processes driving fire regimes across multiple scales and how local landholders perceive and respond to changing wildfire risks. At regional and subregional levels, I used modelling and quantitative analysis to identify drivers of fire regimes and wildfire risks. I complemented this work with local scale analysis, using participatory methods to investigate local fire use and associated knowledge as well as adaptation to changing wildfire risks across three communities of Capoto/Jarina Indigenous land. The results show that fires are becoming increasingly frequent in remote parts of the region, but protected areas and Indigenous lands play a key role in safeguarding remaining forests. While climate change increases wildfire risk over the region, land governance shapes local wildfire risks related to land use change and affects the potential for landholders to adapt to wildfire risk, with Indigenous land experiencing the most stable fire regimes. Indigenous communities of Capoto/Jarina mobilise extensive local ecological knowledge to support their fire use and adapt to changing conditions through a wide array of individual and collective adaptation strategies. These results highlight how the interplay between climatic, ecological and socio-economic factors operating at various scales shape not only fire regimes but also adaptive pathways to changing wildfire risk adopted by local landholders. This thesis demonstrates the importance of cross-scalar research efforts and the integration of local fire knowledge for fair and efficient fire management in an increasingly flammable world.

Resumo

A floresta amazônica está ameaçada simultaneamente pelas mudanças climáticas, pelo desmatamento e pela degradação florestal. Embora as taxas de desmatamento tenham diminuído nas últimas duas décadas, os incêndios florestais continuam sendo uma das principais fontes de degradação ambiental, colocando em risco a biodiversidade, o estoque de carbono e o bem-estar das comunidades locais. Embora historicamente os fogos estivessem principalmente ligados ao desmatamento, eles são cada vez mais impulsionados por fatores climáticos e ecológicos que alteram o risco de incêndios florestais. Nesta tese, pretendo compreender os processos que impulsionam os regimes de fogo em várias escalas e como os proprietários locais percebem e respondem às mudanças nos riscos de incêndios florestais. Nos escalões regional e sub-regional, utilizei modelagem e análise quantitativa para identificar os fatores que impulsionam os regimes de fogo e os riscos de incêndios florestais. Complementei este trabalho com uma análise à escala local, utilizando métodos participativos para investigar o uso local do fogo e os conhecimentos associados, bem como a adaptação às mudanças nos riscos de incêndios florestais em três comunidades da terra indígena de Capoto/Jarina. Os resultados mostram que os incêndios estão se tornando cada vez mais frequentes em partes remotas da região, mas as áreas protegidas e as terras indígenas desempenham um papel fundamental na proteção das florestas remanescentes. Embora as mudanças climáticas aumentem o risco de incêndios florestais na região, a governança da terra molda os riscos locais relacionados à mudança no uso da terra e afeta o potencial dos proprietários de terras para se adaptarem ao risco de incêndios florestais, com as terras indígenas apresentando os regimes de incêndios mais estáveis. As comunidades indígenas de Capoto/Jarina mobilizam um vasto conhecimento ecológico local para apoiar o uso do fogo e se adaptar às condições em mudança por meio de uma ampla variedade de estratégias de adaptação individuais e coletivas. Esses resultados demonstram como a interação entre fatores climáticos, ecológicos e socioeconômicos que operam em várias escalas molda não apenas os regimes de fogo, mas também os caminhos de adaptação às mudanças no risco de incêndios florestais

adotados pelos proprietários de terras locais. Esta tese demonstra a importância dos esforços de pesquisa em múltiplas escalas e da integração do conhecimento local sobre o fogo para uma gestão justa e eficiente do fogo em um mundo cada vez mais inflamável.

Table of Contents

Statement of Originality 2
Copyright declaration 2
Acknowledgement 3
Abstract 5
List of figures 10
List of tables 13
Chapter 1: Introduction 14
1.1 Introduction 14
1.2 Research background 16
1.3. Research gaps and research objectives 33
1.4. Conceptual framework 36
1.5. Thesis outline 41
1. 6. Positionality statement 43
Chapter 2: Regional drivers of fire regimes in the Brazilian Amazon between 2009 and 2021 47
2.1. Main contributions 47
2.2 Introduction 48
2.3 Methods 51
2.4. Results 57
2.5 Discussion 64
2.6. Conclusion 70
Chapter 3: Comparative analysis of wildfire risks and fire regime dynamics in the North of Mato Grosso 71
3.1 Main contributions 71
3.2 Introduction 72
3.3 Methods 74
3.4 Results 82
3.5 Discussion 99

3.6 Conclusion 103
Chapter 4: Understanding fire use and management in an Indigenous land: a case study from the Capoto/Jarina 104
4.1. Main contributions 104
4.2. Introduction 105
4.3. Methodology 107
4.4 Results 117
4.5 Discussion 130
4.6. Conclusion 134
Chapter 5: Adaptation to changing wildfire risks: insights from Capoto/Jarina 136
5.1 Main contributions 136
5.2 Introduction 137
5.3 Methodology 139
5.4. Results 143
5.5 Discussion 165
5.6 Conclusion 169
Chapter 6. Discussion 171
6.1 Research gaps 171
6.3 Limitations 190
6.4 Future Research Directions 192
6.5 Summary 194
References 197
Appendices 224
Appendices from Chapter 2 224
Appendices from Chapter 3 253
Appendices from Chapter 4 269
Appendices from Chapter 5 297

List of figures

Figure 1. Satellite image of a deforestation frontier in the state of Acre main types of fires - page 18

Figure 2. Deforestation rate in the Legal Amazon (PRODES) and annual numbers of active fires detected by satellites (MODIS Active-Fires) – page 23

Figure 3. Conceptual framework develops to describe the adaptation of local communities to changing wildfire risk, mobilising local ecological knowledge – page 37

Figure 4. Map of the most frequent types of fires detected within 5km cells across the Brazilian Amazon – page 52

Figure 5. Number of active fires detected by MODIS and annual deforestation detected by PRODES in km^2– page 57

Figure 6. Posterior means and 95% credible intervals for explanatory variables of the models for deforestation fires in the Brazilian Amazon (log scale) – page 59

Figure 7. Posterior means and 95% credible intervals for explanatory variables of the models for agricultural fires in the Brazilian Amazon (log scale) – page 61

Figure 8. Posterior means and 95% credible intervals for explanatory variables of the models for forest fires in the Brazilian Amazon (log scale) – page 63

Figure 9. Map of the region of interest with the main land tenure categories and number of fires detected by MapBiomas Fire (collection 3) – page 75

Figure 10. Change in the number of days with FWI exceeding extreme value (FWI 95), annual maximum FWI (FWI max), length of the fire season (FSL) and maximum 90-day mean value (FWI mean) – page 83

Figure 11. Histogram of the percentage of land use for each land governance type between 1985 and 2023 – page 84

Figure 12. Violin plots of the 4 forest fragmentation metrics in 2022 (left) and boxplot of the rate of change across each decade (right) – page 86

Figure 13. Total areas burned each year (bars) and number of Active Fires detected across 1000 km^2 in each of the land governance types between 2002 and 2023 – page 87

Figure 14. Histogram of the percentage of the total burned area in different classes of fire size – page 88

Figure 15. Daily percentage of the total of active fires detected (top) and percentage of the total of burned areas detected each month across the land governance types (bottom) – page 90

Figure 16. Smooth estimate of the effect of day on the percentage of active fires according to the land governance type – page 91

Figure 17. Percentage of the total annual burned areas covered by different categories of land cover between 2002 and 2023 – page 92

Figure 18. Smooth estimate of the effect of maximum FWI (FWI max) on the percentage of total area burned in fires of less than 100 hectares – page 94

Figure 19. Smooth estimate of the effect of 90-day mean FWI (FWI mean) on the percentage of total area burned in fires between 100 and 1000 hectares – page 95

Figure 20. Smooth estimate of the effect of 90-day mean FWI (FWI mean) on the percentage of total area burned in fires of more than 1,000 hectares – page 95

Figure 21. Smooth estimate of the effect of 90-day mean FWI (FWI mean) on the percentage of total area burned in fires of more than 10,000 hectares – page 96

Figure 22. Smooth estimate of the effect of maximum FWI (FWI max) on the percentage of burned areas covered by forest – page 97

Figure 23. Smooth estimate of the effect of maximum FWI (FWI max) on the percentage of burned areas covered by other natural vegetation – page 98

Figure 24. Smooth estimate of the effect of 90-day mean FWI (FWI mean) on the percentage of burned areas covered by cropland – page 98

Figure 25. Map of the Capoto/Jarina Indigenous land, with the three communities visited – page 109

Figure 26. Illustration derived from participatory mapping exercise in the village of Piaraçu including all the sites discussed in the results. This illustration groups the insight from the participatory map of both women and men (credit: Manini Bansal) – page 125

Figure 27. Participatory map from fire use and wildfire risks in Mupa – page 128

Figure 28. Participatory map of fire use and wildfire risks in Capoto done by male participants and grouping insights from male and female participants – page 129

Figure 29. Band plots showing the daily average Canadian Weather Fire Index for the 2001-2022 period (bottom) and the average change compared to the reference period between 1951 and 2000 – page 144

Figure 30. Alluvial plot of the land use change in the 10km buffer zone of unprotected land around Capoto/Jarina and map of the land cover in 2023 – page 146

Figure 31. Alluvial plot of the land use change in the 3km buffer zone around Piaraçu and map of the land cover in 2023 – page148

Figure 32. Alluvial plot of the land use change in the 3km buffer zone around Mupa and map of the land cover in 2023 – page 149

Figure 33. Alluvial plot of the land use change in the 3km buffer zone around Capoto and map of the land cover in 2023 – page 150

Figure 34. Secondary forest in process of restoration to reduce wildfire risks – page 159

List of tables

Table 1. Drivers and explanatory variables identified, associated data sources and categories used – page 54

Table 2. Results of the ANOVA test comparing the performance of the model with and without interactions between the explanatory variable and the land governance types – page 93

Table 3. Table resuming the first and second levels of codes used for the analysis of the corpus of text compiled – page 117

Table 4. Typology of the different use of fires developed based on interviews from Piaraçu, Mupa and Piaraçu – page 120

Table 5. Resuming the representation of different processes influencing fire regimes obtained through different methods and scales of analysis through the thesis – page 185

Chapter 1: Introduction

1.1 Introduction

Across the world, fire regimes have been changing rapidly over recent decades: there has been a marked decline in the total burned area detected through satellites, mainly driven by a decrease in savannas and grasslands (Andela et al. 2017). However, analyses of land cover burned, and global fire emissions show that this decline in overall burned area is concomitant with an increase in fires in forested landscapes and associated carbon emissions (Zheng et al. 2021; Jones et al. 2024). The number of very large fire events has also increased, leading to the emergence of the concept of "mega-fire" in the literature (Linley et al. 2022; Armenteras and De La Barrera 2023). Changing fire regimes currently jeopardise approximately 15% of the species classified as threatened (Kelly et al. 2020), and also has important economic cost through infrastructural damages, affects respiratory and cardiovascular health and could result in fatalities (Reid et al. 2016; Ribeiro et al. 2020; Chen et al. 2021).

These global shifts in fire regimes result from the compounded impact of climate, land use and fire use and suppression practices (Rogers et al. 2020; Pausas and Keeley 2021). Globally, climate change has led to an increase in fire-prone conditions (Jolly et al. 2015; Abatzoglou et al. 2019). South America in general, and the Amazon region in particular, have experienced a particularly acute increase in wildfire risks. While El Niño continues to drive interannual variability in rainfall (Feron et al. 2024), climate change has increased the number of nights when climatic conditions permit fires to continue burning, thereby increasing wildfire risks, night-time fire intensity, and the size of wildfire events (Balch et al. 2022).

In addition to climatic changes, human activities have long shaped fire regimes, both directly through fire use and suppression, and indirectly through land-use change (Bowman et al. 2011; Pausas and Keeley 2021). Recent advances in the representation of anthropic influence on fire regimes suggest that about half of the global burned area is composed of managed fires, lit and controlled to achieve specific landscape management objectives (Perkins et al. 2025). However, the proportion of areas burned through managed fires is declining steeply over Latin America, and inter-annual variation in burned areas is mainly due to unmanaged fires (Perkins et al. 2025).

Most of these controlled fires are set by local and Indigenous communities to manage their landscapes, despite frequent legislative restrictions on fire use (Smith et al. 2025). Traditional and local communities are mobilising extensive local ecological knowledge to decide when, where and how to burn to minimize unwarranted impact on ecosystems (Huffman 2013). Regardless, many policies initiatives have attempted to reduce or control fire use by local communities, often with little effectiveness and sometimes with severe adverse effects on local fire use and the community depending on them (Smith et al. 2024; Smith et al. 2025). The failure of approaches to fire management focusing on fire criminalization and wildfire suppression has encouraged the recent emergence of alternative strategies based on the proactive management of wildfire risks and the inclusion of local fire knowledge, such as integrated fire management or intercultural fire management (Tedim et al. 2020; Bilbao et al. 2021; Oliveras Menor et al. 2025).

Like other regions of the world, the Brazilian Amazon has experienced extensive changes in its fire regimes due to the combined impact of land use change, growing human activity, and an increasingly dry climate (Ferreira Barbosa et al. 2021; Jones et al. 2022; Silveira et al. 2022). While the relative influence of the drivers of fire regimes has changed over the last 20 years (Silveira et al. 2020; Libonati et al. 2021), fires are affecting an increasing proportion of forests (Tavares et al. 2022) and have become the major source of regional carbon emissions (Tyukavina et al. 2017; Csillik et al. 2024).

Faced with recurring wildfire crises in the Amazon and other biomes of the country (Fidelis et al. 2018; Garcia et al. 2021), the Brazillian federal government enacted an Integrated Fire Management law in 2024 (Oliveras Menor et al. 2025). Integrated Fire Management is based on fire prevention, response and recovery and aims to integrate local fire ecology and cultural fire practices (Myers 2006; Oliveras Menor et al. 2025). However, there is still limited understanding of how drivers of fire regimes operating across multiple scales are shaping local fire regimes, as well as the diversity of relationships between people and fire in this ecologically and culturally mega-diverse region. Understanding the adaptation strategies adopted by diverse type of landholders could support not only the implementation of Integrated Fire Management in Brazil but also inform broader global efforts to respond to a changing and increasingly flammable planet.

In this chapter, I will give some context about fire regimes and their drivers in the Brazilian Amazon, the region of study of this thesis[1], in section 1.2. In Section 1.3, I will explain the research gaps and research objectives that will be addressed through this thesis, before presenting in Section 1.4 the conceptual framework developed to understand the adaptation of local communities to changing wildfire risk. I will present an outline of the thesis in Section 1.5, before concluding this chapter with my positionality statement in Section 1.6.

1.2 Research background

In the following sections, I will describe the most common types of fires used across the Brazilian Amazon and review their main impacts on the ecosystems, climate and inhabitants of the Brazilian Amazon. Then, I will discuss the main drivers of fire regimes that have been identified in the literature and future projections of regional fire activity. Finally, I will describe the main environmental policies that influenced fire regimes in the region.

[1] Throughout the thesis, I will use the term Brazilian Amazon as a synonym for the intersection between the Brazilian national territory and the Amazon biogeographic province. Thus, it excludes areas of the Pantanal and Cerrado biomes included in the Legal Amazon definition used by the Brazilian government.

1.2.1 Fires in the Brazilian Amazon and their impact

Different types of fires in the Brazilian Amazon

In the Brazilian Amazon, fires are used for a wide range of purposes by various actors, from deforestation to land management and traditional practices(Barlow et al. 2020; Xu et al. 2021; Andela et al. 2022). Different fire types can be driven by distinct processes and have unique impacts on the environment and society (Sorrensen 2009; Mistry and Bizerril 2011). Deforestation fires are lit on areas where the vegetation has been previously felled during the dry seasons to clear new fields, while agricultural fires are employed to remove crop residues and regrowing vegetation, as well as to manage soil fertility on pastures and croplands (Morton et al. 2008; van Vliet et al. 2013). On the same parcel of land, deforestation fires will be used during the initial clearing process, while agricultural fires will be used on already deforested parcels, regardless of whether fires were used or not during land clearing. Fires are important for assuring the economic viability of agricultural systems among landholders with restricted access to agricultural inputs, such as labour, fertilizers and pesticides (Bowman et al. 2008; Carmenta et al. 2019). Deforestation fires are often characterized by longer persistence and higher maximum intensity, due to the large quantity of fuel burned, while agricultural fires are often characterized by shorter duration of burn and lower intensity due to the low quantity of fuel burned (Libonati et al. 2021; Andela et al. 2022). Additionally, many other types of fires are used by traditional and Indigenous communities living in the Brazilian Amazon for diverse objectives, such as traditional hunting practices, cleaning of the landscape, honey collection or stimulating tree fructification (Pivello 2011; Eloy et al. 2021). Controlled fires used for deforestation and subsequent management of agricultural land frequently escape into nearby forests and savannas resulting in forest fires (Cano-Crespo et al., 2015, Figure 1). These fires are often burning at a low intensity but can spread over large areas and keep burning for long periods(Libonati et al. 2021; Andela et al. 2022).



Figure 1. Satellite image of a deforestation frontier in the state of Acre (Google Earth 2022) showing recently deforested areas (stump logo) and potential location of agricultural fires (yellow fire logo), deforestation fires (red fire logo) and forest fires (green fire logo).

Impact of fires in the Brazilian Amazon

Fires in the Brazilian Amazon have ecological, socio-economic and climatic impacts. When fires spread into the forest, they result in substantial post-fire tree mortality, especially when they occur during drought conditions (Balch et al. 2011; Brando et al. 2014). The mortality of large trees is frequently delayed by several years after the fire event (Barlow et al. 2003a) but nonetheless severely affects forest structure due to foliage loss, increased light penetration, and dead fuel accumulation (Brando et al. 2014; Balch et al. 2015; East et al. 2023). The alteration of the canopy also modifies the recruitment dynamics and can lead to shifts in the floristics composition, towards an increase in the dominance of pioneer species (Barlow and Peres 2008) and grass species (Silvério et al. 2013). Fires also impact faunal abundance and diversity, including essential ecological guilds such as the dung beetle community,

which may affect the functional diversity of burned forest (Silveira et al. 2016; de Andrade et al. 2020). Wildfires in the Amazon had a significant impact on the biodiversity of threatened species in the region: around 80% of the threatened species have part of their distribution range affected by fires in the last 25 years (Feng et al. 2021).

Past fires tend to have a cumulative impact, and the occurrence of several fires on the same site severely affects the diversity of the vegetal community and structural integrity of the forest (Xaud et al. 2013; Balch et al. 2015). The repeated impact of fires on forests can lead to substantial losses in forest coverage across large regions (Flores et al. 2016; Silvério et al. 2022). However, frequent fires can also reduce the intensity of future burning by reducing litterfall and the quantity of fuel available for future burning (Balch et al. 2008).

Post-fire tree mortality rates vary considerably across different regions of the Brazilian Amazon and could be affected by past fire regimes and resulting selective pressures, which favour forests with fire-resistant species in some areas (Barlow et al. 2003b; Balch et al. 2011; Xaud et al. 2013). The impact of fires on tree mortality also depends on the type of forest: seasonally flooded forests experience a mortality rate up to four times higher than upland forests (De Resende et al. 2014), as well as a stronger reduction in leaf area index than uplands forest (Almeida et al. 2016). Similarly, the biodiversity hosted in different types of forests and regions of the Brazilian Amazon seems to have distinct responses, with certain vegetation types being more resilient (Barlow et al. 2012).

Fires burning in the region emit significant quantities of carbon, counteracting potential absorption by the remaining forest (Aragão et al. 2018; Celis et al. 2023). During droughts years and in the most degraded areas of the Brazilian Amazon, fires are already representing the highest sources of carbon emissions in the region (Tyukavina et al. 2017; Gatti et al. 2021; Csillik et al. 2024). Fires used for the management of old pastures also prevent both the accumulation of humus in the soils and the regrow of secondary forests, which could represent a large carbon pool at a regional level (Stahl et al. 2017; Heinrich et al. 2021). There are considerable carbon emissions originating from post-fire tree mortality, their

decomposition and the eventual burning of this biomass during repetitive fire events (Vasconcelos et al. 2013; Silva et al. 2018).

Uncontrolled fires also bear an economic cost for landholders involved in pastoralism and agricultural production, as they can destroy agricultural assets such as fences used for pastoralism in the region and subsistence crops (de Mendonça et al. 2004; Campanharo et al. 2019; Carmenta et al. 2021). The recurrence of wildfires also prevents landholders from investing in fire-sensitive intensive agricultural systems that are vulnerable to wildfires, such as agroforestry (Cammelli et al. 2019; Cammelli et al. 2020). The concern about escaped fires and the potential damage to agricultural systems and infrastructure can also generate constant feelings of anxiety (Carmenta et al. 2021). Uncontrolled fires burning in the forest damage timber trees, which are sold at reduced prices and induce significant economic loss, especially in areas close to milling centres (De Oliveira et al. 2019), as well as reduce the availability of non-timber forest products that can represent an important source of income and contribute to livelihoods (Costa et al. 2023). Finally, fires in the region release important quantities of smoke that affect the health of local communities, especially children and elderly people, and results in increased hospitalization rates and premature deaths (Jacobson et al. 2014; Rocha and Sant'Anna 2022; Ribeiro et al. 2024).

1.2.2 Main drivers of fire regimes in the Brazilian Amazon and predicted future fire regimes

Fire regimes can be defined as a "group of parameters describing which fires occur when and where according to frequency, size, seasonality, intensity and type", such as their seasonality, fire return interval or fire size (Krebs et al. 2010). Fires regimes of the Brazilian Amazon are driven by climatic, socio-economic and ecological patterns that influence both the use of fires and the flammability of the natural ecosystems (Libonati et al. 2021, Silveira et al. 2020). I will provide an overview of the literature looking at the impact of climatic conditions, deforestation dynamics, agricultural land use, forest degradation and local fire governance on fire regimes and predictions of future fire activity build on this understanding of fire regimes

While past research identified a wide diversity of drivers influencing fire regimes across different scales, most of the literature presents one of the two challenges preventing a comprehensive understanding of process driving fire regimes: either they assess the impact of a driver of fire regimes in isolation, which might overlook some important interactions with other drivers of fires regimes (Morton et al. 2008, Silva-Junior et al. 2022,) or they look at different types of drivers but conflate different types of fires into one category (Fonseca et al. 2017, Silveira et al. 2020).

Climatic conditions

There is a strong association between annual precipitation and fire occurrence within the Brazilian Amazon (Aragão et al. 2007; Soares-Filho et al. 2012; Fonseca et al. 2017; Silveira et al. 2020). Rainfall is an important determinant of understory humidity, which affects the likelihood of rainforests burning in the presence of ignition points (Ray et al. 2005). While most of the rainforests in the region are usually too humid to burn, El Niño events, the Pacific Decadal Oscillation, and the Atlantic Multidecadal Oscillation trigger periodic droughts, considerably increasing their flammability and associated number of fires (Aragão et al. 2007; Fonseca et al. 2017; Aragão et al. 2018). Dense rainforests appear to be more susceptible to these extreme climatic events than transitional and open forests located in the drier parts of the region (Alencar et al. 2015). Moreover, the intensity and duration of the fire seasons have rapidly increased across the region, raising the number of days when forests are susceptible to catch fires (Abatzoglou et al. 2019; Jones et al. 2022), and the flammability during the night-time (Balch et al. 2022). Prolonged droughts also have an indirect effect on fire likelihood and intensity, as trees lose branches and leaves, resulting in an accumulation of fuel, an opening of the canopy, greater penetration of solar radiation and ultimately more intense fire and higher post-fire mortality (Nepstad et al. 2008; Brando et al. 2014). High-intensity fires triggered by dry climatic conditions can increase the likelihood of invasions by exotic grasses, further increasing wildfire risks (De Faria et al. 2021). However, a chronic water deficit may also limit the regrowth of the vegetation and contribute to fuel scarcity, potentially limiting subsequent wildfires (Silveira et al. 2020).

Deforestation dynamics

Deforestation fires have played and continue to play a key role in the fire activity of the region, as they are still used to burn felled vegetation (Libonati et al. 2021; Andela et al. 2022). The number of fires used for deforestation may be related to future land use, with more frequent fires during direct conversion from forest to cropland (Morton et al. 2008). Recent analysis also points to an increased fire probability prior to the land-clearing process in some regions of the Brazilian Amazon, especially when the land is converted to pasture (Ribeiro et al. 2024).

Large-scale deforestation in the Brazilian Amazon started in the 1970s, with the construction of the first highways, and increased rapidly in the 1990s when the land reform attracted a massive influx of migrants (Laurance et al. 2002; Fearnside 2005; Bidone and Kovacic 2018). In 2004, deforestation rates reached a peak at 27,000km^2 before a decrease due to a mix of initiatives from the private sector, policies to slow down deforestation and a drop in the price of soy and beef (Nepstad et al. 2014; Assunção et al. 2015). Since 2012, deforestation has been increasing again due to the erosion of environmental protection, accelerating since 2016 and reaching more than 11,600 km2 in 2022, and declining again (PRODES, 2024). Deforestation in the Amazon is linked to the expansion of agricultural frontiers and associated land speculations, and most of it occurred along the so-called “arc of deforestation” comprising the states of Maranhão, Tocantins, Pará, Mato Grosso, and Rondônia, though new agricultural frontiers are opening within the Central and Western Amazon (Barona et al. 2010; Sathler et al. 2018; Schielein and Börner 2018).

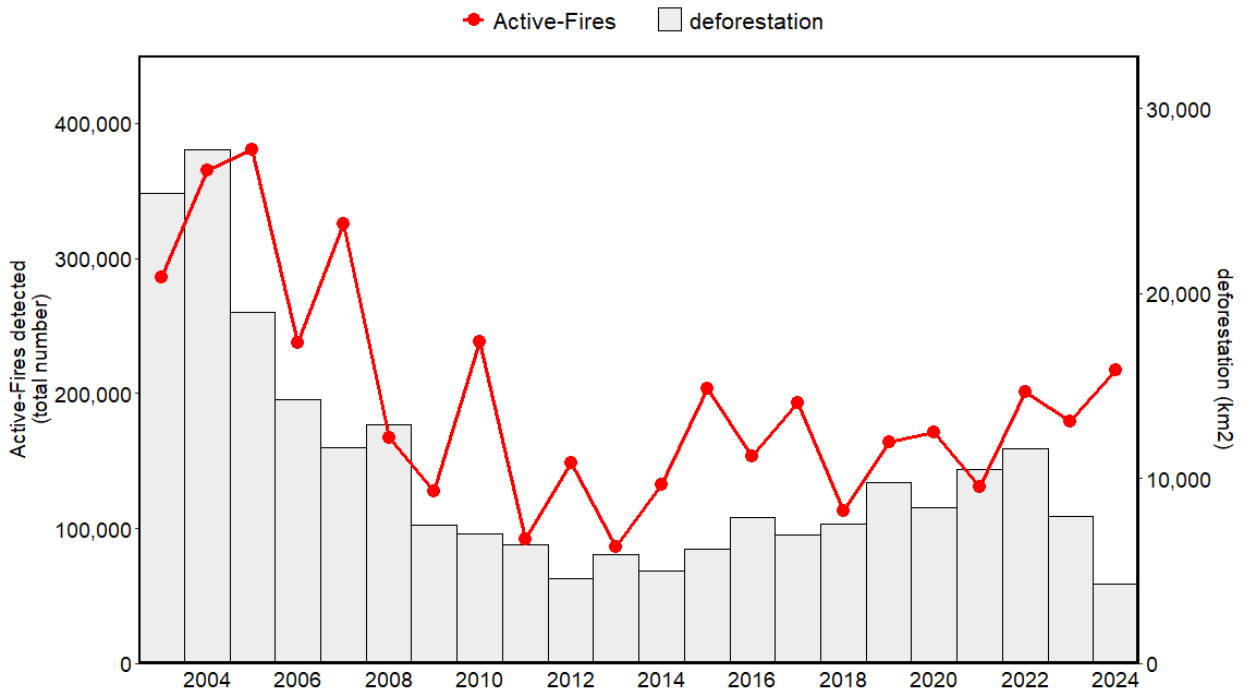


Figure 2. Deforestation rate in the Legal Amazon (PRODES) and annual numbers of active fires detected by satellites (MODIS Active-Fires)

Since 2004 and the first policy efforts to reduce regional deforestation, a decoupling between deforestation rate and satellite detection of fires was observed, partly due to the persistence of agricultural fires and the increased flammability of ecosystems (Figure 2, Aragão & Shimabukuro, 2010; Tasker & Arima, 2016; Libonati et al., 2021). However, the impacts of policies for reducing deforestation are more complex than just a reduction of deforestation fires: these policies influence the locations of different types of landholders in the landscape, their agricultural production strategy, as well as the relationship between landholders themselves and external stakeholders (Schwartzman et al. 2010; Schmink et al. 2019).

Agricultural land use

Increasing the profitability of ranching or crop farming, through regional changes in infrastructures or prices, might incentivize commercial agricultural producers and ranchers to set more land under cultivation and use fire in the process (Arima et al. 2007). After land clearing, fires continue to be used, especially in low-intensity farming systems and pastures, to eliminate regrowing vegetation, creating many ignition points that frequently escape into nearby forests (Cano-Crespo et al. 2015; Walker et al. 2022). After the initial land clearing, landholders may decide to convert some of the young secondary forest growing on abandoned pastures into new pastures, or convert pastures to croplands, some of the most common land use transitions in the Amazon (Nunes et al. 2022), using fires in the process (Ribeiro et al. 2024). Increasing commercial demand for certain crops has led to a reduction in the fallow period and more frequent agricultural fires (Jakovac et al. 2016).

Current fire regimes in the Brazilian Amazon are also the result of historical land use changes, land and capital concentration processes, as well as the capacity and willingness of different landholders to adopt fire-free agriculture (Sorrensen 2009; Carmenta et al. 2019). Land exploitation for pastoralism and agriculture requires transitions from natural ecosystems to pastures or croplands, often accompanied by peaks in fire activity just before and after these shifts (Ribeiro et al. 2024). The most frequent land use transition is from natural ecosystems to pastures, typically associated with high fire activity, followed by the frequent conversion of pasture to cropland (Macedo et al. 2012; Nunes et al. 2022). After deforestation, land prices increase on average by 252 per cent in the Brazilian Amazon, and capital transfers from agricultural to pastoral sectors fund deforestation dynamics and fire use (Gasparri and De Waroux 2015; Reydon et al. 2020). Indirect deforestation linked to crop production and the purchase of deforested pasture for crops has intensified since the moratorium, triggering widespread land speculation (Gollnow et al. 2018).

Mechanization and intensification of agriculture reduce the need to use fires and increase the value of fire-vulnerable assets on agricultural land, creating incentives for better fire management (Morello and Falcão 2020). However, the adoption of fire-free agriculture requires important investment, most often inaccessible for small landholders (Carmenta et al. 2019). Some initiatives, such as cocoa agroforestry, allowed small landholders to adopt fire-free agriculture and generate good incomes but still represent a small proportion of agricultural lands (Venturieri et al. 2022). Investments in fire-vulnerable assets, such as agroforestry plantations, are risky in fire-prone landscapes: high accidental fire risks can trap small landholders in low-intensity fire-resistant land use such as swidden cultivation or ranching (Cammelli et al. 2020). Large landholders can invest in already-deforested land and in mechanised agriculture, enabling the adoption of fire-free systems, whereas small landholders lack the resources for such transitions and remain dependent on fire-based practices (Schielein and Börner 2018; Carmenta et al. 2019; Cammelli et al. 2020). Despite the strong connections between older and newer deforestation frontiers, most of the blame for the recurrent wildfire crisis falls on small landholders (Eloy et al. 2019; Carmenta et al. 2021).

Forest degradation

Different sources of forest degradation also affect the flammability of the remaining forest cover. Both logging and deforestation cause large-scale fragmentation of the regional forest cover (Broadbent et al. 2008; Montibeller et al. 2020). Statistical analysis across several areas of the Brazilian Amazon shows a strong correlation between both the frequency and intensity of forest fires and forest fragmentation (Armenteras et al. 2013; Silva Junior et al. 2018; Silva-Junior et al. 2022). This is due to edge effects, resulting in fuel accumulation and drier climatic conditions close to the forest edges, creating favourable conditions for fires spread from nearby agricultural lands (Alvarado et al. 2004; Ray et al. 2005). Some of the effects of forest edges decline over time, due to the regrowth of new vegetation sealing off forest edges (Numata et al. 2017).

Selective logging is another important source of forest degradation that influences the flammability of remaining forests (Asner et al. 2006; Broadbent et al. 2008; Matricardi et al. 2013). Selective logging results in the creation of secondary roads, accumulation of wood debris on the forest floor, and damage to the canopy, establishing favourable conditions for more intense and larger wildfires (Brancalion et al. 2018; Hethcoat et al. 2020; Barni et al. 2021), as well as post-fire recovery (de Andrade et al. 2020).

The legacy impact of past forest fires is also a key determinant of the flammability of forests, as vegetation feedback can quickly lead to the transition from fire-resilient forests to fire-adapted grass-dominated ecosystems (Balch et al. 2015). Forest fires, even of low intensity, can result in high mortality rates of trees, especially if the burning happens during dry climatic conditions (Brando et al. 2014). This leads to damage in the canopy, accumulation of dead fuel on the ground and favours the invasion of forest edges by grass species (Silvério et al. 2013; Brando et al. 2014; Balch et al. 2015). The mortality of different trees is dependent on some morphological traits, such as bark thickness or absence of buttress roots, and younger classes of ages tend to be more affected: this could lead to a modification of the vegetal community (Balch et al. 2011).

Local governance of fires regimes

The significant damages associated with uncontrolled fires and their associated costs are encouraging collective governance of wildfire risk (Cammelli et al. 2019). Measures to prevent the accidental spread of fires, such as the creation of fire breaks, waiting for the first rains or monitoring the burns, impose a cost on landholders while the outcome depends on collective investment to reduce fire occurrence (Carmenta et al. 2013; Morello and Falcão 2020). The capacity to reach collective governance of fire use is affected by characteristics of the community such as the convergence of interests and values within the community, turnover rates of community members or interventions by external stakeholders (Bowman et al. 2008; Morello and Falcão 2020). However, drought conditions and the frequency of wildfires originating from distant places could undermine the willingness of local

communities to invest in fire control measures due to perceived risks of failure (Cammelli and Angelsen 2019).

Future fires regimes

Efforts at a regional level to assess the drivers of fire regimes in the Brazilian Amazon, as well as the existence of regional land-use models predicting future land-use changes, have led to several predictive models examining future fire regimes in the Brazilian Amazon. Future changes in climate will likely affect the extent of areas across the region susceptible to fires and significantly increase the areas of forest burned (De Faria et al. 2017; Le Page et al. 2017; Santana et al. 2022). This increase would be partly due to the increased dryness of the understory, as well as the indirect impacts of a drier climate on forest structure and fuel load, creating favourable conditions for intense forest fires (De Faria et al. 2017). Areas susceptible to wildfires would also change, with the central and western parts of the Amazon becoming increasingly vulnerable to forest fires (Silvestrini et al. 2011; De Santana et al. 2024). Changes in climate could also affect fire seasonality and increase the probability of fires occurring later in the season (Fonseca et al. 2019). Under the most pessimistic climate change scenario, even controlling deforestation might have only a limited impact on future fire activity (Le Page et al. 2017). However, future land use will also play an important role in shaping future fire activity. If deforestation continues, this could considerably increase future fire activity regardless of the extent of climate change and result in fires spreading across protected areas and Indigenous lands in the region (Fonseca et al. 2019; Brando et al. 2020).

1.2.3 Policy solutions to the fire issue

Faced with the high social and environmental costs of repetitive wildfires in the region, there have been several policy responses coordinated by the federal government. During the early 21st century, most of the policy responses to the wildfires crisis focused on the reduction of deforestation, due to the importance of these drivers in fire regimes (Sorrensen 2009).

However, some of the policies also aimed to address fire management and adoption of fire-free agricultural systems. In the following sections, I will present the most important policy interventions that constitute the backbone of the conservation reform in the Amazon and indirectly influence the fire regimes, as well as the main fire management policies implemented by the federal government.

Policies aiming to reduce deforestation

Over the 2005-2015 period, the Action Plan for the Prevention and Control of Deforestation in the Legal Amazon (PPCDam) coordinated the actions of different Brazilian ministries aiming to decrease deforestation by more than 80%. Initially, the PPCDAm focused on the improvement of satellite monitoring and law enforcement capacities as well as the demarcation of new protected areas. The delimitation of new protected areas succeeded in reducing deforestation rates and fire frequency, but their effectiveness depends on the type of protection system, deforestation pressures faced and managing authorities (Soares-Filho et al. 2010; Nolte et al. 2013; Carmenta et al. 2016; Herrera et al. 2019). Indigenous lands, often located in high-pressure areas, tend to be the most effective protection regime, followed by integral protection areas and then sustainable use areas, allowing many types of human activities (Soares-Filho et al. 2010; Nolte et al. 2013). The creation of a near real-time satellite monitoring system of deforestation to guide law enforcement on the ground was also a crucial point of the PPCDAm (Assunção et al. 2015; Börner et al. 2015). However, the size of the average deforestation patch decreased over the 2005-2014 period to avoid detection and subsequent punishment by environmental authorities (Rosa et al. 2012; Richards et al. 2017; Kalamandeen et al. 2018).

In later phases, the PPCDAm also emphasized the promotion of sustainable economic development and reducing deforestation on private lands (West and Fearnside 2021). In 2008, the critical county program started to publish a “blacklist” of municipalities experiencing an increase in deforestation. The first list published included the 36 Brazilian municipalities responsible for 45% of the deforestation detected by PRODES in 2007 (Assunção and Rocha 2019). The blacklisted municipalities are subject to stricter

administrative requirements for further forest clearing, and suffer from a bad reputation, which could reduce business opportunities, and increase monitoring and enforcement actions by IBAMA. Further restrictions can be adopted by the state government such as restricted access to government-sponsored agricultural credits. However, these municipalities are also receiving increased support from state actors and NGOs to reduce their deforestation rate.

The private sector also adopted the Amazon Soy Moratorium in 2006 and the Cattle Agreements in 2009, when major actors of both supply chains engaged to stop purchasing products originating from recently deforested lands (Gibbs et al. 2016; Heilmayr et al. 2020). Amazon Soy Moratorium decreased the rate of soy-related deforestation, especially for municipalities directly monitored by the Amazon Soy Moratorium or properties registered in the national rural registry (Gibbs et al. 2015; Kastens et al. 2017; Heilmayr et al. 2020). However, the Amazon Soy Moratorium led to both indirect deforestation, through a loophole authorizing the expansion of soy cultivation on the pasture and the extension of pastures on forests, and leakage of deforestation into the nearby Cerrado biome (Dou et al. 2018; Gollnow et al. 2018; Soterroni et al. 2019). The Cattle Agreements have proved difficult to implement, due to the mobility of cattle and the complexity of the supply chain: beef produced on freshly deforested land could still be either sold to the many small and medium slaughterhouses without monitoring systems for their supply branch, or to properties that are not blacklisted by cattle agreement signatories (Gibbs et al. 2016; Klingler et al. 2018).

Since 2016, politicians linked to agribusiness interest have risen in power within the federal government and dismantled the environmental policies of the previous administrations, a dynamic that accelerated tremendously under Bolsonaro's administration (Dobrovolski et al. 2018; Pelicice and Castello 2021). To achieve this, they used multiple strategies including the dismantlement of key federal institutions through budget cuts and administrative restructuring (de Area Leão Pereira et al. 2019), legislative changes that weaken environmental protection (Vale et al. 2021), granting land titles to

illegally grabbed land (Brito et al. 2019), and pro-deforestation rhetoric (Souza et al. 2022). Large numbers of deforestation fires and the quantity of smoke emitted in the early 2019 burning season led to a political crisis and forced the federal government to react promptly, while the deforestation rate rose steadily during Bolsonaro's mandate (Barlow et al. 2020). Since 2023, the new federal government has reestablished some environmental protection leading to a drop in deforestation rates, nevertheless many deforestation and forest fires are still burning through the region (Mataveli et al. 2024).

After the adoption of the PPCDAm in 2004, large networks of protected areas were delimited to counter the expansion of active deforestation frontiers, reducing the availability of land for agriculture in the region (Assunção et al., 20135 Hayes & Rajão, 2011; Nepstad et al., 2014; Soares-Filho et al., 2006). Some of the new delimited areas were contested by small-scale settlers, commercial agricultural producers and institutions supporting them, leading to their downsizing or degazettement to create new settlement areas or regularise land claims from colonists, while others experienced frequent intrusions and land grabs (de Oliveira et al., 2020; Keles et al., 2020). Frequent regularization of land grabs created an expectation of future regularizations of illegal land occupations, leading to large-scale land speculation in the region, which can prevent the creation of shared norms around fire management (Bowman et al. 2012; Brito et al. 2019; Morello and Falcão 2020; Carrero et al. 2022).

Policies aiming to control fires

In 1987, the Brazilian government created the National Centre to Prevent and Combat Forest Fires (Prevfogo), which coordinates the prevention and suppression of wildfires across Brazil with other federal, state and municipal level institutions (Sorrensen 2009; Morello et al. 2017). The main mandate of the federal program is to hire, train, equip and strategically deploy federal brigades of firefighters across conservation units, Indigenous lands and rural settlements, as well as to undertake training and environmental education programs to reduce the number and impact of wildfires (Morello et al. 2017).

The Prevfogo fire brigade program hires around 2,000 firefighters deployed across protected areas, conservation units and rural settlements of Brazil, with an important proportion of the budget and firefighters attributed to the Amazon (Fonseca Morello et al. 2020; Oliveira et al. 2021). The majority of firefighters in the fire brigades are members of traditional and Indigenous communities, representing an interesting point of inclusion of local fire knowledge (Xerente and Oliveira 2021; Oliveira et al. 2022). Analysis of fire regimes in an Indigenous land has shown that the fire brigade program is successful in reducing the total burned areas and increasing resilience to drought (Oliveira et al. 2022). However, the budget remains small compared to the number of fires and extent of the region, limiting the overall efficiency of the program (Fonseca-Morello et al. 2017; Fonseca Morello et al. 2020).

Most of the budget from the Prevfogo program is dedicated to fire brigade programs and responses to wildfires, leaving little opportunity for wildfire risk prevention (Morello et al. 2017; Oliveira et al. 2021). In the aftermath of the fires in Roraima in 1998, a federal program was implemented to train community leaders in safe fire use practices that could then be disseminated, and despite initial success, the chronic underfunding affected its long-term operationality (Sorrensen 2009). However, investment in fire prevention initiatives has been shown to be more effective and cost-efficient in reducing wildfire risks and lowering the cost of implementation than firefighting (Oliveira et al. 2021).

The adoption of fire-free agriculture is one of the strategies promoted by the government to reduce fire occurrence on private land, but it requires significant investment, most often inaccessible for small landholders (Fonseca-Morello et al. 2017; Carmenta et al. 2019). Governmental institutions bringing financial and technical support for the intensification of agriculture have targeted mainly medium and large landholders, leading to the intensification of agricultural systems in areas well connected to markets and the creation of an “agricultural intensification frontier” (Thaler et al. 2019). There have been local initiatives to subsidize tractors to replace fire for field clearing and land preparation, but the efficiency of such initiatives is often limited by the number of tractors available and their capacity to clear fields in forested landscapes (Carmenta et al. 2019).

In 2019, 2020 and 2021, the federal government adopted a blanket ban on fire use, even though it has not significantly impacted fire regimes (Machado et al. 2024). The use of policy instruments to forbid or limit the use of fires is more deeply rooted in the fire management approach, and after the Roraima wildfires in 1998, the federal government developed the Amazon Fire and Deforestation Monitoring and Control Project (PRODESQUE). It resulted mainly in the modification of the forest code to increase the size of the legal reserve from 50% to 80% of the size of the landholding and the creation of a burning permit system, except for Indigenous land traditional communities (Sorrensen 2009; Eufemia et al. 2022). While still in place, the majority of landholdings through the Brazilian Amazon have a deficit in forest reserve (Sparovek et al. 2010), and the permit system is not commonly used, partly due to the high transaction costs for landholders and fear of increased legal liability in case of escaped fires (Fonseca-Morello et al. 2017; Carmenta et al. 2019). The overlooked role of fires in agricultural systems and constraints met by landholders in the Brazilian Amazon led to a failure of the initiatives aiming to control fire use(Mistry and Bizerril 2011; Carmenta et al. 2013; Carmenta et al. 2019). Moreover, the criminalization of fire use might have prevented the development of local fire governance and the sharing of information on fire use due to the taboo generated (Carmenta et al. 2019).

Analysis of different policy interventions to influence landholders' behaviour highlights the impact of local context on their efficiency. In a choice experiment, Cammelli and Angelsen (2019) showed that command and control policies are more likely to influence the behaviour of participants than payment for environmental services, as they favour local fire governance by removing sanction costs. Meanwhile, Morello et al. (2017) found that command and control measures might have a lower overall efficiency than incentive-based instruments due to the limited spatial outreach of law enforcement and the continuation of fire use in remote parts of the landscape. The most effective instruments identified were subsidies for the mechanization of agriculture, but with potentially significant adverse effects on deforestation as it would increase productivity for land and promote continued deforestation (Morello et al. 2017).

Since 2013, Brazil has experimented with the Integrated Fire Management approach in several protected areas and Indigenous lands of the Cerrado region (Schmidt et al. 2016; Eloy et al. 2019; Barradas and Torres Ribeiro 2021). This paradigm is based on combined fire prevention, response and recovery across different scales while integrating the needs, preferences and knowledge of local communities (Myers 2006; Oliveras Menor et al. 2025). This evolution in Brazilian fire management policies has been accompanied by calls for intercultural fire management, highlighting the importance of including multiple perspectives and insights across different knowledge systems to understand the local use of fires (Eloy et al. 2019; Mistry et al. 2019). In 2024, the Integrated Fire Management approach was sanctioned by a national law that recognised notably the importance of traditional and adaptive use of knowledge by Indigenous and traditional communities as well as the positive role played by certain fire regimes according to local fire ecology.

1.3. Research gaps and research objectives

The persistence of forest fires across the Brazilian Amazon after the regional decline in deforestation, their deleterious impact on ecosystem services and the well-being of inhabitants of the Amazon (section 1.2.1) has led to growing attention to fire management in the region. At a regional scale, research efforts have shown the influence of diverse processes such as climatic conditions, deforestation patterns, forest degradation as well as agricultural land use (section 1.2.2). Many of the processes driving fire regimes across the region are dynamic and have cumulative effects, resulting in high inter-annual variations in fire activity (Silveira et al. 2020; Libonati et al. 2021). While there is a wide diversity of fire types in the Brazilian Amazon, each associated with distinct drivers and impacts on the environment and society (section 1.2.1), regional analysis has frequently conflated different types of fires together. **Thus, the first literature gap I address in this thesis is to identify the drivers of different types of fires across the Brazilian Amazon.**

Fire regimes are driven by the interaction between the geophysical environment, local fire ecology and socio-economic drivers that influence landscape management (Bowman et al. 2013). Insights from ecological experiments have demonstrated the existence of strong feedback between vegetation and fire that varies across the region (section 1.2.1). However, feedback between vegetation, fires and climate is still poorly understood despite their potential high spatial and temporal variability, especially in ecologically diverse regions (Archibald et al. 2018). Much of this feedback is also mediated by human management of the landscape, being through their impact on land use dynamics, associated fire management or landscape domestication practices (Bowman et al. 2011; Levis et al. 2018; Perkins et al. 2025). **Thus, the second literature gap I address in this thesis is to comprehend how climatic, ecological and social drivers of fire regimes interact to shape local fire regimes.**

The compounded impact of climate change, forest degradation and land use change will increase wildfire risks across the Brazilian Amazon, both intensifying fire activity in places already experiencing wildfires and exposing new parts of the region to wildfire risks (Silvestrini et al. 2011; Fonseca et al. 2019; Brando et al. 2020). This will exacerbate the impact already observed on both ecosystems and the communities of the Brazilian Amazon (section 1.2.1). Despite the importance of adapting to changing fire regimes, a priority recently recognised in the Brazilian legislation, there is a lack of studies exploring how different type of landholders of the region already adapted their behaviour to changing wildfire risks (Cammelli and Angelsen 2019). Understanding some of the adaptive pathways in areas experiencing significant changes, as well as their potential shortcoming, could inform the design of future policies and interventions to support landholders in their fire management efforts. **Therefore, the third research gap I will address in this thesis is to document existing adaptation strategies to changing wildfire risks adopted by different types of landholders of the Brazilian Amazon.**

The use of remote-sensing approaches and analysis at a broad geographical scale have often overly simplistic representations of anthropic influence on fire regimes (Ford et al. 2021). In the Brazilian Amazon, most of the literature on fire regimes focuses on the use of remote-sensing approaches and overlooks the influences of social processes on fire use (Carmenta et al. 2011), despite their importance for understanding fire regimes and pathways to reduce their occurrence (section 1.2.3). The development of methodologies to capture cross-scalar dynamics of fire regimes and understand both fire use and adaptation preferences of local communities could support the implementation of fair fire management policies (Stoof and Kettridge 2022; Schinko et al. 2023; Oliveras Menor et al. 2025). **Consequently, the last literature gap I will address in this thesis is to understand how the integration of insights collected across multiple scales and research approaches can enrich our understanding of fire regimes, their evolutions and potential adaptive pathways.**

In this thesis, I set out to understand processes driving fire regimes across multiple scales and how local landholders perceive and respond to changing wildfire risks. In order to do this, I will address the following research objectives:

- **Research objective 1.** To identify the main drivers shaping fire regimes across the Brazilian Amazon.

- **Research objective 2.** To examine how interactions between climatic, ecological and socio-economic factors influence local fire regimes.

- **Research objective 3.** To investigate how different types of landholders in the Brazilian Amazon adapt to changing wildfire risks.

- **Research objective 4.** To explore how cross-scalar and interdisciplinary approaches can enhance understanding of fire regimes, changing wildfire risks, and adaptive pathways.

1.4. Conceptual framework

To understand how local communities across the Brazilian Amazon are adapting to changing fire regimes, I developed a conceptual framework drawing on literature on local ecological knowledge, adaptive governance and resilience to climate change. This provides a theoretical basis to understand how local ecological knowledge is mobilised to observe change in fire regimes and decide about adaptation strategies, that will have a broader impact on local socio-ecological systems (Figure 3).

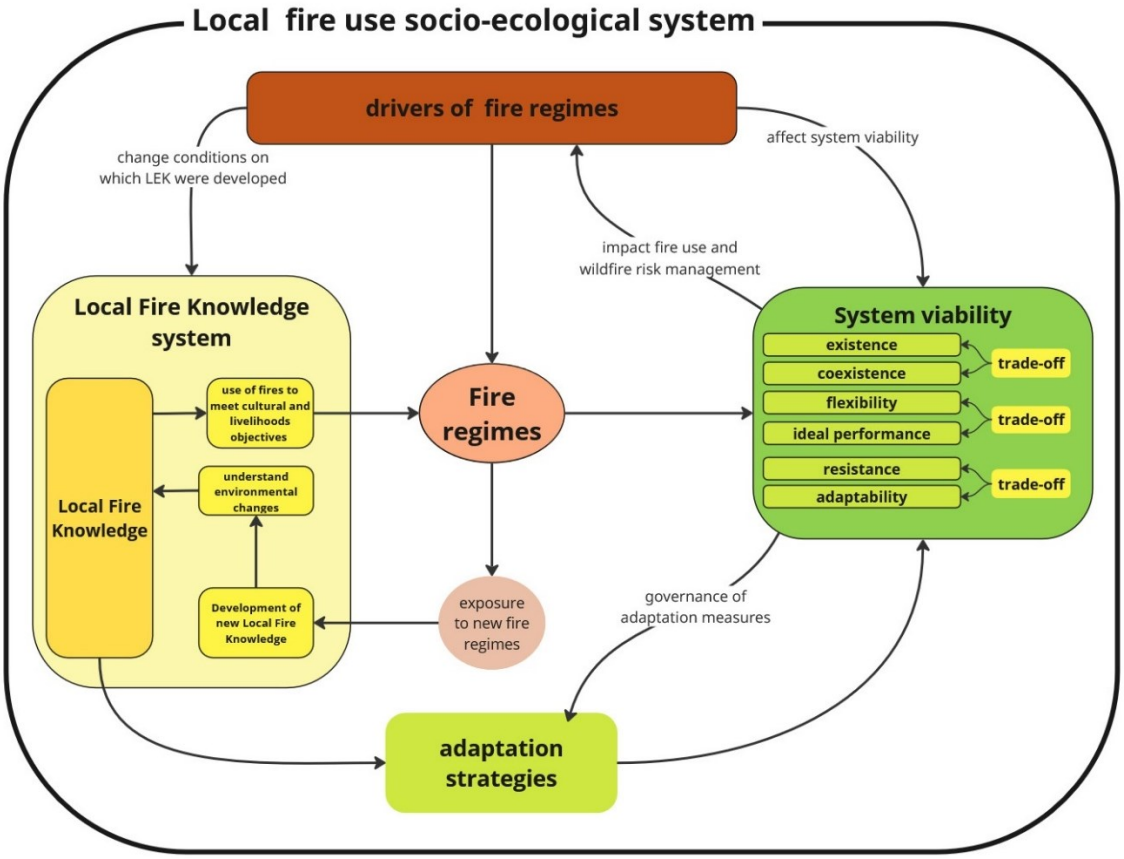


Figure 3. Conceptual framework develops to describe the adaptation of local communities to changing wildfire risk, mobilising local ecological knowledge

1.4.1 Fire use socio-ecological systems and changes in conditions

I will focus on socio-ecological systems related to local fire use, including elements associated with fire use by local inhabitants and their associated wildfire risk management strategies. The geographical boundaries of the local socio-ecological systems are defined by the territory where a community can use fire to achieve livelihoods or cultural objectives.

To manage the socio-ecological system towards a desired state, traditional communities use local ecological knowledge, defined as "*A cumulative body of knowledge, practice and belief, evolving by adaptive processes and handed down through generations by cultural transmission about the relationship of living beings (including humans) with one another and with their environment.*" (Berkes 2018). While this definition initially corresponds to the definition of Traditional Ecological Knowledge, I preferred to use the analogous term of local ecological knowledge to avoid the stigma sometimes associated with traditional practices and associated knowledge, that could be perceived as static (Janke and Sentina 2018). To use fire in specific landscapes and meet their livelihoods and cultural objectives while limiting potential damages, communities are mobilising local fire knowledge, a subset of local ecological knowledge related to fire management (yellow box in Figure 3, Huffman, 2013).

To sustain a desirable state of the socio-ecological system amidst changes in environmental conditions, communities can engage in adaptive governance, defined as a form of *"environmental governance that allows emergence of collective action capable of facilitating adaptation to change and surprise as well as the capacity to itself evolve"* (Cosens et al. 2018). Evans, Brown and Allison (2011) defined three essential capacities for adaptive governance: understanding environmental changes, using this new knowledge for decision-making and adapting the decision-making process to sustain the long-term resilience of SES.

1.4.2 Role of local ecological knowledge to understand environmental changes

Local ecological knowledge plays a key role in supporting adaptive governance: constant interactions between resource users and users' systems they depend on enable communities to monitor and understand environmental changes and their impact (Folke et al. 2009; Evans et al. 2011; Savo et al. 2016; Reyes-García et al. 2024). In the case of my fire use socio-ecological system, the environmental changes relate to changes in drivers of fire regimes (dark red box, Figure 3) and its impact on fire regimes (red box, Figure 3).

In situations of high social agreements, local ecological knowledge can be developed and shared at a community level, reducing the individual cost of observation and experimentation to generate new local ecological knowledge (Reyes-García et al. 2016). Past experiences of environmental disturbances or radical changes in environmental conditions can enhance local communities' adaptive capacity: they can mobilize ecological and social memories accumulated to adapt to present challenges (Folke et al. 2009; Gómez-Baggethun et al. 2012).

Local ecological knowledge is dynamic: certain factors erode some local ecological knowledge, while others will be preserved or generated if communities keep using and developing them (Gómez-Baggethun et al. 2013; Gómez-Baggethun and Reyes-García 2013; Fernández-Llamazares et al. 2015). Berkes & Turner (2006) proposed two non-mutually exclusive models for the development of local ecological knowledge by small-scale societies. First, the resources depletion model in which local ecological knowledge is developed through cycles of resource depletion and subsequent adjustment of community-based conservation strategies (Berkes and Turner, 2006). Second, the ecological understanding model, where local ecological knowledge is developed through observation and experiences in nature leading to incremental learning, their integration in a beliefs system avoiding resource depletion, communications of these incremental learning and beliefs systems leading to the development of institutions and rules consolidating environmental knowledge and practices (Turner and Berkes 2006).

1.4.3 Using new knowledge for decision-making regarding fire management

Globally, Indigenous communities are adapting to climate change and mitigating its adverse impact on local socio-ecological systems through diverse strategies, such as individual changes in practice and strengthening local institutions (Petzold et al. 2020). McWethy (2019) identified two types of adaptation strategies for improving the resilience of socio-ecological systems to new wildfire risks: adaptive changes, which influence fire behaviour and reduce the negative impacts of wildfires, and transformative changes, which fundamentally alter the relationship between local communities and wildfires (light green box, Figure 3).

While most livelihood adaptation strategies to climate change are adaptive changes, a significant proportion are transformational changes (Zant et al. 2023), notably in response to severe wildfires (McWethy 2019; Copes-Gerbitz et al. 2022). Environmental crises offer opportunities to reorganise local socio-ecological systems through transformative changes and improve their resilience (Folke et al. 2009). However, some transformational changes resulted in unintended and harmful effects on the local socio-ecological system and their resilience (Zant et al. 2023).

The appropriate balance between adaptive and transformative changes for fire use socio-ecological systems depends on human vulnerability to wildfires, wildfire severity and the novelty of fire regimes (McWethy 2019). Moreover, historical, institutional, socio-cultural and political factors shape decision-making and influence resilience strategies adopted by local communities (Evans et al. 2011; Rutherford and Schultz 2019). This highlights the importance of contextualizing community resilience to wildfire risks and identifying factors that support or hinder adaptation efforts (Copes-Gerbitz et al. 2021).

1.4.4 Impact on system viability

A review of past policies initiatives aimed at controlling fire regimes demonstrated frequent unintended and deleterious effects on fire use socio-ecological systems (Smith et al. 2024). Similarly, transformational changes to adapt to climate change could lead to mis-adaptation when communities invest in resilience strategies that are inadequate or counterproductive (Zant et al. 2023). Thus, it is essential to evaluate the long-term impact of resilience strategies on the fire use socio-ecological system so they can be adjusted to meet evolving environmental and social conditions.

To this end, I employed the concept of system viability (dark green box, Figure 3), as presented by Mistry et al. (2010) Berardi et al. (2015) for the evaluation of integrated conservation and development projects. This concept is rooted in the orientor theory of Bossel (1999), which defines system viability as a system's ability to "survive, be healthy, and develop in its particular environment". Viability is determined by six survival strategies:

- **Existence:** ability to secure resources for basic survival under a normal environmental state
- **Coexistence:** the ability to coexist with other interdependent systems
- **Ideal performance:** ability to make the best use of limited resources
- **Flexibility:** the ability to have a range of options in an environment where there is a high variety
- **Resistance:** the ability to cope with temporary variability in the environment
- **Adaptability:** the ability to change practices to deal with inevitable changes in the environment

Communities can respond to environmental changes by investing in one or more of these survival strategies. However, tensions exist between these survival strategies, and investment of limited resources requires trade-offs across survival strategies (Berardi et al. 2015). For example, the concentration of resources to ensure the safe use of fires within one community (investment in the existence strategy) could undermine resources available for other communities and increase overall wildfire risks (negative impact on coexistence strategy).

1.5. Thesis outline

To fulfil the different research objectives presented in section 1.3, I will present the research done in four results chapters, detailing research results that have been conducted over different scales and using distinct methodologies.

In Chapter 2, I analysed the **drivers of deforestation fires, agricultural fires and forest fires across the Brazilian Amazon** from 2009 to 2021 using spatio-temporal models. The main results were the persistence of agricultural fires after initial land clearing, the increased occurrence of fires in remote parts of the Amazon and forests further away from agricultural land, and the important inhibitory effects of protected areas and Indigenous lands on all types of fires. These results showed that different fire types are driven by distinct processes and that fire regimes are shaped by the cumulated impact of many climatic, ecological and socio-economic drivers. This chapter is in the process of publication with the co-authorship of Dr Ioannis Kountouris, Prof. Jeremy Woods, Dr Anna Freni Sterrantino and Dr Morena Mills.

In Chapter 3, I assessed the **change in wildfire risks and fire regimes across the three main land governance types of the North of Mato-Grosso**. I found that while all land governance types were exposed to a similar increase in wildfire risks due to climate, each governance type experienced a distinct change in wildfire risks due to land use change. Indigenous land had distinct fire regimes, characterised by a low overall burned area, predominance of large fires over natural vegetation and stability over time. Despite experiencing the quickest increase in wildfire risks, rural settlements experienced an important decrease in the intensity of their fire regimes. This showed the importance of land

governance types on both land use dynamics and social factors affecting the use and control of fires.

In Chapter 4, I presented the results from participatory research **exercises exploring the local fire knowledge of habitants of three communities in Capoto/Jarina**, an Indigenous land of the state of Mato Grosso. I found that local fire use was underpinned by rich local fire knowledge that helps to use fires to reach livelihoods and cultural objectives while reducing wildfire risks. The local fire use and associated knowledge were highly diverse across the communities visited due to a mix of ecological and cultural factors and included elements that could not be captured through remote-sensing approaches. These results highlighted the need to consider the diversity of local fire use and knowledge for effective fire management.

In Chapter 5, I presented both results from the **participatory research exercise and quantitative analysis of wildfire risks on the perception and adaptation of people from Capoto/Jarina to the changing wildfire risk**. I found that the local communities are perceiving the detailed impacts of both climate change and change in local vegetation and adopting a wide array of individual and collective measures to adapt to it depending on wildfire risks novelty and severity. These results highlighted the diversity of adaptation strategies that are adopted by local landholders. Some of the results of this chapter and chapter 4 are in the process of publication with the co-authorship of Mr. Roiti Metuktire, Prof. Jayalaxshmi Mistry, Mr Kokokrotire Metuktire, Mr Yarika Juruna, Ms Juliana Martins, Dr Rafael Chiaravalloti, Ms Juliana Bonanomi and Dr Morena Mills.

In Chapter 6, I will summarise how the results described from Chapter 2 to Chapter 5 contributed to answering the different research objectives, before discussing some of the limitations of my doctoral work and the future research directions.

1. 6. Positionality statement

As researchers' positionality inevitably shapes the selection of research questions, design, and analytical approaches, it is essential to explicitly situate one's positionality within the research process (Hopkins 2015).

I am a non-Indigenous, white male researcher who grew up in a rural, predominantly fire-free, forested region of France. I pursued a Master's degree in tropical ecosystems management, focusing on tropical forest conservation, and worked on forest restoration initiatives prior to starting my PhD. My long-held perception of fire as an unequivocal driver of ecosystem degradation was challenged through extensive engagement within the Leverhulme Wildfire Centre, a multidisciplinary research centre where most researchers work in fire-dependent ecosystems. This perception shift was reinforced by participating in the Fire-Adapt project, which showcased integrated fire management in the fire-dependent ecosystems of Brazil.

Growing up in a rural region of France, I was exposed to the local ecological knowledge of my community in activities such as gardening, managing orchards, forestry, wild foraging, and fishing. This early recognition of the value and depth of local ecological knowledge was reinforced by years of living in countries of the Global South, where I explored the detailed knowledge held by communities managing complex tropical ecosystems, particularly in relation to tree planting and management.

The hierarchisation of evidence coming from different knowledge systems, and the lack of recognition of the importance of local ecological knowledge by certain influential stakeholders, pushed me to explore the complementarity between knowledge traditionally recognised and valued by Western academic institutions, including my host university, and local traditional and ecological knowledge. Being part of the Leverhulme Wildfires Centre, a research centre working across multiple scales and geographical areas, allowed me to engage in numerous presentations and discussions on the limitations of various research instruments and methods. This context enabled me to develop a detailed understanding of

the constraints of widely used tools in environmental management, such as remote sensing imagery and dynamic global vegetation models.

I have formal training in natural sciences and used mainly positivist and quantitative methods in previous research projects. While I developed a strong interest in the social aspects of environmental sciences, my earlier experiences using social science methods focused predominantly on quantitative tools such as surveys. This influenced my qualitative data analysis process and guided my decision to adopt a grounded, inductive approach to coding, enabling the development of an analytical framework that remained closely aligned with empirical evidence rather than predefined theory.

My doctoral research began during the COVID-19 pandemic when Bolsonaro's administration was dismantling federal environmental institutions in Brazil. This political context influenced my doctoral work in two ways: (i) it led me to focus my initial research on the relationship between deforestation dynamics and fire regimes, as deforestation rates surged and fire was increasingly used as a land-grabbing tool; (ii) it required me to adopt various research methods, some of which did not rely on fieldwork data due to uncertainties surrounding travel possibilities.

Throughout my doctorate, I adopted critical realist ontology and constructivist epistemology (Moon and Blackman 2014). From a critical realist perspective, I recognise the existence of an external reality but also acknowledge the limitations of the human capacity to fully perceive it. This stance encourages critical examination of any claims about reality and supports interdisciplinary approaches when investigating complex phenomena. Constructivism holds that we construct knowledge through our engagement and interpretation of the world. I embraced this epistemological stance as it aligns with my view that different knowledge systems emerge from distinct interactions with and interpretations of the environment. This position is particularly relevant for engaging with local ecological knowledge without placing it in a hierarchy beneath traditional scientific knowledge.

Faced with the urgency of addressing the coupled biodiversity and climate crises and supporting the adaptation of forest-dependent communities to their adverse effects, I adopt a pragmatic approach to research. I actively draw from various disciplinary perspectives to gather the information necessary to answer my research questions and provide credible evidence for stakeholders. I believe that my ontological, epistemological and philosophical perspectives naturally led me to interdisciplinary research, mixed methods, and the adoption of the Multiple Evidence Base Approach (Tengö et al. 2014). This decision is rooted in a strong conviction that combining quantitative and qualitative data, drawn from different disciplinary silos and knowledge systems, can reveal complementary dimensions of complex phenomena, such as shifts in fire regimes and community adaptation pathways.

While I had previous experience working in tropical forests and savannas, this was my first experience conducting research in Brazil. During this thesis, I learned Portuguese and engaged extensively with literature in both English and Portuguese on environmental governance and fire management in the Brazilian Amazon. I also presented preliminary results to research participants, local Indigenous partners, and other researchers with extensive experience across Brazil, including several Brazilian nationals, to receive feedback from partners familiar with the national context.

During fieldwork, I was always accompanied by a national, non-Indigenous Brazilian research assistant who assisted with translation between Portuguese and English, as well as a local Indigenous facilitator who supported translation between either Mẽbêngôkre or Yudjá and Portuguese. In cross-cultural and cross-language research contexts, it is important to reflect on the positionality of research assistants and their role in knowledge production (Caretta 2015). The choice of a research assistant fluent in English was driven by the need to ensure my Portuguese skills would not limit my comprehension of participants' responses and to have a woman in the research team, thereby facilitating women's participation. Nonetheless, the presence of a local facilitator remained vital in acting as a gatekeeper, facilitating introductions to the community, coordinating research activities, and helping us respect local cultural norms. Our local facilitators' involvement in the community fire brigade initiatives may have influenced participant selection, potentially

favouring individuals either involved with or supportive of these initiatives (Caretta, 2015). While both my research assistant and I were perceived as outsiders, the participation of a local facilitator connected to the fire brigades, our partnership with the Raoni Institute, and the presentation of preliminary results helped overcome certain barriers by creating the expectation that research findings would be returned to and used by the community.

Chapter 2: Regional drivers of fire regimes in the Brazilian Amazon between 2009 and 2021

2.1. Main contributions

- Agricultural land use is an important determinant of fire regimes and determines both the use of fires during and after land clearing, with more fires associated with pastures and fewer with perennial crops.

- The occurrence of forest fires increases in remote locations and forests far away from agricultural land over time, indicating an "interiorisation" of fire regimes in the Brazilian Amazon.

- Protected areas and Indigenous land were experiencing fewer fires than unprotected areas, especially in areas more than 10 kilometres from unprotected lands. Indigenous lands were more effective against deforestation and agricultural fires, while integral protection areas were more effective against forest fires.

- Rural settlements and municipalities that were removed from the priority list experienced a decrease in the occurrence of different types of fires, showing the potential role played by policy aiming to control land use.

2.2 Introduction

Either through managed burns for land clearing, subsequent land management or ignitions that have escaped human control, fires place substantial pressure on tropical rainforests (Van Wees et al. 2021). Fires occur in 41% of areas that have experienced tropical rainforest loss, while forest fires in particular contribute an increasingly large proportion of global fire-related carbon emissions (Van Wees et al. 2021; Zheng et al. 2021). Fire-related disturbances can trigger shifts in vegetation composition, enabling woodland conversion to savanna and grassland that host different biodiversity assemblages and are more vulnerable to subsequent cycles of drought and wildfire (Armenteras et al. 2021). Even when forests are not later converted to agricultural land, fires can prevent their regeneration, hinder the recovery of essential ecosystem functions, and limit their resilience to external disturbances (Drüke et al. 2023; Zheng et al. 2021). The ecological impact of uncontrolled fire use is complemented by its negative effect on local populations through its damaging influence on agricultural assets, infrastructure, access to forest products, and health (Carmenta et al. 2021).

Despite its ecological, environmental and human impacts, fire use is common throughout tropical rainforest landscapes for land clearing and management. Andela et al. (2017) link the global decrease in burned areas to the intensification of agriculture and increased management of the landscape to protect agricultural assets and infrastructure. However, tropical rainforest basins are often characterized by the co-existence of intensified agricultural systems in most accessible areas, alongside extensive agricultural systems in remote areas (Schielein and Börner 2018). For landholders with limited access to capital, fires often represent the most viable tool for land management due to their low cost (Carmenta et al. 2021). Moreover, the prevalence of fires in the landscape may hinder agricultural intensification and the transition toward fire-free agricultural systems due to the risks of accidental fires and their associated damages (Cammelli et al. 2020). These dynamics are also fuelled by the coupling between old and new deforestation frontiers, as landholders with high capital buy and invest in already deforested land while pushing

landholders with little capital to the margin of the landscape and creating economic incentives for deforestation, extensive ranching and fewer fire control (Bowman et al. 2008, 200; Bowman et al. 2012; Schielein and Börner 2018). Fire regimes are expected to intensify as climate change induces longer and more intense dry periods (Abatzoglou et al. 2019), and tropical rainforests are becoming increasingly fragmented (Ma et al. 2023).

The Brazilian Amazon, which contains more than a quarter of the world's remaining tropical rainforest and harbours an estimated 11,210 tree species, including 3,248 rare species (Hubbell et al. 2008), is severely impacted by fires. Burning, along with other sources of forest degradation including logging, has transformed the region into a carbon source over the last two decades (Tyukavina et al. 2017; Gatti et al. 2021). Between 2001 and 2019, fires affected the distribution range of about 95% of plant and vertebrate species in the Amazon Basin, many of which lack adaptation to fire (Feng et al. 2021; Brando et al. 2012). Nevertheless, fires continue to be used in the Brazilian Amazon for clearing new fields, disposing of crop residues and vegetation regrowth, and managing soil fertility (van Vliet et al. 2013). Deforestation and land management fires frequently escape their intended confines and turn into forest fires (Silvério et al. 2013; Cano-Crespo et al. 2015). Fire control measures, including monitoring, fire breaks, and burning under low-risk meteorological conditions are costly, while their effectiveness depends on the collective will and effort of local communities for fire use regulation (Bowman et al. 2008; Morello and Falcão 2020).

The evolving fire crisis in the region has historically been blamed on deforestation patterns. Government efforts to manage it have primarily focused on conservation policy and the criminalization of fire use (Sorrensen 2009). While the rapid decrease in deforestation after 2004 was correlated with a significant decline in satellite-detected fires across the Brazilian Amazon, deforestation rates and fire occurrence later decoupled, suggesting an increasing influence of fire regime drivers unrelated to deforestation (Aragão and Shimabukuro 2010). However, the recent spike in deforestation, driven by the regional weakening of environmental policies, along with a reanalysis of fire trends, again suggests that deforestation remains a key factor in the region’s fire regimes (Libonati et al. 2021). The

continued use of fires in agricultural landscapes after deforestation, especially in low-intensity agricultural systems, may contribute to the persistence of fires in the Brazilian Amazon (Aragão and Shimabukuro 2010; van Vliet et al. 2013).

Understanding the typology and the drivers behind every type of fire is crucial for assessing the impact of policies focused on reducing deforestation and criminalizing fire use, as well as for bridging the gap between policies targeting deforestation, forest degradation, and integrated fire management (Sorrensen 2009; Mistry and Bizerril 2011). Identifying areas that are, or will become, susceptible to different types of fires can help prioritize them for wildfire prevention interventions, such as the deployment of fire brigades. Additionally, understanding the drivers of different types of fires provides insights for developing initiatives that address their root causes and mitigate potential harmful impacts on ecosystems, the economy, and human health in the region.

Here I present a typology of fire incidence in the Brazilian Amazon, characterize key drivers for each fire type, and examine their stability over time. Using datasets on deforestation and land use, I begin by classifying fires detected by satellites into deforestation, agricultural, and forest fires. I then use spatio-temporal statistical models to estimate the relationship between each fire type and a series of socio-economic and environmental drivers from 2009 to 2021. This publication contributes to the literature on fire regime drivers in tropical forests (Fonseca et al. 2017; Silveira et al. 2020; Jones et al. 2022) and the impact of conservation policies on fire regimes in the Brazilian Amazon (Adeney et al. 2009; Sorrensen 2009; Libonati et al. 2021).

2.3 Methods

2.3.1 Outcome variables

We derived the response variables by classifying satellite-detected fires into deforestation fires, agricultural fires and forest fires from 2009 to 2021. To obtain fire detections within the Brazilian Amazon, I combined the Thermal Anomalies/Fire Daily datasets from MODIS Terra (MOD14A1 v6.1) and Aqua (MYD14A1 v6.1) satellites which consist of 1 km grids containing the daily occurrences of fires. I removed low-confidence fire detections before creating annual grids with the number of days when thermal anomalies (hereafter referred to as fires) were detected.

Each year, all the grid cells were associated with a percentage of deforestation derived from the PRODES dataset (PRODES 2022), forest cover and pastoral/agricultural land cover derived from MapBiomas collection 7 (Souza et al. 2020). I classified fires into three categories, excluding fires that didn't fit in any of them:

Deforestation fires: more than 100 m^2 of deforestation (the detection threshold for the MOD14A1/MYD14A1 datasets) detected in the same year than fire. About 14.7% of the fires (221 580 fires) were classified as deforestation fires.

Agricultural fires: less than 100 m^2 of deforestation detected on the grid cell in the same year than fire and more than 80% of pastoral/agricultural land cover. About 18.0% of the fires (271 150 fires) were classified as agricultural fires.

Forest fires: less than 100 m^2 of deforestation detected on the grid cell in the same year and more than 80% of forest cover. About 13.2% of the fires (199 287 fires) were classified as forest fires.

In total, 45.87% of the daily fires with medium to high confidence detected were classified as deforestation, agricultural or forest fires (Figure 4). Due to the coarse resolution of satellite products, it was not possible to confirm that all deforestation fires result from burning after vegetation felling, however, they are in proximity to recently deforested areas.

Additionally, fires classified as agricultural and forest fires may be associated with deforestation events smaller than 6.5ha, the threshold of detection of the deforestation product used.

We used the pixel centroid as the reference point for the fires detected, thus creating a new dataset of points that was used to fit a point process using the INLA-SPDE approach described later.

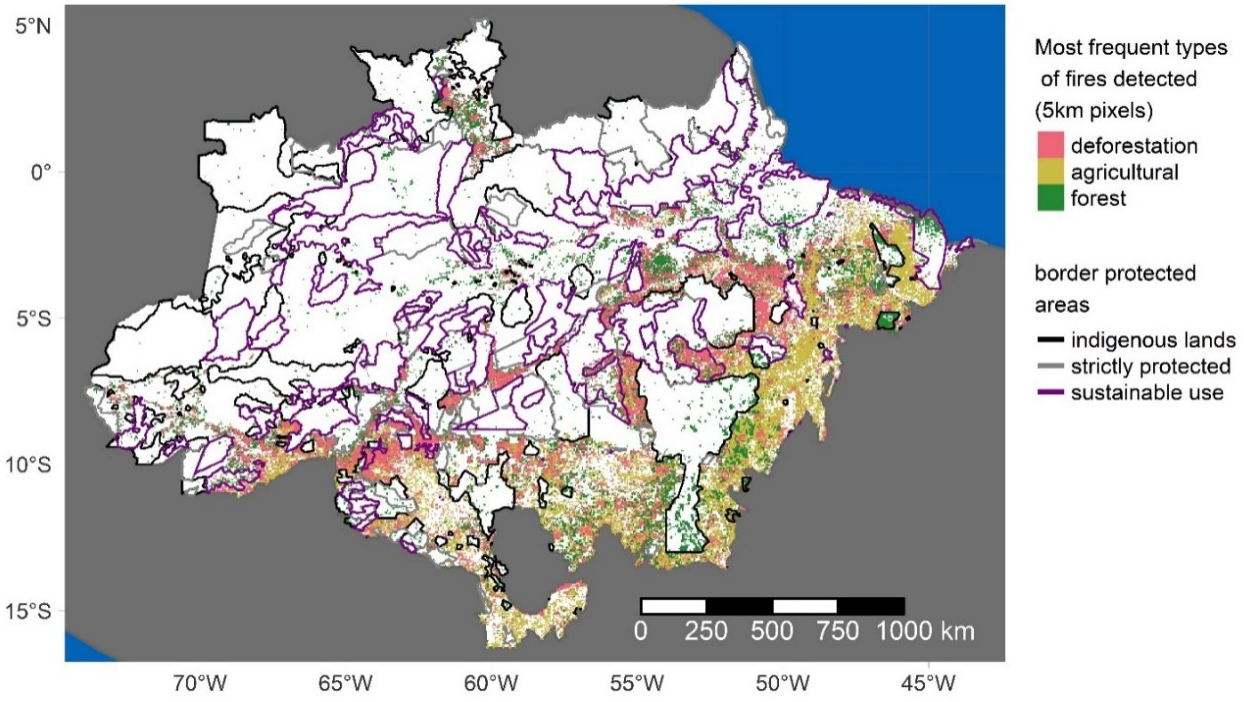


Figure 4. Map of the most frequent types of fires detected within 5km cells across the Brazilian Amazon.

2.3.2 Explanatory variables

We identified potential direct and indirect fire drivers in the Brazilian Amazon from a review of 51 relevant papers published between 2004 and 2022. Of these, 24 studies presented results from quantitative analyses of drivers and impacts of fires, 17 presented results from quantitative analyses of drivers and impacts of deforestation, four presented results from quantitative analyses looking at the driver and impact of other sources of forest degradation. Additionally, 6 publications provided broader context on the link between

drivers of deforestation, fires and other sources of forest degradation (Appendix 2.1 for more details on the literature review).

This led to the identification of 5 main categories of drivers of fires in the Brazilian Amazon and 14 drivers for fire regimes. From these, I identified 22 variables for inclusion in my models and identified associated data sources to model them over the 2009–2021 period, prioritizing the highest possible spatial and temporal resolution. After evaluating redundancy and correlation with other covariates, I only retained 17 variables. The explanatory variables were aggregated into a 1 km grid to correspond to the resolution of my response variable (Table 1 for the final selection of variables). To address highly skewed distributions of some variables, all explanatory variables were categorized. This categorization helps in the interpretability of the model outputs (see Appendix 2.2 for more information on data processing and distribution of response variables).

Drivers	Proxy variable	Data sources	Variable categories
Climate			
Precipitation	Maximum Cumulated Water deficit (MCWD)	CHIRPS	No deficit/1-100mm/101-200mm/201-300mm/301-400mm/401mm and more
Forest degradation			
Fragmentation	Edges density	MapBiomas collection 7	<1m.Ha^{-1}/1-25m.Ha^{-1}/25-50m.Ha^{-1}/ >50m.Ha^{-1}
Forest degradation	Distance forest edges	MapBiomas collection 7	Adjacent to non-forest land/ 0-1km/1-3km/3-5km/5-10km/10-25km/>25km
Infrastructure and development			
Access to market	Transport cost for exportation	Victoria et al. (2021)	<5K RU/5-10 RU/10-20RU/20-30RU/>30 RU
Rural settlements	Rural settlements	INCRA	Undesignated land/rural settlement
Agriculture and pastoralism			
Agriculture	Annual crop cover Annual crop cover increase Perennial crop cover Perennial crop cover increase Domiant agricultural land use	MapBiomas collection 7	<1%/1-50%/51-100% no increase/ increase <1%/1-50%/51-100% no increase/increase Pasture/annual crops/perennial crops
Pastoralism	Pastoral land cover Pastoral land cover increase	MapBiomas collection 7	<1%/1-50%/51-100% No increase/increase
Environmental policies			
Protected areas and Indigenous land	Integral protection areas (IUCN cat. I to III) Sustainable use areas (IUCN cat. IV to VI) Indigenous lands	World Database on Protected Areas FUNAI	Undesignated land/ periphery (<10km from border)/ core (>10km from border)
Protected areas downsizing or degazettement	Protected areas downsizing or degazettement (PADDD)	PADDD trackers	Undesignated land/ areas no longer protected
Priority municipalities	List of priority municipalities	Ministry of environment (MMA)	Never on priority list/ on priority list/ on priority list in the past

Table 1. Drivers and explanatory variables identified, associated data sources and categories used

2.3.3 Statistical Methods

We modelled fire occurrences on a yearly basis, accounting for spatial and temporal dependence. I fitted the model using a Bayesian approach that used Integrated Nested Laplace Approximation (INLA). I defined a Log-Gaussian Cox process and accounted for the spatial component with the stochastic partial differential equation (SPDE) approach. The SPDE provides a Markovian representation of the Matérn covariance (Lindgren et al. 2011, Appendix 2.3 for more details on the method). Additionally, I accounted for the temporal component by including a first-order autoregressive random effect across years (AR1).

We developed separate statistical models for each fire type, including the following variables:

- Deforestation fires: MCWD+ edges density + transport cost for exportation + rural settlements + annual crop cover + perennial crop cover + pastoral land cover + annual crop cover increase + perennial crop cover increase + pastoral land cover increase + integral protection areas + sustainable use areas + indigenous land + PADDD + priority list

- Agricultural fires: MCWD + transport cost for exportation + rural settlements + dominant agricultural land cover + annual crop cover increase + perennial crop cover increase + pastoral land cover increase + integral protection areas + sustainable use areas + Indigenous land + PADDD + priority list

- Forest fires: MCWD+ distance forest edges + transport cost for exportation + rural settlements + integral protection areas + sustainable use areas + Indigenous land + PADDD + priority list

For each fire type, I fit four separate models to investigate the changes in the relationship between variables and fires over time. I decided to use the different phases of the PPCDAm to define different periods, as each phase is characterized by different deforestation

patterns and the deployment of distinct policy instruments to reduce deforestation and environmental degradation in the region (West and Fearnside 2021), as well as a final period from 2019 to 2021, to capture the effects of the weakening of environmental policies and the resulting surge in deforestation rates during this time (Menezes and Barbosa 2021, Figure 5). I defined the following time periods:

- Period 1 (2009-2011): phase 2 of the PPCDAm, accounting for 5 640 ± 297 km^2 of annual deforestation.
- Period 2 (2012-2015): phase 3 of the PPCDAm, accounting for 4 958 ± 725 km^2 of annual deforestation.
- Period 3 (2016-2018): part of phase 4 of the PPCDAm, accounting for 6 864 ± 169 km^2 of annual deforestation. I excluded 2019 to ensure this period reflects pre-Bolsonaro administration environmental policies.
- Period 4 (2019-2021): phase of the dismantlement of environmental policies, accounting for 11 020± 951km^2 of annual deforestation.

We reported results as log-linear posterior means with 95% credible intervals. I used the package inlabru v2.5.2 of the software R 4.3 (R Core Team 2022).

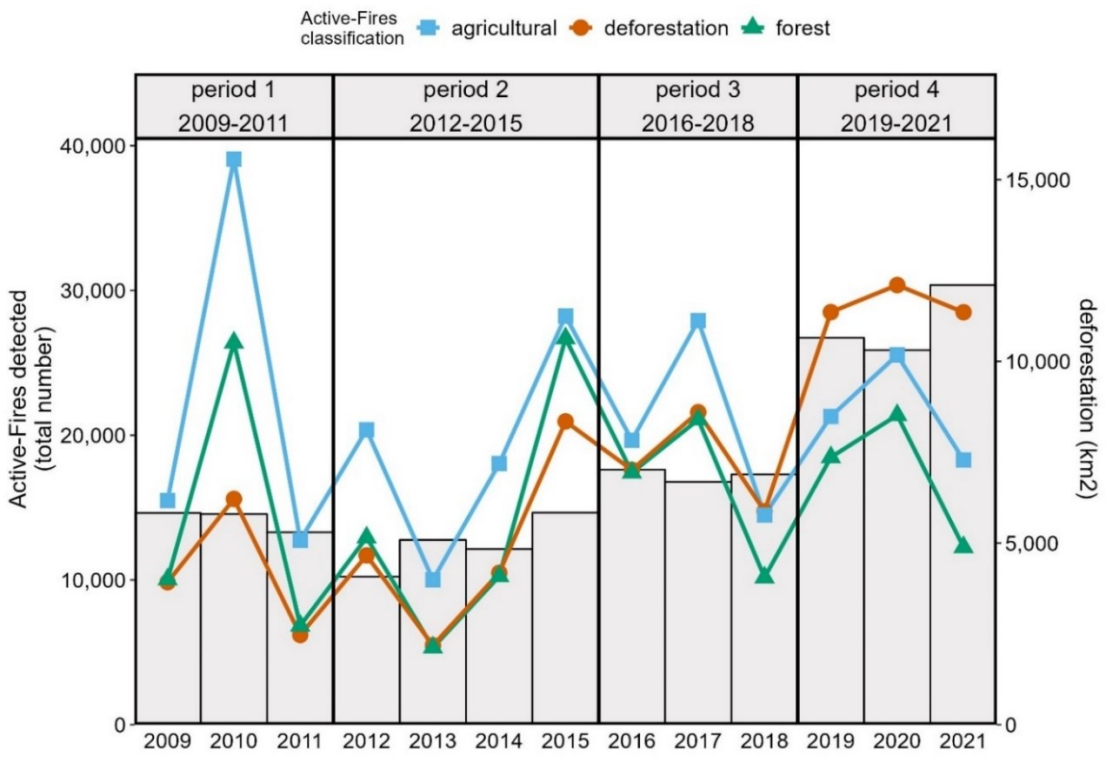


Figure 5. Number of active fires detected by MODIS and annual deforestation detected by PRODES in km2.

2.4. Results

Overall, all types of fires were positively associated with droughts and transport costs and negatively associated with the establishment of protected areas and Indigenous land. Different types of agricultural land use were associated with distinct patterns of deforestation fires and agricultural fires, with more fires in pastoral landscapes. Between 2009 and 2021, areas covered by perennial crops and rural settlements were associated with a decreasing number of fires, but areas with high transport costs and forests far from agricultural land were associated with an increasing number of fires.

2.4.1 Climatic factors

More intense droughts, measured by the Maximum Cumulated Water Deficit (MCWD), were associated with increasingly high numbers of agricultural fires and forest fires, up to an MCWD of 300 mm (Figures 6 and 7). A similar pattern was observed for deforestation fires and forest fires but with a lower threshold of 100mm of MCWD (Figures 7 and 8). Areas experiencing extreme droughts (MCWD >400mm) tended to experience the highest number of forest fires from 2009 to 2015. However, after 2016, areas with an MCWD higher than 300 mm saw fewer forest fires than in the earlier period.

2.4.2 Forest degradation

Forest fragmentation was positively associated with deforestation fires for all periods (Figure 6), except in areas with low to moderate forest fragmentation (1-50m edges/ha) between 2012 and 2015. Between 2009 and 2018, areas with low to highly fragmented forest (>1m edges/ha) were associated with similar numbers of deforestation fires. However, between 2019 and 2021, areas with highly fragmented forest cover (>50 m edges/ha) were associated with more deforestation fires.

For all the periods, forests within 1 km of agricultural land were positively associated with forest fires, while forests located more than 1 km away from agricultural land were negatively associated with forest fires (Figure 8). The number of forest fires generally decreased with increasing distance from forest edges, with this trend extending up to five kilometres between 2009 and 2018 and up to ten kilometres from 2019 onward. From 2016, areas situated three to five kilometres from agricultural land experienced more fires than in previous years. This pattern intensified after 2019, when forested areas within one to three kilometres of agricultural land saw a further increase in fire occurrences.

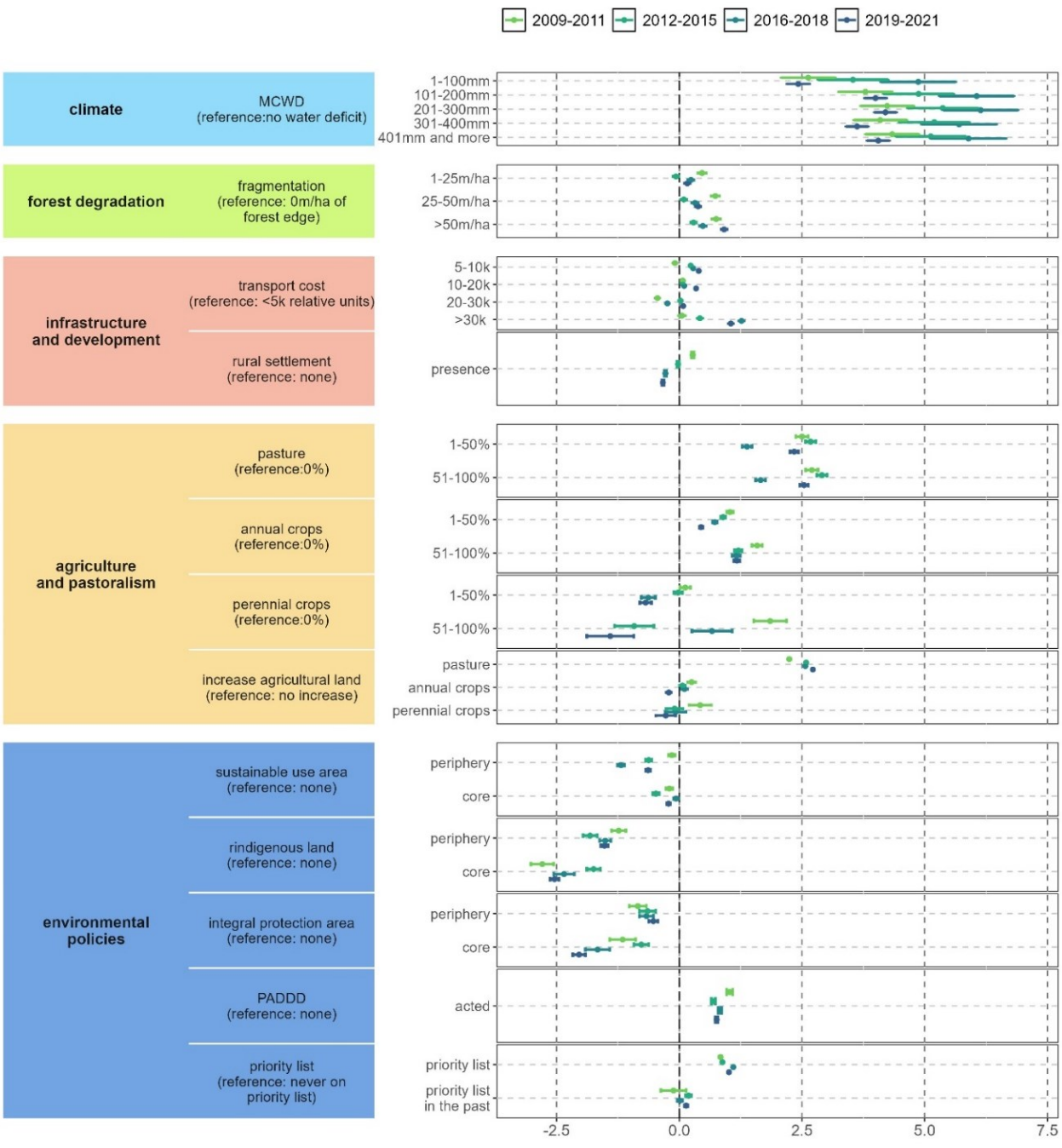


Figure 6. Posterior means and 95% credible intervals for explanatory variables of the models for deforestation fires in the Brazilian Amazon (log scale). Intervals lower than 0 indicate a variable negatively associated with deforestation fires, and intervals higher than 0 indicate variables positively associated with deforestation fires.

2.4.3 Infrastructure and development

Overall, all types of fires became increasingly frequent in the areas with higher transport costs between 2009 and 2021. From 2012 to 2021, low transport costs (5-10K RU) were positively associated with deforestation fires, and from 2019 onwards moderate transport costs (10-20K RU) were also positively associated with deforestation fires (Figure 6). Across almost all periods, areas with low to high transport costs (5-30K RU) were associated with similar numbers of agricultural and forest fires (Figures 7 and 8). From 2009 to 2011, areas with high transport costs (>30K RU) were associated with a similar number of deforestation fires, agricultural fires and forest fires as elsewhere. But from 2012 onwards, they became associated with higher numbers of agricultural fires and from 2016 onwards they became associated with higher numbers of deforestation fires and forest fires.

Rural settlements were positively associated with deforestation fires and agricultural fires from 2009 to 2012 and with forest fires from 2012 to 2015 but were negatively associated with deforestation fires and agricultural fires from 2019 onwards (Figures 6, 7 and 8).

2.4.4 Agriculture and pastoralism

Pastures were associated with the highest number of deforestation fires regardless of the period, as well as higher numbers of agricultural fires than annual crops from 2009 to 2019, and perennial crops from 2016 onwards (Figures 6 and 7). Areas covered by less than half of perennial crops were negatively associated with deforestation fires from 2016 onwards, along with areas covered by more than half of perennial crops during the 2012-2015 and the 2019-2021 periods. Although areas dominated by perennial crops were associated with similar or fewer agricultural fires than annual crops and pastures from 2016 onwards, they were associated with the highest number of agricultural fires from 2012 to 2015 (Figure 6).

Pasture expansion was associated with the highest number of deforestation and agricultural fires (Figures 6 and 7). While expansions of perennial crops and annual crops were positively associated with deforestation fires from 2009 to 2011, they were either unrelated or negatively associated in all later time periods. Expansions of all types of agricultural land

were positively associated with agricultural fires from 2009 to 2015, but from 2016 onwards, only the expansions of pastures and annual crops maintained this association.

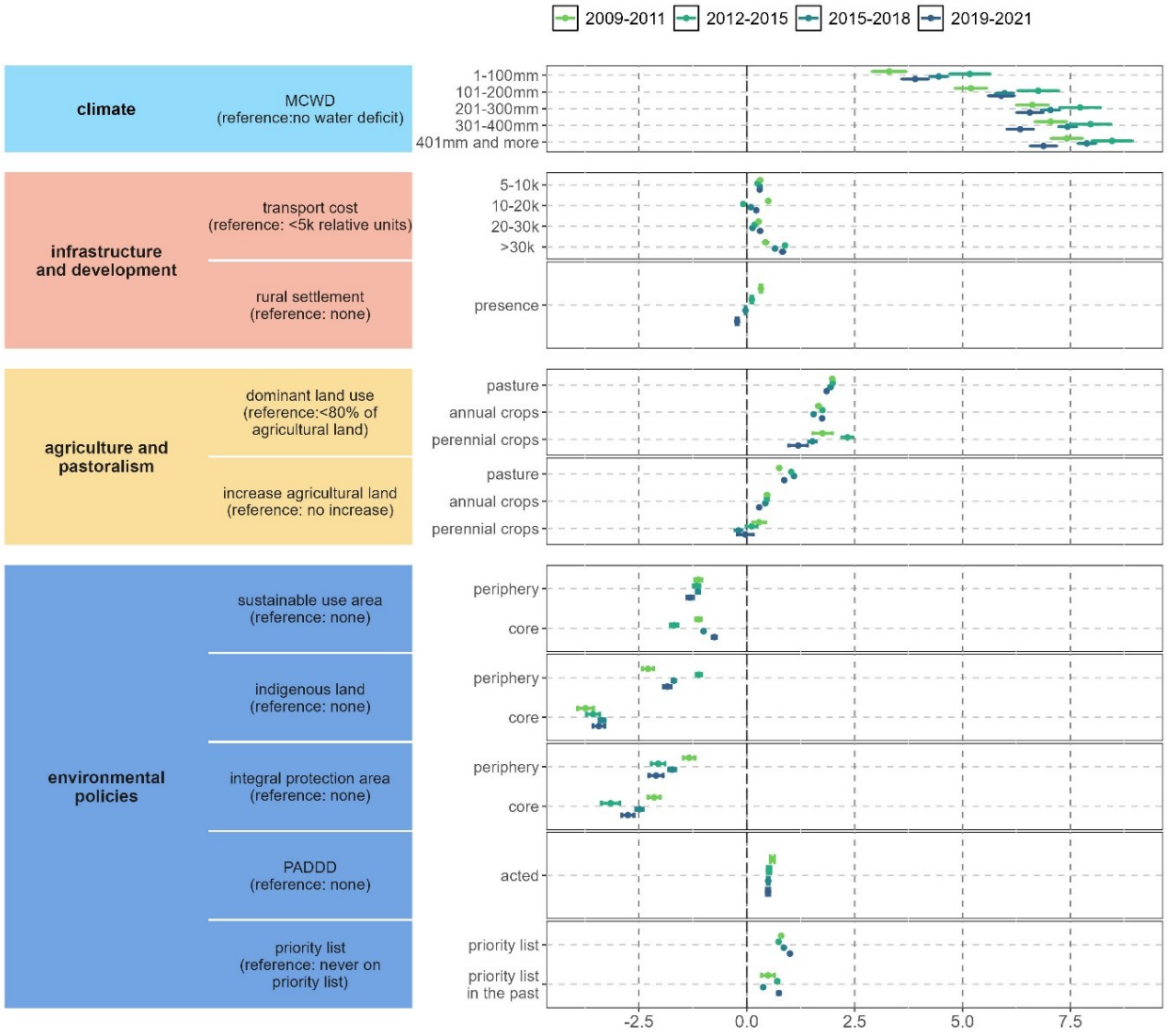


Figure 7. Posterior means and 95% credible intervals for explanatory variables of the models for agricultural fires in the Brazilian Amazon (log scale). Intervals lower than 0 indicate a variable negatively associated with agricultural fires, and intervals higher than 0 indicate variables positively associated with agricultural fires.

2.4.5 Environmental policies

All types of protected areas were associated with fewer fires than unprotected lands, regardless of the period or fire type. Indigenous lands were associated with the lowest number of agricultural fires and deforestation fires, while integral protection areas were associated with the lowest number of forest fires (Figures 6, 7 and 8). Sustainable use areas were associated with more fires of all types than integral protection areas and Indigenous lands.

The peripheries of all types of protected areas, defined as the portions of their territory within 10km of unprotected lands, were associated with more forest fires than the core of the protected areas (Figure 7). The periphery of Indigenous lands was associated with similar numbers of forest fires to non-protected areas from 2012 to 2018, as did the periphery of integral protection areas from 2012 to 2015.

Periphery of Indigenous lands and integral protection areas were consistently associated with more agricultural fires than their core areas (Figure 7), and more deforestation fires in their periphery than their core from 2016 to 2021, and from 2009 to 2012 for the Indigenous lands (Figure 6). The core of integral protection areas is associated with fewer deforestation fires and forest fires between 2016 and 2021 than previously. Downsizing and degazettement of protected areas were associated with more deforestation fires, agricultural fires and forest fires compared to non-protected areas across all periods (Figures 6, 7 and 8).

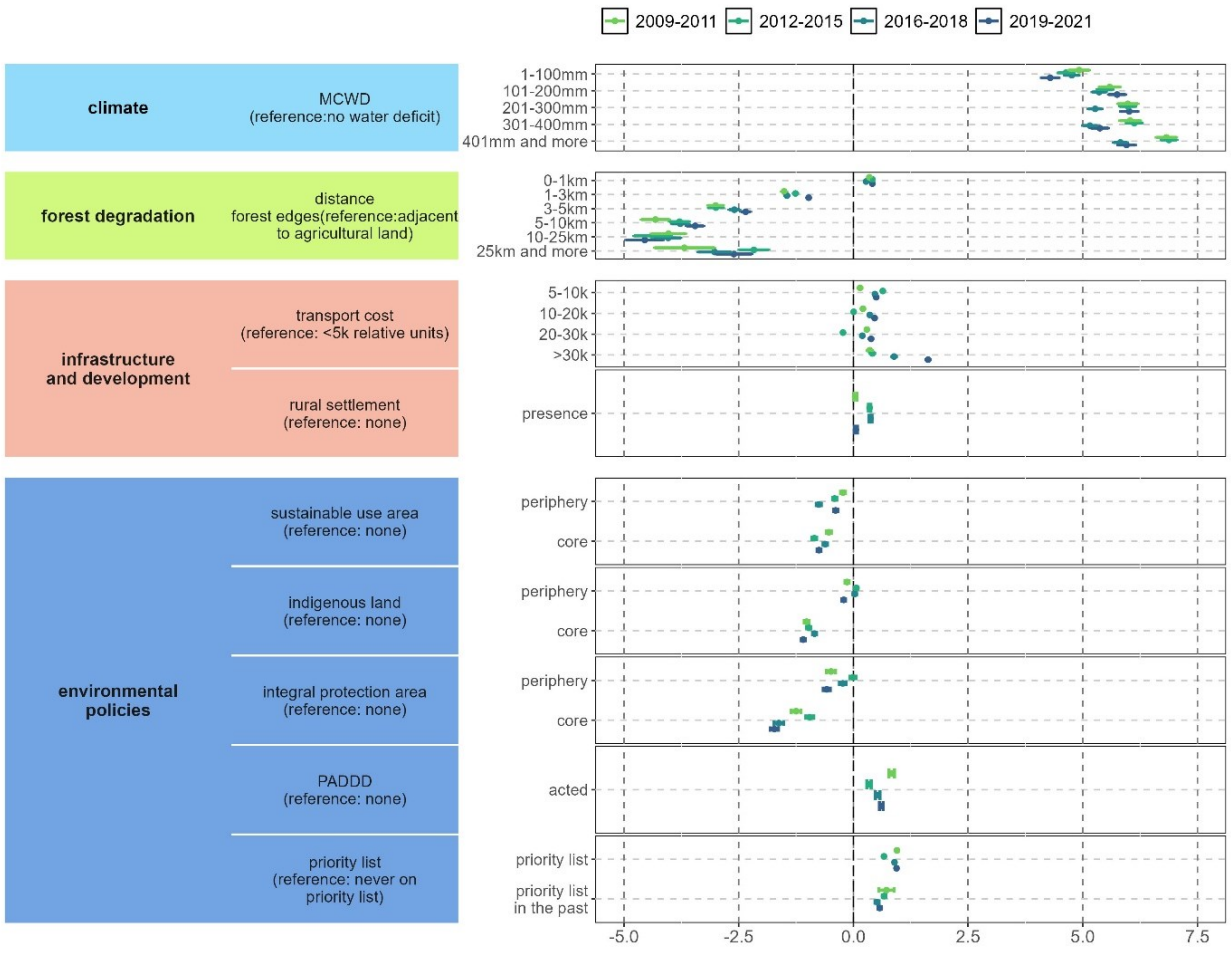


Figure 8. Posterior means and 95% credible intervals for explanatory variables of the models for forest fires in the Brazilian Amazon (log scale). Intervals lower than 0 indicate a variable negatively associated with forest fires, and intervals higher than 0 indicate variables positively associated with forest fires.

Municipalities included on the priority list were associated with higher rates of deforestation fires, agricultural fires and forest fires than other municipalities (Figures 6, 7 and 8). However, once removed from the priority list, these municipalities were associated with similar numbers of deforestation fires as municipalities that were never on the priority list, and from 2016 onwards to lower numbers of agricultural and forest fires than municipalities still on the priority list.

2.5 Discussion

This study found that, in the Brazilian Amazon, the relationship between droughts and fires is non-linear and depends on fire type, that agricultural land use is a key driver of fire regimes, that fires are becoming increasingly frequent in the remote areas and forests, and that the network of protected areas and Indigenous lands is essential for safeguarding the remaining forests from fires. By categorizing MODIS Active Fire detections into several fire types, I gain a more nuanced understanding of the relationship between fire use for land clearing, subsequent land management, and forest fires at a regional scale. The three fire types are driven by different processes and, at times, follow distinct dynamics, which can inform policy interventions and help tailor support to either limit fire use or improve fire control measures, ultimately reducing the incidence of forest fires. Additionally, this study sheds light on the increasing spread of fires within the Brazilian Amazon and the exposure of new areas to wildfire risk.

My analysis shows that droughts were associated with more fires of all types until a certain threshold, depending on the fire types and period. This aligns with other studies that show complex and non-linear relationships between weather and fire regimes across the Amazon, with many fires during relatively wet years (Fonseca et al. 2019; Libonati et al. 2021). The changing relationship across the 4 periods could highlight the fluctuating importance of climate-related drivers. The drought intensity threshold until which deforestation fires and forest fires increased (100mm of maximum water deficit across most periods) was lower than for agricultural fires (300 mm of maximum water deficit). Framed field experiments showed that command and control policies, similar to policies currently used to limit deforestation in the Brazilian Amazon, could reduce wildfire risks during droughts by promoting the use of fire control measures and alternative land management (Cammelli and Angelsen 2019). A similar dynamic might occur across the Brazilian Amazon, with agricultural producers limiting their use of fires for deforestation and better controlling their agricultural fires during more intense droughts, to avoid escape fires and potential sanctions. A more detailed analysis of the relationship between fire use and climatic

conditions throughout the year could provide greater insights into the complex interactions between climate and different fire types than my analysis relying on the driest month.

My analysis shows that fire use characteristics vary with agricultural land uses, with pastures being more prone to deforestation fires and, to a lesser extent, agricultural fires. The higher number of deforestation fires associated with pastures compared to other land use is consistent with other work (Santos et al. 2021; Nunes et al. 2022). An analysis of fire regimes in the Amazon basin from 2003 to 2005 identified more frequent fires during and after land clearing for cropland than for pastures (Morton et al. 2008). However, after the implementation of the Amazon Soy Moratorium in 2007, direct deforestation related to soy culture, the main annual crop in the Brazilian Amazon, dropped drastically and expanded on already-deforested areas (Gollnow et al. 2018). A similar dynamic was identified in a palm oil expansion frontier in the state of Para: while conversion of primary forest to palm oil was common before 2005, it decreased steeply afterwards when palm oil plantations expanded on already-deforested farmland (De Almeida et al. 2020). My results, showing continuous use of deforestation fires on pastures, suggest that indirect deforestation due to crop expansions is likely to be still contributing to fire regimes across the Brazilian Amazon.

My results also highlight the frequent use of fires after initial land clearing, especially in areas with expanding pastures and annual crops. Agricultural fires identified in this study not only include fires purposefully lit for land management but also escaped fires burning on agricultural lands and secondary forests, when they cover less than 20% of a pixel. Across the state of Pará in the Brazilian Amazon, all types of ranchers use fires to manage pasture productivity, though large-scale ranchers burn less frequently (Carvalho et al. 2020). In some places, increasing demand for local agricultural products, such as cassava flour, has also led to local intensification of the use of fires and a reduction in fallow periods by smallholders (van Vliet et al. 2013). These agricultural fires could hamper carbon sequestration in the region, by reducing carbon accumulation in soil (Stahl et al. 2017) and

preventing the regeneration of secondary forests, a carbon pool that could contribute up to 5.5% of Brazil's 2030 net emissions target (Heinrich et al. 2021).

The decline in deforestation fires, and agricultural fires from 2016 onwards, in areas with perennial crops aligns with previous studies highlighting the antagonism between fire use and perennial crops (Gutiérrez-Vélez et al. 2014; Cammelli et al. 2020). Adopting perennial crops could incentivize landholders to reduce wildfire risks in the landscape to protect valuable assets (Cammelli et al. 2020). In a study in the Peruvian Amazon, Gutiérrez-Vélez et al. (2014) found that contrary to young palm oil plantations, older palm oil plantations were negatively associated with fires, potentially due to changes in vegetation structure and lower susceptibility to escaped fires. A similar dynamic, with ageing perennial crops and reduced risks of roaming fires, could explain the declining number of fires associated with perennial crops after 2016.

While potentially limiting the occurrence of fires in the long term, large-scale adoption of perennial crops faces significant constraints, such as the access to necessary output and low economic return during first years, and some perennial crops monocultures have other harmful ecological impacts (Mendes-Oliveira et al. 2017). Moreover, there is a need to consider indirect deforestation dynamics that may arise from the adoption of intensified perennial crop systems in the region. In an analysis of the land sparing hypothesis in tropical landscapes, Pratzer et al. (2023) found that land sparing is less likely to occur when agricultural intensification involves crops destined for the international agricultural commodity market. In Indonesia, forest fires were associated with land clearing for palm oil expansion and the transfer of land ownership from local communities to private companies (Purnomo et al. 2019). This process is similar to the transfer of land ownership between small landholders and cereal farmers after the adoption of the soy moratorium in the Brazilian Amazon (Gollnow et al. 2018). Without appropriate controls on indirect deforestation and land ownership transfers, agricultural intensification and adoption of

perennial crops such as palm oil could shift the location of fire activity related to deforestation and agricultural land use rather than reduce it overall (Reydon et al. 2020).

In recent years, my analysis shows all types of fires have occurred more frequently in the areas with higher transport costs of the Brazilian Amazon. Andela et al. (2017) propose agricultural intensification drives a decrease in fire activity for tropical forest regions, due to increased mechanization and the higher value of fire-vulnerable assets. However, the process of agricultural intensification is not uniform across the Brazilian Amazon: well-connected areas have intensified their agricultural systems while marginal areas continue to rely on low-intensity agriculture (Schielein and Börner 2018; Carvalho et al. 2020). Numerous case studies show that limited access to agricultural inputs and mechanization, and a scarcity of fire-vulnerable assets, could favour the use of fires for land management (Cammelli et al. 2020; Morello and Falcão 2020). Both agricultural intensification increasing the price of already-deforested land and frequent regularization of illegal land occupation have encouraged the opening of new deforestation frontiers for land speculation, leading to increased fire risks in increasingly remote areas (Gollnow et al. 2018; Carrero et al. 2022). In an analysis at a coarser scale, Tavares et al. (2022) showed that, between 2012 and 2019 in the Brazilian Amazon, fires occurred more frequently on pixels with higher forest cover and more croplands, potentially indicating the indirect impact of agricultural intensification on fire regimes. The divergent results for the impact of cropland between my study and the study of Tavares et al. (2022) could be due to my finer-scale analysis illustrating a more localized impact of land use change on fire regimes.

My analysis shows that deforestation fires are occurring increasingly frequently in areas with highly fragmented forests, more vulnerable to escaped fires and conducive to large forest fires (Alencar et al. 2015), while forest fires are burning farther from agricultural lands. The increasing prevalence of fires in remote forests could contribute to the widespread loss of resilience of forests to disturbances already observed across the Amazon basin, especially in areas with higher anthropic disturbance (Boulton et al. 2022). The “interiorization” of fires in the Brazilian Amazon has important implications from an evolutionary ecology

perspective: forests that were previously not exposed to fire disturbance, due to wetter climatic conditions and the absence of sources of ignition, are now burning. These ecosystems can experience higher post-fire mortality than seasonally dry Amazonian forests, because of the differences in species composition and lack of selective pressure for fire-resistant traits in the past (Balch et al. 2011). Fires alter species composition and structures of forests, creating favourable conditions for future fires and shifting the vegetal community toward increased dominance of fire-adapted species (Silvério et al. 2013; Balch et al. 2015; Prestes et al. 2020).

The lower number of fires identified in all types of protected areas is consistent with previous analyses of deforestation and fires in the Brazilian Amazon (Adeney et al. 2009; Nolte et al. 2013). My analysis highlights the increase in all types of fires after the downsizing and degazettement of protected areas, highlighting the importance of maintaining their existence to prevent both deforestation and forest degradation. Indigenous lands, associated with the lowest number of deforestation fires and agricultural fires, serve as a crucial barrier against deforestation, especially in high-pressure frontiers (Soares-Filho et al. 2010). Some of the forest fires identified within Indigenous land could be related to Indigenous land management, including small-scale clearing of secondary forests for food production or burning of other vegetation types (Schwartzman et al. 2013). Extensive traditional knowledge of fire management may allow some Indigenous communities to use fire while preventing accidental large-scale wildfires, such as through the use of controlled fires to reduce fuel load (Bilbao et al. 2010).

When compared to other types of protected areas, the relatively higher frequency of fires in sustainable use areas could be explained by several factors: sustainable use areas are inhabited, livelihood activities relying on fires are allowed, and they generally receive less funding than integral protection areas for fire management and other activities (Oliveira et al. 2021). While there are legal requirements for conducting fires in sustainable areas, such as the acquisition of burning permits or the clearing of wide fire breaks, many are unrealistic given the constraints met by landholders and are frequently disregarded (Carmenta et al.

2013). Certain sustainable use areas also have loose regulations on land ownership, which can lead to extensive deforestation (Jesus and Catojo 2020). Capitalizing on fire management experience in two sustainable use areas, Nóbrega Spínola et al. (2020) proposed different strategies to reduce fire in sustainable use areas, including the involvement of local communities in fire management, support to community fire-fighters, development of alternative livelihoods to swidden cultivation and better forecasting of wildfire risks.

My models highlighted a consistently higher occurrence of all types of fires close to the border of protected areas and Indigenous lands throughout the whole Brazilian Amazon. Dos Santos et al. (2021) found that fires within Indigenous lands in the state of Rondônia were partly explained by fire occurrence and land use in their immediate vicinity, while Walker et al. (2020) identified similar dynamics in the state of Mato Grosso. Silva et al. (2022) revealed a higher prevalence of fires in Indigenous lands either cut by highway or located within 10 km of a highway. Complementary measures to avoid forest fires and control fire use around protected areas could help reduce the occurrence of fires within protected areas (Walker et al. 2020). Analysis relying on medium to high-resolution mapping of fires within and close to Indigenous lands and protected areas, like the one done by Walker et al. (2020) in Mato-Grosso, could help shed light on the complex relationship between fires outside and within these areas.

My analysis also suggests that the Priority List Program, aiming to reduce deforestation at the municipality level through incentives and disincentives, has had positive outcomes on the use of fires, especially regarding deforestation fires and, to a lesser extent, on agricultural and forest fires. This adds to conclusions from previous studies showing that the priority list program was a cost-efficient way to reduce deforestation (Assunção and Rocha 2019). I also identified a general decrease in deforestation and agricultural fire use in rural settlements, a type of land governance covering about 8% of the Brazilian Amazon and supporting the livelihoods of approximately 600,000 families. While part of this decline could be due to a

decreasing amount of remaining forest available to burn, I need to better understand what is driving this trend, given that the landscape is often highly fragmented, and settlers rely on low-intensity farming practices on small landholdings.

2.6. Conclusion

Climate, forest degradation, infrastructure, agriculture and environmental policies influence deforestation, agricultural and forest fires. Pastures remain strongly associated with deforestation fires while all agricultural land use contributes to agricultural fires. This showcases the need to not only alleviate deforestation but also support better land management strategies on already-deforested land to prevent fires from escaping and further degrading the remaining forests.

The fire regime intensified in remote regions of the Brazilian Amazon and within forests located farther from agricultural lands. This “interiorisation” of fires in the Brazilian Amazon, occurring in the context of a drying climate and a loss of resilience of forests to external disturbances, raises concerns about potential forest diebacks in the region.

Protected areas experienced fewer deforestation fires and forest fires than unprotected lands, and within protected areas fires were more frequent in areas near unprotected land. Over time, rural settlements and municipalities that were removed from the priority list were associated with fewer fires, highlighting the potential of policies aiming to control land use outside of protected areas. Strengthening environmental policies, ensuring adequate resources for agencies responsible for fire governance, and securing the land rights of traditional and Indigenous communities are crucial for reducing fire pressures and tackling deforestation and forest degradation simultaneously.

Chapter 3: Comparative analysis of wildfire risks and fire regime dynamics in the North of Mato Grosso

3.1 Main contributions

- Regional climate change created a longer and more intense fire season over the North of Mato Grosso, while each land governance category experienced different trajectories in land use change and associated wildfire risk

- Despite experiencing the highest increase in wildfire risks, rural settlements quickly reduced their overall burned areas and areas of forest burned

- Indigenous lands have a different fire regime than rural settlements and undesignated lands, characterised by a lower percentage of burned areas, large fires burning mainly natural vegetation types, greater stability over time and vulnerability of forests to very intense fire seasons

- Analysis of the dynamics of wildfire risks and fire regimes suggests different challenges and adaptive potentials of different types of landholders across land governance types, supporting the need for tailored approaches to support adaptation to changing wildfire risks

3.2 Introduction

Fire regimes in tropical forest landscapes result from the" i*nteraction of social, economic and biophysical factors operating with feedback and interactions across spatial scales*" (Carmenta et al. 2011). In the Brazilian Amazon, social and economic drivers shape the spatial distribution of agricultural systems and associated fire use, while climatic and ecological factors determine natural ecosystems' flammability (Cano-Crespo et al. 2015; dos Reis et al. 2021). These interacting factors create highly dynamic systems where feedback loops influence fire regimes over time (Archibald et al. 2018). Over the 21st century, there has been a decoupling between deforestation and fire activity, with the ratio of active fires to annual deforestation increasing (Libonati et al. 2021). Fires are also occurring with increasing frequency in areas of high forest cover (Tavares et al. 2022) and farther from forest edges (Morton et al., 2013, previous chapter). Additionally, new regions of the Amazon are increasingly exposed to extreme climatic conditions and shifting deforestation frontiers, resulting in abnormally high fire activity (Silveira et al. 2020).

In response to these changes in fire regimes, Integrated Fire Management has been proposed as a strategy combining wildfire risk prevention, response, and recovery (Oliveras Menor et al. 2025). Integrated Fire Management remains highly context-dependent and requires wildfire risk assessments tailored to local socio-ecological conditions while also being broad enough to be embedded in territorial planning(Moore 2019; Oliveras Menor et al. 2025). Subregional level assessments could prove useful in Brazil, due to the differences in fire regimes and processes driving them across Amazonian states (Fanin and Van Der Werf 2015; Silveira et al. 2020; Xu et al. 2021), as well as the operational capacities of state-level institutions for land use and fire management.

Mato Grosso offers a valuable case study for examining adaptations to shifting wildfire risks. Historically, it has experienced some of the highest deforestation rates and fire activity in the Brazilian Amazon, largely driven by the expansion of export-oriented agriculture during the early 2000s driven by agricultural technology improvements, integration into global commodity markets, and major infrastructure projects (Morton et al. 2008; DeFries et al. 2013). This land-use change not only increased ignition sources (Cano-Crespo et al. 2015) but also contributed to regional climatic changes, including hotter and drier conditions (Dubreuil et al. 2012). Despite significant reductions in deforestation rates since the late 2000s, forest fires have persisted at high levels (Morton et al. 2013; Volpato et al. 2023).

Policy interventions have struggled to address this challenge. Temporary fire bans implemented in Mato Grosso during the last three fire seasons have achieved limited success, reflecting similar shortcomings in broader biome-wide restrictions (Schelly et al. 2021; Machado et al. 2024). This limited impact partly stems from the reliance of some landholders on fire as a land management tool and the difficulty of enforcing bans in remote areas (Sorrensen 2009; Morello et al. 2017; Carmenta et al. 2019). While economic losses and livelihood impacts could motivate landholders to adopt better fire control practices or transition to fire-free agricultural systems, these decisions depend on enabling factors such as capital availability and reduction of local wildfire risk (Morello et al. 2017; Cammelli and Angelsen 2019; Cammelli et al. 2020).

An important yet underexplored dimension of this problem is the role of land governance categories. Different governance regimes influence land-use processes, access to resources, fire management capacities, and policy enforcement (Schielein and Börner 2018; Reydon et al. 2020). They also shape important social dynamics such as land tenure security and the establishment of collective norms for ecosystem management (Morello and Falcão 2020). Despite their importance, there is currently no analysis of how wildfire risks and fire regimes differ across land governance types, nor how these differences might affect adaptation strategies to changing wildfire risks. Currently, most public fire management programmes in Brazil focus on protected areas and Indigenous lands (Oliveira et al. 2021),

despite a higher prevalence of fires in other types of land governance (results from previous chapter)

Recent advances in medium to high-resolution satellite imagery provide new opportunities to capture detailed spatial patterns of fire regimes and their associations with land use (Xu et al. 2021; Walker et al. 2022; Da Silva Arruda et al. 2024). In this chapter, I explore the simultaneous trajectory of wildfire risk and fire regimes across different land governance types to understand the challenges of adapting to changing wildfires. I employ medium-resolution satellite data to conduct a comparative analysis of changes in wildfire risks, fire regimes, and the relationship between climatic conditions and fire activity across the three main land governance types in North Mato Grosso. This approach aims to identify the specific challenges and opportunities faced by different landholder groups, providing insights to inform locally tailored, integrated fire management strategies both in Mato Grosso and other Amazonian regions.

3.3 Methods

3.3.1 Region of interest

I defined a region of interest, including all or part of 12 microregions of the state of Mato-Grosso, corresponding to the overlap between the Southern Amazonian Forests bioregion and the state of Mato-Gross (Figure 9). The region of interest includes dense rainforest, ecotonal forest and savanna, typical of the Cerrado-Amazon transition zone where part of the region is located (Marques et al. 2020). The climate is characterised by an average annual precipitation of approximately 2,000 mm, with a marked dry season from May to September and pronounced inter-annual variations due to the ENSO cycle (Barbieri et al. 2019). I focused on the Southern Amazonian Forest bioregion in Mato Grosso to ensure similarities in the ecological processes linking land use, climate and fire regimes.

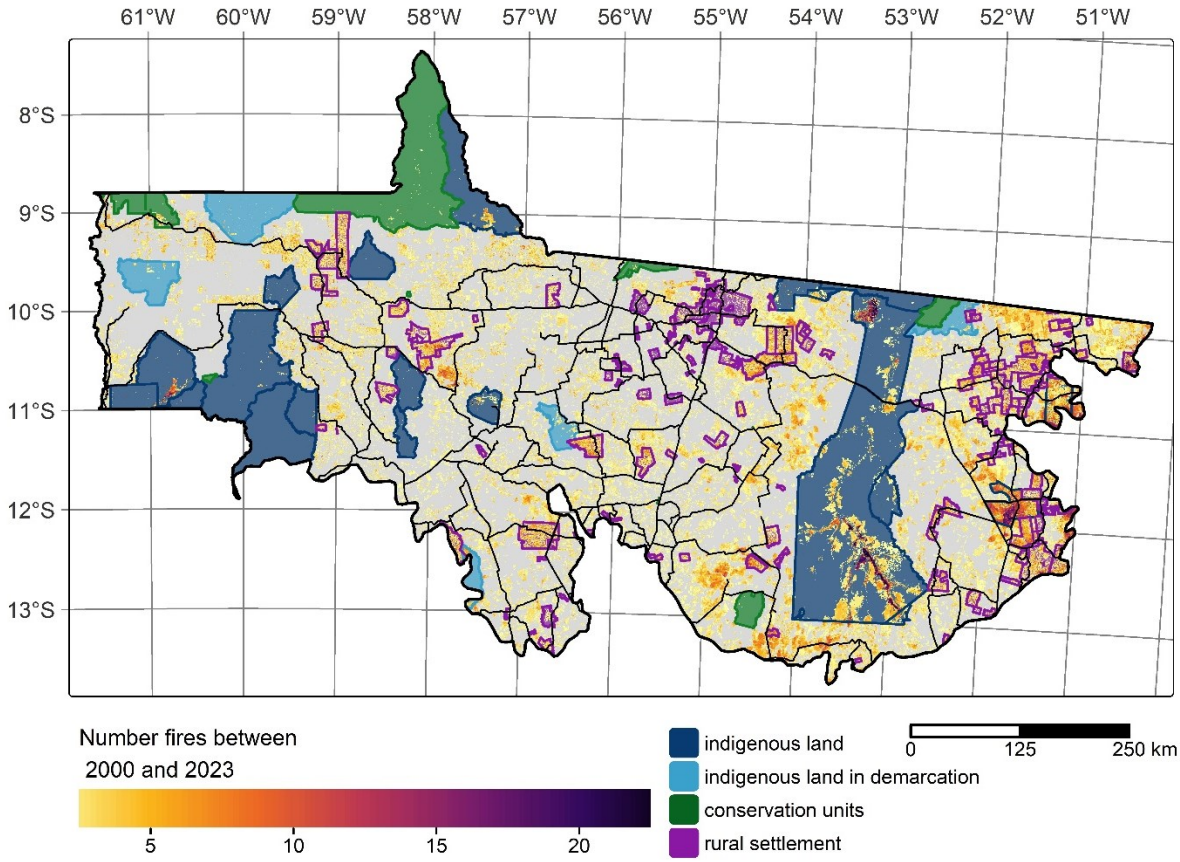


Figure 9. Map of the region of interest with the main land tenure categories and number of fires detected by MapBiomas Fire (collection 3).

The region is divided into different land governance types influencing the type of landholders that occupy them, their land tenure security, associated legislation (including agricultural and environmental regulations) and the governmental institutions supporting landholders. The major land governance types are:

- **Indigenous lands (17.3% of the region of interest):** including part or totality of 23 Indigenous lands and reserves that have been homologated by the FUNAI and hosting 31 Indigenous groups.
- **Rural settlements (6.5% of the region of interest):** consisting of 161 rural settlements delimited by the INCRA between 1979 and 2014 for the implementation of agrarian reform. These areas are dominated by small and medium landholders (mean area registered in the rural land registry: 71± 828 hectares).

- **Undesignated lands (69.6% of the region of interest):** remaining land with no specific protection regimes, mostly privately owned by a mix of small, medium and large landholders (mean area registered in the rural land registry: 517 ± 3 623 hectares)

Part of the region of interest is also covered by Indigenous lands in the process of demarcation (2.3% of the region of interest) and conservation units (4.2% of the region of interest). However, due to the small proportion of the region of interest covered by these land governance types, their low population density and the limited number of fires detected within them, I excluded these categories from the analysis.

3.3.2 Analysis of wildfires risks

Climatic conditions

To examine the influence of climate on wildfire risks, I adopted the Canadian Fire Weather Index (FWI), which uses daily surface temperature, precipitation, relative humidity and wind to estimate the moisture content of different fuel categories, and the likelihood of fires spread (Wagner 1974). FWI has shown good performance in predicting fire risks in diverse contexts, including the Eastern Amazon (Cavalcante et al. 2021). I used the dataset described in Vitolo et al. (2020), built from an analysis of ERA5 weather datasets, which provides a daily estimate of the different components of the Fire Weather Index at a 0.25° scale since 1940.

To assess recent changes in fire weather, I picked a reference period from 1940 to 1989 to have a reference period of 50 years, aligning with the methodology of Abatzoglou et al. (2019). I then assessed changes in FWI for the 1990-2023 period based on four annualised metrics of FWI described in Abatzoglou et al. (2019). Two of these relate to the risks of extreme fire weather: annual maximum FWI (FWImax) and number of days exceeding the 95th percentile of the FWI value during the reference period (FWI95), while the other two relate to wildfire risks across the season: the maximum 90-day mean value (FWImean) and the length of fires season (FSL). I used the Man-Kendall trend analysis and Thiel-Sen

estimator for each cell of my dataset to assess the existence and magnitude of any trends between 1990 and 2023 for each of the metrics.

Land use

To assess the land use changes across the different land governance types, I used MapBiomas version 9, a dataset based on Landsat imagery and automatic classification to create annual land use and land cover maps at a 30-meter resolution (Souza et al. 2020). This dataset was chosen as it provides the highest spatial and temporal resolution for land use in Brazil while showing good performances (Souza et al. 2020). To understand land use change, I grouped forest and flooded forest into a single "forest" category, other temporary crops, sugar cane and cotton into a single "other crops" category and mosaic of uses, urban areas, other non-vegetated areas, and mining into a unique "other" category. This was done to simplify the analysis and focus on the transition from forest to the main types of natural vegetation or agricultural land use, rather than multiplying categories covering only a small percentage of the region of interest. I used a PERMANOVA (Anderson 2017) and post-hoc Dunn's test (Dinno 2015) to assess the difference in land use composition across land governance types, and Man-Kendall trend analysis and Thiel-Sen estimator (Hamed and Ramachandra Rao 1998) to assess the existence and magnitude of any trends between 1985 and 2023 for each land use category.

Forest fragmentation

To assess forest fragmentation across land use types, I've selected four metrics based on the analysis done by Soares-Filho (2012) of the best metrics to predict understory fires in the Xingu headwater, a subset of the region of interest: (1) largest patch index, (2) perimeter-area ratio, (3) nearest-neighbour distance and (4) index of contagion. Divergent trends across metrics could highlight forest fragmentation processes with similar magnitudes but different impacts on landscape structure (Cushman et al. 2008) and potential wildfire risks. MapBiomas datasets were reclassified for 1992, 2002, 2012 and 2022 to assess fragmentation of forest and flooded forest, before computing the metrics at a 1-km scale using the landscapemetrics package in R (McGarigal et al. 2023). For each of the

fragmentation metrics, I derived the annual rate of change by comparing the fragmentation index across each period. I used a Kruskal-Wallis (McKight and Najab 2010) and post-hoc Dunn's test to assess the difference in forest fragmentation and forest fragmentation rate across the land governance type. I also used a Friedman test and post-hoc Dunn's test to assess the difference in forest fragmentation rate over the same pixels across the three periods.

3.3.3 Analysis of fire regimes

I investigated the fire regimes across the different land governance types, based on the strict sense definition of fire regimes given by Krebs et al. (2010) which include the temporal distribution, spatial distribution and fire characteristics. Given the regional change in land use dynamics from 2000 onwards in Mato-Grosso and the emergence of MODIS satellite imagery, I restricted my analysis to the period from 2002 to 2023. To investigate the fire regimes, I used two different datasets as response variables:

- **MapBiomas burned area collection 3:** this dataset provides monthly images of burn scars at a 30-meter resolution between 1985 and 2023. It also provides an estimate of the size of the fire that burned on each pixel. Comparisons with other products showed that it performed well in the Brazilian Amazon compared to other burned areas datasets (Ferro et al. 2024).

- **MODIS Active-Fires collection 6.1:** this dataset provides daily images of Active-Fires detected at a 1-km resolution. I selected the period between 2002[2] and 2023 to combine data from Terra and Aqua satellites, in order to maximize opportunities to detect fires and avoid detection bias due to the diurnal cycle of fire use (Giglio 2007). While the coarser resolution of the Active-Fires data affects the possibility of

[2] The Aqua satellite was launched in May 2002, before the onset of the fire season, thus I included data from 2002 in my analysis.

analysing precisely land use burned, the high temporal resolution makes this dataset useful for understanding the seasonality of fire use.

Spatial distribution

To assess the spatial distribution of the fire regimes, I used a Kruskal-Wallis test and post hoc Dunn's test to verify significant differences in the percentage of burned areas and active fires detected.1000 km^2 across the land governance types. I also calculated the relative contribution of different fire sizes to the total burned areas, reclassifying the fire size given by MapBiomas fire in the following categories: <10 hectares/10-100 hectares /100-1000 hectares /1000-10000 hectares />10000 hectares. I used Kruskal-Wallis test and post hoc Dunn's test to assess the difference across land governance types, as well as Man-Kendall trends analysis and Thiel-Sen estimator to assess the existence and magnitude of any trends for each of the fire size categories.

Temporal distribution

To assess the inter-annual dynamics of fire regimes in each land governance type, I used the percentage of total burned areas identified by MapBiomas burned areas and the number of active fires detected per 1000km^2 each year. I then used the Man-Kendall trends analysis and Thiel-Sen estimator to assess the existence and magnitude of any trends between 2002 and 2023 for both burned areas and active fires.

To assess the seasonality of fire used in each land governance type, I used the percentage of annual active fires detected each day identified by MODIS Actives-Fires. I then ran alternative GAM models to predict the frequency of fires according to the day of the year:

$$freq_i = \beta_0 + f(time_i) + \varepsilon_i$$

and

$$freq_i = \beta_0 + f_{land_gov}(time_i) + \varepsilon_i$$

$freq_i$ = percentage of total annual Active-Fires detected by day

β_0= Intercept

$f(time_i)$ = smooth, cyclic function of time unit

$f_{land_gov}(time_i)$ = smooth, cyclic function of time unit fitted separately for each land governance type

ε_i= residual error term

I used a Chi squared test to verify significant differences between the two models' specifications and selected the best model, based on AIC and lowest variance identified.

Fire characteristics

To assess the characteristics of the fires burned across land governance types, I evaluated the land use of the area burned each year. To do this, I mask each annual MapBiomas land use with the corresponding MapBiomas fire map for each year. I then calculated the proportion of the total burned areas for each year that occurred within each land use category. I used Permanova and post hoc Dunn's test to test differences across land governance types, as well as Man-Kendall trends analysis and Thiel-Sen estimator to assess the existence and magnitude of any trends for each of the land use categories.

3.3.4 Relationship between fire regimes and climatic conditions

To assess the relationship between the climatic conditions and fire regimes, I used GAM models with different specifications to assess the relationship between the annual Fire Weather Index metrics described in section 3.3.2 and the different characteristics of the fire regimes. GAM models were used due to their flexibility in capturing non-linear relationships between climate and fire regime variables. The two tested model specifications were:

$$fire_regime_i = \beta_0 + f(weather_i) + \beta_1\ land_gov_i + b_{year_i} + \varepsilon_i$$

and

$$fire_regime_i = \beta_0 + f_{land_{gov}}(weather_i) + \beta_1\ land_gov_i + b_{year_i} + \varepsilon_i$$

$fire_regime_i$= metrics of the fire regimes measured

β_0= Intercept

$f(weather_i)$ = smooth, cyclic function of one of the four Fire Weather Index metrics

$f_{land_gov}(weather_i)$= smooth, cyclic function of Fire Weather Index metrics fitted separately for each land governance type

$\beta_1\ land_gov_i$ = effect of the land governance type

b_{year_i} is a random intercept for year

ε_i= residual error term

I then used ANOVA tests to evaluate if significant differences existed between the model and, when the results were significant and the model with different land governance types showed better performance, I presented the results of the $f_{land_gov}(weather_i)$ function.

The metrics of fire regimes tested were chosen as they could be symptomatic of either adaptation measures or challenges related to fire weather. The metrics of fire regimes tested included the percentage of burned areas, percentage of burned areas covered by forest, percentage of burned areas covered by agricultural land, percentage of burned areas covered by other natural vegetation, percentage of burned areas in fires more than 100 hectares, percentage of burned areas in fires more than 1000 hectares, percentage of land burned after the 1st of September and percentage of land burned after the 1st of October.

3.4 Results

In the following sections, I will first present an analysis of the change in wildfire risks focusing on climate and land use related factors. I will then evaluate the fire regimes and their trends across the three land governance types and finally assess the relationship between change in climate and fire regimes.

3.4.1 Changes in Wildfires risks

Climatic conditions

The trend analysis shows that between 1990 and 2023, the number of days with extreme FWI increased by 18.6 days (± 12.4), the maximum FWI increased by 15.5 (± 3.8), the length of the fire season increased by 43.8 days (± 8.9), and the maximum 90-day mean FWI increased by 15.9 (± 4.5). The Tau's value from the Man-Kendall test indicates that the trends for the maximum FWI values are less monotonic than for other metrics. The spatial output of the analysis (Figure 10) shows that different parts of the region of interest experience these changes at different rates: the western part of the region experienced a faster increase in the number of days with extreme FWI and maximum 90-day mean FWI, while both the southern and western parts of the region experienced a higher increase in the fire season length and maximum FWI. The northeastern part of the region, corresponding broadly to the Xingu River Basin, experienced less pronounced changes across the four metrics.

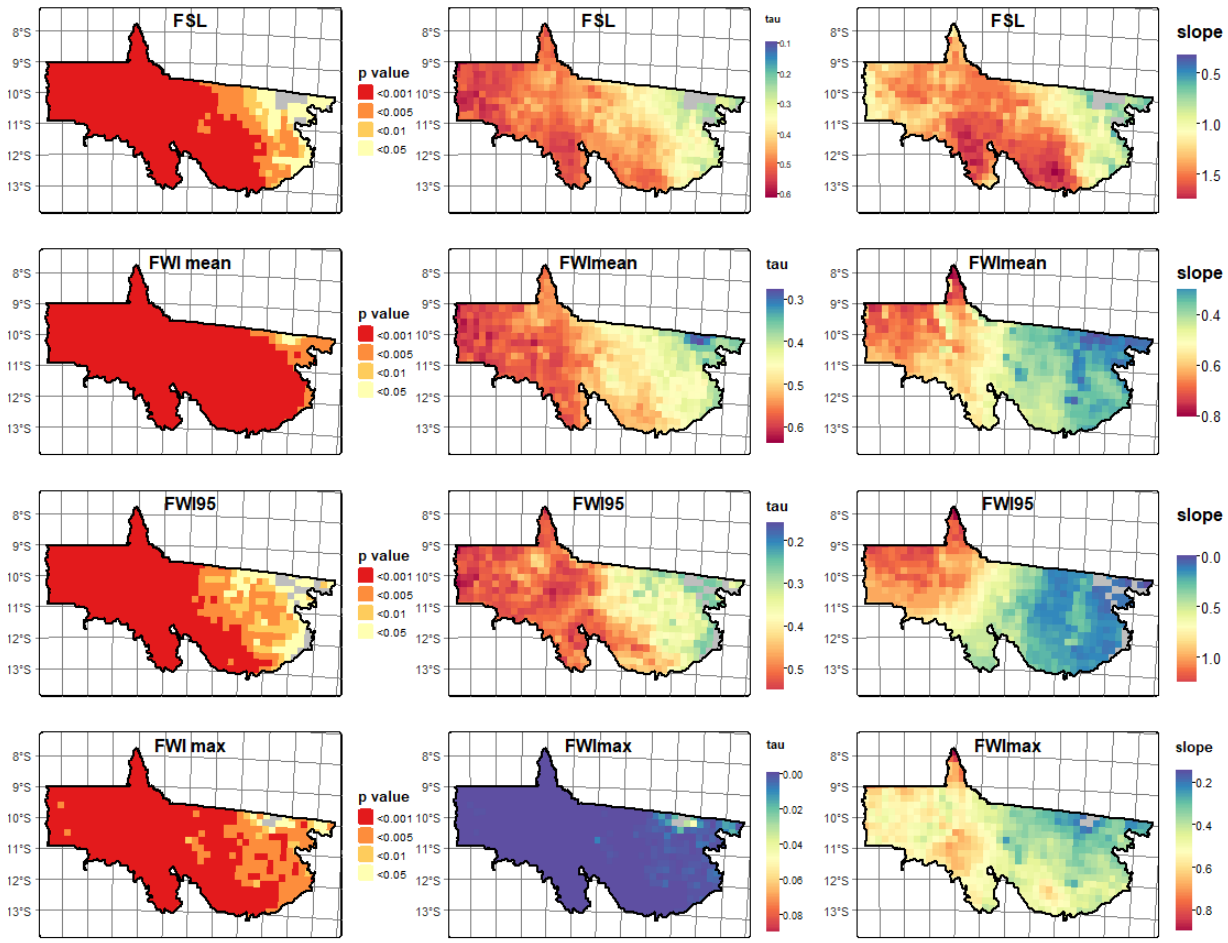


Figure 10. Change in the number of days with FWI exceeding extreme value (FWI95), annual maximum FWI (FWImax), length of the fire season (FSL) and maximum 90-day mean value (FWImean).

Land use change

Land use composition differed significantly between the three land governance types (PERMANOVA, R^2 = 0.60, p = <0.005; Table S2). Overall, rural settlements have the highest proportions of land covered by pastures and significant areas of land covered by soybean and other temporary crops (Figure 11). Undesignated lands have a lower proportion of pasture and other crops, but a larger proportion of soy plantations. Indigenous lands are the

most pristine land governance type, covered mainly by natural vegetation, but have some areas covered by pastures[3].

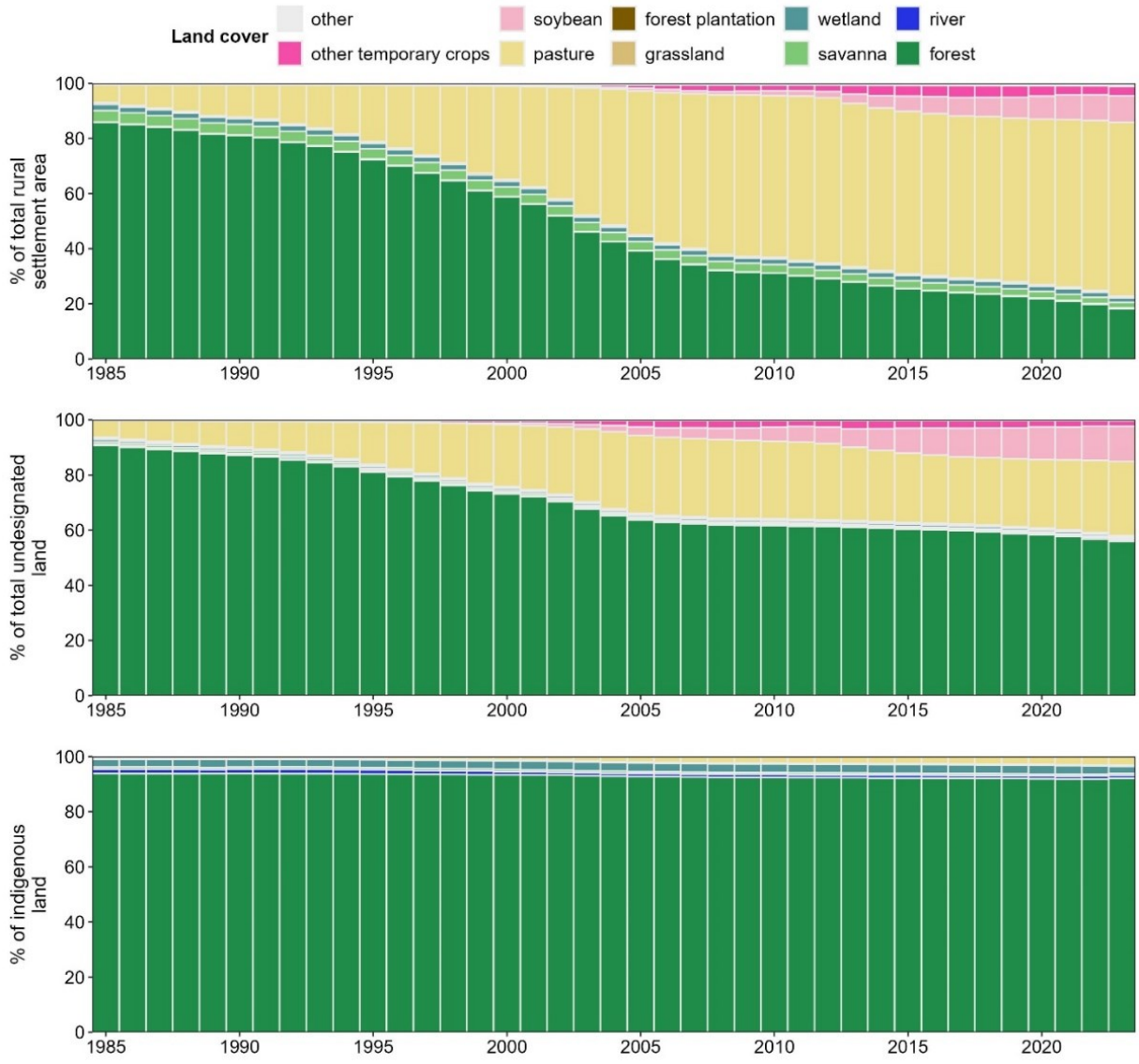


Figure 11. Histogram of the percentage of land use for each land governance type between 1985 and 2023

[3] These areas are believed to be covered by invasive *Brachiaria spp.* that are categorised as pastures by the MapBiomas algorithm, as Indigenous communities in this region don't engage in cattle ranching. More information on this in chapter 4.

Land use change was also different between the land governance types: while Indigenous lands experienced little change between 1985 and 2023, rural settlements and undesignated lands saw their forest cover reduce drastically while pasture and cropland expanded. Trend analysis identified the following trends with a rate of change higher than 0.10% of the total area yearly (Appendix 3.1 for detailed results):

- **Rural settlements:** decrease of forest (-2.06%. $year^{-1}$), increase in pasture (1.75%. $year^{-1}$), soybean (0.18%. $year^{-1}$) and other temporary crops (0.12%. $year^{-1}$)
- **Undesignated lands:** decrease of forest (-1.00%. $year^{-1}$), increase in pasture (0.60%. $year^{-1}$) and soybean (0.37%. $year^{-1}$)
- **Indigenous lands:** no significant land use change at a rate higher than 0.10% per year.

Forest fragmentation

In 2022, rural settlements were the land governance type with the most fragmented forest: they have the lowest large patch index and contiguity index and the highest perimeter-area ratio and nearest-neighbour distance. Conversely, Indigenous lands have the metrics reflecting the least fragmented forest (Figure 12). Differences in fragmentation metrics across all land governance types were highly significative (Appendix 3.2 for detailed results).

I also identified divergent dynamics in each landscape metric over the 1992-2022 period across the different land governance types. Between 1992 and 2022, there were highly significant differences in the rates of change of the largest patch index, perimeter-area ratio, nearest-neighbour distance and contiguity index across the three land governance types (Appendix 3.2 for detailed results). Rural settlements experienced the steepest change across the four metrics, indicating faster fragmentation of the forest than in other land governance types. In contrast to other periods and other fragmentation metrics, where undesignated lands experienced quicker fragmentation than Indigenous lands, no significant difference was detected in the nearest neighbour distance between these two land governance types during the 2002-2022 period. Regarding the dynamics over time of the rate of change, the only non-significant result identified was for the change of nearest

neighbour distance, that has not changed significantly between 2002 and 2022 in the rural settlements.

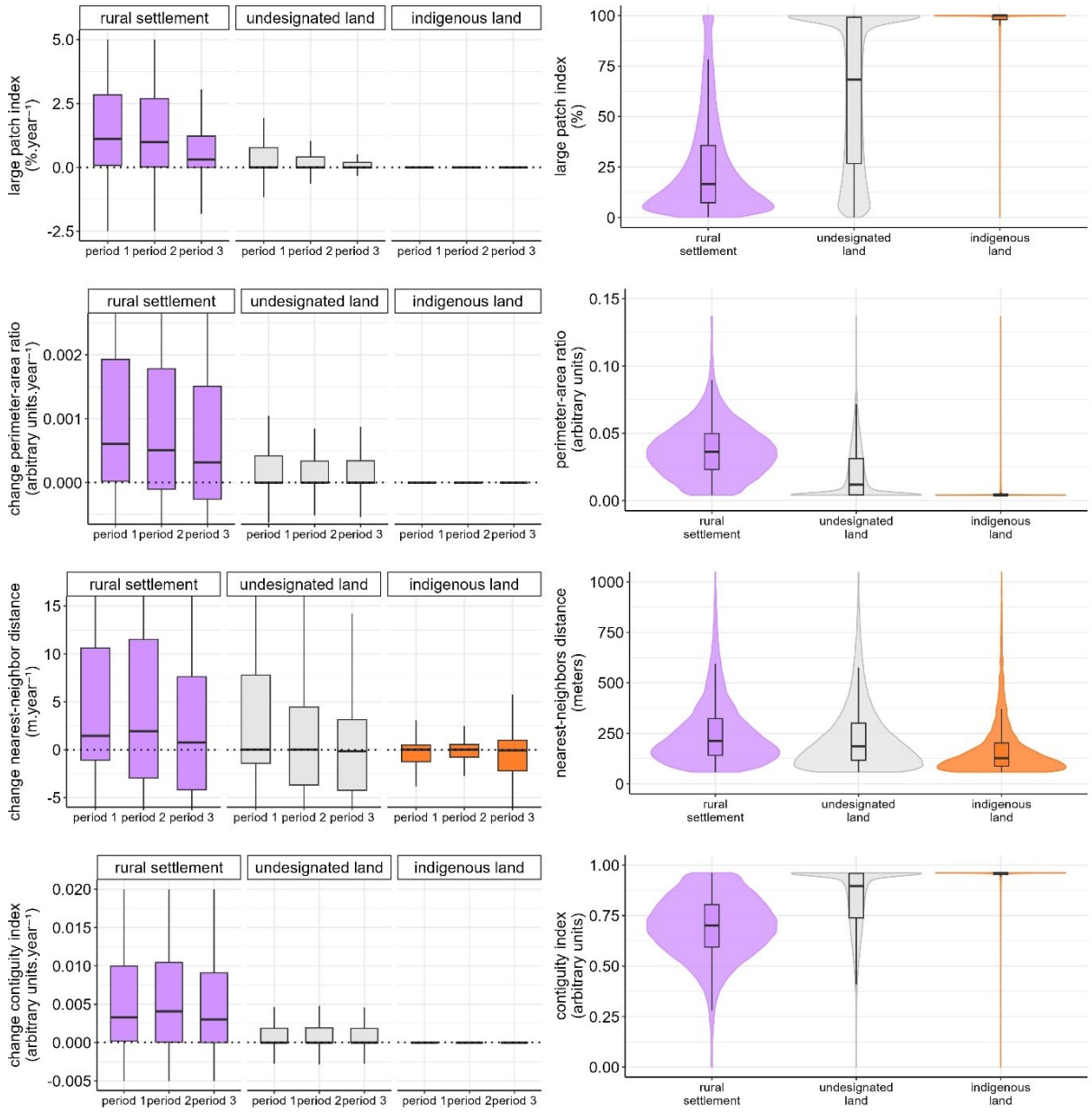


Figure 12. Violin plots of the 4 forest fragmentation metrics in 2022 (left) and boxplot of the rate of change across each decade (right).

3.4.2 Fire regimes and their dynamic

Spatial distribution

I observed that the density of active fires detected, and the percentage of total area burned varied across the three land governance types (Figure 13). The Indigenous lands have a significantly lower density of active fires detected than other land governance types, and rural settlements a significantly higher proportions of burned areas (Appendix 3.1 for detailed results).

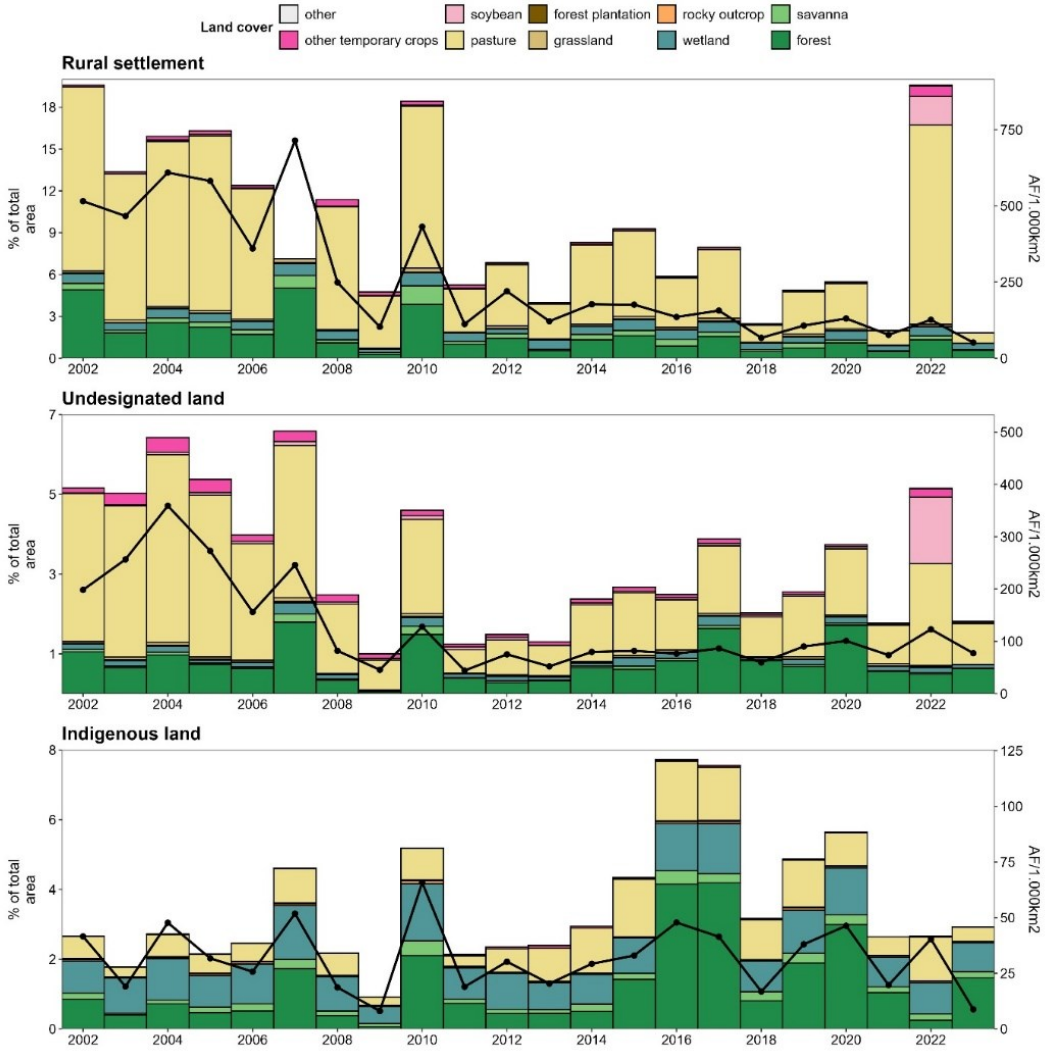


Figure 13. Total areas burned each year (bars) and number of Active Fires detected across 1000 km^2 in each of the land governance types between 2002 and 2023 (dots).

There are significant differences in the proportion of burned areas in different fire size categories across the different land governance types, except for the fires between 1000 and 1000 hectares (Figure 14, detailed results in Appendix 3.3). Rural settlements have a higher proportion of their burned area in fires of less than 100 hectares than other types of land governance, while rural settlements and undesignated lands have similar proportions of their burned areas in fires of more than 1000ha. Indigenous lands have a higher proportion of burned areas in fires of more than 10000 hectares.

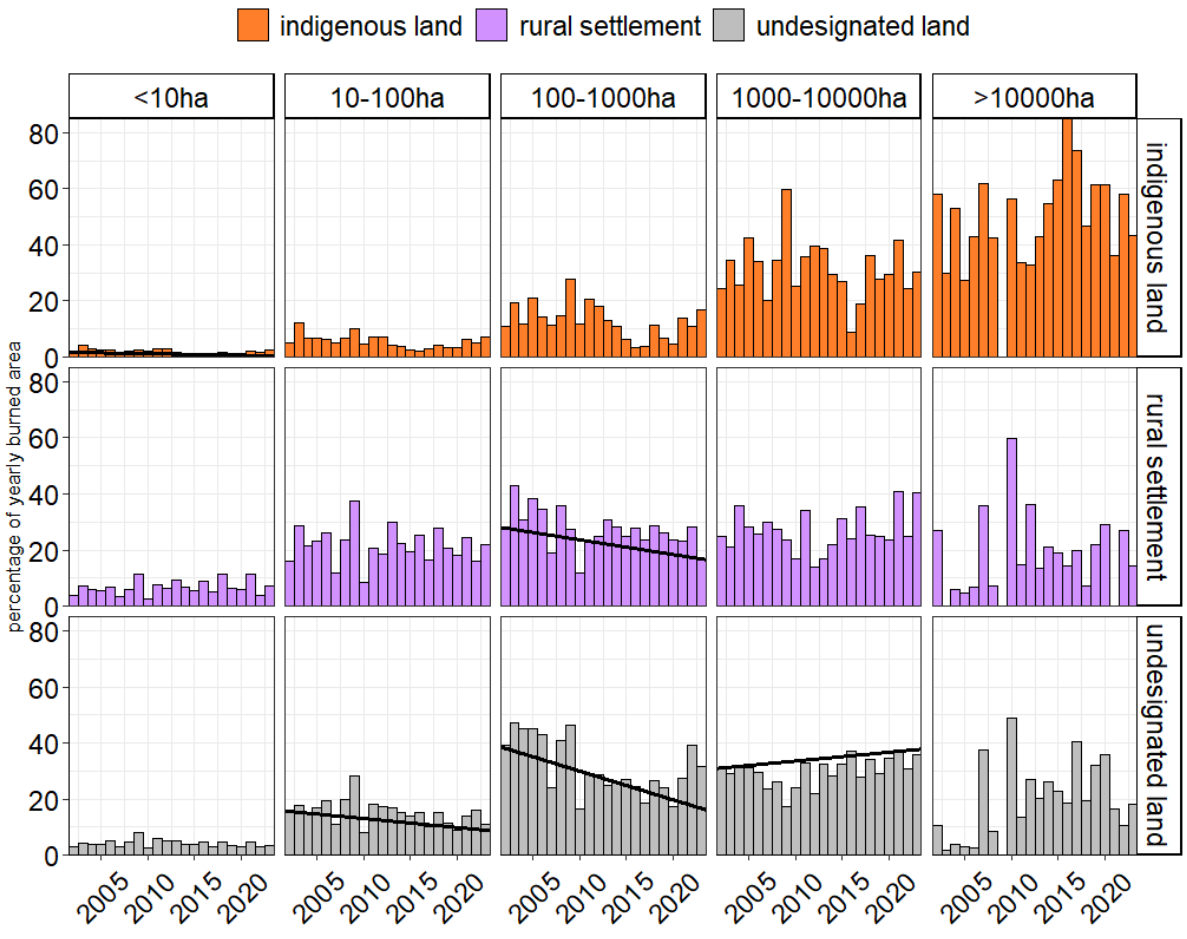


Figure 14. Histogram of the percentage of the total burned area in different classes of fire size. The black regression lines represent the monotonic trends identified with Mann-Kendall test.

Trends analysis also highlighted some different dynamics regarding the relative contribution of fires of different sizes to the fire regimes, with a decreased contribution of small to medium-sized fires for all land governance types, and a simultaneous increase in the contribution of large fires for undesignated lands. Trend analysis identified the following trends with a rate of change higher than 0.10% of the total area yearly (Appendix 3.3 for detailed results):

- **Rural settlements:** decrease contribution of fires between 100 and 1000 hectares (-0.52%.year^{-1})
- **Undesignated lands:** decrease contribution of fires between 10 and 100 hectares (-0.32%.year^{-1}) and between 100 and 1000 hectares (-1.03%.year^{-1}) and increase contribution of fires between 1000 and 10000 hectares (0.31%.year^{-1})

Temporal distribution

Trend analysis of the burned area and active fires density highlighted a decrease in the density of active fires detected in rural settlements (-21.24 AF/1000km^2.year^{-1}), and trend analysis of the percentage of total burned areas showed declining trends in rural settlements (-0.66% of total area burned.year^{-1}) and undesignated lands (-0.06% of burned area.year^{-1}).

The seasonality of fire use across the three types of land governance follows different modalities (Figures 15 and 16). The GAM model including an interaction between day of year and land governance type performed better than the GAM excluding land governance type in predicting the daily percentage of active fires detected, confirming different seasonality of fire regimes across land governance types.

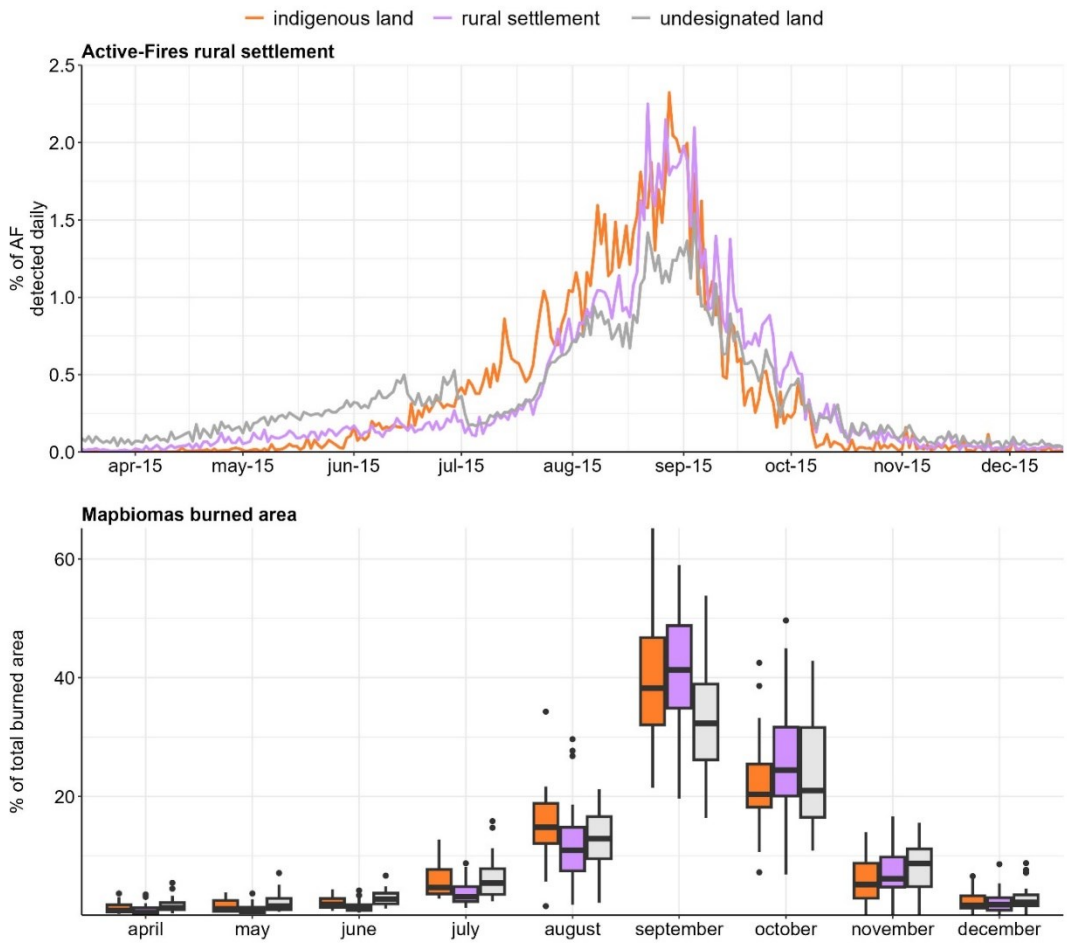


Figure 15. Daily percentage of the total of active fires detected (top) and percentage of the total of burned areas detected each month across the land governance types (bottom).

Smooth estimates of the effect of days fitted for each land governance type indicates that undesignated lands have an early peak in fire activity in July and have a more even distribution of fire activity through the season than other land governance types (Figure 16). Indigenous lands have more fire activity concentrated during the mid to late-dry season, before September. Rural settlements have more fire activity detected from October onwards than other types of land governance. These observations are consistent with the patterns observed in the daily percentage of Active Fires detected and the percentage of burned areas each month (Figure 15).

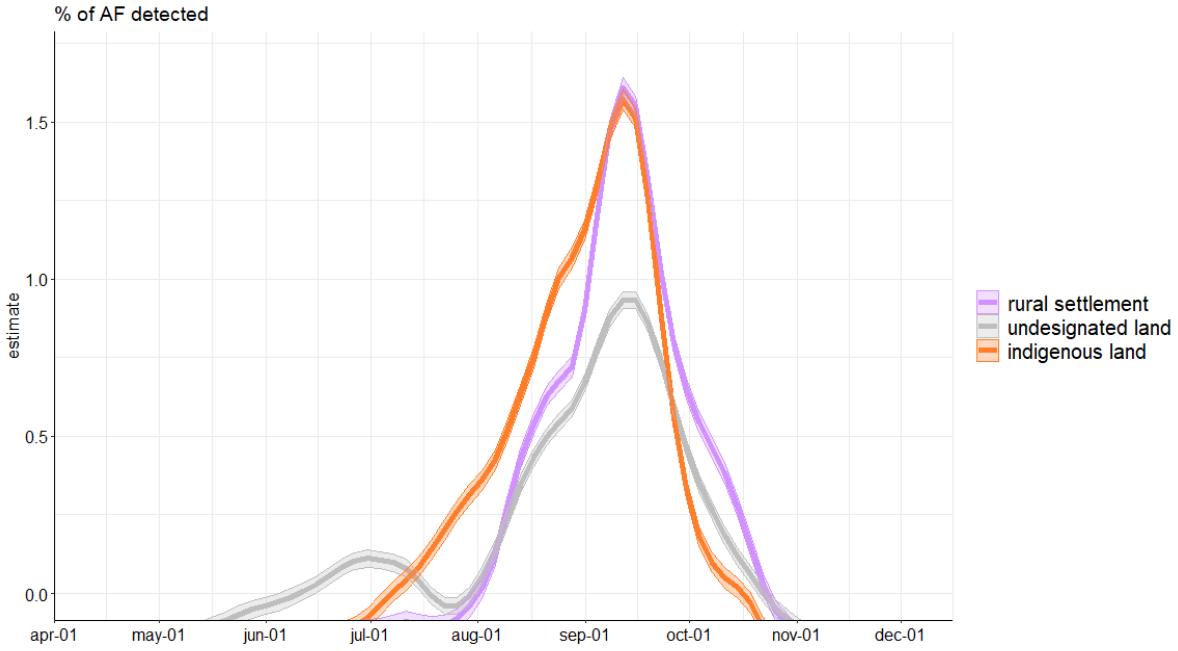


Figure 16. Smooth estimate of the effect of day on the percentage of active fires according to the land governance type.

Fire characteristics

The analysis of the land use burned each year reveals differing composition across the land governance types (PERMANOVA, R^2 = 0.64, p = <0.005; Table S4), with some similarities between undesignated lands and rural settlements and singularities of the land use burned in Indigenous lands. Post-hoc analysis revealed that the burned area across the three land governance types has a similar percentage of forest. Rural settlements and undesignated lands tend to burn a higher proportion of grassland, other temporary crops, pastures, forest plantation and soybean than Indigenous lands (Figure 17). Meanwhile, Indigenous lands tend to have a higher proportion of rocky outcrop, wetland and savanna (see Appendix 3.4 for detailed results).

Trends analysis also highlighted some different dynamics regarding the types of land use that were burned with a rate of change higher than 0.10% of the total area yearly:

- **Rural settlements:** decrease in pasture (-1.17%.year^{-1}), increase in wetland (0.46%.year^{-1})
- **Undesignated lands:** decrease in pasture (-1.28%.year^{-1}) and other temporary crops (-0.23%.year^{-1}), increase in forest (-1.11%.year^{-1})
- **Indigenous lands:** decrease in wetland (-1.09%.year^{-1})



Figure 17. Percentage of the total annual burned areas covered by different categories of land cover between 2002 and 2023.

3.4.3 Relationship between fire regimes and climatic conditions

When assessing if land governance types influence the relationship between climatic conditions and characteristics of fire regimes, including an effect of climatic variables fitted separately for each land governance type, improves the model performance for 14 out of the 44 models (Table 2). Some of the FWI indices, such as the maximum FWI and 90-day FWI mean, were more frequently improving the model performance when including land governance type. While including an effect of climatic variables fitted separately for each land governance type improves performance to predict variables related to the spatial occurrence of fire or fire characteristics, this was not the case for variables related to temporal distribution, which indicate a similar response across all land governance types.

Response variable	**explanatory variable**			
	FWI 95	**FWI max**	**FSL**	**FWI mean**
Spatial occurrence				
Proportion of total area burned	ns	ns	ns	ns
Proportion of area burned by fires of less than 100 hectares	ns	<0.0001	ns	<0.001
Proportion of area burned by fires between 100 and 1.000 hectares	ns	ns	ns	<0.005
Proportion of area burned by fires larger than 1.000 hectares	ns	<0.01	ns	<0.0001
Proportion of area burned by fires larger than 10.000 hectares	ns	<0.05	ns	<0.01
Temporal distribution				
Percenatge burned area after September	ns	ns	ns	ns
Percentage burned area after October	ns	ns	ns	ns
Fires characteristics				
Proportion of burned area covered by forest	ns	<0.05	ns	<0.05
Proportions of burned area covered by other natural vegetation	ns	<0.05	<0.0005	<0.0005
Proportions of burned area covered by pasture	ns	ns	ns	ns
Proportions of burned area covered by cropland	ns	ns	<0.0005	<0.0005

Table 2. p-values of the ANOVA test comparing the performance of the model with and without interactions between the explanatory variable and the land governance types. All reported results indicate that the model including interactions performed better.

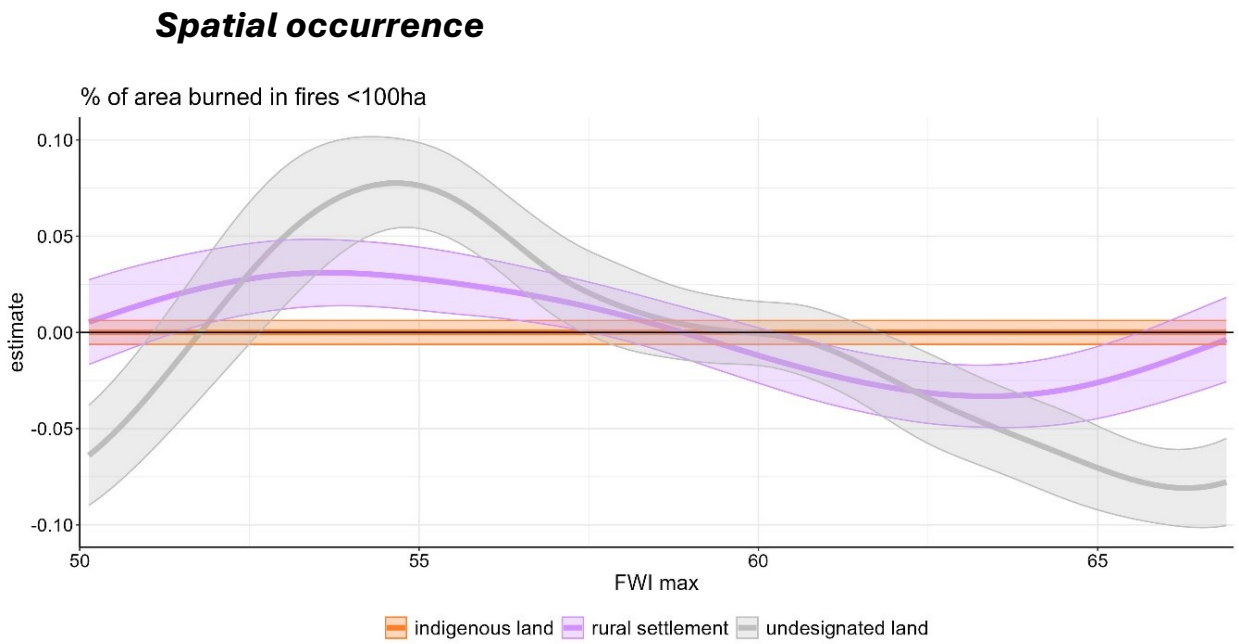


Figure 18. Smooth estimate of the effect of maximum FWI (FWI max) on the percentage of total area burned in fires of less than 100 hectares.

Compared to Indigenous lands, rural settlements, and undesignated lands have a higher proportion of fires of modest size (<100 hectares) during low to moderate intensity fire season (90-day mean FWI<37 and maximum FWI<57, Figure 18 and Appendix 3.5). However, the proportion of fires of modest size decreases during high intensity fire season (90-day mean FWI between 42 and 47, maximum FWI between 62 and 65), as well as during extreme fire season for undesignated lands (90-day mean FWI >47 and maximum FWI >65). This pattern is more pronounced for the undesignated lands than for the rural settlements. A similar pattern is observed for fires between 100-1000 hectares in rural settlements when accounting for 90-day mean FWI, but not for undesignated lands (Figure 19).

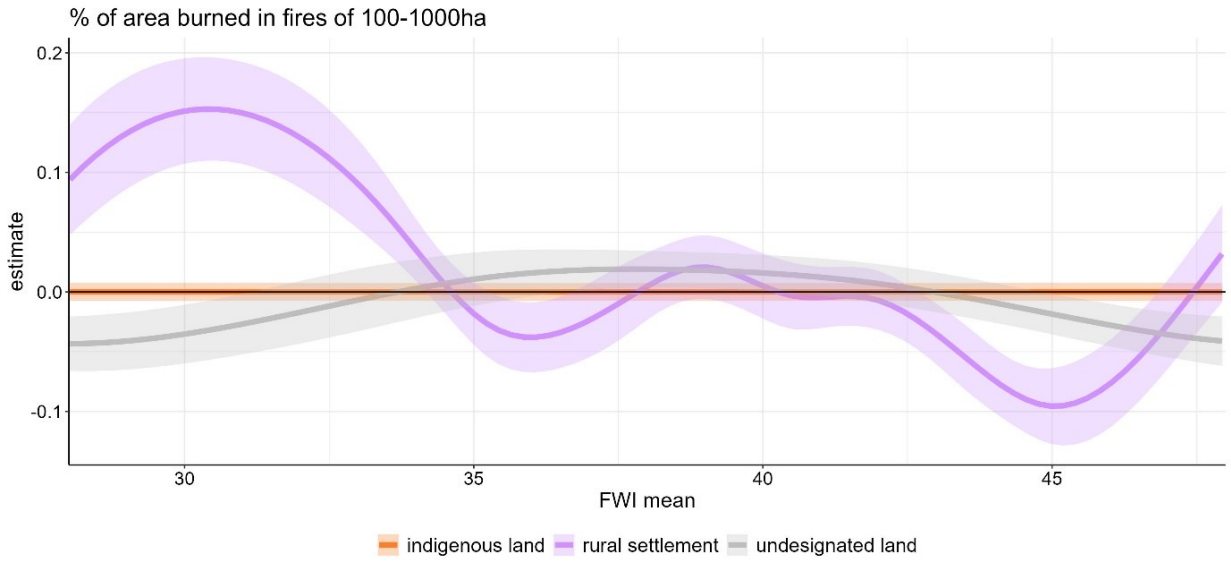


Figure 19. Smooth estimate of the effect of 90-day mean FWI (FWI mean) on the percentage of total area burned in fires between 100 and 1000 hectares.

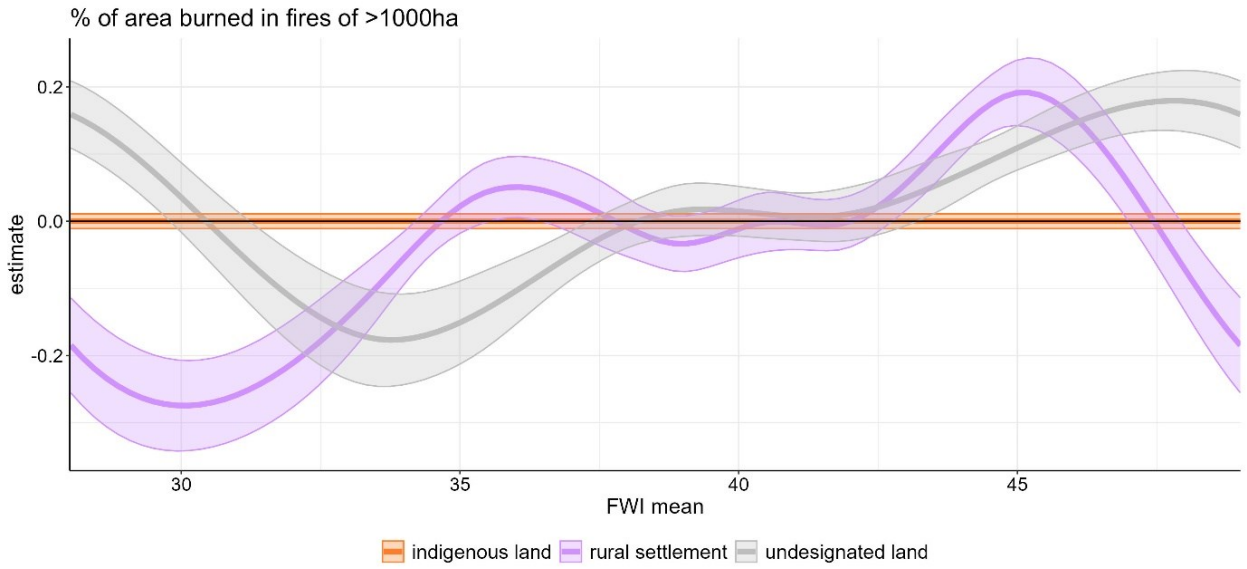


Figure 20. Smooth estimate of the effect of 90-day mean FWI (FWI mean) on the percentage of total area burned in fires of more than 1,000 hectares

Compared to Indigenous lands, rural settlements, and undesignated lands tend to have a lower proportion of fires of large size (>1000 hectares) during low to moderate intensity fire season (90-day mean FWI <34 for rural settlements and <37 for undesignated lands, maximum FWI<57, Figure 20 and Appendix 3.5). However, the proportion of fires of large size

increases during high intensity fire season (90-day mean FWI between 42 and 47, maximum FWI between 62 and 65), as well as during extreme fire season for undesignated lands (90-day mean FWI >47 and maximum FWI >65) when the proportion of large fires in rural settlements drop. The relationship between the FWI index and very large fires (>1000 hectares) is quite similar across land governance types, but Indigenous lands experience fewer of them during years with moderate intensity fire season (90-day mean FWI <32) and high intensity fire season (90-day mean FWI >45, Figure 21).

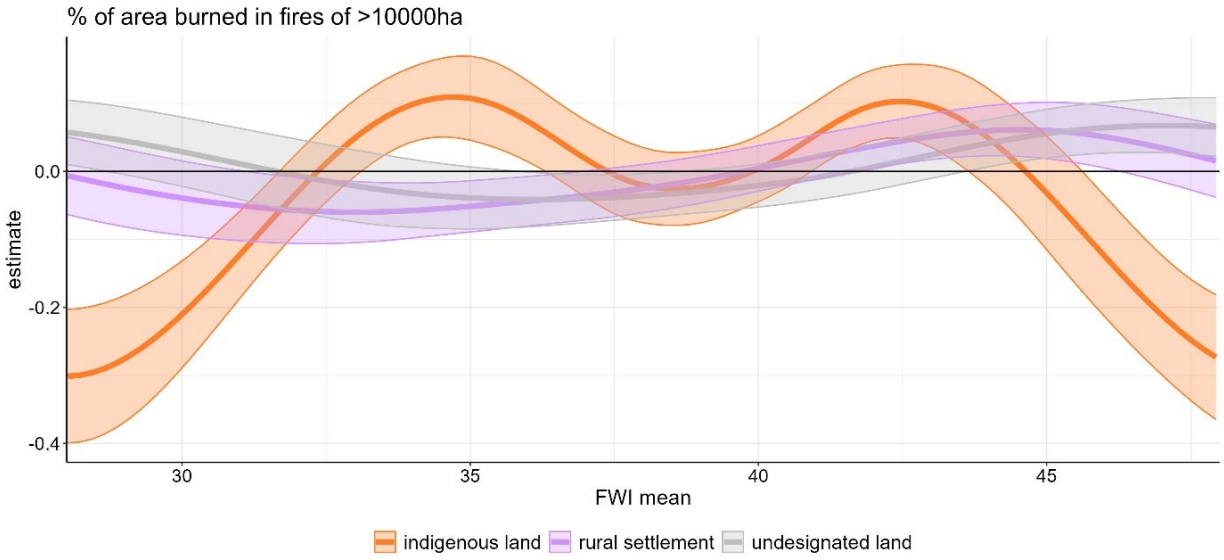


Figure 21. Smooth estimate of the effect of 90-day mean FWI (FWI mean) on the percentage of total area burned in fires of more than 10,000 hectares

Fire characteristics

Compared to rural settlements and undesignated lands, Indigenous lands tend to have a lower proportion of their burned areas covered by forest during years with moderate intensity fire season (90-day mean FWI <34, maximum FWI <57, Figure 22 and Appendix 3.5). However, the proportion of burned areas covered by forest in Indigenous lands increases during high intensity fire season (90-day mean FWI between 42 and 47, maximum FWI between 60 and 63).

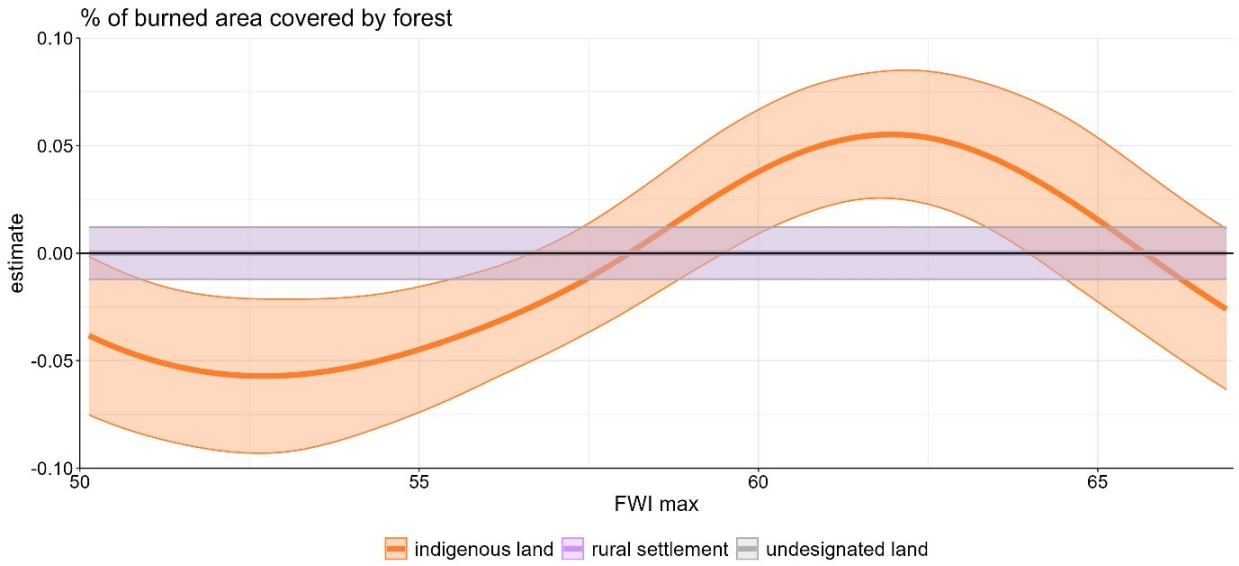


Figure 22. Smooth estimate of the effect of maximum FWI (FWI max) on the percentage of burned areas covered by forest.

Compared to rural settlements and undesignated lands, Indigenous lands tend to have a higher proportion of their burned areas covered by other natural vegetation during years with moderate intensity fire season (90-day mean FWI <32.5, maximum FWI <53, fire season lengths <35 days, Figure 23 and Appendix 3.5), as well as years with extreme fire season (90-day mean FWI >46, maximum FWI >65, FSL>75). The relationship for moderate to intense fire season was unclear and depended on the metrics.

Compared to Indigenous lands and undesignated lands, rural settlements tend to have a higher proportion of their burned areas covered by cropland during years with medium intensity of fires season (90-day mean FWI between 40 and 43, fire season length between 55 and 65 days, Figure 24 and Appendix 3.5) and a lower proportion during high intensity fire season (90-day mean FWI >43 and fire season length>65 days).

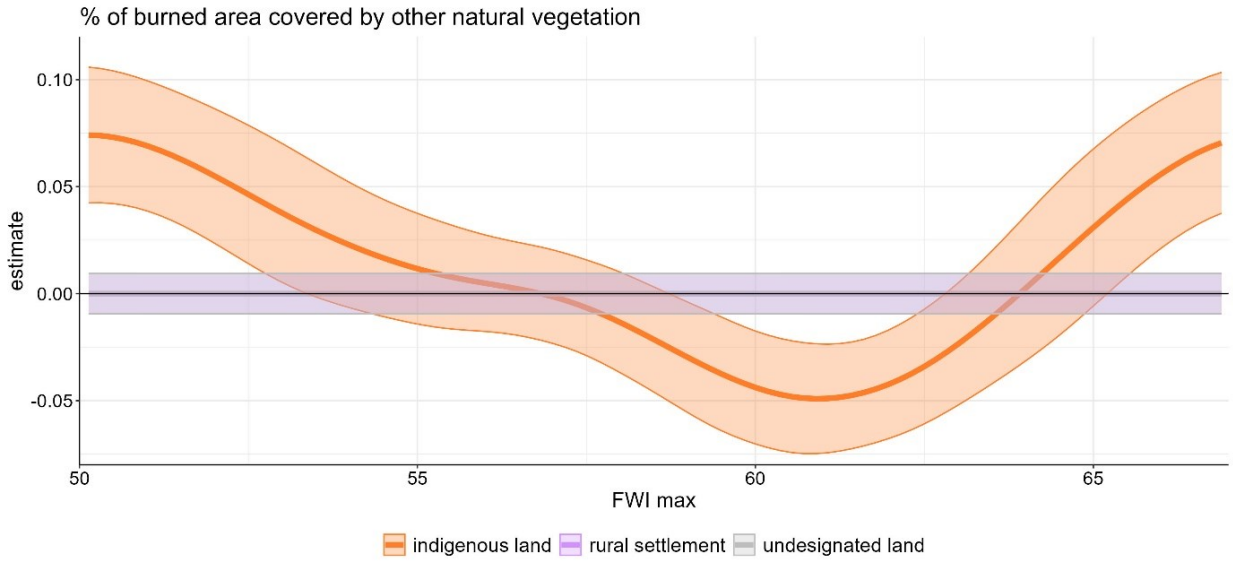


Figure 23. Smooth estimate of the effect of maximum FWI (FWI max) on the percentage of burned areas covered by other natural vegetation.

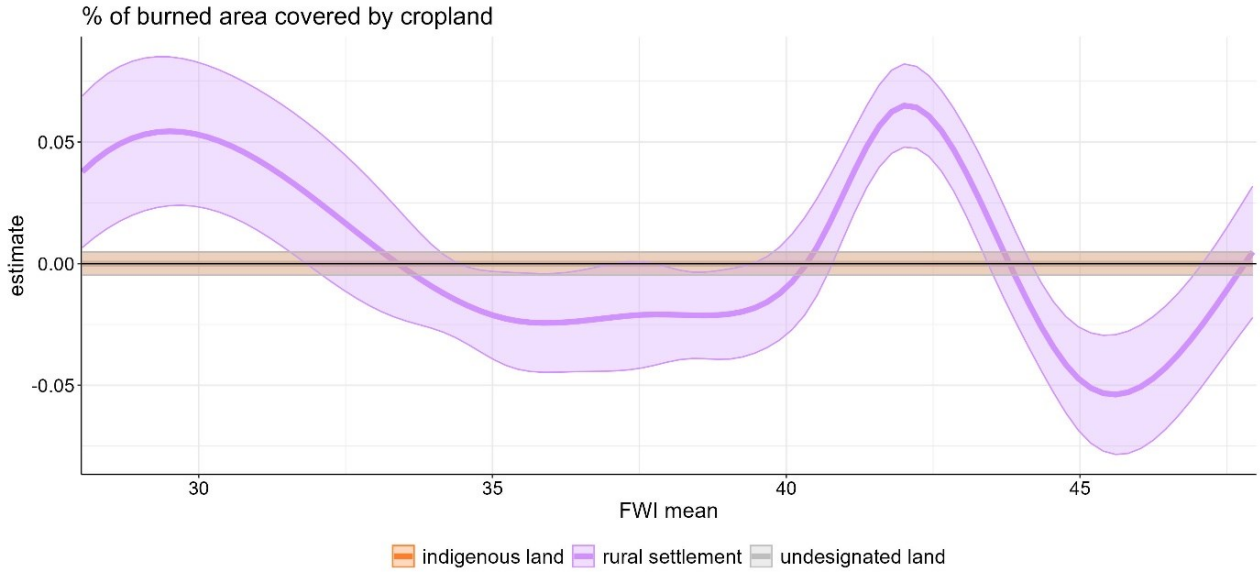


Figure 24. Smooth estimate of the effect of 90-day mean FWI (FWI mean) on the percentage of burned areas covered by cropland.

3.5 Discussion

The results of my study show that all types of landholders in North Mato Grosso have faced a similar increase in wildfire risk driven by climate change, which has been compounded by land-use changes specific to each land governance type. Fire regimes also differ across governance types; however, undesignated lands and rural settlements share many similarities. The dynamics of fire regimes differ considerably. Despite experiencing the highest increase in wildfire risks, rural settlements have seen significant reductions in overall fire extent, individual fire size, and the proportion of burned areas covered by pasture. Meanwhile, undesignated lands experienced a smaller reduction in fire extent and an increase in the size of individual fires and in the proportion of burned areas covered by forest. Indigenous lands experienced few changes over the study period. Finally, each land governance type exhibits a distinct relationship between fire regime characteristics and climate, which affects fire size in rural settlements and undesignated lands, and the proportion of forest burned in Indigenous lands. These findings highlight the diversity of wildfire risk and fire regime dynamics across land governance types in northern Mato Grosso.

The highest percentage of deforestation and pastures in the rural settlements of North Mato-Grosso (Figure 11) is consistent with results from Yanai et al. (2020). High population densities in rural settlements have driven the demand for land for pastoralism, particularly in the initial years following their establishment (Schneider and Peres 2015; Pereira et al. 2022). This land conversion process has resulted in higher fire rates within rural settlements compared to surrounding areas, especially during the first decade after their creation (Schneider and Peres 2015; dos Reis et al. 2021). The highly fragmented forests within rural settlements, combined with the predominance of pastures where fire is routinely used for land management, increase the vulnerability of remaining forests to escaped fires (Cano-Crespo et al. 2015; Silva Junior et al. 2018).

Public policies can influence deforestation extent and forest fragmentation during the establishment of rural settlements. Some categories of rural settlements, such as "environmentally differentiated" settlements that prioritise forest conservation and non-timber forest product extraction, experience lower deforestation rates(Alencar et al. 2016; Yanai et al. 2017; Pereira et al. 2022). The design of land clearing and the establishment of farming families influence the forest fragmentation metrics, with some designs minimising fragmentation for similar deforestation rates (Alves et al. 2021).

In a study of fire management in rural settlements of the neighbouring state of Pará, participants attributed most wildfires to escaped agricultural fires, often difficult to control due to high forest fragmentation, the proximity of many small parcels and overgrown vegetation (Cammelli et al. 2019). High social capital, shared fire use norms, similar agricultural production strategies and a sense of agency over local fire regimes could facilitate collective fire management efforts within rural settlements (Cammelli et al. 2019; Morello and Falcão 2020). Choice experiments have shown that small landholders, the predominant landholder type in rural settlements, are willing to bear certain costs of fire control measures when they believe these measures are likely to be effective (Cammelli and Angelsen 2019). The higher fire rates in rural settlements compared to other land governance types in my study suggest that wildfire risks in North Mato Grosso's rural settlements are largely endogenous, with the perceived decline in fire regime intensity potentially resulting from intra-settlement initiatives. Localised, community-based efforts may partly explain the observed decline in fire extent and frequency within rural settlements.

Undesignated lands were characterised by lower deforestation (Figure 11) and fragmentation rates than rural settlements (Figure 12), but greater expansion of monoculture, particularly soy. Undesignated areas of Mato-Grosso have been at the forefront of the expansion of intensive agriculture in the Brazilian Amazon (DeFries et al. 2013). The initial phase of monoculture expansion was characterised by intense deforestation accompanied by heavy fire use (Morton et al. 2008), but the reinforcement of environmental regulation quickly reduced deforestation in areas targeted for soy cultivation (Garrett et al. 2018). The use of fire on monocultures after initial land clearing is less

frequent, which may reduce overall wildfire risk (Silveira et al. 2020; Ribeiro et al. 2024). Deforestation patterns associated with large properties produce a lower degree of forest fragmentation compared to those characteristics of rural settlements (de Filho and Metzger 2006). However, the overall impact of the agricultural intensification in the undesignated lands of Mato Grosso also results in spillover of deforestation, both within Mato Grosso and in neighbouring states (Gollnow et al. 2018; Kuschnig et al. 2021).

Despite the smaller increase in wildfire risks in undesignated lands, fire regimes have experienced a more modest decrease in the total extent of burned areas (Figure 13), alongside an increase in the proportion of land burned in large fires (Figure 14) and burned areas covered by forest (Figure 17). Several characteristics of undesignated land could undermine agricultural producers' willingness to reduce fire use or invest in fire control. High land turnover, a common situation in Mato-Grosso where there is high land speculation (Reydon et al. 2020), can impede the development of collective fire management strategies (Morello and Falcão 2020). Additionally, the frequent occurrence of wildfires originating from distant places and drought conditions can discourage investment in fire control measures, as agricultural producers perceive these efforts as likely to fail, resulting in a preference for external interventions (Cammelli et al. 2019; Cammelli and Angelsen 2019). The spread of fires from distant sources may be more frequent in undesignated lands than in rural settlements due to larger fires and land parcels, and a greater extent of connected forest facilitating fire spread. However, several studies have highlighted the limited effectiveness of command-and-control instruments widely deployed across the Brazilian Amazon (Carmenta et al. 2013; Morello et al. 2017; Machado et al. 2024). These factors may help explain the divergent fire regime trends observed between rural settlements and undesignated lands.

Indigenous lands have not experienced significant land use change and are still covered by mainly unfragmented forest (Figure 12), a trend consistent with other analyses at a broader scale (Jusys 2018; Herrera et al. 2019), as well as no significant change in their fire regimes over the study period. Indigenous lands present a unique fire regime shaped by long-standing cultural practices and local ecological knowledge, which could explain their

capacity to do large burns over natural vegetation while maintaining similar land use burned over the years. In the Xingu basin and other Amazonian regions, fires have been employed both to manipulate soils, leading to the creation of nutrient-rich Amazonian dark earth, and to increase the occurrence of useful plant species (Heckenberger et al. 2003; Heckenberger et al. 2007; Maezumi et al. 2018). Fire use, along with the traditional knowledge systems underpinning it, has persisted and adapted to changes in regional climate, ecology, and social organisation (Pivello 2011). Past experiences of environmental disturbances or radical changes in environmental conditions strengthen the adaptive capacity of local ecological knowledge (Folke et al. 2009). Memories of past environmental crises help mobilise ecological and social knowledge accumulated during these episodes to cope with and adapt to the current crisis (Folke et al. 2009; Gómez-Baggethun et al. 2012).

Unlike other land governance types, more intense droughts on Indigenous lands were associated with a higher proportion of burned areas covered by forest (Figure 22). While low forest fragmentation helps preserve humid microclimates that suppress fire spread, regional climate drying may undermine this resilience and increase the proportion of forests becoming flammable and the occurrence of large wildfires (Brando et al. 2020). Compounding these environmental pressures is high demographic growth across many Indigenous lands, increasing the demand for land for swidden cultivation and associated fire use. Evidence already indicates accelerating forest loss and intensifying fire regimes in the Xingu Indigenous Park, the largest Indigenous land in the region, with flooded forests being particularly vulnerable (Silvério et al. 2022).

3.6 Conclusion

Our results reveal that while climate change has driven a general increase in wildfire risk, distinct patterns of land-use change associated with each land governance type have shaped unique wildfire risk. Fire regimes also displayed divergent trajectories. Notably, rural settlements, despite experiencing the highest increase in wildfire risk, showed the strongest reduction in fire regime intensity, potentially reflecting successful collective efforts to reduce and control fire use. Indigenous lands exhibited distinct fire regimes compared to other land governance types, primarily facing challenges during years with severe fire-prone weather, when larger and more intense forest fires can occur. These results highlight the importance of land governance types and subsequent effects on local fire management to modulate the impact of changing wildfire risks on local fire regimes.

It also highlights the need for governance-specific strategies to support the implementation of the Integrated Fire Policy. In rural settlements, public policies could limit deforestation and forest fragmentation and support existing collective fire management efforts. In contrast, undesignated lands feature less fire-prone landscapes but higher opportunity costs for changing land-use practices due to the extent of soy cultivation. Given the lower likelihood of collective fire management, alternative fire management initiatives, such as command-and-control measures or incentive-based schemes, could prove more effective. Indigenous lands may struggle with increasing forest flammability during drought events and require periodic support. Together, these findings underscore the need for diversified fire management strategies in the Brazilian Amazon that account for the socio-ecological diversity and governance structures shaping wildfire risks.

Chapter 4: Understanding fire use and management in an Indigenous land: a case study from the Capoto/Jarina

4.1. Main contributions

- Participants use fires for a diversity of objectives related to their livelihoods and cultural needs while mobilising extensive local fire knowledge to prevent wildfires.

- Wildfire risk management strategies are embedded in a detailed understanding of their ecosystems based on frequent observations of weather, vegetation, animals and the impact of past wildfires.

- Local fire use and associated knowledge differ considerably across villages depending on the objectives of fire use and local ecosystems.

- The diversity of fire use objectives and wildfire management strategies only partly align with broader Integrated Fire Management objectives and approaches in the biome.

4.2. Introduction

Indigenous lands cover 24% of the Brazilian Amazon, providing essential ecosystem services across South America and constituting territories vital for the survival and cultural preservation of 305 ethnic groups (Nelson and Chomitz 2011; Nogueira et al. 2018; Walker et al. 2020). This cultural diversity, along with associated land management practices such as tree planting, soil management and fire use, has shaped the biodiversity of the region through a process of landscape domestication (Heckenberger et al. 2007). Since pre-Columbian times, the use of fires has been an integral part of daily life for Amazonian Indigenous communities, both for swidden agriculture and other livelihoods and cultural activities, such as hunting or cooking (Pivello 2011; Schwartzman et al. 2013; Rodrigues 2022). Extensive fire use has influenced soil fertility, supported the abundance of fire-adapted species and contributed to the maintenance of vegetation types providing important resources for Indigenous livelihoods (Levis et al. 2018).

Indigenous Fire management differs markedly from Western fire management. It is based on distinct ontologies, embedded within multi-scalar and complex governance environments, supported by local ecological knowledge developed for specific landscapes and aims to simultaneously achieve ecological, social and cultural benefits (Nikolakis and Roberts 2020). Central to Indigenous fire management is local fire knowledge, a subset of local ecological knowledge that focuses on the use and management of fires and includes elements such as the influence of soil, vegetation and weather on fire (Huffman 2013).

Drawing upon this local knowledge is essential for ensuring that fire management strategies are contextually appropriate. Over-reliance on ecological knowledge developed in different environmental settings risks creating mismatches with local fire ecology and fire management (Ray et al. 2012). This issue is particularly relevant for fire management in complex and dynamic socio-ecological systems, given the wide diversity of adaptation traits of plant communities to fires and their varied responses to environmental and climatic stressors (Archibald et al. 2018). Moreover, local ecological knowledge plays a crucial role in interpreting environmental changes and informing adaptive decision-making processes

that contribute to the resilience of these socio-ecological systems (Folke et al. 2009; Evans et al. 2011; Ruiz-Mallén and Corbera 2013).

In response to recurrent wildfire crises, their environmental and social consequences, and the failure of previous policies focusing solely on fire criminalisation and firefighting efforts, the Brazilian government has adopted an Integrated Fire Management approach (Schmidt et al. 2018). This new management approach, formally recognised by a national law (law 14.944 of 31st of July 2024) in 2024, acknowledges the positive role of "traditional and adaptive use of fire" in Indigenous and traditional communities and the unique ecological knowledge associated with it. This evolution of the Brazilian legislation is part of a wider global change of paradigm, embracing the ecological benefits and adaptive capacities of fire use by Indigenous and local communities (Russell-Smith et al. 2013; Maclean et al. 2023; Oliveras Menor et al. 2025).

Nevertheless, the integration of Indigenous fire management and local ecological knowledge in fire management policy remains a recent development in Brazil (Eloy et al. 2019; Mistry et al. 2019). Policy interventions aimed at wildfire reduction have historically overlooked the preferences and practices of local communities(Smith et al. 2024). Embedding local knowledge into fire management policies and practices requires addressing potential disconnects between Indigenous fire use preferences and state-led fire management frameworks (Eriksen and Hankins 2014; Petty et al. 2015). Failure to do so risks fostering tensions within communities, limiting the intergenerational transmission of local ecological knowledge, and weakening existing governance structures (Sletto and Rodriguez 2013; Eriksen and Hankins 2014; Smith et al. 2024).

Despite growing recognition of the importance of Indigenous fire practices, there remains limited empirical research on traditional fire use and associated knowledge in the Indigenous lands of the Brazilian Amazon. Addressing this knowledge gap is crucial for supporting the development of fire management strategies that are ecologically effective, socially acceptable and culturally appropriate. Integrating social data through local case studies can improve understanding of why and how people use fire, helping to align national

policy objectives and local realities (Mistry and Bizerril 2011). This is particularly relevant regarding the low intensity of fire regimes despite environmental changes identified in the previous two chapters.

In this chapter, I aim to examine the specificities of fire use practices and associated knowledge systems within Indigenous lands, and to explore their implications for the implementation of Integrated Fire Management in Brazil. To this end, I present a case study from the Capoto/Jarina Indigenous Land in northern Mato Grosso, investigating the fire use and wildfire risk management strategies across three villages.

4.3. Methodology

4.3.1 Sites of study

Capoto/Jarina Indigenous Territory is a 635,000 hectares Indigenous land in northern Mato Grosso, in the Cerrado-Amazon Transition zones characterized by a complex mosaic of dense forest, ecotonal forest and savanna (Marques et al. 2020). Capoto/Jarina is home to around 1,600 inhabitants, predominantly from the Metyktire people living along with smaller groups of Juruna, Tapayuna, and Trumai peoples. The territory is characterized by diverse vegetation: lowlands are primarily covered by rainforests, including significant areas of seasonally flooded forests along major rivers, and a plateau in the northwest is dominated by savanna.

The history of Capoto/Jarina is marked by persistent conflicts with invading farmers, prompting collective action and strategic alliances to defend the territory and the high political power of the local indigenous leaders[4] (Oliveira 2017). The southern and northern borders of the Capoto/Jarina Indigenous land are adjacent to two larger Indigenous lands,

[4] Since the episode of the "ferry war" during which inhabitants of Piaraçu sized control of the ferry crossing the Xingu and forced to the demission of the director of the FUNAI in 1984, chief Raoni and leaders from the Capoto/Jarina have held a central place in the Indigenous movement of Brazil and succeeded to deter invasions by neighbouring farmers (see Oliveira et al. 2017 for more details on this episode). An annual expedition is still organised to inspect the border of the indigenous land and deter land invasions.

while the eastern and western border is adjacent to privately owned farmlands covered by pastures and cereals monocultures with only a few remnants of rainforest, as well as an Indigenous land in process of demarcation. The Raoni Institute, an Indigenous association created in 2001 to defend the interests of people living in the Capoto/Jarina, Menkragnoti and Parana Indigenous lands, is coordinating diverse projects in the region and managing relationships with external stakeholders. Since 2004, a voluntary fire brigade has operated in Capoto/Jarina, carrying out diverse activities to reduce wildfire risks throughout the dry season, and responding to wildfire events when possible. The fire brigade was professionalised in 2019 and was integrated into the national Prevfogo fire brigade programme in 2024 [5].

Capota/Jarina currently has ten villages along the main rivers, two villages along the Western border and one village on a savanna plateau on the north of the Indigenous land. I visited three villages for this study (Figure 25):

- **Piaraçu:** a large, long-established village located near a ferry that controls regional transport and a road marking the boundary between Capoto/Jarina and Parque do Xingu, the neighbouring Indigenous land. The surrounding vegetation includes degraded forests and grasslands covered by *Brachiaria spp.*, an invasive species commonly used by ranchers in planted pastures.
- **Mupa:** a small and recent village established in 2020, next to the Xingu River and 20 km from the road. The village is close to well-preserved forest, with patches of grassland and savanna on the shore of the Xingu.

[5] The Prevfogo fire brigade programme seasonally hires, equips and trains fire practitioners in Indigenous lands and protected areas of Brazil to conduct wildfire risk prevention activities and fight wildfires. In Indigenous land, this program is co-managed by two governmental institutions, the FUNAI (National Indigenous People Foundation) and IBAMA (Brazilian Institute of the Environment and Renewable Natural Resources)

- **Capoto:** a large, long-established village near a plateau covered with fire-dependent vegetation, including a savanna-grassland mosaic. It is more than 90 km from the main road but experiences deforestation pressure in areas near the territory borders.

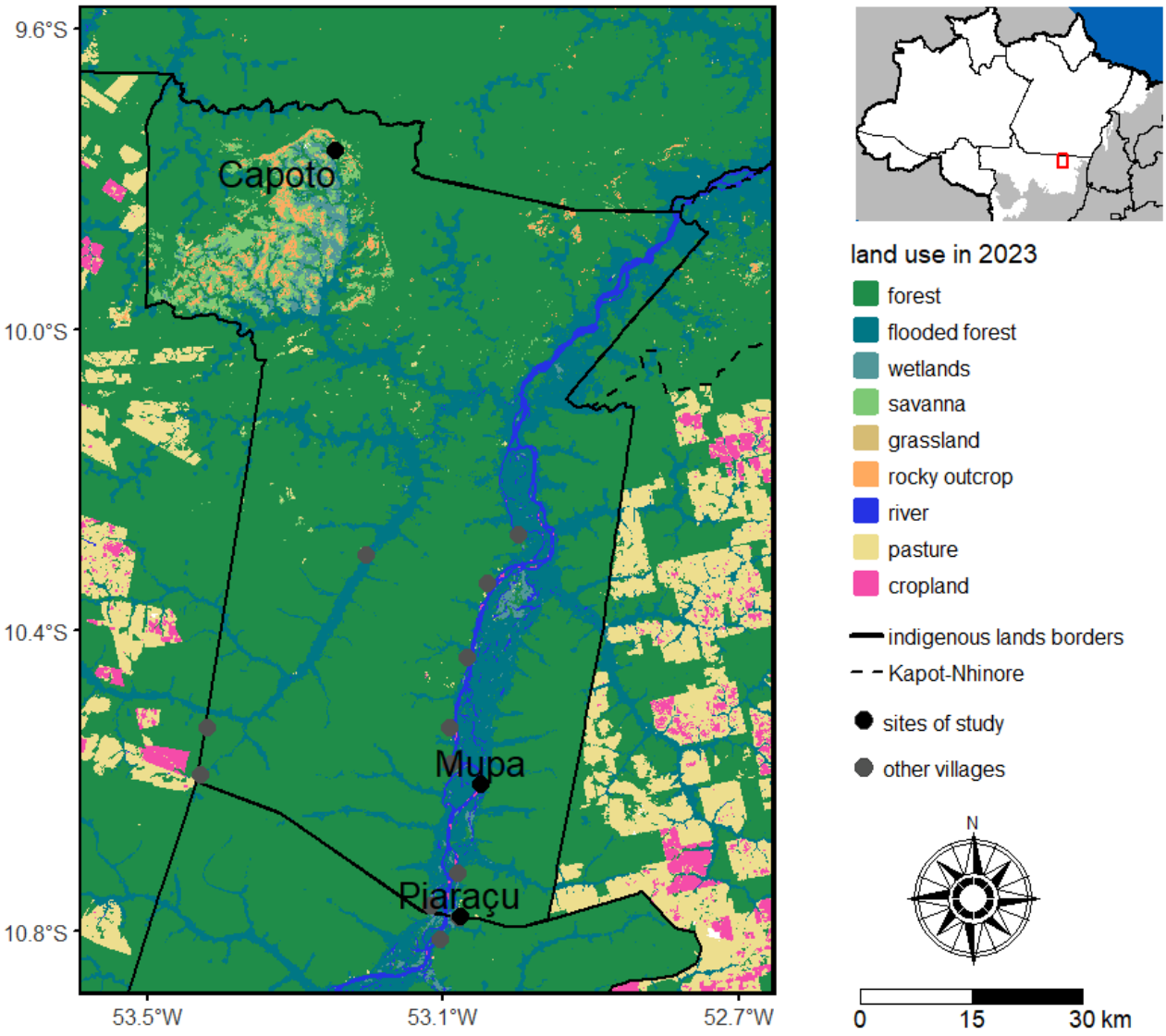


Figure 25. Map of the Capoto/Jarina Indigenous land, with the three villages visited.

I selected these three villages following discussions with the Raoni Institute because they have different types of landscapes and associated risks of wildfires representative of the diversity of challenges encountered in Capoto/Jarina. Piaraçu has recently experienced a significant increase in wildfire risk and is surrounded by degraded fire-sensitive vegetation,

Mupa is a recently opened village and is surrounded by well-preserved fire-sensitive vegetation, whereas Capoto has historically faced consistently high wildfire risk and is surrounded by fire-dependent vegetation.

4.3.2 Ethical considerations

Indigenous epistemologies have unique characteristics, such as their grounding in a holistic understanding of local socio-ecological systems and their high dynamism, based on frequent observations, experimentation and collective interpretation (Turner and Berkes 2006; Nikolakis and Roberts 2020). Following several major wildfire crises resulting from the failure of the fire suppression paradigm, rooted in colonial and post-colonial views of landscape management, there has been a growing recognition of the relevance of Indigenous fire knowledge for coexisting with fire and adapting to changing wildfire risk conditions (Trauernicht et al. 2015; Bowman and Sharples 2023). However, the integration of this knowledge into fire management by Western-led institutions raises important questions about their potential alienation. The isolation of Indigenous fire knowledge from the knowledge systems in which it is produced and adapted, the oversimplification of complex fire management and the increased monetarisation of fire management actions have created tensions and disagreements within communities (Eriksen and Hankins 2014; Petty et al. 2015). Thus, it is important to situate Indigenous fire knowledge within its socio-environmental context, including the dynamics of power across stakeholders in fire management, the hierarchisation of different knowledge systems supporting decision making, and the agency of local communities over their socio-ecological systems (Copes-Gerbitz et al. 2021).

This study was co-developed with the Raoni Institute and academics from UK-based institutions to understand recent changes in fire regimes and potential wildfire risk adaptation strategies in the Capoto/Jarina indigenous land. While the role of Indigenous and traditional communities in ecosystem conservation is increasingly recognised and valued, severe procedural and ethical challenges persist in conducting cross-cultural research

projects (McKemey et al. 2022; Newing et al. 2024). These challenges stem partly from a history of extractive research practices in conservation science, hierarchization of knowledge systems with dismissal or disrespectful validation of local ecological knowledge and power imbalances between researchers and participants (Roue and Nakashima 2018; McKemey et al. 2022; Newing et al. 2024). To address ethical concerns during the planning and implementation of this research, the research team followed the International Society of Ethnobiology's Code of Ethics, which outlines 16 principles and practical guidelines for researchers (International Society of Ethnobiology 2006). Working with local facilitators, can also raise ethical dilemmas due to their insider positions and the differing expectations of the research team and community members (Mistry et al. 2015). I sought to alleviate these challenges by adhering to the ethical principles proposed by Mistry et al. (2015).

In agreement with Brazilian legislation, the research team obtained project approval from the National Research Council (process n° 01300.012906/2023-50), ethical approval from local and national ethical committees in Brazil (process n° 77514423.0.0000.5165) and from the FUNAI (process nº 08620.012356/2023-03). The research team also obtained ethical approval from the Imperial College of London ethical committee (n° 6632578).

The research team, composed of both Indigenous researchers and academics from UK-based institutions, developed a Free and Prior Informed Consent Protocol with the Raoni Institute, the institution representative of communities of the Capoto/Jarina, to obtain community and individual-level approval for participation in the research project. An oral presentation of the research project, expected outcomes, participants' rights and potential harms and benefits was given in Portuguese translated into local languages (Mẽbêngôkre or Yudjá) as needed, based on participants' preferences. Free and Informed Prior Consent was systematically obtained, with at least one local community leader signing a document in Portuguese and an oral recording made in the native language of all participants. At least one copy of a document detailing all participants' rights was left in each community for consultation, including information on how to contact the research team and withdraw from the study.

Discussions have been engaged with the local members of the research team from the initial steps of the project to ensure both the co-construction of the project around issues relevant to local stakeholders and the respectful integration of insights from different knowledge systems (see section 5.3.1 for more details on this issue). Efforts were made to include diverse stakeholders affected by fire use and wildfire risk management. Thus, the initial sampling design included some villages that are more remote and are less frequently involved in research efforts, as well as a minority ethnic group in the Capoto/Jarina. There was also an effort to include women's groups in the research, adapting the methods to enable their participation and organising separate Focus Group Discussions for women and men whenever possible.

For the dissemination of the research results, the research team organized a second visit and presented the main preliminary research results using a poster (Appendix 4.1). This allowed participants, including men and women from different villages of the Capoto/Jarina as well as the local fire brigade, to provide feedback and contribute to the cross-validation of insights from different knowledge systems. A report on wildfire risks was also compiled for the local leaders and the fire brigade (Appendix 4.2) and a research summary was compiled to be included in the Territorial and Environmental Management Plan of Capoto/Jarina (Appendix 4.3). The publication of any material related to the research activity is subject to the approval of the Raoni Institute, which has the authority to restrict information deemed sensitive or valuable.

4.3.3 Data collection

For the collection of qualitative data, I adopted a flexible approach based on semi-structured interviews with key informants, focus group discussions organised around the creation of a fire typology, a seasonal burning calendar, and a participatory map of fire use and wildfire risks, as well as field walks (Newing et al. 2011). These methods were deemed

appropriate for understanding the diversity of fire use and associated local fire knowledge, along with the spatial and temporal distribution of fire use and wildfire risk. In each of the villages, the research team were introduced by a local facilitator who introduced the research team and project on fire management to inhabitants of the village and organised meetings for interviews and focus group discussions. Depending on their availability, they would sometimes assist the research team during the research activity and help with translation between Mẽbêngôkre or Yudjá and Portuguese or identify someone who could help with translation.

Semi-structured interviews were conducted in each village with key informants to collect baseline information about fire use and wildfire risks prior to organising three focus group discussions (see Appendix 4.4 for the guide of the semi-structured interviews). To select the key informants for semi-structured interviews, I used purposive sampling (Kelly 2010), asking the local facilitator for potential participants with extensive experience in fire use and wildfire risk management.

Then, three Focus Group Discussions were organised with each participant group. While participants could decide not to participate in the second or third Focus Group Discussion, I didn't allow participants that haven't attended the first Focus Group Discussion to join the participant groups afterwards. This aims to ensure there is a common understanding among the participants about the main use of fires and landscape spatiality.

In the first focus group discussion, participants were asked about the different fires they used, before drawing an initial version of the participatory map, including their villages and different characteristic features of the landscape such as roads, rivers and hills. Finally, they indicated the different places where they used each type of fire on the map.

In the second focus group discussion, participants were asked if they wanted to bring any corrections to the map created during the first session and if they remembered more uses of fires. Then, they were asked a series of questions for each type of fire, to know who lit them, how and when sites are prepared, during which months they are usually burned, if participants relied on other seasonal indicators, how participants decide which day to burn,

the time of the day when fires are lit, fire control measures deployed, the likelihood and potential damages of escaping fires. The participants were then asked if the way some of these fires are used changed over time. To conclude the second focus group discussion, facilitators used the base map to review different locations where fires are used over a season.

In the third focus group discussion, participants were initially asked if they wanted to correct anything on the map and the fire typology created during the first two sessions. Then participants were asked to discuss previous wildfire events they had witnessed, and specify when they happened, the origin of the fires, associated damages and efforts to control wildfires. Participants were then asked to indicate different locations with significant wildfire risks and/or the presence of resources vulnerable to wildfires, rank them from having a low to high wildfire risk, before discussing specific wildfire risks and their change over time for each location.

After the final focus group discussion, researchers ask if participants are willing to bring the research team to some parts of the landscapes that have been identified during the group discussion. These field walks could be in parts of the landscapes that are managed by fires, have experienced past fires or have been identified as having a high wildfire risk. During these field walks, facilitators asked follow-up questions on different points evoked during the focus group discussions, to triangulate and enrich results from the semi-structured interview and focus group discussion (see Appendix 4.5 for detailed guides for the Focus Group Discussion and field walks).

The sequence of research activities was adapted to opportunities and constraints inherent to fieldwork. While Focus Group Discussions were considered appropriate due to the collective nature of information-sharing and decision-making in Mẽbêngôkre culture, they require a significant time commitment for participants. To avoid research fatigue, and depending on participants' availability and group size, the three focus group discussions

were sometimes combined into two sessions: one focusing on the creation of the fire typology and seasonal burning calendar and the other on the participatory map.

A pilot phase was organized in Piaraçu with 10 women and 13 men to test the research methods. The main change adopted after the pilot phase concerned the participatory mapping exercise: during the pilot phase, participants experimented with freehand drawings and making maps based on printed satellite images. The research team noticed that some participants, especially women, experienced difficulties in using the satellite images. Thus, the research team adopted freehand mapping, as it provided more flexibility for creating narratives based on individual experiences of fire use and wildfire risks, even for participants less familiar with satellite imagery. Regarding the absence of major change in the research methods used, results from the pilot study were used in the data analysis.

The research team organised the following activities in each village:

- Piaraçu: one semi-structured interview, two field walks, and seven Focus Group Discussion including three with a group of five women, three with a group of 13 men (including nine that have been involved in the fire brigade initiative) and one with a group of four elderly women and three elderly men (using the guide from semi-structured interview guide as it was possible to organise only one session).
- Mupa: two semi-structured interviews, one field walk, and two Focus Group Discussions with a group of three women and three men
- Capoto: one semi-structured interview, two field walks, and five Focus Group Discussions, including two with a group of ten women and three with a group of ten men

Additionally, I organised four feedback sessions in Piaraçu at the beginning of my second visit to present the early results of the research project. During the sessions, I asked follow-up questions and included notes from the participation of five new participants from Piaraçu (including three women) and one new participant from Capoto. I also conducted semi-structured interviews with three participants in Tothnhore (including one woman) and one

participant in Metyktire, two villages located around the Xingu. In Piaraçu, I also interviewed four participants visiting from the neighbouring Menkragnoti indigenous lands. The contributions from outside of Piaraçu, Mupa and Capoto were used for the analysis of strategies to adapt to increasing wildfire risks and their limitations and provide contextual data. In total, I used data coming from 65 participants, including 26 women.

In Piaraçu and Capoto, participants were divided into men's and women's groups, whereas this was not the case for Mupa due to the small size of the village and the limited number of participants available. In Piaraçu, a supplementary group of elders were interviewed, using a modified focus group discussion guide focusing on change in fire use and wildfire risk. In total, participants were divided into three groups in Piaraçu, one group in Mupa and two groups in Capoto. All activities were carried out with at least one local facilitator and conducted either in Portuguese, Mẽbêngôkre or Yudjá depending on the participant's preferences.

4.3.4 Data analysis

After the research activities, all semi-structured interviews and focus group discussions were integrally transcribed and translated into English, while notes were compiled for the feedback sessions (which were not audio-recorded to encourage participation and feedback from participants) and for the field walks. Photographs were taken of any visual outputs that were produced, and field notes were added to the corpus of text. Integral transcription and field notes from the KremKrem research project, a previous research project focusing on the creation of a participatory video on fire management in Capoto/Jarina between 2015 and 2017, as well as the 2024 Activity Report from Prevfogo fire brigade were used as contextual data.

The research teams used a grounded and inductive approach to coding the qualitative data corpus (Glaser and Strauss 1967). After the first visit to Capoto/Jarina, I read the corpus of data obtained, identifying themes aligned with the research question. I then developed first-level codes, systematically applying them to the dataset before refining a second-level coding scheme. This preliminary coding structure and initial findings were reviewed by three

other research team members, leading to minor refinement. The final version of the coding framework included five main categories: local change in climate, environmental conditions, fire use, Wildfire risks and the role of the fire brigade. Each first-level code was further divided into four to height second-level codes (Table 3, see Appendix 4.6 for definitions for each code). Once the final coding framework was agreed upon, I developed the theoretical framework outlined in Chapter 1.

1st level of code	**2nd level of code**
Local change in climate	Rainfall / temperature / wind / impact of climate change / relevance of local ecological knowledge / uncertainty about the future
Environmental conditions	Local ecological knowledge / importance of ecosystems to support livelihood / abundance and scarcity of natural resources / external environmental degradation
Fire use	Objectives of different fires / seasonality / time of the day / decision to use fire or not / preparation of the sites / fire control measures / gender / specificity village
Wildfires risks	Good and bad fires / meteorological conditions / combustible / ignitions sources / neighbouring villages / damages / past wildfires / change in fire use and management
Emergence of fire brigades	Response to growing wildfire risks / diffusion of the fire brigade initiative / introduction of firebreaks / coordination of the fire brigade and the rest of the village / delegation of wildfire risk management / limits of firefighting / need for the firefighting brigade

Table 3. Table resuming the first and second levels of codes used for the analysis of the corpus of text compiled.

4.4 Results

4.4.1. Fire use and associated local fire knowledge

In the following sections, I will describe the objectives of different uses of fires across the three villages, associated fire control measures and decisions about when to burn these fires to prevent wildfires.

Fires uses

I reported all the types of fires that are used outside of the villages and that participants used on their own at least once (see Table 4). While several other uses of fires were discussed during the research activities, such as fires for roasting cassava flour or for crafts, these are small fires lit next to habitations and under constant monitoring, thus having a low risk of turning into wildfires.

In total, I identified 9 main types of fires frequently used in the Capoto/Jarina indigenous land (see Table 4). In Piaraçu and Mupa, each fire use has a well-defined objective. Meanwhile, in Capoto some fire uses have well-defined objectives, but the most extensively used fire, savanna burning, serves multiple objectives. Savanna fires were described as an important driver for the regeneration of the savanna, essential for reducing wildfire risk by controlling fuel load. They were also important for resource collection, as they reduce the risks of injury and encounters with venomous animals while revealing the location of important resources, such as the seeds of *Attalea speciosa* used to make cosmetic oil, beehives or animal hides. Additionally, several participants groups from Piaraçu and Mupa reported the use of fires similar to the savanna fires in the region of Piaraçu for hunting parrot chicks, but I didn't include it in the typology as none of the participants in the research activities have directly done it in Piaraçu.

Fire type and objectives	who	seasonality	preparation sites	fires control measures	Time of the day
Swiddens fire burning secondary vegetation to open news swiddens	**Piaraçu:** men (women help to choose the location, preparation and second fires)	Preparation: May to July Burning: late August to late September, just before/after the first rainfall	cut down vegetation, gather it close to the centre (sometimes in piles) and let everything dry	Fire-breaks, assessing weather conditions, monitoring fires, burning in teams, burning by the fire brigades	noon or afternoon
	Mupa: men and women	Preparation: June to July Burning: September, after the first rain		Fire-breaks, assessing weather conditions, monitoring fires, burning in teams, burning with the fire brigades for challenging fires	before: around 11a.m. now: after 3 p.m.
	Capoto: men (women help choosing location, preparation and second fires)	Preparation: June to August. Some wait for the flowering of yellow ipé Burning: August in the past, now September. Wait for the song of cicadas, the fall of yellow ipé flowers and the first thunder to burn just before rainfall		Fire breaks, assessing weather conditions, monitoring fires, burning in teams	early afternoon (from 1 p.m. onwards)
Waste fire burning all waste products	**Piaraçu:** women	none	clean the place before	monitoring fires	afternoon/night
	Mupa: women and men		make a hole to burn the rubbish inside		afternoon/night
	Capoto: women and men		clean the place before		anytime
Village opening fires opening an area for building a new village	**Mupa:** women and men	Preparation: June Burning: September	cut down vegetation, gather it close to the centre (sometimes in piles) and let everything dry	Fire-breaks, monitoring fires, burning in teams	after 3 p.m.
Bonfires for fish and game Cook fish and game caught during expeditions outside of the villages	**Piaraçu:** men (outside of the village)	none	clean the place before, especially close to grasses	have water to extinguish fires	any time
	Mupa: men (outside of the village)			have water to extinguish fires, posting signs for fishermen from other villages	
	Capoto: men (outside of the village)				

Fire type and objectives	who	seasonality	preparation sites	fires control measures	Time of the day
Collecting honey produces smoke to calm bees and collect honey	**Piaraçu:** men	June to August	clean the place before, especially close to grasses	have water to extinguish fires, spread the burned fuel, using only herbs as combustible	any time
Cleaning paths opening and keeping paths clear of vegetation	**Piaraçu:** women	March to September	clean the place before, especially vines	have water to extinguish fires, particularly on burning tree roots	any time
Savanna fires open landscape for collections of diverse products (seeds used for craft, seeds of *Attalea speciosa* used for cosmetics, honey, parrot chicks, games, medicinal plants, poisonous vines used for fishing), reduce risks of encountering venomous animals, reduce fuel load and risks of wildfires	**Capoto:** men and women	From March to September. Partly dictated by resource requirements of different cultural festivals	none	Strategic use of fires with the most dangerous places burned at the beginning of the season. Gradual mosaic burning patches. Avoid strong wind. Sometimes monitoring fires	As early as possible during the day. Often burn and come back after a few days/weeks to collect resources.
Clearing fires breaks burn fuel to quickly clean firebreaks	Members of the fire brigade	April to September	none	Work in teams, one participant using a drip-torch and five other fire-beaters	any time
Back-fires burn fuel before a wildfire arrive to control intensity	Members of the fire brigade	July to September (during wildfires events)	none		any time

Table 4. Typology of the different use of fires developed based on interviews from Piaraçu, Mupa and Piaraçu

For clearing the surroundings of habitations or new paths around the villages, participants interchangeably use fire or alternative techniques, such as manual clearing and using vegetation leftovers for mulching. Preferences regarding fire use or alternative techniques varied across villages and participant groups. In Piaraçu, men usually avoid using fire except for opening and clearing paths wider than 2 meters, while women prefer to use fire as it produces cleaner paths. In Mupa, participants prefer manual cutting and avoid using fire for path clearing. In Capoto, path creation is one of the objectives of frequently used savanna fires by both women and men, as these fires improve visibility and reduce the risks of encounters with venomous animals.

Fire control measures

Across the three villages, participants deploy a wide array of strategies to control the fires they set and prevent them from escaping into nearby vegetation.

When preparing a site for burning, either for swiddens or opening paths, participants from Piaraçu and Mupa highlighted the importance of clearing all the vegetation, especially vines that can transfer heat from one area to another and being cautious about buried roots that could smoulder for a long time. In the case of swiddens, the vegetation is felled towards the centre of the plots to reduce fire intensity near the edges of the burn sites. Participants in all villages are also now using firebreaks to prevent escaped fires (this will be discussed further in Chapter 5). For other types of fires, such as roasting fish or collecting honey, participants also clear vegetation around the fire site, especially in areas with abundant grass that could easily ignite.

In most cases, participants monitor fires until they are extinguished to ensure they do not escape. While not being used by everyone, some participants in Piaraçu and Mupa suggested always bringing a water container to extinguish fires. In Capoto, participants do not always monitor savanna fires until extinguishment, especially at the beginning of the dry season when wildfire risk is considered low. Their burning practices help to prevent intense

fires by controlling fuel load, and fires usually cease burning near forest edges, where the humidity increases.

Seasonality

The seasonality of fire use is strategically planned to help achieve different livelihoods and cultural objectives while minimizing wildfire risk.

Participants from different villages reported distinct seasonality for the preparation of swiddens, beginning in May or June depending on the village, but they all burn their swiddens in September, and in the case of Piaraçu and Mupa after the first rainfall. This timing was deemed an ideal balance between the need to burn when the fuel is dry enough to ensure complete combustion, and the likelihood of rainfall in the days following the burn to extinguish any fire that might escape into the forest, favour the penetration of ashes into the ground and germination of planted seeds. In Capoto, they also rely on the song of a species of cicada and the fall of yellow ipé (*Handroanthus spp.*) flowers to indicate the right time to burn.

In Capoto, savanna fires were used during the whole dry season, but their locations depended on the perceived wildfire risks and resources that needed protection. Fires were lit in the savanna close to the village and cassava plantations during the early dry season, when participants declared the moisture content of the fuel was higher and fire easier to control, to reduce fuel load and potential damage caused by wildfires. After these early-season fires, participants chose different locations to burn based on resource requirements for various festivities. Participants said they were more careful when burning savanna fires during drought episodes and at the end of the dry season when they would light smaller fires and monitor more closely the fires lit.

Other fire seasonality was also driven by resource availability and the need for livelihood objectives. For example, fires for honey collection in Piaraçu were lit from June to August during the time of the year when bees produce the most honey, and fires for roasting fish and game or burning waste throughout the year.

The use of fire or alternative techniques for clearing paths and the surroundings of habitations is influenced by seasonality and the associated risk of wildfires. While some participants from Piaraçu perceived fire use for clearing paths and the surroundings of habitations as safe during the early dry season, they felt it became increasingly dangerous from July onwards and therefore opted for manual clearing.

Time of the day

Participants strategically chose the time of the day when they lit fires to avoid risky conditions and reduce wildfire risks.

Weather conditions are important for determining when fires are used. Most participants reported avoiding burning during windy or hot weather, as these conditions increase wildfire risk. They prefer to wait until later in the day, when wind and temperature decrease, or postpone burning to another day. However, this attitude was not unanimously adopted, and two wildfire incidents caused by fires lit during windy conditions were reported by participants from Piaraçu.

Preferences regarding the time of day for using fires varied across study sites. In Mupa, participants burn their swiddens in the afternoon, after 3:00 p.m., as wind conditions are more stable, and this leaves a few hours before dusk when wind and temperature drop while relative humidity increases, helping to control fires. In Piaraçu, all participants agreed that burning in the early morning or late afternoon was preferable for similar reasons. However, it was reported that some people, especially among the older generation, preferred to burn around noon for a “cleaner” burn. This was perceived as both dangerous and ineffective by some participants. In Capoto, there was a preference for burning swiddens in the early afternoon. For savanna fires, participants preferred to burn as early as possible to have enough time to gather resources from nearby patches of savanna and carry out other activities planned for the day.

4.4.2 Wildfires risk and associated local fire knowledge

In the following section, I will discuss wildfire risks identified related to recent changes in vegetation and interactions between traditional fire use and vegetation identified during participatory mapping, as well as potential damages from wildfires. I will then discuss the wildfire risk management strategy discussed by participants.

Changes in vegetation

Several recent changes in vegetation surrounding the village were identified as emerging drivers of wildfire risks, based on local observations by participants. During the session in Piaraçu, participants described the recent invasions by *Brachiaria spp.*, an invasive grass species planted for pastures outside Indigenous lands as the most important driver of wildfire risks (Figure 26, locations 1 to 8). *Bracharia spp*. were accredited for the rapid increase in wildfire risks due to their extreme flammability. Efforts to reduce *Brachiaria spp.* biomass were hampered by the grass' rapid regrowth and the significant workload required for mechanical cutting, leading to its expansion across the landscape. Similarly, participants mentioned the spread of another grass species, *Paspalum virgatum*, in one of the freshwater lagoons that had dried up over time due to the declining water levels of the Xingu River (Figure 26, location 9). They also highlighted the long-term impact of wildfires, such as tree mortality, increased flammability in previously burned forests and further grass invasion (Figure 26, location 10). During a field walk, one participant described how some of the remaining trees in the *Brachiaria spp*. grasslands attract lightning during the early rainy season, when rainfall is still irregular, and grassland can catch on fire.

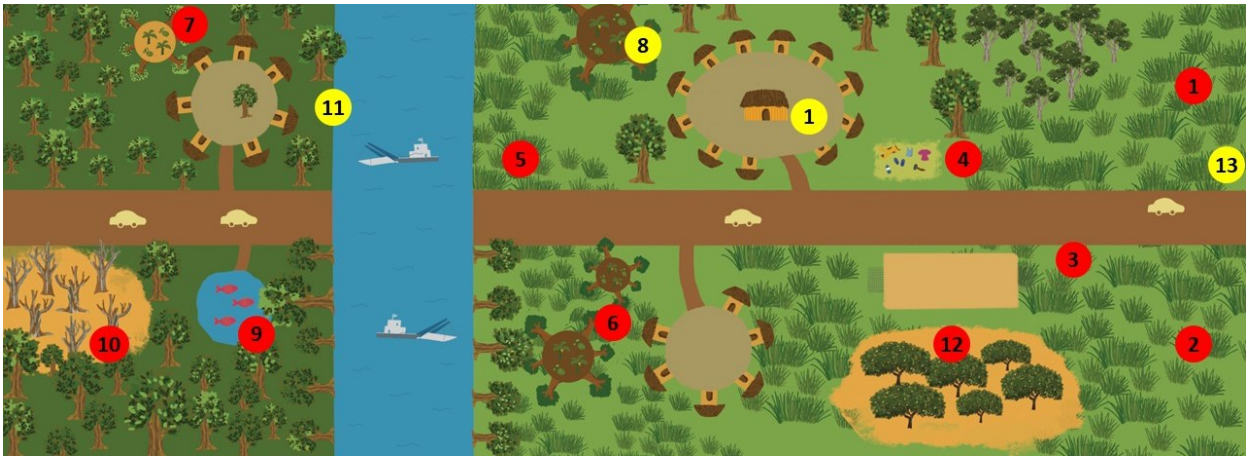


Figure 26. Illustration derived from participatory mapping exercise in the village of Piaraçu including all the sites discussed in the results. Number in yellow indicate low wildfire risk and in red high wildfire risk. This illustration groups the insight from the participatory map of both women and men (credit: Manini Bansal).

In Mupa, one old swidden was identified as having a high wildfire risk (Figure 27, location 1) partly due to the legacy impact of past fires and the presence of dead trees increasing fuel load. In Capoto, participants talked about the increased wildfire risks around the village, as the fruit trees planted during the establishment of the village grew (Figure 28, location 1).

Interactions between the use of fire and vegetation

Participatory mapping provides insights into the interactions between vegetation and traditional use of fires, which are often difficult to detect accurately using satellite imagery due to the small size and short duration of many fires. In Piaraçu, participants are using small fires for various livelihoods and cultural activities, such as the collection of honey (Figure 26, location 9), hunting or roasting fish (Figure 26, locations 9 and 11), or burning trash (Figure 26, location 4), especially in resource-rich locations. These frequent small fires were described as increasingly dangerous when burning close to *Brachiaria spp.,* as embers escaping from these fires can be sufficient to ignite them and start a wildfire. Participants also identified high wildfire risk in areas where new swiddens were being opened, attributing this risk to both the presence of degraded vegetation nearby, invasion by *Brachiaria spp.* and

the high intensity of fires used by local communities due to the biomass accumulated within forests (Figure 26, location 6). Participants also discussed the wildfire risks posed by the road and potential ignitions in areas with *Brachiaria spp*. (Figure 26, location 3) and fires originating from farmlands outside of the Indigenous land (Figure 26, location 13).

In Mupa, participants classified an old swidden (Figure 27, location 1) and several grassland patches along the Xingu River (Figure 27, location 2) as having a medium wildfire risk, due to the frequent use of fires in these areas for hunting and roasting fish by members of neighbouring villages. The increasingly dry soils of seasonally flooded forests, associated with frequent wildfires originating from villages south of Mupa, and the capacity of wildfires to travel several kilometres over weeks, were described as a major risk of wildfires. In Capoto, participants also discussed the risks of fires originating from farmlands outside of the Indigenous land as the vegetation close to the border is usually not burned (Figure 28, location 7).

Wildfire damages

Across all participant groups and the three villages, wildfires were described as problematic and jeopardising Capoto/Jarina's ability to provide the resources required to sustain the livelihood and cultural practices of the community in the long term.

Wildfire directly threatens participants' health and safety, due to smoke exposure and damage to their habitations. This risk is particularly critical in Capoto where houses are thatched with banana leaves, a material more flammable than the palm fronds used in Piaraçu and Mupa.

Wildfires can also damage swidden and tree plantations, severely impacting food security for the following year. In Piaraçu, participants discussed the wildfire vulnerability of a *pequi* plantation (*Caryocar brasilienses,* Figure 26, location 12) which is important for nutrition. In Capoto, participants discussed the vulnerability to wildfires of a cassava plantation close to the village (Figure 28, location 2), as well as fruit trees planted around the village (Figure 28, location 1).

Across all villages, participants reported that wildfire also jeopardises some natural ecosystems and reduces the availability of resources essential for various activities. Wildfires can destroy plants used for crafts and medicine, timber trees used for construction and fruit trees.

> *"We take care of the forests because they're disappearing, and every time it burns, it reduces the trees, medicines, and animals even more. Where there used to be a lot of animals, ... there is no animal anymore because the fire is destroying the forest. The animals are no longer there, they are farther away. " elderly woman from Piaraçu*

In Mupa, participants identified two resource-rich locations with high wildfire risks during participatory mapping: an old swidden hosting different valuable species, such as *Caryocar brasilienses* (Figure 27, location 1), as well as a forest patch with valuable timber trees used for house construction (Figure 27, location 3).

In Capoto, where the landscape comprises a mix of fire-dependent and fire-sensitive vegetation, participants described several types of vegetation and areas that they deliberately should not burn to conserve valuable natural resources. These include riparian forests and regenerating fallows, as they are important for the reproduction of various animal species and the obtention of some medicinal plants (Figure 28, locations 4, 5 and 6). Regenerating swiddens are also important for the sustainability of swidden cultivation near the village (Figure 28, locations 5 and 6).

Some participants also reported that wildfires threaten animal populations, especially slow-moving species such as sloths, tapirs, armadillos, tortoises and snakes. In Mupa, participants were particularly concerned about the negative impact of wildfires on fishing, the village's primary food source. Wildfires along the Xingu Riverbank have reduced fruit production, thereby affecting fishery productivity. A wildfire in 2024 also destroyed part of the palm used to make arrows for traditional fishing practices (Figure 27, location 1). In Capoto, participants described avoiding burning several types of forest as providing food and shelter to animal populations that could then be hunted in the grasslands.



Figure 27. Participatory map from fire use and wildfire risks in Mupa. Number in yellow indicate low wildfire risk, in orange medium wildfire risk and in red high wildfire risk.

During field walks in Piaraçu, participants pointed out the impact of wildfires on the *Brachiaria spp*. grasslands, which prevent tree regeneration and trap ecosystems in a fire-adapted and grass-dominated stage. Some participants from Piaraçu expressed concern about wildfires escaping from Capoto/Jarina to nearby farmlands, and the potential conflicts it could trigger.

Wildfire risk management strategies

In each village, participants described some strategies for reducing the risks of wildfires near the village going beyond individual fire control measures. In Piaraçu, some of the swidden cultivations were classified as having a low wildfire risk despite the proximity of *Brachiaria spp*. (Figure 26, location 8), while other swiddens in similar locations were classified as having a high wildfire risk (Figure 26, locations 6 and 7). This lower wildfire risk was attributed to the actions of the fire brigades, a professional team of trained and equipped fire practitioners who systematically clear wide firebreaks and burn the swiddens following best

practices for other community members (more details in Chapter 5). During the field walk, a participant showed several fire breaks that were cleared by the fire brigades and a natural stream that act as a barrier against wildfires originating from the neighbouring villages.

In Mupa, participants ranked locations where they use fires for their swidden cultivation (Figure 27, location 4) as having a low risk of wildfire due to the extent of fire control measures, they deploy, such as clearing large fire breaks or burning in teams of several people. They also highlighted the importance of one location with a stream (Figure 27, location 3) that serves as a natural barrier to fight fires coming from neighbouring villages.

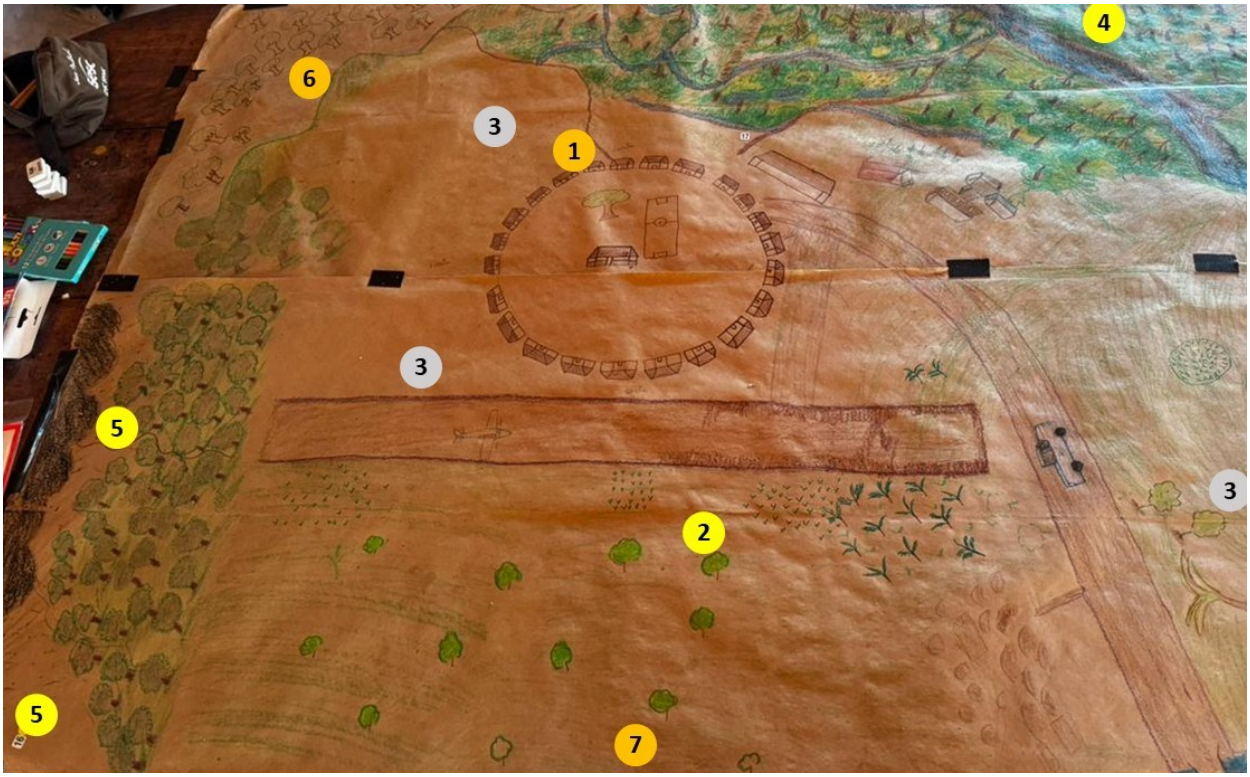


Figure 28. Participatory map of fire use and wildfire risks in Capoto done by male participants and grouping insights from male and female participants. Number in yellow indicate low wildfire risk, in orange medium wildfire risk and in grey areas that need to bue burned.

In Capoto, participants described to me how they are using fires in several fire-dependent vegetation types, such as savannas and grasslands (Figure 28, locations 2 and 3) while reducing wildfire risks in areas they wish to avoid burning (Figure 28, locations 4, 5 and 6). In the early dry season, fires are set near the village (Figure 28, location 1) to reduce fuel loads and prevent intense fires that could damage fruit trees planted around the village and homes, which are vulnerable to fires due to the roofing material used. During this period, the grasslands near a large cassava plantation are also burned to protect the crop from intense late-season fires (Figure 28, location 2). During the rest of the dry season, mosaic burning is carried out across grasslands and savannas to collect a variety of resources (Figure 28, location 3). Following the first rains, participants use fire at swidden sites to prepare land for planting new crops (Figure 28, location 6). This area was ranked as having a medium wildfire risk due to the risks of fires escaping into nearby secondary forests. This wildfire risk was increasing due to difficulties in predicting the onset of the rain season: some fires are lit before extended periods of droughts and, if they escape, can burn over large areas before being extinguished by rainfall.

4.5 Discussion

Across the three villages, participants demonstrated distinct fire use and wildfire risk management strategies, shaped by extensive local fire knowledge related to soils, vegetation types, weather, fire behaviour, and their ecological impacts. The elements of the local fire knowledge were deployed to maximise resources obtained while reducing wildfire risk and associated resource depletion.

Fire use and associated local fire knowledge were partly linked to the livelihoods, cultural needs, and environments of each village. Participants from Capoto live on a plateau dominated by fire-adapted grassland and savanna, analogous to the ancestral territory of the Metyktire people, which enables the continuation of traditional burning practices embedded in their culture and cosmology (Rodrigues 2022). In contrast, Piaraçu is located amidst fire-sensitive vegetation and experienced the highest increase in wildfire risk,

primarily due to the invasion of *Brachiaria spp.* used on pastures surrounding Capoto/Jarina. While the inhabitants of Piaraçu historically employed diverse types of fires supported by local fire knowledge, they have adapted their fire use and associated governance system in response to escalating wildfire risks (Rodrigues 2022, more details in Chapter 5).

Inhabitants of Mupa use fewer fires and deploy more fire control measures, a pattern that may be partly explained by their distinct cultural and historical background, as well as the environmental conditions of their surroundings. Unlike the Metyktire people, the Yudjá people traditionally inhabited riverine forests along the lower Xingu, with fishing forming a central component of their livelihood strategies (Indigenous Peoples in Brazil, 2025). Thus, fires for hunting and agriculture play a less prominent role in their livelihood, while the potential damage of wildfires to fisheries productivity may encourage more cautious fire use.

Understanding the rationales behind local fire use and wildfire risk management strategies is crucial to aligning policies aiming to reduce wildfire risk with local contexts and community preferences (Mistry & Bizerril, 2011; Smith et al., 2024). However, engaging effectively with the diversity of traditional fire use and associated knowledge can be challenging, especially in regions with high ecological and cultural diversity such as the Xingu headwaters (Schwartzman and Zimmerman 2005; Marques et al. 2020).

Participatory maps were central to understanding fire use and wildfire risk management, displaying preferred locations for using different types of fires, vegetation types relevant to fire management, presence of fire-vulnerable resources and natural barriers to wildfire. Some important features shaping local wildfire risks were not detectable using satellite imagery, such as small fires below the minimum detection threshold, which burn over short periods and are frequently hidden by the forest canopy (Giglio et al. 2020).

Classification of vegetation types and their fire sensitivity, based on local fire knowledge, is valuable for understanding feedback between fires and vegetation as well as the cultural value of different parts of the landscape. In the neighbouring Xingu Indigenous Park, complex classification systems of secondary forests partly based on the use of indicator species and their resistance to fires can help prioritise the protection of certain forest patches to support the natural regeneration of the forest (Schmidt et al. 2021). Even in fire-prone landscapes where fire is extensively used, Indigenous communities frequently leave patches of unburnt habitats to promote the regeneration of plant and animal species (Posey 1985; Mistry et al. 2005; Trauernicht et al. 2015).

Results show that local fire knowledge supports fire use and wildfire risk management strategies that protect fire-sensitive vegetation. Participants from all villages rely on seasonal and short-term weather indicators to determine the best time to use fires, integrating their future use within a broader calendar that includes factors such as wildfire risks, species phenology and resource availability, a practice widespread among Indigenous fire practitioners (Eriksen 2007; Falleiro et al. 2021; McKemey et al. 2021). Not only does the use of seasonal indicators allow flexibility in burning dates that could be useful in the context of strong inter-annual variations in seasonality (Armatas et al. 2016), but it also supports the conservation of valuable resources vulnerable to fires (Falleiro et al. 2021; McKemey et al. 2021).

The use of fire in savanna vegetation in Capoto, aiming to burn fire-dependent vegetation types while reducing wildfire risk in fire-sensitive vegetation types, aligns with other studies from the Brazilian Cerrado (Posey 1985; Mistry et al. 2005; Welch et al. 2013) and other savanna landscapes across the world (Laris 2002; Bilbao et al. 2010; Sletto and Rodriguez 2013). In Northern Australia, indigenous burning is now supported by a carbon abatement program that promotes early dry season burning to avoid carbon loss caused by late dry season burning (Russell-Smith et al. 2013). While there is discussion for emulating this model across other savanna landscapes (Russell-Smith et al. 2017), the results show that wildfire risks are controlled through the frequency of burns rather than their seasonality. In

the savanna landscape of Venezuela, Bilbao et al. (2010) identified a similar mechanism, and demonstrated experimentally the importance of the number of years since the last burn to prevent wildfires in nearby fire-sensitive forests.

Exploring local fire knowledge and integrating it into the decision process prevents wildfire risk management strategies from being based on ecological knowledge developed in different environmental contexts (Ray et al. 2012). This might be especially crucial in ecologically complex areas, such as the Cerrado-Amazon transition zone, where dense forests, ecotonal forests, and savannas coexist, each with distinct sensitivities to fire (Marques et al. 2020). From an administrative standpoint, the whole of Capoto/Jarina is classified as part of the Amazon biome, resulting in a risk-based approach to wildfire risk management premised on the assumption that most fires are environmentally damaging. While this approach could be relevant for Piaraçu and Mupa, this is not the case in Capoto, where many extensive fires are burned to increase the provision of different ecosystem services, including wildfire risk reduction. Nevertheless, savanna fires in Capoto lit by inhabitants of the villages were classified as “wildfires’ in the annual report of the Prevfogo fire brigade (Appendix 4.7).

This highlights potential scale mismatches arising from centralised wildfire risk management initiatives, as the ecological variability of the landscape does not necessarily align with the scale of the social organisation responsible for its management (Cumming et al. 2006). Mismatches between the scale of social institutions involved in fire management and the associated ecological processes can lead to ecosystem mismanagement due to a limited understanding of key challenges, such as the definition of wildfires, or an inability to identify relevant solutions (Cumming et al. 2006).

The risk of scale mismatch is mitigated by the high degree of agency that the inhabitants of Capoto/Jarina exercise over fire management and organisation within their territory, as well as the fire brigade members’ recognition of the distinctiveness of Capoto inhabitants' relationship with fire use and wildfire risk management strategies. This contributes to a

growing corpus of evidence demonstrating that local community involvement, engagement with local knowledge, and collaborative governance can reduce issues related to scale mismatches (Ray et al. 2012; Lyver et al. 2015; Bodin 2017). In co-managed socio-ecological systems, collaboration between customary and governmental institutions, alongside mutual learning, has been shown to support adaptation to a changing climate (Ruiz-Mallén and Corbera 2013).

However, the integration of local ecological knowledge in wildfire risk management is affected not only by scale mismatches but also by the hierarchisation of different management objectives and associated knowledge (Eriksen and Hankins 2014). Fires used in the savanna of Capoto both reduce wildfire risks and serve other important functions, some of which may not align with federal environmental policies, such as increasing the dominance of some useful species, a practice that shaped the present biodiversity of the Amazon (Levis et al. 2018). The multiplicity of objectives behind fire use and their co-benefits are key characteristics of indigenous fire use across the world (Trauernicht et al. 2015; Nikolakis and Roberts 2020). The institutionalisation of fire management programmes, and the associated focus on one or a few objectives, can result in a disconnect with traditional fire use, cause disagreements within the community, and prevent the transmission of part of local fire knowledge (Eriksen and Hankins 2014; Petty et al. 2015; Perry et al. 2018).

4.6. Conclusion

Fine-scale differences in the environment of each village, as well as their socio-environmental history and culture, result in different preferences for fire use and approach to wildfire risk management. The use of participatory methods to explore local fire knowledge yielded information that could not be leveraged using satellite imagery. Some of this information is pivotal in understanding the interactions between regional drivers of wildfire risk, such as climate change or biological invasions, and local drivers of wildfire risk, such as the use of traditional fires.

Flexible, locally tailored strategies that recognise ecological diversity across villages in a territory and the multiple objectives of fire use and wildfire risk management are crucial for effective wildfire risk management in the Xingu headwaters and similar socio-ecological systems. While some of the objectives of traditional use of fires align well with current management objectives at a broader level, some of them don't necessarily fit within national objectives of wildfire risk reduction and biodiversity optimisation. Detailed information about fire-vulnerable resources could help prioritise wildfire risk reduction strategies to minimise the negative impacts of wildfires on livelihoods and cultural needs.

Chapter 5: Adaptation to changing wildfire risks: insights from Capoto/Jarina

5.1 Main contributions

- Inhabitants of Capoto/Jarina perceived a change in weather conditions consistent with analysis of global climatic models, as well as changes in the vegetation that affected the flammability of the landscape.

- While climate-related change in wildfire risks was consistent, changes in vegetation differed vastly across the three communities, resulting in different changes in wildfire risk.

- Participants adapted to these changing wildfire risks by adopting diverse individual and collective adaptation strategies, according to the severity and novelty of wildfire risks encountered.

- Local fire brigades proved pivotal for the adaptation to changing wildfire risk, but their actions encountered some challenges that could have long-term impacts on the adaptive capacity of communities.

5.2 Introduction

Indigenous land in the Brazilian Amazon faces multiple environmental threats (Rorato et al. 2021), and among them, the increasing frequency of wildfires is particularly concerning. Fires originating from outside these territories frequently cross into Indigenous lands, damaging ecosystems and disrupting local livelihoods (de Oliveira et al. 2020; Walker et al. 2020; dos Santos et al. 2021). Meanwhile, the populations inhabiting these lands are growing (Begotti and Peres 2020), which in some cases heightens pressures for land clearing and increases the use of fires (Silvério et al. 2022).

These challenges are compounded by climate change, which is already impacting Indigenous lands and protected areas across the Brazilian Amazon (Feeley and Silman 2016). Changing climatic conditions are shifting vegetation composition, reducing the prevalence of drought-intolerant taxa (Esquivel-Muelbert et al. 2019) and are projected to trigger localised rainforest die-offs, particularly in the Southern and Eastern Amazon (Lapola et al. 2020; Parry et al. 2022). These changes in vegetation, associated with rising temperature and prolonged dry seasons, create favourable conditions for the expansion of invasive grass and abrupt intensification of fire regimes (Pausas and Keeley 2014; Archibald et al. 2018; Mano et al. 2023).

The increased prevalence of large wildfires threatens the socio-ecological systems of Indigenous and traditional communities (Silvério et al. 2022). Wildfires can damage agricultural assets and infrastructure, reduce the availability of forest resources, and create health and safety risks (Carmenta et al. 2016; Ribeiro et al. 2024), as well as generate canopy gaps that facilitate grass invasions, further increasing long-term wildfire risks (Silvério et al. 2013). In some cases, the intensification of fire regimes has led to forest loss and scarcity of secondary forest available for swidden agriculture, jeopardizing food security (Schmidt et al. 2021; Silvério et al. 2022). To limit the deleterious impact of intensifying regimes on their socio-ecological systems, it is primordial to adapt fire use and wildfire risk management strategies to new conditions.

Local ecological knowledge, also frequently called traditional ecological knowledge (Berkes, 2018), plays a key role in understanding ongoing environmental changes. Through constant observations and monitoring of landscapes and resources, local communities develop and integrate new knowledge into decision-making systems that ensure biodiversity conservation and strengthen the resilience of their socio-ecological systems (Evans et al., 2011; Folke et al., 2009; Ruiz-Mallén and Corbera, 2013). Local ecological knowledge can be developed and shared at a community level, reducing the individual cost of observation and experimentation to generate new knowledge (Reyes-García et al. 2016). For example, local ecological knowledge related to phenology can be used to determine the ideal seasonality of fire use to minimise their impact on ecosystems and risks of wildfires in a changing climate (Armatas et al. 2016).

The recent recognition of the positive role of "traditional and adaptive use of fire" in Brazilian national Integrated Fire Management legislation (law 14.944 of 31st of July 2024) highlights the expected contribution of Indigenous fire knowledge to building fire-resilient landscapes. Fire brigades deployed in Indigenous land and protected areas, which primarily recruit members from Indigenous and traditional communities, have demonstrated their capacity to mitigate the effects of drought on fire regimes (Oliveira et al. 2022). The detailed understanding of the impact of climate change on ecosystems resulting from frequent observation by Indigenous and traditional communities (Reyes-García et al. 2014) can also inform wider policy development for adaptation to changing climate (Brondízio et al. 2021).

Nevertheless, the rapid pace of ecosystem changes and broader social changes within Indigenous communities could challenge their adaptive capacity (Fernández-Llamazares et al. 2015). The extent and type of adaptation measures required to maintain fire-resilient landscapes will depend on multiple factors, including the severity and changes in wildfire risk (McWethy 2019). Furthermore, fire resilience strategies are embedded in complex social and historical contexts that can shape which adaptation pathways are adopted and preferred (Copes-Gerbitz et al. 2021). Therefore, it is crucial to understand the role of Local

ecological knowledge in perceiving wildfire risk changes, how this knowledge informs adaptation measures, and the trade-offs that may result from these decisions.

To address this, I investigate whether and how Indigenous communities in the Amazon perceived change in wildfire risks, how they are adapting their use of fire and wildfire risk management practices, and the challenges they encounter in doing so. Using a mixed-methods approach, I use Capoto/Jarina, an Indigenous land in the North of Mato-Grosso as a case study and build on the results presented in the previous chapter about their local fire knowledge.

5.3 Methodology

In the following sections, I will present the research design of this study, justifying the integration of different sources of data for exploring the relevant research questions, as well as a summary of the methods employed for qualitative data collection and analysis and a detailed description of the methods for quantitative data collection and analysis. For more detailed information regarding the sites of study, the ethical considerations and the methods employed for qualitative data collection and analysis, refer to Section 4.3.

5.3.1 Research design

The co-production of new knowledge based on insights from different knowledge systems compelled the research team to carefully consider their integration, due to a history of dismissal of Local ecological knowledge and its disrespectful validation through other knowledge systems (Roue and Nakashima 2018). Thus, attention was taken to co-develop research questions and discuss both data collection and analysis with local partners. Drawing from an initial concern expressed by members of the Raoni Institute about the deleterious impact of changing wildfire risks, the research team co-designed an interdisciplinary research project exploring this issue.

Prior to the first visit to Brazil, I organised a series of meetings with my primary contact from the Raoni Institute (the coordinator of territorial management) and my thesis supervisor and created a discussion thread. This allowed us to discuss the desired output of the research project, methodologies considered and share results from preliminary quantitative analysis. The research methods were adapted according to the feedback received. Prior to the pilot study, a detailed presentation of the research activity, ethical aspects, and design of the data collection activity was organised with my primary contact from the Raoni Institute.

The integration of local ecological knowledge with physical and natural science knowledge to investigate change and adaptation to wildfire risks was deemed relevant for several reasons. Firstly, the reliance of local ecological knowledge on frequent, place-based observations of the weather can overcome some limitations of global climatic models, especially in data-scarce regions or areas with fine-scale climatic variations that are not well captured by these models (Savo et al. 2016; Chanza and Musakwa 2022). Secondly, local ecological knowledge includes many indicators of the impacts of environmental changes on both natural and social systems (Reyes-García et al. 2024), which can then affect local fire regimes (Pausas and Keeley 2014). Thirdly, local ecological knowledge is developed in a specific location and reduces the over-reliance on ecological knowledge from other regions and environmental contexts, which may be misaligned with local conditions (Ray et al. 2012).

While these specificities of local ecological knowledge made them relevant to understanding changes in wildfire risk, there are also benefits to integrating and cross-validating results with data originating from remote-sensing products and climatic models. Satellite imagery allows tracking of forest and land-use changes over large areas, including remote places and private lands outside of Capoto/Jarina, which may be difficult to access (Fuller 2006). This can provide useful information on regional drivers of wildfire risk and issues related to wildfires from outside of Capoto/Jarina. Moreover, long-term changes in weather patterns are susceptible to memory bias and various processes affect perceptions of climate change (Mello-Théry et al. 2020; Arora and Feng 2024), especially in environments

with significant inter-annual variability such as in North Mato-Grosso (Dubreuil et al. 2012). The use of Global Climatic Models can help mitigate these biases and enable robust analysis of trends in climate-related wildfire risk.

I adopted the Multiple Evidence Based Approach, focusing on the complementarity between different knowledge systems to create an enriched picture of an environmental issue and generate useful knowledge about it (Tengö et al. 2014). After an initial agreement to investigate fire use, wildfire risk management and its changes over time, the research team agreed on the use of a concurrent nested mixed-methods research design (Creswell et al. 2003). The study includes (i) a predominantly qualitative approach for understanding fire use and wildfire risk management by communities in Capoto/Jarina, based on data collected during field visits, and (ii) a complementary quantitative approach to analyse the long-term evolution of wildfire risks in and around the Capoto/Jarina, based on land cover and global climate model data.

Before the initial visit to Capoto/Jarina, I compiled a report with the results from quantitative analysis, presented it to partners from the Raoni Institute and, after the research activity, to the members of the local fire brigade. The preliminary results from the main qualitative and quantitative analysis were presented during the second visit to participants of the research project, as well as inhabitants from different villages of the Capoto/Jarina gathered in Piaraçu to discuss the Territorial and Environmental Management Plan. This ensured a multidirectional evaluation of the results and insights gathered across knowledge systems, facilitating the generation of new knowledge on fire management and wildfire risk adaptation (Tengö et al. 2014) After the second field visit, a discussion thread with my primary contact from the Raoni Institute served to share the results of data analysis. A second version of the report with main results from both quantitative and qualitative analysis was prepared in early 2025 and presented by the members of the Raoni Institute during a visit to Capoto, Piaraçu and a village from the neighbouring Mekragnoti indigenous land.

5.3.2 Qualitative data collection and analysis

For the collection of qualitative data, the research team employed a flexible approach that combined semi-structured interviews with key informants, focus group discussions centred around the creation of a fire typology, a seasonal burning calendar, and a participatory map of fire use and wildfire risks, as well as field walks. In total, 65 participants (26 women and 39 men) were involved, including 30 from Piaraçu, 6 from Mupa, 21 from Capoto, 3 from Totnohre and 5 from other villages. All activities were carried out with at least one local facilitator and conducted either in Portuguese, Mẽbêngôkre or Yudjá, depending on the participant's preferences. All recordings were integrally transcribed and translated into English. An inductive approach was used for doing a thematic analysis of the corpus of data. Further details on the collection and analysis of qualitative data can be found in Section 4.3.

5.3.3 Quantitative data collection and analysis

The quantitative data collection and analysis focused on two thematics identified as relevant for complementary analysis: land use change and climate change.

For land-use change analysis, I used MapBiomas 9 to examine changes in land use around the three villages of Piaraçu, Mupa and Capoto between 1985 and 2023. My results include an alluvial diagram and results from the Man-Kendall trends analysis and the Thiel-Sen estimator for buffer areas with 3km, 5km and 10km radii around each village. I conducted a similar analysis for unprotected land outside of the Capoto/Jarina, in a buffer zone of 5km, 10km and 20km. Land uses representing less than 1% of the total area covered were excluded from the analysis. I also analysed the land use changes of each pixel in the 3km zone (or 10km for the Capoto/Jarina) between 1985 and 2022 and reported all results representing more than 1% of the total area.

To evaluate the influence of climate on wildfire risk, I used the Canadian Fire Weather Index (FWI), which incorporates daily surface temperature, precipitation, relative humidity, and wind to estimate the moisture content of different fuel categories and the likelihood of fire

spread. FWI has shown good performance in predicting fire risk in diverse contexts, including the Eastern Amazon (Cavalcante et al. 2021). I used the dataset described in Vitolo et al (2020), built from ERA5 weather data analysis, which provides daily estimates of the different components of the Fire Weather Index at a 0.25° resolution since 1940.

To assess recent changes in fire weather conditions, I selected a reference period from 1950 to 1999. I then analysed changes in the FWI for the 2000-2022 period based on four annualized FWI metrics. Two of them indicate risks of extreme fire weather: annual maximum FWI (FWImax) and the number of days exceeding the 95th percentile of the FWI value during the reference period (FWI95). The other two relate to the wildfire risks throughout the season: the maximum 90-day mean value (FWImean) and fire season length (FSL). I used Man-Kendall trends analysis and Thiel-Sen estimators to assess trends in these four metrics between 1990 and 2022. Additionally, I applied Mann-Kendall trend analysis and Thiel-Sen estimation to evaluate trends in daily FWI throughout the fire season.

5.4. Results

5.4.1. Increasing wildfires risks and impact on local fire knowledge

Both qualitative and quantitative analyses point to a similar increase in wildfire risks due to the change in rainfall seasonality and land use changes outside of the Capoto/Jarina, while each village is experiencing different changes in vegetation affecting their local wildfire risks. Certain changes in seasonality and vegetation render some of the local fire knowledge less reliable.

Changes in local climate

Across the three villages visited, all the participants described changes in the local climate with a high degree of consensus. Across all villages, participants reported that the onset of the rainy season was harder to predict than in the past and was often characterized by moderate rainfall. Participants from Piaraçu and Capoto described an overall reduction in precipitation. Participants from Piaraçu also reported more frequent droughts during the rainy season and more frequent rainfall during the dry season, as well as an increase in temperature. Participants from Piaraçu and Capoto also noted an increase in wind speed.

Quantitative analyses of the Fire Weather Index are concordant with these observations. The Fire Weather Index annual metrics indicate a longer and more intense fire season. Since 2000, fire season length increased by 21.8 (± 3.1) days and the average daily FWI increased by 6.0 (±0.6) during the 90 days of the season with the highest FWI. There are also 4.8 (±1.5) supplementary days with very high wildfire risks, and the maximum annual FWI became higher by 4.5 (±1.5).

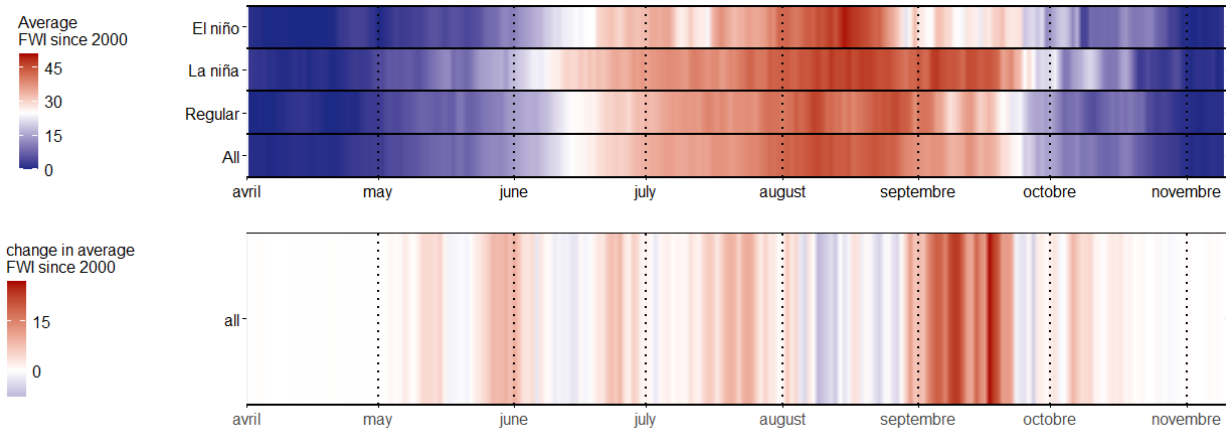


Figure 29. Band plots showing the daily average Canadian Weather Fire Index for the 2001-2022 period (bottom) and the average change compared to the reference period between 1951 and 2000.

Analysis of the change in the daily Fire Weather Index corroborates the trends observed by the local community: there is an increase in the overall FWI during the dry season and an acute increase in FWI over the first three weeks of September (Figure 29). There is also a modest decrease in the FWI in August, which might correspond to the rainfall described by some participants (Figure 29).

When looking at the FWI between normal, El Niño and La Niña years, I can see a clear pattern: during El Niño years, the FWI increased strongly over the month of August compared to other years but decrease quickly by the end of August. During La Niña years, the average FWI is slightly higher than for other years from mid-June and continues to increase until late September. None of the participants mentioned the El Nino cycle directly during the interviews, however, they recognized the more important drought during certain year corresponding to strong La Niña events, such as in 2020 and 2022.

Change in land cover and vegetation

Change in land cover and vegetation outside of the Capoto/Jarina

Participants from all groups and villages described a past situation in which a larger territory, with forest and savanna extending beyond the borders of present-day Capoto/Jarina, provided a better climate and greater resources such as fish, game, medicinal plants, fruits and construction materials. One consequence of Capoto/Jarina's delimitation and past conflicts has been the territorialisation of the region: forests and Indigenous peoples remain within Indigenous land boundaries, while surrounding areas have been impacted by logging, deforestation, pastoralism and, more recently, monoculture expansion.

> *"But today, things are changing, and now there are many farms around our area. It wasn't like this, there was no land boundary, it was all ours, the forest, the forest was free, and we used to walk a lot in the forest" elderly woman from Piaraçu*

Participants from all villages associated these changes outside the territory with the increasing scarcity of essential resources, drying of the local climate and heightened wildfire risk. This reduction of the territory was particularly impactful for the participants from Capoto, as they rely on the exploitation of large areas of savanna and deforestation has increased close to their village over the last few years.

Quantitative analysis indicates significant land use changes in the unprotected areas surrounding Capoto/Jarina between 1985 and 2023. Areas covered by pasture and soy cultivation have increased within the 5, 10, and 20 km buffer zones, and other temporary crops have expanded within the 20 km buffer zone, while forest area has decreased within the 5, 10, and 20 km buffer zones. Additionally, the area of floodable forest has decreased within the 20 km buffer zone (see Appendix 5.1 for the detailed results of all Mann-Kendall trend tests). Land transition analysis within the 10 km buffer zone shows that parts of the forest have been converted into pasture (12,4% of total area) and soy plantations (1,5% of total area), while some pastures have been converted to forest (1,2% of total area, Figure 30).

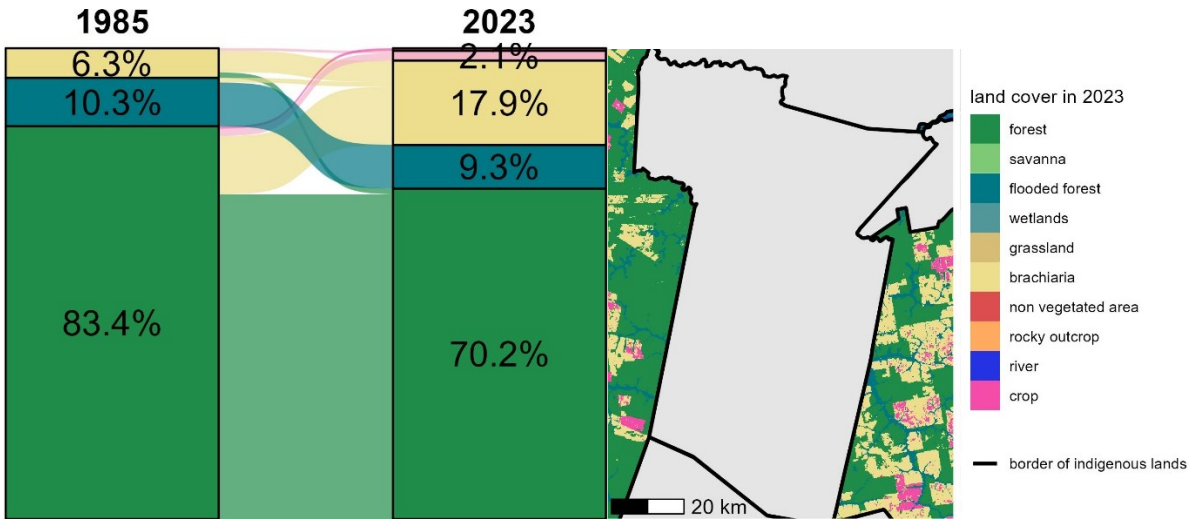


Figure 30. Alluvial plot of the land use change in the 10km buffer zone of unprotected land around Capoto/Jarina and map of the land cover in 2023

Change in land cover and vegetation in Piaraçu

All participants from Piaraçu discussed the recent invasion by *Brachiaria spp.*, and both men's and women's groups reported the spread of razorgrass (*Paspalum virgatum L.*) in a drying pond. All participants also discussed the degradation of certain forest patches due to the cumulative impact of fires and droughts. Participants from the men's groups also observed lower water levels in the Xingu River during certain years, an increasing number of visible sandbanks and the disappearance of small streams. During field walks, a key informant also showed a restored forest area on a former *Brachiaria spp.* grassland that had regrown for about 30 years.

> *"After the road, problems entered the villages. Cattle trucks cross here, and the cattle eat the grass and defecate. The grass falls to the ground, seeds sprout, and start spreading there, and there is a lot of grass around us now. Nowadays, it is worrying us a lot, as this is destroying the forest. This pasture grass kills trees and destroys the forests. And the grass dries up a lot, and in the fire, they burn the forest, and it spreads, and every year we have this problem. We didn't suffer from these things in the past."*
> *elderly woman from Piaraçu*

Quantitative analysis indicates a significant increase in *Brachiaria spp.* grassland within the 3km and 5km buffer zones and for forest and floodable forest within the 10km buffer zone. Meanwhile, river and wetland areas declined within the 3km, 5km and 10km buffer zones. A land transition analysis within the 3km buffer zone revealed that part of the river turned into wetlands and floodable forests (respectively 1,9% and 1,3% of total area), some of the wetlands transformed into floodable forests and forest (respectively 4,3% and 1.1% of total area) and some wetlands and forest converted into *Brachiaria spp*. grassland (respectively 1,3% and 1,8% of total area, Figure 31). This corroborates local observations of an increase in areas covered by *Brachiaria spp.*, a decline in Xingu River levels, and the drying up of some freshwater lagoons.

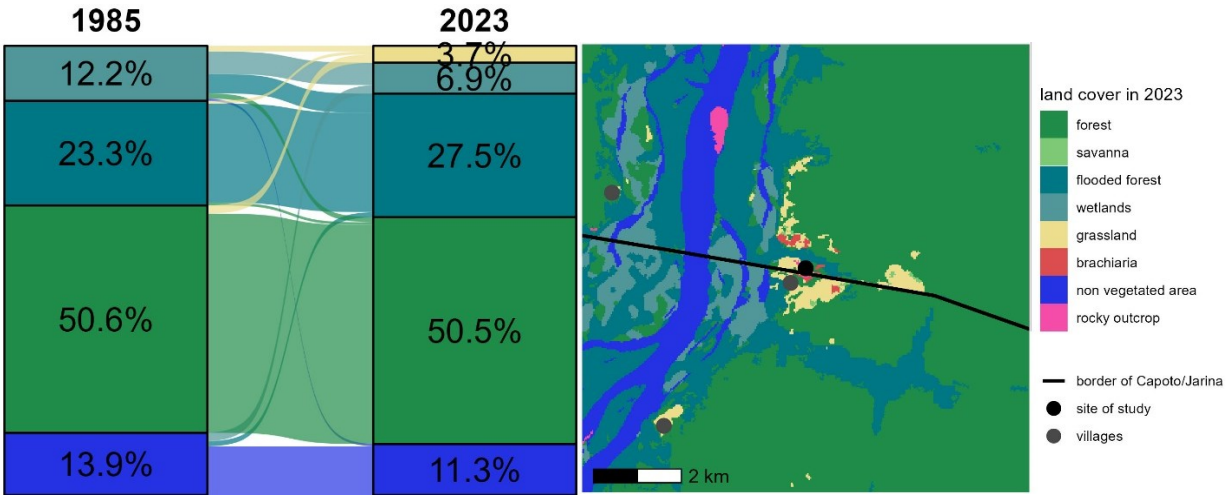


Figure 31. Alluvial plot of the land use change in the 3km buffer zone around Piaraçu and map of the land cover in 2023

Change in land cover and vegetation in Mupa

In Mupa, participants described the drying up of the forest, with more dry leaves accumulating on the ground [6]. Participants also observed lower water levels in the Xingu River during certain years, and an increasing number of visible sandbanks. Participants mentioned changes in water colour and increased pollution.

> *"Unfortunately, when we start burning, the fire far spread far away. And before it wouldn't go far, it couldn't go because the layer of leaves was very thick and damp, before it was more humid. When it's very dry, it starts drawing water too. Because the water gets absorbed into the leaves, it doesn't spread far. Here, actually, it spreads far: if the fire starts spreading, it spreads far." men from Mupa*

Quantitative analysis indicates a significant increase in floodable forest within the 3km, 5km and 10km buffer zones, and for forests within the 10km buffer zone, while river and wetland areas declined across 3km, 5km and 10km buffer zones. Land transition analysis within the 3km buffer zone revealed that part of the river and wetland turned into floodable forests

[6] Results on long-term change in Mupa should be taken with caution as the village was only recently opened and these observations rely on frequent visits during fishing expeditions in the past.

(respectively 3,3% and 6,5% of total area, Figure 32). This aligns with local observations of a declining Xingu River water level.

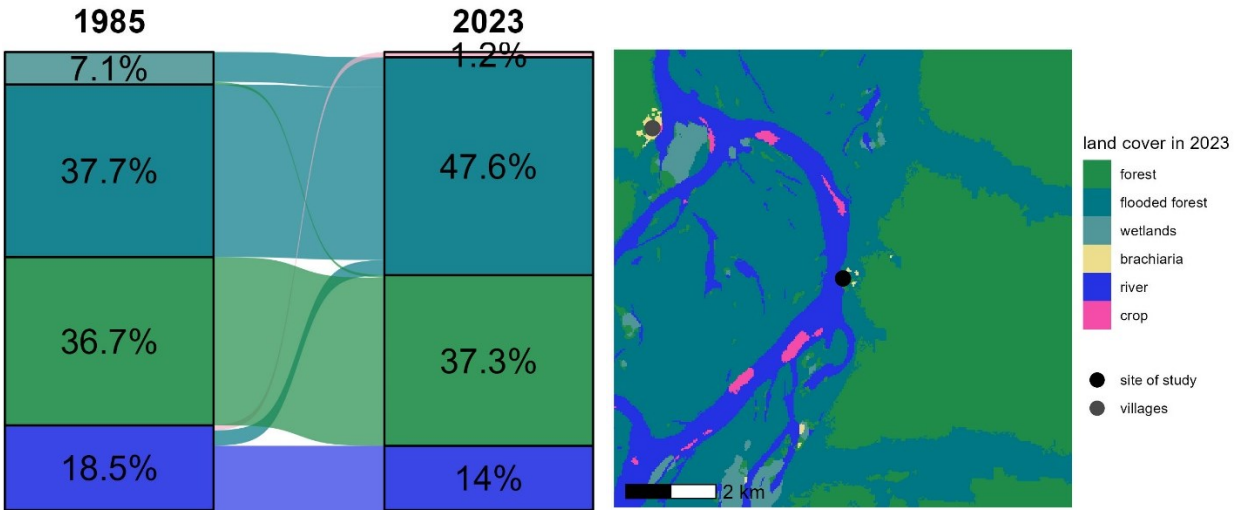


Figure 32. Alluvial plot of the land use change in the 3km buffer zone around Mupa and map of the land cover in 2023

Change in land cover and vegetation in Capoto

In Capoto, there was a consensus amongst participants about the growth of fruit trees planted around the village since it was established, including fruit species such as mango (*Mangifera indica*) and pequi (*Caryocar brasilienses*). However, opinions differed regarding woody encroachment in the savanna: while an elder and a participant from Menkragnoti who frequently visit relative in Capoto reported woody encroachment, men's and women's groups noted no significant changes in savanna vegetation.

> *"It changed a lot where I lived in the countryside because there was only the savanna, right? There wasn't any vegetation, just that kind of small palm tree, which grows in some savannah near the river, in the mountains. But as I grew older, sometimes I passed by there, that savanna turned into a forest. It's turning into a forest now" men from Menkragnoti who visit frequently Capoto*

“It hasn't changed at all, nor has the savannah, but the plants are the same, the savannah hasn't changed much, it hasn't changed much. “ woman from Capoto

Quantitative analysis indicates significant increases in floodable forest and *Brachiaria spp.* grassland within the 3km buffer zone and savanna within the 3km and 5km buffer zones. Meanwhile, forest coverage declined within the 3km, 5km and 10km buffer zones. Land transition analysis within the 3km buffer zone shows that part of the forest, wetland and grassland turned into savanna (respectively 4,3%, 4,9%, and 1.1% of total area) and some wetland converted into floodable forest (1.5% of total area, Figure 33).

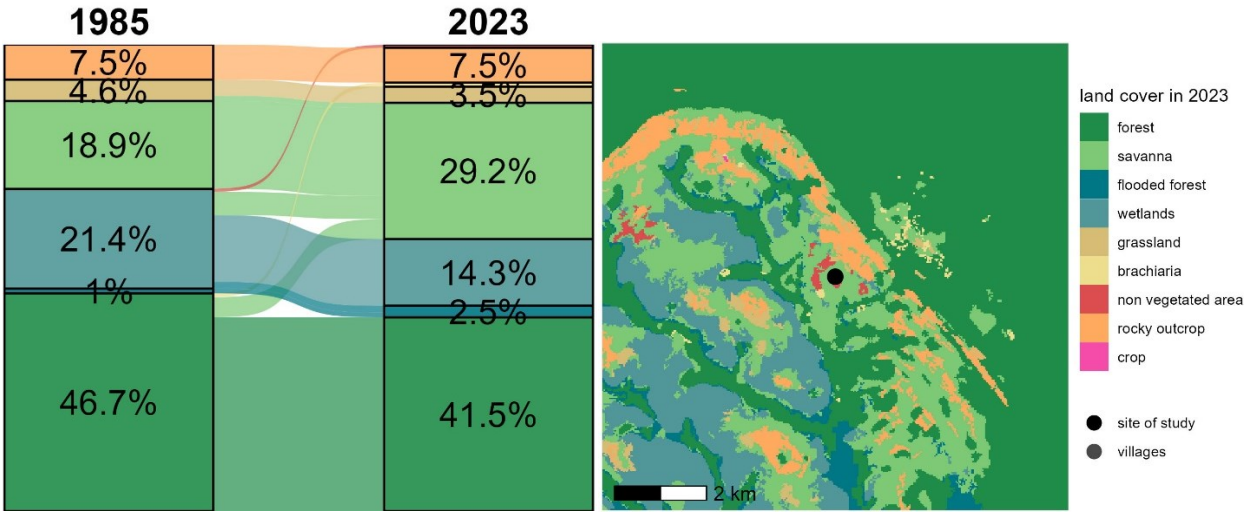


Figure 33. Alluvial plot of the land use change in the 3km buffer zone around Capoto and map of the land cover in 2023

Impact of climate and environmental change on wildfire risk

Across all three villages visited and participant groups, there was a consensus that overall wildfire risk is increasing. However, the drivers and extent of this change varied: participants from Capoto observed a modest increase in wildfire risks, while those from Piaraçu described a fundamental shift in wildfire risks and their relationship with fire use [7].

Across the three villages, changes in rainfall seasonality have caused difficulties in determining the best time to burn the swiddens and plant crops. Participants want to minimize the number of rains before burning their swiddens to have a clean burn but also desire to have abundant rainfall just after the burn. The different phenological indicators traditionally used across the three villages to indicate the beginning of the rainy season, such as changes in the song of a cicada's species or the flowering of the yellow ipé were considered increasingly unreliable.

> *"Before, we did things at the right time, we didn't know the right time, but we planted, we didn't know the date, the month, but we knew when flowers, fruits would come, we knew the right time! The elders know well, and they guide us, and that's why we understand things. But we are losing the elders, and we lose knowledge over time, and with climate change too. We planted and bore fruit, and today we have things that we plant that don't bear fruit, climate, time, plants die, some bloom but don't bear fruit" elderly men and traditional chief from Piaraçu*

Participants from Piaraçu and Mupa also reported a loss of foliage in the canopy of the forest resulting from the impacts of droughts and, in certain sites, past wildfires. This loss of foliage increased the amount of solar radiation reaching under the canopy, creating drier conditions favouring wildfires. They also discussed the increasing dryness of the soil, which becomes flammable at the end of the dry season. This poses several new challenges for managing wildfire risks: small fires, such as those used to roast fish, can smoulder in the soil and start

[7] *The village of Mupa was created 3 years ago, and while the habitants of Mupa visited frequently this landscape, the observations on the long-term evolution of wildfires reported here correspond to the wider Piaraçu-Mupa areas (including several villages along the Xingu River), in which participants lived before the creation of Mupa.*

wildfires, sometimes reigniting several days later. Moreover, wildfires burning through the soil can be particularly difficult to combat and may blaze over long distances, with several respondents evocating fire travelling dozens of kilometres over the course of several weeks.

All groups interviewed in Piaraçu described a transition from a past with 'good fires', when fires were used with few fire control measures and rarely escaped, to a present situation with 'bad fires', when fires are used with many fire control measures and still frequently escape. In Piaraçu, there was a consensus that the shift from 'good fires" to 'bad fires' was predominantly due to the increased presence of *Brachiaria spp.*, creating a layer of dry fuel that could easily catch on fire and burn intensely.

> *"Nowadays, we take great care not to burn the forest, but still, we burn. It burns the animals, the forest medicines, the fruits, fire destroys, and that's why it's very worrying. But in the past, there wasn't so much fire that burned the forests" elderly men from Piaraçu*

Participants from Capoto described a modest increase in wildfire risks due to land use changes outside Capoto/Jarina. Both women and men participants spoke about the increasing risks of wildfires coming from neighbouring farming communities. Capoto is located close to an active-deforestation frontier, and several farms have been cleared near the border of Capoto/Jarina in recent years. This has resulted in the abandonment of certain patches of land, due to concerns about potential conflicts, and participants attributed at least one large wildfire to farms outside the Indigenous land.

5.4.2. Adaptation measures to increasing wildfire risks

Faced with increasing wildfire risks, participants have adapted their fire use and wildfire risk mitigation strategies, ranging from subtle changes in their fire management practices to transformative change deeply modifying the relationship between local communities and fires.

Indigenous-led fire brigade

Since 2004, a voluntary fire brigade has operated in Capoto/Jarina, conducting environmental education during the early dry season, carrying out activities to reduce wildfire risks throughout the dry season, and responding to wildfire events when possible. The brigade originated from a collaboration between the inhabitants of Capoto/Jarina and the Mato Grosso military fire department, was professionalised in 2019, and was integrated into the national Prevfogo fire brigade programme in 2024[8].

Each year, a training session is organised in Piaraçu for all inhabitants of Capoto/Jarina who are willing to join the fire brigade, during which they learn about basic wildfire prevention and firefighting techniques at a dedicated training site. The best candidates are recruited into the professional brigade, while others who meet the required standards join the voluntary fire brigade, which can be mobilised during wildfire events. The fire brigade possesses substantial expertise in fire use and wildfire control, acquired through a combination of individual experience, formal training, and operational practice.

[8] The Prevfogo fire brigade programme seasonally hires, equips and trains fire practitioners in Indigenous lands and protected areas of Brazil to conduct wildfire risk prevention activities and fight wildfires. In Indigenous land, this program is co-managed by two governmental institutions, the FUNAI (National Indigenous People Foundation) and IBAMA (Brazilian Institute of the Environment and Renewable Natural Resources)

Replacement of traditional burning practices

Due to the higher wildfire risk and associated damages compared to other areas, Piaraçu's region is designated as the fire brigade's priority area of intervention. Most inhabitants of Piaraçu have delegated the responsibility to burn the swiddens to the fire brigade, so they could do it following best practices and prevent wildfires. During the dry season, each family clears their swidden and registers it with the fire brigade. From August onwards, the fire brigade visits the swidden at least three times: the first time to estimate the best day to burn according to fuel dryness and check that firebreaks are properly prepared, the second time to burn the swidden, and the third time to assure no wildfire escaped.

When they burn the swiddens, members of the fire brigade ensure that sufficient fire control measures are deployed to prevent fires from escaping into nearby *Brachiaria spp*. grasslands. The cleanliness and width of firebreaks as well as fuel dryness are checked and fires are set after 3 pm, when the wind starts to decrease and relative humidity increases. Moreover, the burn is carried out by a team of physically fit and well-equipped members of the fire brigade, who can quickly respond to escaping fires. Finally, they also checked the area near the burning site the following day to ensure no fires escaped.

While the fire brigade does not use fire as extensively in other regions of Capoto/Jarina as in Piaraçu, they reported being called upon in other villages to assess and sometimes assist with burning. They helped burn a swidden in Mupa in 2023 that was adjacent to a house and conducted a prescribed burn in savanna formations near the border between Capoto/Jarina and external farmers.

Support role for fire use

In Piaraçu, participants from the fire brigade emphasised the importance of their role as advisors on fire management, especially for people who prefer to burn on their own or who live in other villages. The members of the fire brigade are sometimes consulted before someone lights a fire, which provides an opportunity to discuss how and when to burn, visit the site beforehand, or even help them burn if the person is alone or if the fire is considered

dangerous. It also informs the fire brigade about the expected location of fires, so they do not worry about the smoke plume and can quickly intervene in case of escaped fires. They detailed several important recommendations: (i) clearing firebreaks that are at least 3 meters wide, (ii) monitoring fires until they are extinguished, (iii) using water to extinguish fires, especially when outside of the village, (iv) warning neighbours before using fires, to make sure they can help if needed, (v) checking weather forecasts, especially to avoid strong winds, and (vi) avoid burning between 10 am and 4 pm, preferring the late afternoon.

> *"You'll come across someone who's determined to start a fire no matter what. You'll tell them not to, but they might get upset, saying you don't have the authority to tell them what to do. They'll argue that it's their land, their field, and they'll do as they please. But yes, they can do it their way, as long as we're there together to oversee things. To prevent it. That's what I believe, and I think it's really important" Men from Piaraçu and member of the fire brigade*

Fire fighting

The fire brigade has a mandate to fight wildfires and protect important infrastructure and resources. They respond to wildfire events and attempt to contain them through different techniques, such as direct attacks on small fires, the use of counter-fires and the creation of firebreaks. Sometimes, they use fires to clear firebreaks in response to wildfires, a practice that is quicker than manual clearing but technically demanding and riskier. However, logistical challenges have hampered the capacity of the fire brigade to respond to wildfire events (see Section 5.4.3). Firefighting was described as particularly challenging during windy days or when wildfires start to burn in the soils of the seasonally flooded forest, creating smouldering fires.

Individual-level adaptation measures

Across the different villages, people have adapted the seasonality and time of the day when they use fires, started clearing fire breaks around sites for burning and reduced the presence of B*rachiaria spp.* to reduce the risks of wildfires.

Seasonality and time of burning

Across the three villages, people have shifted the seasonality of their use of fires for burning swiddens due to the unpredictability of rainfall during the early rainy season. In Mupa and Piaraçu, participants also used to observe the weather and, on the day they believed the first rainfall would occur, burned their swiddens in the morning. They now prefer to wait until after the first rainfall to ensure there is enough humidity to control the fires and that another rainfall is likely to occur shortly after the burn. Now, they also prefer to burn during the afternoon to benefit from the decreased wind and increased humidity in the evening, allowing better control of the fires.

In Capoto, participants still prefer to burn before the first rainfall but now rely on both local bioindicators and the Gregorian calendar to determine when to burn swiddens. While traditionally they burned swiddens in August after the fall of the flower from the yellow Ipé, they now continue to observe this sign but burn only after the 1st of September. This adjustment helps ensure that rainfall will likely follow soon after which could extinguish potential fires escaping into nearby secondary forests and regenerating fallows. Unlike participants from Piaraçu and Mupa, they continue to burn in the early afternoon.

> *“Now today we follow the calendar, which is yours, the White Man's, and we're getting used to it, we burn the crops in the month of September, but we used to do it in August, our ancestors burned in August, even in the drought they planted and when the first rain falls, it water and the plants come out, now we follow the calendar and count the days, so, usually in the month of September” men from Capoto*

Firebreaks

In all three villages, participants have adopted firebreaks as a standard practice for the clearing of swiddens. In the past, participants across the three sites would just fall trees towards the centre of the plot, thus reducing the fuel load close to the edges and the intensity of the flaming front near the surrounding forest. Nowadays, they deliberately clear a band free of combustible material to help contain fires. The clearing of firebreaks was initially introduced by the fire brigade, but the practice is also spreading outside of their influence: one participant from Piaraçu learned how to clear firebreaks from a person who was never part of the fire brigades.

The fire brigade helps to clear firebreaks around swiddens of inhabitants of the village who are not able to do it themselves. They also clear larger firebreaks around Piaraçu and nearby swiddens, to create several lines of defence helping to control large wildfires. In Piaraçu, the fire brigade borrowed a tractor from the municipality to help clean these large firebreaks, more than 3 meters wide, which are also used as trails by the local community.

The fire brigades also assure quality control of firebreaks in Piaraçu, through inspection of finished firebreaks and advice on the best way to clear them. However, there is no consensus across participants for the ideal width, as larger firebreaks are safer but also require a more significant time commitment. While members of the fire brigade who were interviewed recommend firebreaks to be 3 meters wide, most participants estimate their firebreaks to range between 1 and 2 meters. All participants, including from the fire brigade, also declare that the minimum widths should depend on the fuel load and fire risks specific to the locations.

Sometimes, the fire brigades use fire to quickly clear a firebreak. While being technically demanding and requiring a well-trained and equipped team, this allows clearing firebreaks on all types of grounds, including wetlands which could be challenging otherwise. It is also quicker than manual clearing of firebreaks, which could be important when they are cleared as an emergency response to an incoming wildfire.

Vegetation Management

Across all villages, there are ongoing efforts to control the fuel loads, especially in the areas within and near the village to protect houses and planted trees. In Piaraçu, the fire brigade regularly limits *Brachiaria spp*. growth through controlled burning and mechanical cutting, depending on the seasonality and associated wildfire risk. However, efforts to reduce *Brachiaria spp*. biomass is hampered by the grass' rapid regrowth and the significant workload required for mechanical cutting. Participants from all villages highlighted the importance of cleaning the village and its surroundings to reduce the risks of wildfires associated with the daily use of fires for different tasks such as cooking, burning trash or roasting cassava flour.

During a field walk close to Piaraçu, a key informant pointed to an area of forest undergoing restoration, in a place formerly invaded by *Brachiaria spp*. (Figure 34). The objective was to recover a vegetation type that was not favourable to *Brachiaria spp*. establishment and could be resilient to wildfires. While the process was described as challenging due to pervasive wildfires preventing tree regeneration, the key informant described a steep reduction of wildfire risk after 20 years, when the canopy was restored, fuel structure was modified and humidity increased. The area chosen was considered strategic because it could protect some villages from wildfires coming from the south of Capoto/Jarina.



Figure 34. Secondary forest in the process of restoration to reduce wildfire risks.

Community-level adaptation measures

Elder participants from Piaraçu reported that not only has the community delegated the burning of their swiddens to the fire brigade, but also that certain types of fires, such as those traditionally used on grassland patches for hunting, are now rarely burned. When interrogated about fire for hunting on grassland, no male participants reported having used it on their own around Piaraçu.

> *"We used to set fire to the armadillo hole, the smoke would kill it, and we would take the armadillo to eat. Fire is our survival: without fire, we don't survive, we cook food and use it for warmth, but nowadays fire is a threat to us. With climate change, we have changed a lot, we don't play with fire anymore" elderly man from Piaraçu*

In Mupa, while participants have always tried to contain swidden fires to individual plots, they traditionally burned their swiddens individually. After witnessing several incidents in other villages, they now prefer to burn in teams. Participants from Mupa also have set up signposts in strategic locations frequented by members of other villages who come fishing or hunting, to remind them to extinguish fires afterwards.

In Capoto, there are also increasing efforts to control fires used for clearing swiddens. Male participants stated that in the past they didn't mind if a fire escaped from a swidden plot and burned neighbouring ones, as it could save time for them. Nowadays, they try to have greater control over these fires to reduce the risks of fires escaping and damaging nearby secondary forests and regenerating fallows. However, no significant changes in the way to conduct savanna fires were reported.

5.4.3. Challenges and limitations in adapting to wildfires risks

While inhabitants of Capoto/Jarina have adopted a wide range of strategies to adapt to the emerging wildfire risk on their territories, several limitations and trade-offs of these adaptation strategies have been identified.

Delegation of fire management responsibility

In Piaraçu, there was a consensus amongst all participants that the fire brigade initiative contributed to a reduction in the wildfire risk. This was also shown in the participatory maps, where swiddens managed by the fire brigades were ranked as having lower wildfire risk (Chapter 4, section 4.4.2, Figure 26, location 8) than similar locations without fire brigade action (Chapter 4, section 4.4.2, Figure 26, location 6 and 7). This change in the relationship between inhabitants of Piaraçu and fire management also resulted in a delegation of wildfire risk management to a specific group of people within the community. Both women and elderly participants expressed a lower sense of responsibility for fire and wildfire risk management compared to the past.

"Today we have firefighters, and I want them to work, that's why they are learning these things, for them to burn our fields, for them to take care of the forest, now they burn the fields and now we leave it in their hands" elderly women from Piaraçu

Loss of local fire knowledge

In Piaraçu, a few participants discussed the implications of the changing and decreasing use of fires on the conservation of local fire knowledge. Several elders discussed former fire control practices that were unfamiliar to the younger participants, such as consulting a shaman to know the best time to burn the swiddens, dances and songs that were performed before and during the burn, and the application of "medicine" along the edge of the burning site to prevent fires from escaping.

"And when everything is dry, and there are times when we see that it won't rain, we wait for thunder to make noise, we know it will rain soon. The men prepare torches to burn the fields. There's also a dance that men do before burning the fields, they dance, sing, and then start setting fire to the fields" elderly men from Piaraçu

Most elderly participants claimed that these fire control measures are not appropriate anymore given the current wildfire risk. However, a few participants were concerned that the loss of this knowledge, and more broadly the decrease in fire use, could impact Metyktire cultural heritage where fire holds a central position.

Centralised fire brigade logistics

The centralised model of the fire brigade, based in Piaraçu, could hamper its ability to support local communities in fire use and wildfire risk management in other villages of Capoto/Jarina. During a visit to Tothnhore, a village close to Mupa, a key informant discussed the limited opportunities for people of his village to partake in the training, due to the long distance to Piaraçu and the time commitment it required. He also expressed concern about the fact that no one from the village was part of the fire brigade and could support fire management at a local level, a situation similar to Mupa.

> *“Last year there was brigade training, and nobody informed us here in the village. We still want to participate in the brigade training, and no man from our village is also working with the brigades. So, I wanted women or men from our village to participate in the brigade training to take care of our forest here within our village” men from Tothnhore*

Participants from the fire brigade spoke about the limited funds dedicated to travel and sometimes late hiring in the season, preventing them from dedicating enough time to wildfire risk prevention. During the early dry season, the low number of wildfire incidents allows the fire brigade to dedicate more time to other activities, such as discussing fire management with the local community or clearing firebreaks. But the hiring process can sometimes occur as late as June, and the fire brigade could already need to dedicate time to firefighting.

Participants from Mupa and members of the fire brigades also stated that they often take several hours to arrive at a wildfire event due to the distance between villages. The key informant from Tothnhore recalled a past wildfire incident during which the fire brigade took a long time to intervene. This long delay between the detection of a wildfire and the initial attack was deemed problematic by participants from Piaraçu and Mupa, as they acknowledge that the fire brigade can only effectively fight small fires. To overcome this limitation, participants of Mupa and a key informant from Tothnhore suggested leaving some equipment in the villages and providing basic training, so they could respond quickly to wildfires. Participants from the fire brigades declared that the issues related to their limited number and location in Piaraçu were especially critical in September, as multiple wildfires could burn in different villages across the Capoto/Jarina.

> *“I had told the brigadistas, like, their boss, to send at least five for us, so that we can use it when there's a fire, something, an accident, we can use it. Because since they stay far away from us, by the time we get them, by the time they come here, the fire is already burning, like, it's already going fast, it's already burning a lot of things. A lot of plants, and like what happened now, right, it's already burned our pequi plants and our arrows.” Men from Mupa*

Diverging perspectives around fire use and adaptation to wildfire risk

The diversity of views on fires and wildfire risk management across villages and groups of participants sometimes led to disagreements regarding wildfire risk mitigation strategies and their trade-offs.

Participants from Mupa felt that most of the wildfire risks they experienced originated from other villages, partly due to the use of fires for hunting on patches of grassland along the Xingu, and they advocated for a reduction in the use of these fires. Meanwhile, some of the older participants in Piaraçu were concerned about the reduction in the use of these fires and its impact on Metyktire cultural heritage.

Participants from the fire brigade acknowledged that the use of fires and wildfire risk management differed in Capoto compared to other villages within Capoto/Jarina, due to the different types of vegetation and local cultural use of fires. Participants from Capoto also claimed to use fire and manage wildfires differently than other villages of Capoto/Jarina and generally had a different attitude towards the fire brigades, as some participants declared that they already knew how to manage high wildfire risk.

> *"Now, the region is different, right? What you see there in Piaraçu, that the region is different, the vegetation here is already cerrado, so... If we were to stay in the region that is the forest, we wouldn't use fire anymore. Since we're in the cerrado, we need fire for this benefit, right? Hunting, handicrafts, other materials that you can see, right? The Xingu region is different from us. And it's also a practice that my own people used to do." Men from Capoto*

Participants from Capoto also evoked some tensions with the Prevfogo fire programme, which confused their deliberate use of fires with wildfires. Consultation of the Prevfogo 2024 report corroborates this, as only prescribed fires done by the fire brigade during the early dry season were not considered wildfires (Appendix 4.7).

Participants from the fire brigade also discussed diverging views and sometimes disagreements between them and the local community in Piaraçu. The elderly group interviewed agreed that climate has changed over the last decades and, as a result, they should adapt their use of fires. However, participants from the fire brigade reported several older people were not willing to change their fire use practices compared to the past. Their willingness to burn at the same time of the year as previously could lead to low-quality burn when done too early, but also wildfires burning long before the arrival of rainfall that will extinguish fires.

> *"But for me, I think in this changing scenario, they need to understand what the risk is. It's not that we can prohibit, but they have to understand that now it's different, it can't be the same as before. Because the forest here is thinning out and already decreasing, so you have to understand that it can't be like in the past" Men from Piaraçu and member of the fire brigade*

Disagreements between the fire brigade and some inhabitants of Capoto/Jarina also arose regarding the ideal time of the day for burning. Traditionally in Piaraçu, and still nowadays in Capoto, people prefer to burn during the early afternoon to benefit from strong winds and high temperatures, ensuring good combustion. However, the local fire brigades viewed this as dangerous and ineffective, as the wind can reduce combustion completeness on the plot and could displace the fire outside of the burning site. They instead prefer to burn later in the afternoon and promote this practice. The transition of the fire brigade towards the role of environmental education was described by the participants of all villages as a way to reduce these conflicts, by convincing people to benefit from their support or adopting best burning practices through dialogue.

5.5 Discussion

The results presented highlight the detailed observation of the local community on wildfire risk changes, which were concordant with the quantitative analysis of climate and land use change. In response to these changes in wildfire risks, people are adopting various adaptation strategies, depending on the severity and novelty of wildfires across the three villages. While these strategies allow for responding to emerging wildfire risks, they also present some trade-offs and potential limitations that could limit their applicability and lead to long-term adverse effects.

Participants from all villages monitor local climate indicators to guide decisions about fire use, particularly for the timing of swidden burning, but participants reported that indicators are becoming unreliable. This finding aligns with other studies, which show that local communities across the world observe changes in their climate, especially those related to precipitation regimes and their effects on physical and biological systems (Savo et al. 2016; Reyes-García et al. 2024).

It also highlights how these observations are being integrated into fire management, as participants adapt the seasonality of their fire use to this declining confidence in traditional seasonal and weather indicators: participants from Piaraçu and Mupa now wait until the first rainfall before burning their swiddens, while participants from Capoto combine local phenological knowledge and the Gregorian calendar to decide when to burn their swidden. This adaptive management strategy can support fire management in a dynamic environment by identifying the safest and most effective burning periods despite high uncertainty and year-to-year variability. Seasonal calendars, informed by local fire knowledge, provide a flexible approach to the seasonality of fire use and related activities (Armatas et al. 2016). When local climate indicators prove unreliable (Savo et al. 2016; Bauer et al. 2022), the Gregorian calendar and burning dates based on long-term analysis can serve as important safeguards.

Global Climatic Model analyses highlight the strong influence of the El Niño cycle on seasonal wildfires, but none of the participants mentioned it. This might be partly explained by memory bias affecting perceptions of past precipitation patterns (Arora and Feng 2024). Another potential explanation could be the marked drought during La Niña years in Mato Grosso (Barbieri et al. 2019), a tendency that contrasts with the prevailing media narrative associating El Niño years with severe droughts in the Amazon region. Integrating local fire management practices with local climatic data and predictive models forecasting wildfire risk on short to medium timescale could support decisions about the seasonality of fires in a changing climate (Singh and Srivastava 2024).

Changes in vegetation resulting from past environmental degradation, biological invasion and drying climate also influence local wildfire risk, but its impact differs considerably across villages. In Piaraçu, participants observed the transformation of forests into grass-dominated vegetation, aligning with the result of a burn experiment in the Southern Amazon, which identified strong feedback between post-fire tree mortality, the establishment of herbaceous species, changes in fuel load and fire intensity (Silvério et al. 2013). In Mupa, participants worried about the change in canopy integrity and drying of soils that increase the risks of smouldering fires. Similar intensification of fire regimes due to loss of foliage and smouldering combustion in seasonally flooded forests have been identified by others in the region (De Resende et al. 2014; Balch et al. 2015; Silvério et al. 2022).

In addition to these locally observed changes, participants highlighted several processes occurring outside Capoto/Jarina, such as extensive land use changes and propagation of invasive species, which further compound wildfire risk within their villages. While global climate change contributes to drier conditions, extensive regional land use change intensifies its effects (Dubreuil et al. 2012). The expansion of agricultural land near the border of Capoto/Jarina degrades forests through edge effects, such as increased wind, sunlight penetration or litter accumulation (Broadbent et al. 2008). It also introduces frequent ignition sources, increasing fire escape risks (Cano-Crespo et al. 2015). However, the impact of export-oriented agriculture and pastoralism extends beyond the forests near

the Indigenous land's border: the invasion of *Brachiaria spp.* likely originates from pastures outside Capoto/Jarina and past land invasions.

New elements of wildfire risks identified by inhabitants of Capoto/Jarina are being integrated into local decision-making processes to select appropriate adaptation measures. Due to the specificity of vegetation change around each village, each community has encountered significant differences in the magnitude and characteristics of emerging wildfire risk and has embraced different adaptation strategies. McWethy (2019) proposed a gradient of basic, adaptive and transformative wildfire resilience strategies that can be adopted based on fire severity, novelty and human exposure. In Piaraçu, rapidly intensifying fire regimes driven by biological invasions are severely affecting livelihoods and the safety of inhabitants. In response, the community now delegates the most dangerous burns and wildfire control to the local fire brigade, constituting a transformative change to increase resilience to this novel and severe wildfire risk. In Mupa, which experiences fewer fires with lower impacts on livelihoods, participants have adopted adaptation measures such as stricter fire control to adjust to new climatic conditions and enhance their capacity to respond to external fires. Capoto, which has extensive experience in dealing with wildfire risk and where wildfires have a more modest impact on livelihoods, maintains basic resilience by mobilising local fire knowledge to avoid burning sensitive areas despite changing rainfall seasonality.

It is essential to consider the trade-offs and decisions made to reduce wildfire risks and to assess their potential long-term impact on fire use systems, as changes in livelihood systems can have severe adverse effects (Zant et al. 2023). In Piaraçu, the replacement of the traditional fire use system by the fire brigade system carries important implications for the generation, transmission, and adaptive capacity of local fire knowledge. As the fire brigade primarily recruits young participants, it risks widening the intergenerational divide in fire perception and local fire knowledge. This disconnect may be further reinforced by the integration of the fire brigade into a national programme that, until recently, classified fires in the Brazilian Amazon region as a source of environmental degradation (Sorrensen 2009; Carmenta et al. 2019). Discourses criminalising fire use have been shown to generate

tensions between younger and older generations, disrupting the transmission and development of local fire knowledge (Rodríguez 2007; Carroll et al. 2021). Even when fires are promoted, the institutionalisation of fire management, changes in fire practices and the pursuit of different management objectives can lead to disagreement within the community (Petty et al. 2015).

While the fire brigade does not adopt a punitive approach within the community, the consensus for reducing wildfire risk and complying with guidelines from the fire brigade could discourage community members who continue burning from sharing information about their practices. These issues are common in approaches aiming to control, reduce or suppress fire use, which can result in more secretive use of fires that prevents the sharing of experiences from individual fire use (Johansson et al. 2019; Smith et al. 2024). These dynamics could affect the collectivisation of individual experiences with the use of fires, an essential component for the gradual updating of local fire knowledge in the face of changing environmental conditions (Turner and Berkes 2006).

The fire brigade initiative and its recent professionalisation have also increased the community's reliance on external stakeholders. Wildfire risk management responsibilities have been delegated to a part of the community that depends on external funding. The limited funding of the federal programme for fire brigades has been identified as a constraint on their deployment at a national level (Fonseca-Morello et al. 2017) despite their effectiveness in reducing overall burned areas and mitigating the impacts of climate change on wildfire (Oliveira et al. 2021; Oliveira et al. 2022). An overreliance on the fire brigade risks deepening dependency on federal funding and external support, both of which have proven vulnerable to shifts in federal government policy in recent years (de Area Leão Pereira et al. 2019; Menezes and Barbosa 2021).

Even when funding is maintained, its amount and release schedule affect the operational efficiency of the fire brigade, while logistical challenges inherent to fire management over a large territory remain. This influences the fire brigade's capacity to conduct wildfire prevention, swidden burning, and firefighting. In the context of increasing wildfire risks and systems depending on external funding, the coordination between local preferences and supporting institutions to prioritise wildfire management around key resources is essential for maximising the effectiveness of wildfire risk management strategies (Rutherford and Schultz 2019). The use of participatory maps has proved to be a useful tool for integrating the priorities of different community members directly affected by fire use and wildfire risk management in Capoto/Jarina by indicating the location of valuable resources (see section 4.4.2). Experience from Canada has shown that participatory mapping allows for the inclusion of knowledge of practitioners and local communities regarding valuable infrastructure, natural assets needing protection, and wildfire probability to guide wildfire risk management (Sherry et al. 2019).

5.6 Conclusion

The observation and integration of new elements of wildfire risk related to climate and vegetation change into local fire knowledge illustrates the adaptive capacity of local fire use within Capoto/Jarina. Frequent interactions with the environment enable these communities to monitor changes in key elements of their fire knowledge, such as shifts in fire behaviour, the legacy effects of fires, and the desynchronisation of seasonal indicators with weather conditions. Detailed knowledge of the impacts of regional drivers of wildfire risk highlights the potential contribution of local fire knowledge in informing research agendas and fire management policies at broader scales.

These observations are then integrated into the decision-making process to determine appropriate adaptation measures for reducing wildfire risks, depending on the severity and novelty of the wildfire risk. While the emergence of responses to wildfire risks can push towards transformational changes in fire use and wildfire risk management, there are

potential limitations and adverse effects that need to be considered to maintain the adaptive capacity of fire use and associated local fire knowledge. The use of methodology showcasing local communities' preferences and local fire knowledge could be useful for optimising fire management efforts.

Chapter 6. Discussion

In this final chapter, I will discuss how the results from the different chapters are advancing our understanding of processes driving fire regimes across multiple scales and adaptive pathways to changing wildfire risks. In Chapter 2, I presented the results of a regional analysis of the main drivers of fire regimes in the Brazilian Amazon based on coarse-resolution satellite detection of fires. In Chapter 3, I delved into more detail and presented an analysis of changes in fire risks and fire regimes across different land governance types based on medium-resolution satellite detection of fires. In Chapter 4, I presented the fire use and wildfire risk management strategies of three villages of Capoto/Jarina, before detailing their perceptions and adaptations to changing wildfire risks in Chapter 5. Taken together, these chapters provide insights into the drivers of wildfire risks and fire regimes, as well as local fire knowledge and adaptation strategies to changing wildfire risks. In the following sections, I'll detail how the combined results from the different chapters contribute to answering the research objectives outlined in section 1.3 of the thesis, before discussing the limitations and future research directions that emerge from this work.

6.1 Research gaps

6.2.1 Research objective 1. What are the main drivers of fires regimes across the Brazilian Amazon?

In recent decades, increasing attention has been paid to the drivers of fire regimes in the Brazilian Amazon and their relative contributions. Following the reduction in deforestation rates after 2010, several studies have examined the decoupling between deforestation and fire activity (Aragão and Shimabukuro 2010; Aragão et al. 2018; Morgan et al. 2019). Several mechanisms have been proposed to explain this trend, including the widespread use of maintenance fires on pastures and agricultural lands and their frequent escape into neighbouring forests (Aragão and Shimabukuro 2010; Cano-Crespo et al. 2015), increasing

forest fragmentation and associated edge effects (Armenteras et al. 2013; Silva Junior et al. 2018; Silva-Junior et al. 2022), and the cumulative impacts of environmental degradation and drought events (Alencar et al. 2015; Fonseca et al. 2017; Barni et al. 2021). The recent resurgence in deforestation has coincided with a marked increase in fire activity linked to land clearing (Silveira et al. 2020; Libonati et al. 2021), raising important questions about the extent to which decoupling between deforestation and fire regimes persists. The dynamic and cumulative nature of fire drivers in the region continues to challenge efforts to fully understand the processes shaping fire regimes. Additionally, the frequent conflation of distinct fire types into a single category has contributed to an oversimplified framing of the 'fire problem', limiting the capacity of fire researchers to identify and address the diverse social and ecological drivers of fire in the region (Sorrensen 2009).

My results highlight the close relationship between resource exploitation patterns and fire use within the landscape across scales. In Chapter 2, I combined deforestation and land-use data to classify the MODIS Active-Fires dataset before analysing the drivers of the fire regime, producing the first regional analysis of the drivers of different types of fires. These analyses shed light on the association between land use dynamics and fires: pastoral landscapes still heavily use deforestation fires, while landscapes dominated by cropland rely mostly on agricultural fires (Figures 6 and 7). My results help explain why tractor-lending schemes that propose tractors incapable of clearing vegetation have had limited success in promoting fire-free agriculture in pastoral landscapes (Carmenta et al. 2019). Morello et al. (2017) warned about the spillover effects of efforts to promote mechanised agriculture, which could strengthen the interiorisation of fires observed in my results (Chapter 2). In Chapter 4, the fine-scale fire typology obtained demonstrated how cultural factors such as traditional hunting and non-timber forest product harvesting shape distinct fire use across each village (Table 4). Neighbouring communities were also mobilising different local ecological knowledge to control their fires, corroborating the work of Schmidt et al. (2021) illustrating the diversity of local ecological knowledge used in the neighbouring Xingu Indigenous Park to manage post-fire forest recovery.

Nevertheless, reducing the frequency of agricultural and forest fires might have limited effects on regional fire regimes, as results from Chapters 2, 3 and 5 indicate that forests are becoming increasingly vulnerable to escaped fires. In Chapter 2, all types of fires became increasingly frequent in remote areas and forests further from agricultural land, while Chapter 3 showed a concomitant decrease in the proportions of agricultural land burned and either stable or increasing proportions of forest burned outside indigenous lands (Figure 17). In Chapter 5, participants described more frequent escapes of fires used for swiddens in nearby forests than in the past. This suggests that the proportions of fires escaping from agricultural land to forest, a phenomenon identified by Cano-Crespo et al. (2015), might be increasing and burning over larger areas. These results align with analyses showing a decreasing resilience of the Amazon forest to external disturbances over the last two decades due to the cumulative impact of deforestation, forest degradation and repetitive droughts (Machado-Silva et al. 2021; Boulton et al. 2022). The results from Chapter 4 highlight several mechanisms contributing to this decreasing resilience of the forest that could occur at a regional scale, such as the legacy impact of past wildfires that affect large tracts of forest across Mato Grosso and Pará (Morton et al. 2013), important forest degradation across protected areas and indigenous land in the Amazon basin (Walker et al. 2020) and large areas of the Southern and Central Amazon that have suitable climatic conditions for invasion by *Brachiaria spp.* (Lopes et al. 2023; Mano et al. 2023).

Beyond climatic, ecological and socio-economic factors, governance structures also play a critical role in shaping fire regimes across the Amazon. Land governance types affected fire regimes and the types of challenges encountered by different types of landholders. In Chapter 2, protected areas and indigenous lands were associated with fewer fires of all types (Figures 6, 7 and 8), but their periphery was consistently associated with higher fire rates. This result demonstrated for the first time that the contagion of fires from undesignated land to Indigenous land, a phenomenon previously identified at the state scale (dos Santos et al. 2021; Walker et al. 2022), is also occurring at a regional scale and also affecting protected areas. This contributes to calls for the reinforcement of buffer zones around Indigenous land

to limit the risks of environmental degradation (Rorato et al. 2021). In Chapter 3, the analysis of medium-resolution satellites showed that differences in fire regimes across land governance types in Mato-Grosso go beyond differences in fire frequency: they also exhibited unique patterns of fire size (Figure 14), composition of land use burned (Figure 17), stability over time and seasonality (Figure 13). These results allow us to go beyond the broad-scale analysis of areas with high wildfire risks (Fonseca Morello et al. 2020) and understand both seasonality and potential challenges encountered by the diversity of landholders living in the region to design tailored support initiatives (Oliveras Menor et al. 2025).

This thesis focused mainly on the climatic, environmental and behavioural dimensions of fires. However, broader political and economic factors such as changes in governance and political context at different scales, impact of agricultural commodities prices or international trade agreements, and associated environmental expectations are also important drivers of deforestation and fire regimes (Assunção et al. 2015; West and Fearnside 2021). Bolsonaro's administration's dismantling of environmental regulation and its impact on both deforestation dynamics and fire use shed light on the importance of the broader political context due to its severity and immediate impact (Deutsch and Fletcher 2022). Changes in broader environmental governance in the region and the adoption of different discourses have dictated the locations and extent of environmental degradation that have been shaping fire regimes for the past decades (Pokorny et al. 2021; West and Fearnside 2021). These dynamics are essential to account for today, as the national discourse is moving from environmental regulation dismantling towards the push for agricultural intensification and zero deforestation commitment to continue exportation on the international market (Fuchs et al. 2024). However, there is limited evidence suggesting that it can prevent indirect deforestation in agricultural landscapes oriented towards exportation, especially if there are not enough safeguards, such as protection of Indigenous rights and territories (Goulart et al. 2023; Pratzer et al. 2023).

Taken together, these findings demonstrate that fire regimes in the Brazilian Amazon are shaped by diverse climatic, socio-economic and ecological drivers. The availability of different resources accessible through burning and local fire governance determines patterns of fire use, while environmental degradation and climate alter landscape flammability and associated wildfire risks. My thesis shows how the drivers of fire regime drivers vary spatially and across scales, reinforcing the importance of locally tailored fire management and policy interventions.

6.2.2 Research objective 2. How do the interactions between climatic, ecological and socio-economic factors shape local fire regimes?

At a global scale, climate change is impacting fire weather, resulting in longer and more intense fire seasons (Jolly et al. 2015; Abatzoglou et al. 2019). This increase in fire-prone conditions is especially severe in the Amazon region, where climatic conditions have been identified as an important determinant of burned areas (Jones et al. 2022). Fire regimes in the region are also strongly influenced by anthropogenic activities, notably deforestation, subsequent agricultural land use and the fires associated with these management practices (Jones et al., 2022; Libonati et al., 2021; Cano-Crespo et al., 2015, results from Chapter 2). Human influence on fire regimes operates at multiple levels: regional dynamics influence land use trajectories and associated wildfire risks, while local dynamics affect the motivations and capacity to use and control fires. Important feedback also exists between fire and vegetation: frequent fires, sometimes exacerbated by dry climatic conditions, can lead to shifts in vegetal community composition (Pausas and Keeley 2014; Balch et al. 2015; Archibald et al. 2018).

The instruments used to study the climatic, ecological, and anthropogenic drivers of fire use originate from different disciplinary fields, which traditionally work at distinct spatial scales

(Bowman et al. 2013). However, understanding the interactions between these factors operating at various scales is essential for supporting sound fire management in a changing climate (Bowman et al. 2013; Oliveras Menor et al. 2025). The cultural and ecological diversity of the Brazilian Amazon offers opportunities to deepen our comprehension of these interactions in tropical rainforest landscapes.

Analysis conducted at a sub-regional level (Chapter 3) demonstrated that different land governance types are associated with distinct wildfire risks. While all types of land governance experienced a similar increase in wildfire risks due to climate change (Figure 10), there were significant differences in land use composition and forest fragmentation (Figures 11 and 12). The results build on the land use literature, which had already identified higher deforestation rates and pasture prevalence in rural settlements and higher forest cover in indigenous lands (Herrera et al. 2019; Yanai et al. 2020; Alves et al. 2021). The analysis also examined the impact of forest fragmentation, an essential driver of vulnerability to wildfire in the region (Armenteras et al. 2013; Silva-Junior et al. 2022). Local land governance and types of landholders can substantially influence forest fragmentation metrics, even for similar deforestation rates (de Filho and Metzger 2006; Alves et al. 2021).

Different land governance categories are not only associated with unique landscape configurations but also with distinct fire regimes and specific challenges during drought years. Analyses at a regional level (Chapter 2) and sub-regional level (Chapter 3) indicated that rural settlements were associated with fewer fires on pastoral land over time, but a similar rate of forest fires. Moreover, drought conditions in rural settlements and undesignated lands were associated with a higher proportion of land burned in large fires, exceeding the typical size of controlled fires (Figures 20 and 21). Cammelli and Angelsen (2019) showed that non-indigenous landholders were less likely to invest efforts in controlled fires during drought events due to the perceived risks of a collective failure to control wildfires risks, which might result in large wildfires burning through a mix of pastures and neighbouring forest (Cammelli et al. 2019).

Results from analyses at a regional level (Chapter 2) showed that Indigenous land was associated with the highest reduction in deforestation fires and agriculture fires (Figures 6 and 7) but was outperformed by integral protection areas for forest fires (Figure 8). This aligns with existing literature demonstrating the effectiveness of Indigenous lands in deterring deforestation (Nepstad et al. 2006; Jusys 2018), but persistent issues due to frequent fire contagion from surrounding areas (Rorato et al. 2021; Santos et al. 2021; Walker et al. 2022). Results from analyses at a sub-regional level (Chapter 3) and local level (Chapters 4 and 5) indicated that indigenous land could also face different challenges during drought years. Contrary to rural settlements and undesignated land, drought years were associated not with larger fires, but with a higher proportion of burned land covered by forest (Figure 22). Modelling studies have shown that large tracts of forest can become flammable during particularly drought years (Soares-Filho et al. 2012; Brando et al. 2020). The results obtained from local interviews confirmed that, while usually the humidity of the forest is sufficient to prevent wildfires from spreading in neighbouring forests, dry climatic conditions can make these forests flammable and result in wildfires spreading over large areas of unfragmented forest.

To complement these broader-scale patterns, analyses at a local scale (Chapter 4) revealed how local fire use practices, associated knowledge, and the diversity of vegetation types further shape fire regimes. While understanding these details is critical for effective and fair fire management policies (Mistry and Bizerril 2011; Smith et al. 2024), they are missing for the other analysis (within Chapter 2 and 3). Across the three villages, the decision to burn swiddens was based on the estimation of the short-term likelihood of rainfall that could extinguish escaped fires and ensure good fertilisation of the soils but relied on different seasonal indicators. As different seasonal indicators have distinct sensitivity to changes in climate, this might impact the potential to use local phenological knowledge to manage fire (Armatas et al. 2016). Results from participatory maps also highlighted the close interactions between vegetation types and wildfire risks, identifying some vegetation types that are highly sensitive to fires (Figures 26,27 and 28). Participatory methods have shown

the potential to develop a rich typology of vegetation types including key information from local ecological knowledge systems, such as the value of certain species or their resistance to fires (de Robert et al. 2012; Schmidt et al. 2021; Böhlen et al. 2022). Such typologies could help to develop locally relevant wildfire risk mapping strategies, integrating local knowledge on fire-vegetation feedback and informing post-fire recovery strategies, all important elements of the Integrated Fire Management paradigm (Schmidt et al. 2021; Oliveras Menor et al. 2025). They could also inform the regional research agenda by supporting the understanding of fire-vegetation feedback occurring at a larger scale based on plant trait distributions (Archibald et al. 2018; Brondízio et al. 2021).

The difference in fire regimes across land governance types is symptomatic of the wider impact of colonial history, conflicts around land tenure, access to capital and support from the state across different actors in the region. Local Ecological Knowledge, on which Indigenous communities rely to manage their territories, has been impacted by forced relocations within Brazil and their experience with different ecosystems (Athayde and Silva-Lugo 2018). Moreover, traditional fire management in some patches of fire-dependent landscapes has been disrupted due to past land invasions, a situation analogous to the current exclusion of the Metyktire and Juruna communities from Kapot-Nhinore, which has led to an increase in large wildfires (Welch et al. 2013). Fires have also been associated with land invasions in protected areas and Indigenous lands, especially along their borders (de Oliveira et al. 2020; Walker et al. 2020). Some of the emerging drivers of wildfires identified, such as the presence of brachiaria grasses in Capoto/Jarina, stem from land use practices occurring outside these areas and possibly from past land invasions. Deforestation rates and associated use of fires are also greater in areas that have experienced land conflicts and the eviction of large landholders, due to the need to demonstrate productive use of lands and to secure support from governmental institutions (Aldrich et al. 2020). The strength of environmental regulation is pivotal in controlling fire regimes, and the environmental dismantling undertaken by Bolsonaro's administration coincided with a surge in fire activity in the contested region of the Brazilian Amazon (Jakimow et al. 2023).

Collectively, these insights show that complex interactions between climatic, ecological, and socio-economic factors shape both local wildfire risk and fire use practices. This cross-scalar perspective highlights how regional land use patterns, local governance, forest cover, and local fire knowledge interact with climate-driven fire risk. Integrating local fire knowledge and its impact on fire practices could enrich our understanding of fire-vegetation-climate feedback and contribute to a more detailed understanding of pyro-geography.

6.2.3 Research objective 3. How do different types of landholders of the Brazilian Amazon adapt to changing wildfire risks?

Climate change has already extended the length and increased the intensity of the fire season across the Amazon, heightening the region's vulnerability to wildfires (Jones et al. 2022). The impact of climate change exacerbates a general loss of resilience in the Amazon forest driven by the cumulative effects of past droughts, logging, fires and deforestation (Davidson et al. 2012; Boulton et al. 2022). Models of future fire regimes predict increases in the frequency, severity and seasonal shifts of fires in the region, including within forested areas and regions previously untouched by fires (Silvestrini et al. 2011; Fonseca et al. 2019; Brando et al. 2020). This increase in frequency and extent of wildfires will be particularly severe if deforestation trends are not controlled (Fonseca et al. 2019; Brando et al. 2020).

The rise of increasingly frequent and destructive fires in South America has led to questions about the effectiveness of management approaches focused on fire suppression and criminalisation (Carmenta et al. 2019; Garcia et al. 2021; Armenteras and De La Barrera 2023). The Integrated Fire Management approach, which relies on proactive wildfire risk management grounded in local contexts, has been proposed as an alternative paradigm to adapt to changing fire regimes (Oliveras Menor et al. 2025). In South America, this has notably materialised through efforts to better integrate local fire knowledge and local

perspectives on fire management (Eloy et al. 2019; Mistry et al. 2019). However, there is still little empirical evidence examining the pathways through which local communities perceive and adapt to rapidly changing wildfire risks or capturing the diversity of adaptation pathways that could be adopted across ecologically and culturally diverse landscapes.

The results of analyses across all scales show that wildfire risks and fire regimes are changing across the Brazilian Amazon. At a regional level (Chapter 2), I demonstrated how fires are increasingly frequent in remote areas and forests further away from agricultural lands (Figures 6, 7 and 8). This interiorisation of the fire regime in the region is concerning, as there are many uncertainties regarding the resilience of newly exposed forests to fires that have historically lacked selective pressure favouring the abundance of fire-resistant trees (Barlow and Peres 2008; Balch et al. 2011).

At a local level (Chapter 5), participants describe how changes in climate and vegetation have increased wildfire risks in Capoto/Jarina. While there is a growing body of literature showing how indigenous and local communities observe climate change and its impacts (Savo et al. 2016; Petzold et al. 2020; Reyes-García et al. 2024), this study represents the first evidence demonstrating how they integrate their observation into adaptation strategies to wildfire risk. These observations of climate change allowed us to go beyond the identification of trends in fire weather and assess their impact on physical and biological systems, a common characteristic of local observations of global climatic change (Savo et al. 2016; Reyes-García et al. 2024). This highlights the importance of integrating local ecological knowledge into analyses of changing fire regimes, as it enables us to understand specific changes occurring in physical and biological components of the local fire system (Bowman, O'Brien & Goldammer, 2013). Participants also highlighted the important role of *Brachiaria spp.* in one of the communities that fundamentally change wildfire risks, and whose appearance was favoured by past fires. In an ecologically complex landscape, characterised by a high floristic diversity and associated feedback between fire, vegetation and climate (Archibald et al. 2018), local ecological knowledge could play a crucial role in adapting to changing fire regimes. The incremental nature of local ecological knowledge and

the rapid integration of new observations into decision-making may be pivotal for adaptation in fast-changing environments (Turner and Berkes 2006; Folke et al. 2009).

At all levels, I identified not only changes in wildfire risks but also various adaptation strategies to alleviate the impact of wildfires. At the regional level, analysis indicated a constant rate of fires across Indigenous land and a decrease in deforestation and agricultural fires in rural settlements (Figures 6, 7 and 8). These results were corroborated by analysis at a sub-regional level, which indicated that fire regimes across Indigenous land remained stable despite changes in climatic conditions. Despite experiencing the strongest increase in wildfire risk due to land use change and forest fragmentation, rural settlements recorded the steepest reduction in their overall burned areas while undesignated land experienced a more modest decline (Figure 13). While it wasn't possible to establish causal mechanisms, results suggest that small scale settlers in rural settlements have adapted by reducing the use of fires on pastures and croplands. A potential explanation could be the existence of shared norms around fire management and high social capital that favour the emergence of collective fire management (Roos et al. 2016; Cammelli et al. 2019; Morello and Falcão 2020).

This thesis also provided the first empirical evidence of how Indigenous communities in the Brazilian Amazon adapt to changing wildfire risks through individual and collective adaptation measures. While the three villages experienced a similar shift in climate, they encountered distinct changes in surrounding vegetation that affected wildfire risks to varying degrees (Figures 31, 32 and 33). The magnitude and novelty of wildfire risks, as well as the opportunity to access different support systems, shaped the adaptation strategies adopted in each community. These adaptation strategies ranged from minor changes to adapt to a changing climate in Capoto to a comprehensive transformation of the fire governance system in Piaraçu, in response to an abrupt increase in the flammability of the landscape. Members of all villages visited have secure land tenure, high social capital and shared norms around fire management, important characteristics supporting collective fire management identified by Morello & Falcão (2020). This appears to favour the adoption of certain costly

fire control practices, such as waiting for the first rainfall to burn swiddens, clearing firebreaks to prevent fire escape or burning in teams.

The fire brigade played an important role in adaptation to increasing wildfire risks across Capoto/Jarina, but its role was flexible depending on the resources available and the needs of local communities. While Indigenous communities are increasingly integrated into national programs for fire management, it remains critical to ensure sufficient decision-making power is retained at the local level to prevent issues of scale mismatch, as shown by divergent perspectives on savanna burning between the local community and Prevfogo annual report (Cumming et al. 2006; Ray et al. 2012). Efforts to institutionalise fire management and misalignments between the objectives of local communities and agencies overseeing wildfire risk management could lead to internal tensions and limited sharing of information (Eriksen and Hankins 2014; Petty et al. 2015). While evidence of the deleterious impacts of wildfires adaptation strategies is still limited, a review of adaptation of livelihood strategies to changing climate showed that transformative changes are frequent, despite the risks of maladaptation and adverse effects (Zant et al. 2023). Several aspects of the adaptation strategy adopted in Piaraçu such as the reduction of the number of fire users, disruption of some cultural practices and increased exposure to fire management practices developed in a different ecosystem, align with the drivers of erosion of local ecological knowledge identified for the management of other resources (Reyes-García et al. 2014; Fernández-Llamazares et al. 2015; Reyes-García et al. 2016). The long-term impacts of adaptation strategies on local fire knowledge need to be carefully considered to maintain the adaptive capacity of the community, given their importance in enabling communities to perceive and adapt to changing wildfire risks.

The cross-scalar analyses demonstrated the importance of regional drivers of wildfire risks linked to agricultural development outside Capoto/Jarina (Figures 29 and 30), which raises important questions from an environmental justice perspective. While past wildfire crises in the Amazon have been partly blamed on indigenous communities and NGOs supporting them (Hecht 2020), many of the drivers increasing wildfire risks, and thus making some

traditional fire practices more hazardous, originate from processes that benefit the agribusiness sector. Although recent legal developments in Brazil represent an important advancement in the recognition of traditional fire use and associated knowledge, Indigenous communities still bear a disproportionate burden from changes in wildfire risks emanating from processes benefiting other stakeholders. Increasing fire risks are undermining their ability to meet basic livelihood and cultural needs, while adaptation strategies come at a considerable cost. The main external support they receive is a fire brigade, which only became professionalised in recent years and remains underfunded relative to the size of the Indigenous land, a situation common to many fire brigades across the Brazilian Amazon (Fonseca-Morello et al. 2017). These questions about cross-scale drivers of wildfire risk, and the unequal distribution of the costs and benefits of fire, are also relevant to other smallholder communities across the tropics (Carmenta et al. 2021). They are especially urgent given that the spread of invasive grass species is expected to affect an increasing number of Indigenous lands across the Amazon and lead to other drastic shifts in wildfire risks (De Faria et al. 2021; Lopes et al. 2023; Mano et al. 2023).

Taken together, these findings demonstrate that wildfire risks across the Brazilian Amazon are changing rapidly, driven by the cumulative impacts of climate change, forest degradation, and biological invasions. This thesis set out to examine how local different types of landholders perceive and respond to changes in wildfire risks occurring at multiple scales. The results show that communities are mobilising local fire knowledge to develop a wide range of adaptation strategies, shaped both by community-specific characteristics and the nature and novelty of wildfire risks they face. While these strategies appear effective in mitigating some emerging wildfire risks, uncertainties remain regarding their long-term impact on local fire knowledge and the overall adaptive capacity of these communities. Addressing these uncertainties will be crucial to support resilient fire management in the face of continuing environmental change.

6.2.4 Research objective 4. How could cross-scalar and inter-disciplinary research improve our understanding of fires regimes, change in wildfires risks and adaptation pathways?

While controlled fires could represent as much as half of the total burned areas annually, human use of fires and their influence on global fire regimes have remained underrepresented in global fire models (Perkins et al. 2025). The complexity and accuracy of the representation of anthropic influence on fire regimes depend on methodological choices and the scale of analysis, with a greater level of detail offered by local studies that can integrate place-based contextual data (Carmenta et al. 2011; Ford et al. 2021). However, an analysis of tropical forest fire research reveals that the remote-sensing approach is the most frequently employed, especially in Brazil, resulting in biases in both the attribution of wildfire causes and the identification of solutions (Carmenta et al. 2011). Remote sensing continues to be preferred by policymakers due to its perceived credibility, replicability and generality of research outputs (Harwell 2000; Sorrensen 2009), despite studies highlighting consistent biases in fire detection of currently used remote-sensing products (Giglio 2007; Ferro et al. 2024). These limitations reinforce the importance of integrating ecological, climatic and socio-economic processes to understand changing fire regimes and designing fair and effective support measures (Carmenta et al. 2011; Bowman et al. 2013). Nevertheless, this integration requires overcoming the persistent challenges of reconciling data from diverse disciplines and scales (Moore 2019; Shuman et al. 2022).

	Chapter 2	Chapter 3	Chapter 4 and 5
Methods	Modelling of drivers of fire occurrence – regional level	Statistical analysis of wildfire risks and fire regimes – subregional	Mixed-methods analysis of wildfire risks and fire regimes - community
Spatial scale	Biome	State	Community
Diversity of fire types	Three main types of fires, based on land use and land use change in 1-kilometre grid-cell	Burned areas based on the land use in the same year at a 30-metre grid-cell	Typology based on fire objective and including details such as seasonality, preparation of sites, fire control measures, time of the day and people doing the burn
Land governance	Main categories and influence of proximity of Indigenous land border, associated dynamics of occurrence for different types of fires	Wildfires risks related to land use change and forest fragmentation Fire regimes including fire size, land use burned, seasonality, and inter-annual trends	Detailed information on local fire governance arrangements and impact on fire regimes
Infrastructure development	Occurrence of main types of fires in areas with different transport costs across the Amazon	Not evaluated	Impact of the road and associated land use dynamics on local wildfire risk
Land use	Broad classification of agricultural land use and impact on occurrence of different fire types	Classification of land use and their trends, including more detailed classification of natural land use than Chapter 2	Granular classifications of different vegetation types according to their sensitivity to wildfires, based on local fire knowledge
Climate	Unique variable estimating the drought intensity on each grid-cells	Four metrics derived from the Fire Weather Index serving as proxies for fire season intensity and seasonality	Details information on seasonality of rainfall, its change, and impact on different elements of fire use systems such as associated seasonal indicators.
Forest degradation	Unique variable estimating forest fragmentation and distance from agricultural land for each grid cell	Four forest fragmentation metrics assessing different landscape characteristics	Impact of past fires, proximity from biological invasion and changing climatic conditions on Forest and their flammability
Biological invasion	Not possible to segregate from the impact of pastoralism	Can see the extent in Indigenous land, but low percentage. Not possible to segregate from the impact of pastoralism outside Indigenous land	Identification of two invasive species, including one unknown to science, and their impact on the wildfire risks in the landscape. Integration with traditional fire use

Table 5. Resuming the representation of different processes influencing fire regimes obtained through different methods and scales of analysis through the thesis

The results of this thesis demonstrate how the integration of results obtained across different research methods and scales can enrich our understanding of the impact of changing climate on fire regimes (Table 5). In Chapter 2, data from satellite imagery enabled the evaluation of the influence of past climatic conditions on different types of fires (Figures 6, 7 and 8), while in Chapter 3 data from the global climatic model allowed the assessment of changes in climate-related wildfires risk in North of Mato Grosso (Figure 10). These broad analyses at a regional to subregional level can not only help track climate change patterns but also inform the prioritisation of resource allocation for mitigating the impact of climate on wildfires by highlighting areas experiencing quicker changes. In Chapter 5, participants' observations corroborated the results from the Global Climate Model analysis (Figure 29). Yet, participatory methods additionally revealed the impacts of observed climate changes on other elements of local fire use systems, such as the seasonality of bio-indicators and soil moisture. This information, linking changes in climate to their impact on the physical and biological systems, is essential to understanding the complex feedback between social and biophysical systems. However, the fact that the El Niño cycle was not discussed by participants (Chapter 5) despite its role in inter-annual variations in wildfire risk shows the potential role of global climatic models in forecasting annual wildfires risks and supporting decisions about when to burn.

The details obtained about the impact of forest degradation and biological invasions on fire regimes further illustrate the value of cross-scalar analysis of wildfire risk. In Chapter 2, forest fragmentation was represented by a single variable, but the results indicating an increased occurrence of fires in forests further away from agricultural land over time have important implications (Figure 8). In Chapter 3, I used several forest fragmentation indices, capturing different landscape characteristics, to detect potential divergences that might reveal distinct fragmentation patterns across land governance types (Figure 12). This analysis highlighted the types of land governance where forest fragmentation may be a significant issue, such as rural settlements, and those where it is not, such as Indigenous land. Despite the absence of significant forest fragmentation, Chapter 5 identified several

types of forest degradation affecting wildfire risk, including post-fire tree mortality and the drying of seasonally flooded forest soils. Results from Chapters 4 and 5 also documented the presence of two invasive species, including one for which no existing literature could be found linking it to fire risk. The identification of *Brachiaria spp.* as a major local wildfire driver enabled the interpretation of land use change results, which revealed the presence of pasture in many Indigenous lands. While the insight from Piaraçu could help to identify other Indigenous villages with a significant cover of *Brachiaria spp.*, aggregated analysis done in Chapter 3 identified only a small proportion of *Brachiaria spp.* grassland in Indigenous lands of Mato Grosso (Figure 11), showing that locally significant drivers of wildfire risk could be overlooked during analysis at a broader scale.

The granularity of information obtained on fire use across the chapters clearly illustrates the trade-offs between local relevance and the generalisation of results derived from different methods. In Chapter 2, the use of coarse-resolution Active-Fire products provided limited insight into the precise type of land burned and the objectives of different fires (Figure 4). By contrast, the use of medium resolution burned area products in Chapter 3 allowed us to link fire events to specific land uses and fire sizes (Figures 14 and 17). This helped clarify some of the findings from Chapter 2: while frequent agricultural fires were detected in areas with annual crops (Figure 7), annual crops accounted for only a small proportion of burned areas except in 2022 (Figure 17). This discrepancy between the two products' results could be explained by the active management of pastures and natural vegetation surrounding annual cropland, generating high fire activity within 1 km pixels. In Chapter 4, I presented a more detailed typology of fires based on local fire uses and the resources accessible to each (Table 4). Among the nine fire types identified, six are likely undetected by satellite, either due to the time of day they are lit or the small scale of the fires, showing potential important detection bias. Nevertheless, some of these fires, such as those for honey collection, roasting fish, or burning trash, contribute to shaping local wildfire risk (Figure 3).

Cross-scalar analysis and the use of different methodologies also support the development of fair mitigation strategies by integrating local preferences and priorities in wildfire risk management. The identification of climate change and *Brachiaria spp.* as major drivers of wildfire risks in Chapters 4 and 5 show that a major increase in their wildfire risk originates from exogenous processes. Meanwhile, the identification of the consistently higher occurrence of fires in areas close to the borders of Indigenous lands and integral protection areas identified in Chapter 2 suggests a spread of fires across Indigenous land borders (Figure 6, 7 and 8). This also helps to explain the rationale behind different uses of fire, which can vary at a fine scale: while controlled fires typically occur over areas smaller than a few hectares in Piaraçu and Mupa, participants from Capoto use large fires over grassland and savanna to manage the landscape and, among other objectives, reduce wildfire risk (Figure 28). Integration of these results in the region or biome-scale analysis can highlight the potential ecological benefits of some fires in savanna vegetation, quite frequent across the Cerrado-Amazon transition zone. More broadly, participatory methods and the identification of the objectives behind different fire uses could enhance the integration of fire management policies with local livelihood strategies. Participatory mapping and complementary interviews in Chapter 4 enabled us to understand the damages of wildfires, and their impact on the broader livelihoods' strategies, but also identify the location of different important and fire-vulnerable resources that could not be identified by remote-sensing, such as the location of different species of trees. This could help to prioritise the coordination between the needs of the local community and the fire brigade.

This thesis illustrates the value of cross-scalar and interdisciplinary approaches in capturing the complexity of fire regimes, shifting wildfire risks, and local adaptation strategies (Table 5). The results expose persistent biases in fire detection and oversimplification of complex processes when conducting analysis at a regional scale and relying solely remote-sensing and climatic data. Integrating data from local case studies and collected with participatory methods helps to understand how large-scale drivers, such as climate change and forest degradation, intersect with fine-scale, place-based fire uses and management practices.

The integration of local evidence on fire use also highlighted the diverse rationales behind fire practices and uncovered previously undetected fire types and vulnerable resources beyond the reach of satellite-based monitoring. These findings reinforce the need to reconcile broad-scale risk assessments with local fire knowledge to build more just, inclusive, and effective wildfire management strategies.

Modelling of fire regimes at large scales tends to disregard the impact of fire–vegetation feedback, despite the importance of these mechanisms for shaping fire regimes (Archibald et al. 2018). Given the rapid pace of climate change and the potential for abrupt feedback between vegetation and fire (Keeley et al. 2011), local fire knowledge, grounded in frequent landscape observations and incremental learning (Turner and Berkes 2006), could play a pivotal role in improving our understanding of shifting wildfire risks. Insights from local case studies, such as the influence of specific feedback mechanisms and biological invasions, can help set research priorities, for instance, by identifying problematic invasive species and evaluating appropriate management strategies. Multi-scalar, mixed-methods research also enables the analysis of exogenous wildfire risks, including the impacts of regional land-use change and climate trends on specific local contexts. Findings from Capoto/Jarina, for example, raise important questions about the responsibilities of different stakeholders in driving wildfire risk and about who should bear the costs of adaptation measures (Carmenta et al. 2021). Moreover, local case studies and participatory approaches offer opportunities to engage communities in decision-making for integrated fire management, an essential dimension of environmental justice (Schinko et al. 2023) and a critical factor for the success of fire policy interventions (Smith et al. 2024). In my case study, participants from Capoto demonstrated how their fire practices actively contributed to wildfire risk reduction, with minimal risk of fires escaping into neighbouring forests, a finding consistent with broader-scale analyses (Da Ponte et al. 2023).

6.3 Limitations

Several constraints shaped the research conducted during this doctoral thesis. The most significant constraints were limited time and resources, which determined the duration of the fieldwork. The political context creating challenging conditions to conduct fieldwork safely in the Brazilian Amazon around fire management as well as the COVID-19 pandemic prevented the preparation and execution of fieldwork during the first two years of this thesis, while the subsequent process of obtaining research permits also required additional time. Combined with the high costs of travelling to Capoto/Jarina, and a deliberate effort to avoid participant research fatigue, these constraints restricted the possibility of extended or repeated visits.

Collaborating with the Raoni Institute allowed us to quickly organise research activities to explore local fire use and associated knowledge, but longer engagement between the research team and participants would have enabled the exploration of complementary facets of fire use systems. While we initially intended to explore how participants decided when to burn and how this relates to other livelihoods and cultural objectives, this would have likely required participant observation over an entire fire season. Consequently, I developed this aspect of the study to a lesser extent than initially intended and focused only on the general seasonality of fire use and its relationship to wildfire risk. Similarly, some of the cultural aspects of fire use, particularly its connection to the cosmology of different peoples living in Capoto/Jarina and the impact of adaptation measures on these cultural aspects, would have needed longer-term engagement. Finally, both the introduction to the communities through the Raoni Institute and the limited timeframe prevented a more in-depth exploration of the impacts of adaptation to changing wildfire risks on local communities, as well as disagreements surrounding adaptation measures. The introduction within the community by the Raoni Institute helped overcome certain barriers to participation by creating the expectation that research findings would be returned to and used by the community. However, this could have prevented some members of the

community from disagreeing with adaptation strategies supported by the Raoni Institute to engage in the research activity.

Beyond logistical and temporal constraints, the spatial scope of this research also presents a limitation. I conducted fieldwork exclusively within an Indigenous territory with few issues of land invasions, despite identifying a higher frequency of fires close to Indigenous land borders through remote-sensing data. It is probable that the high political power of the Indigenous people of Capoto/Jarina, history of conflicts with outsiders and organization of frequent monitoring missions deter land invasions more effectively than other Indigenous lands of the region. This may be why there are more significant concerns about endogenous risks of wildfires compared to Indigenous land experiencing frequent land invasions and intrusions from outsiders.

I also did fieldwork only in an Indigenous land territory despite identifying notable changes in fire regimes across undesignated lands and rural settlements. While most of the forest remains in protected areas and Indigenous lands, most of the fire activity is concentrated within rural settlements and undesignated lands. Fire regimes, and more broadly land management practices, in these areas are likely vastly different from those within Indigenous lands and could represent distinct pyric phases, as defined by Bowman et al. (2012). However, I was not successful in establishing relationships with local partners who could have provided opportunities to conduct fieldwork in undesignated lands or rural settlements. Establishing such connections is quite difficult, both due to the prominence of fires in international debates surrounding Amazon deforestation in recent years, and the illegality of most fires used in undesignated land and rural settlements. This focus on fire management within Indigenous land reflects the nature of the evidence base on which Integrated Fire Management policies in Brazil have been developed, which has drawn almost exclusively on insights from Indigenous lands and protected areas.

6.4 Future Research Directions

There are opportunities to build on the research of this doctoral thesis and continue to explore different pathways to the adaptation for increasing wildfire risks in the Brazilian Amazon.

The first research direction that I would like to explore is the use of participatory GIS to support the action of the local fire brigade of Capoto/Jarina. Mapping encompasses various approaches, ranging from less structured methods like freehand mapping used in this study to more structured techniques such as participatory GIS that rely on digital formats (Denwood et al. 2022). Using medium- to high-resolution satellite imagery for participatory mapping offers several advantages, such as greater credibility with external stakeholders and better reusability of produced maps for guiding different actions (Eilola et al. 2021). While I chose to do free-hand mapping for the early stage of my study to encourage the participation of all community members, I could now progress to a second round of participatory maps relying on a GIS system or printed-out maps. This could both improve the usability for the local fire brigade of these maps for planning wildfire risk management actions and during emergency response to wildfires, as well as make a stronger case for the complexity and diversity of Indigenous local fire knowledge, thereby contributing to their recognition in political processes. Lessons learned from the participatory GIS experience in Capoto/Jarina could support the establishment of a replicable methodology supporting local diagnosis of wildfire risks and management priorities, an important aspect of Integrated Fire Management (Oliveras Menor et al. 2025).

A second research direction would be to explore the impact of different adaptation measures for increasing wildfire risks on the capacity of local communities to maintain their local fire knowledge and associated adaptive capacity. The extent of adaptation measures varied across villages, depending on wildfire risk severity and novelty. Paradoxically, transformative changes in livelihood systems to adapt to climate changes are more likely to result in unexpected and deleterious impacts (Petzold et al. 2020). However, the potential adverse effect of adaptation strategies to increasing wildfire risks has not been extensively

investigated. I propose using the system viability framework (Berardi et al. 2015) presented in the conceptual framework (section 1.4) to understand relative investment in different community survival strategies. This could highlight trade-offs adopted by communities between different elements of their fire-use systems and unveil some of the potential deleterious impacts on their fire-use systems and associated livelihoods and cultural objectives. These results would contribute to the literature looking at the role of local ecological knowledge in the adaptation to global environmental change (Folke et al. 2009; Fernández-Llamazares et al. 2015; Fernández-Llamazares et al. 2021). It could also inform on the governance of adaptations to growing wildfire risks and help identify members of the community likely to bear the higher cost of adaptation strategies, contributing to the literature on adaptive governance (Chaffin et al. 2014). Preliminary results showed that these differences seem to exist across gender and ethnicity, but they should be consolidated with further data collection and longer engagement. Moreover, it would be important to explore the diversity of differences across complementary dimensions, such as age, the capacity of each village to engage with externally supported initiatives and political power within the communities, to see the distributional justice of adaptation efforts.

A final research direction could be to delve into the adaptation strategies of landholders in undesignated land and rural settlements. Ideally, this could be explored through fieldwork using a similar methodology to that used in Chapter 4 and Chapter 5, as it would allow a comparative analysis of local fire use, associated knowledge, and adaptation strategies. Recent legislation on Integrated Fire Management in Brazil legalised some fire practices in these areas and created an environment more conducive to collaboration between agricultural producers and researchers. It would also be an opportunity to explore the influence of historical and political processes that have affected both direct and indirect drivers of wildfire regimes. In the state of Mato-Grosso, which has been a major landscape of production of agricultural commodities for global trade, it could be crucial to examine the link between the access to capital across different types of landholders, adoption of sustainable agriculture discourse and support by different governmental institutions. In the

absence of fieldwork opportunities, the concurrent emergence of medium-resolution burned area products and the increased rate of land registration in Brazil offers new possibilities for exploring the behaviour of different types of landholders. Based on land use trajectories, I could characterize individual fire events and assess their relative dynamics across different landholder types and rural settlements, before analysing the determinants of increases or reductions in different types of fires. This could help tailor fire management policies and support measures designed for different landholders in the Brazilian Amazon, improving the policy support and integration of landscape planning and fire management process, an important aspect of Integrated Fire Management (Oliveras Menor et al. 2025).

6.5 Summary

In summary, the results presented in this thesis show that regional changes in climate, land use patterns, forest degradation, and biological invasions interact to shape local fire regimes. Fires are already becoming increasingly frequent in remote parts of the region and distant forests, but the network of protected areas and indigenous land continue to protect the remaining forests from all types of fires. Land governance types have a strong influence on fire regimes due to their impact on land use patterns and associated wildfire risks, as well as on the livelihoods and cultural objectives of local landholders and their associated use of fire. In the North of Mato Grosso, landholders of rural settlements and Indigenous lands demonstrated adaptive capacities in the face of changing wildfire risks, respectively decreasing or maintaining the low intensity of their fire. In Capoto/Jarina, extensive and diverse local fire knowledge supports fire use, allowing communities to exploit different resources and manage large tracts of fire-prone vegetation while minimizing wildfire risks. This extensive knowledge has also enabled them to identify changes in climate and vegetation that affected wildfire risk. Across all communities, individual and collective adaptation measures were adopted to address this increasing wildfire risk. The nature and extent of these adaptation measures depended on the severity and novelty of emerging wildfire risks as well as opportunities for adaptation for each community. Collectively, my thesis results shed light on the complexity of the interplay between ecological, climatic, and

socio-economic factors shaping fire regimes and adaptive capacities to wildfire risks. The use of different methodologies and analysis across scales was essential to capture this diversity of factors. The use of participatory methods allowed us to unveil some important aspects of local fire use that are difficult to capture using a remote-sensing approach and unveil quickly changing interactions between vegetation and fires. They also highlighted the diversity of fire uses knowledge existing across neighbouring villages, shedding light on the risks of scale mismatch when designing policy at a conservation unit or Indigenous land level. Overall, our results highlight the importance of cross-scalar and interdisciplinary research approaches to integrate insights from different knowledge sources in a rapidly changing world and ensure that adaptation measures supported today do not compromise the future adaptive potential of local communities.

Sumário

Em resumo, os resultados apresentados nesta tese mostram que as mudanças regionais no clima, os padrões de uso da terra, a degradação florestal e as invasões biológicas interagem para moldar os regimes locais de fogo. Os fogos já estão se tornando cada vez mais frequentes em partes remotas da região e em florestas distantes, mas a rede de áreas protegidas e terras indígenas continua a proteger as florestas remanescentes de todos os tipos de incêndios. Os tipos de governança da terra têm uma forte influência nos regimes de fogo devido ao seu impacto nos padrões de uso da terra e nos riscos associados aos incêndios florestais, bem como nos meios de sobrevivência e objetivos culturais dos proprietários locais e seu uso associado do fogo. No norte de Mato Grosso, os proprietários de terras de assentamentos rurais e terras indígenas demonstraram capacidade de adaptação diante dos riscos crescentes de incêndios florestais, diminuindo ou mantendo a baixa intensidade de seus fogos, respectivamente. Em Capoto/Jarina, o amplo e diversificado conhecimento local sobre o fogo apoia o uso do fogo, permitindo que as comunidades explorem diferentes recursos e manejem grandes extensões de vegetação propensa a incêndios, minimizando os riscos de incêndios florestais. Esse amplo

conhecimento também permitiu que elas identificassem mudanças no clima e na vegetação que afetaram o risco de incêndios florestais. Em todas as comunidades, foram adotadas medidas de adaptação individuais e coletivas para lidar com esse risco crescente de incêndios florestais. A natureza e a extensão dessas medidas de adaptação dependeram da severidade e da novidade dos riscos emergentes de incêndios florestais, bem como das oportunidades de adaptação para cada comunidade. Coletivamente, os resultados da minha tese esclarecem a complexidade da interação entre fatores ecológicos, climáticos e socioeconômicos que moldam os regimes de fogo e as capacidades de adaptação aos riscos de incêndios florestais. O uso de diferentes metodologias e análises em várias escalas foi essencial para capturar essa diversidade de fatores. O uso de métodos participativos nos permitiu revelar alguns aspectos importantes do uso local do fogo que são difíceis de capturar usando uma abordagem de sensoriamento remoto e revelar interações em rápida mudança entre a vegetação e os fogos. Eles também destacaram a diversidade de conhecimentos sobre o uso do fogo existente nas aldeias vizinhas, esclarecendo os riscos da incompatibilidade de escala ao elaborar políticas em uma unidade de conservação ou em terras indígenas. No geral, nossos resultados destacam a importância de abordagens de pesquisa interdisciplinares e entre escalas para integrar informações de diferentes fontes de conhecimento em um mundo em rápida mudança e garantir que as medidas de adaptação apoiadas hoje não comprometam o potencial de adaptação futuro das comunidades locais.

References

Abatzoglou, J. T., A. P. Williams, and R. Barbero. 2019. Global Emergence of Anthropogenic Climate Change in Fire Weather Indices. *Geophysical Research Letters* 46: 326–336. doi:10.1029/2018GL080959.

Adeney, J. M., N. L. Christensen, and S. L. Pimm. 2009. Reserves Protect against Deforestation Fires in the Amazon. Edited by Rob P. Freckleton. *PLoS ONE* 4. doi:10.1371/journal.pone.0005014.

Aldrich, S. P., C. S. Simmons, E. Arima, R. T. Walker, F. Michelotti, and E. Castro. 2020. Agronomic or contentious land change? A longitudinal analysis from the Eastern Brazilian Amazon. Edited by Rodolfo Nóbrega. *PLOS ONE* 15: e0227378. doi:10.1371/journal.pone.0227378.

Alencar, A., C. Pereira, I. Castro, A. Cardoso, L. Souza, R. Costa, A. J. Bentes, O. Stella, et al. 2016. *Desmatamento nos Assentamentos da Amazônia: Histórico, Tendências e Oportunidades.* Brasilia: Ipam.

Alencar, A. A., P. M. Brando, G. P. Asner, and F. E. Putz. 2015. Landscape fragmentation, severe drought, and the new Amazon forest fire regime. *Ecological Applications* 25: 1493–1505. doi:10.1890/14-1528.1.

Almeida, D. R. A. D., B. W. Nelson, J. Schietti, E. B. Gorgens, A. F. Resende, S. C. Stark, and R. Valbuena. 2016. Contrasting fire damage and fire susceptibility between seasonally flooded forest and upland forest in the Central Amazon using portable profiling LiDAR. *Remote Sensing of Environment* 184: 153–160. doi:10.1016/j.rse.2016.06.017.

Alvarado, E., D. V. Sandberg, J. A. De Carvalho Jr., R. Gielow, and J. C. Santos. 2004. Landscape Fragmentation and Fire Vulnerability in Primary Forest Adjacent to Recent Land Clearings in the Amazon Arc of Deforestation. *Floresta* 34. doi:10.5380/rf.v34i2.2390.

Alves, M. T. R., V. J. Piontekowski, E. Buscardo, M. A. Pedlowski, E. E. Sano, and E. A. T. Matricardi. 2021. Effects of settlement designs on deforestation and fragmentation in the Brazilian Amazon. *Land Use Policy* 109: 105710. doi:10.1016/j.landusepol.2021.105710.

Andela, N., D. C. Morton, L. Giglio, Y. Chen, G. R. Van Der Werf, P. S. Kasibhatla, R. S. DeFries, G. J. Collatz, et al. 2017. A human-driven decline in global burned area. *Science* 356: 1356–1362. doi:10.1126/science.aal4108.

Andela, N., D. C. Morton, W. Schroeder, Y. Chen, P. M. Brando, and J. T. Randerson. 2022. Tracking and classifying Amazon fire events in near real time. *Science Advances* 8. doi:10.1126/sciadv.abd2713.

Anderson, M. J. 2017. Permutational Multivariate Analysis of Variance (PERMANOVA). In *Wiley StatsRef: Statistics Reference Online*, ed. R. S. Kenett, N. T. Longford, W. W. Piegorsch, and F. Ruggeri, 1st ed., 1–15. Wiley. doi:10.1002/9781118445112.stat07841.

de Andrade, D. F. C., A. R. Ruschel, G. Schwartz, J. O. P. de Carvalho, S. Humphries, and J. R. V. Gama. 2020. Forest resilience to fire in eastern Amazon depends on the intensity of pre-fire disturbance. *Forest Ecology and Management* 472. doi:10.1016/j.foreco.2020.118258.

Aragão, L. E. O. C., and Y. E. Shimabukuro. 2010. The Incidence of Fire in Amazonian Forests with Implications for REDD. *Science* 328: 1275–1278. doi:10.1126/science.1186925.

Aragão, L. E. O. C., Y. Malhi, R. M. Roman-Cuesta, S. Saatchi, L. O. Anderson, and Y. E. Shimabukuro. 2007. Spatial patterns and fire response of recent Amazonian droughts. *Geophysical Research Letters* 34. doi:10.1029/2006GL028946.

Aragão, L. E. O. C., L. O. Anderson, M. G. Fonseca, T. M. Rosan, L. B. Vedovato, F. H. Wagner, C. V. J. Silva, C. H. L. Silva Junior, et al. 2018. 21st Century drought-related fires counteract the decline of Amazon deforestation carbon emissions. *Nature Communications* 9: 536. doi:10.1038/s41467-017-02771-y.

Archibald, S., C. E. R. Lehmann, C. M. Belcher, W. J. Bond, R. A. Bradstock, A.-L. Daniau, K. G. Dexter, E. J. Forrestel, et al. 2018. Biological and geophysical feedbacks with fire in the Earth system. *Environmental Research Letters* 13. doi:10.1088/1748-9326/aa9ead.

de Area Leão Pereira, E. J., P. J. Silveira Ferreira, L. C. de Santana Ribeiro, T. Sabadini Carvalho, and H. B. de Barros Pereira. 2019. Policy in Brazil (2016–2019) threaten conservation of the Amazon rainforest. *Environmental Science & Policy* 100: 8–12. doi:10.1016/j.envsci.2019.06.001.

Arima, E. Y., C. S. Simmons, R. T. Walker, and M. A. Cochrane. 2007. Fire in the Brazilian Amazon: a Spatially Explicit Model for Policy Impact Analysis. *Journal of Regional Science* 47: 541–567. doi:10.1111/j.1467-9787.2007.00519.x.

Armatas, C. A., T. J. Venn, B. B. McBride, A. E. Watson, and S. J. Carver. 2016. Opportunities to utilize traditional phenological knowledge to support adaptive management of social-ecological systems vulnerable to changes in climate and fire regimes. *Ecology and Society* 21. doi:10.5751/ES-07905-210116.

Armenteras, D., and F. De La Barrera. 2023. Landscape management is urgently needed to address the rise of megafires in South America. *Communications Earth & Environment* 4. doi:10.1038/s43247-023-00964-6.

Armenteras, D., T. M. González, and J. Retana. 2013. Forest fragmentation and edge influence on fire occurrence and intensity under different management types in Amazon forests. *Biological Conservation* 159: 73–79. doi:10.1016/j.biocon.2012.10.026.

Armenteras, D., L. M. Dávalos, J. S. Barreto, A. Miranda, A. Hernández-Moreno, C. Zamorano-Elgueta, T. M. González-Delgado, M. C. Meza-Elizalde, et al. 2021. Fire-induced loss of the world's most biodiverse forests in Latin America. *Science Advances* 7. doi:10.1126/sciadv.abd3357.

Arora, G., and H. Feng. 2024. Assessing systematic biases in farmers' local weather change perceptions. *Scientific Reports* 14. doi:10.1038/s41598-024-76327-8.

Asner, G. P., E. N. Broadbent, P. J. C. Oliveira, M. Keller, D. E. Knapp, and J. N. M. Silva. 2006. Condition and fate of logged forests in the Brazilian Amazon. *Proceedings of the National Academy of Sciences* 103: 12947–12950. doi:10.1073/pnas.0604093103.

Assunção, J., and R. Rocha. 2019. Getting greener by going black: the effect of blacklisting municipalities on Amazon deforestation. *Environment and Development Economics* 24: 115–137. doi:10.1017/S1355770X18000499.

Assunção, J., C. Gandour, and R. Rocha. 2013. DETERring Deforestation in the Brazilian Amazon: Environmental Monitoring and Law Enforcement. *Climate Policy Initiative* 1: 36.

Assunção, J., C. Gandour, and R. Rocha. 2015. Deforestation slowdown in the Brazilian Amazon: prices or policies? *Environment and Development Economics* 20: 697–722. doi:10.1017/S1355770X15000078.

Athayde, S., and J. Silva-Lugo. 2018. Adaptive Strategies to Displacement and Environmental Change Among the Kaiabi Indigenous People of the Brazilian Amazon. *Society & Natural Resources* 31: 666–682. doi:10.1080/08941920.2018.1426801.

Bachl, F. E., F. Lindgren, D. L. Borchers, and J. B. Illian. 2019. inlabru: an R package for Bayesian spatial modelling from ecological survey data. Edited by Robert Freckleton. *Methods in Ecology and Evolution* 10: 760–766. doi:10.1111/2041-210X.13168.

Balch, J. K., D. C. Nepstad, P. M. Brando, L. M. Curran, O. Portela, O. De Carvalho, and P. Lefebvre. 2008. Negative fire feedback in a transitional forest of southeastern Amazonia. *Global Change Biology* 14: 2276–2287. doi:10.1111/j.1365-2486.2008.01655.x.

Balch, J. K., D. C. Nepstad, L. M. Curran, P. M. Brando, O. Portela, P. Guilherme, J. D. Reuning-Scherer, and O. de Carvalho. 2011. Size, species, and fire behavior predict tree and liana mortality from experimental burns in the Brazilian Amazon. *Forest Ecology and Management* 261: 68–77. doi:10.1016/j.foreco.2010.09.029.

Balch, J. K., P. M. Brando, D. C. Nepstad, M. T. Coe, D. Silvério, T. J. Massad, E. A. Davidson, P. Lefebvre, et al. 2015. The Susceptibility of Southeastern Amazon Forests to Fire: Insights from a Large-Scale Burn Experiment. *BioScience* 65: 893–905. doi:10.1093/biosci/biv106.

Balch, J. K., J. T. Abatzoglou, M. B. Joseph, M. J. Koontz, A. L. Mahood, J. McGlinchy, M. E. Cattau, and A. P. Williams. 2022. Warming weakens the night-time barrier to global fire. *Nature* 602: 442–448. doi:10.1038/s41586-021-04325-1.

Barbieri, J. D., R. Dallacort, P. S. L. D. Freitas, D. V. D. Araújo, R. C. Tieppo, and W. Fenner. 2019. Effects of the ENSO on the Variability of Precipitation and Air Temperature in Agricultural Regions of Mato Grosso State. *Journal of Agricultural Science* 11. doi:10.5539/jas.v11n9p91.

Barlow, J., and C. A. Peres. 2008. Fire-mediated dieback and compositional cascade in an Amazonian forest. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363: 1787–1794. doi:10.1098/rstb.2007.0013.

Barlow, J., C. A. Peres, B. O. Lagan, and T. Haugaasen. 2003a. Large tree mortality and the decline of forest biomass following Amazonian wildfires. *Ecology Letters* 6: 6–8. doi:10.1046/j.1461-0248.2003.00394.x.

Barlow, J., B. O. Lagan, and C. A. Peres. 2003b. Morphological correlates of fire-induced tree mortality in a central Amazonian forest. *Journal of Tropical Ecology* 19: 291–299. doi:10.1017/S0266467403003328.

Barlow, J., J. M. Silveira, L. A. M. Mestre, R. B. Andrade, G. Camacho D'Andrea, J. Louzada, F. Z. Vaz-de-Mello, I. Numata, et al. 2012. Wildfires in Bamboo-Dominated Amazonian Forest: Impacts on Above-Ground Biomass and Biodiversity. Edited by Ben Bond-Lamberty. *PLoS ONE* 7. doi:10.1371/journal.pone.0033373.

Barlow, J., E. Berenguer, R. Carmenta, and F. França. 2020. Clarifying Amazonia's burning crisis. *Global Change Biology* 26: 319–321. doi:10.1111/gcb.14872.

Barni, P. E., A. C. M. Rego, F. das C. F. Silva, R. A. S. Lopes, H. A. M. Xaud, M. R. Xaud, R. I. Barbosa, and P. M. Fearnside. 2021. Logging Amazon forest increased the severity and spread of fires during the 2015–2016 El Niño. *Forest Ecology and Management* 500: 119652. doi:10.1016/j.foreco.2021.119652.

Barona, E., N. Ramankutty, G. Hyman, and O. T. Coomes. 2010. The role of pasture and soybean in deforestation of the Brazilian Amazon. *Environmental Research Letters* 5. doi:10.1088/1748-9326/5/2/024002.

Barradas, A. C. S., and K. Torres Ribeiro. 2021. Manejo Integrado do Fogo: Trajetória da Estação Ecológica Serra Geral do Tocantins (2001 a 2020). *Biodiversidade Brasileira - BioBrasil*: 139–152. doi:10.37002/biobrasil.v11i2.1739.

Bauer, T. N., W. De Jong, and V. Ingram. 2022. Perception matters: an Indigenous perspective on climate change and its effects on forest-based livelihoods in the Amazon. *Ecology and Society* 27. doi:10.5751/ES-12837-270117.

Begotti, R. A., and C. A. Peres. 2020. Rapidly escalating threats to the biodiversity and ethnocultural capital of Brazilian Indigenous Lands. *Land Use Policy* 96. doi:10.1016/j.landusepol.2020.104694.

Berardi, A., J. Mistry, C. Tschirhart, E. Bignante, O. Davis, L. Haynes, R. Benjamin, G. Albert, et al. 2015. Applying the system viability framework for cross-scalar governance of nested social-ecological systems in the Guiana Shield, South America. *Ecology and Society* 20. doi:10.5751/ES-07865-200342.

Berkes, F. 2018. *Sacred ecology*. Fourth Edition. New York: Routledge.

Berkes, F., and N. J. Turner. 2006. Knowledge, Learning and the Evolution of Conservation Practice for Social-Ecological System Resilience. *Human Ecology* 34: 479–494. doi:10.1007/s10745-006-9008-2.

Bidone, F., and Z. Kovacic. 2018. From nationalism to global climate change: analysis of the historical evolution of environmental governance in the Brazilian Amazon. *International Forestry Review* 20: 420–435. doi:10.1505/146554818825240656.

Bilbao, B. A., A. V. Leal, and C. L. Méndez. 2010. Indigenous Use of Fire and Forest Loss in Canaima National Park, Venezuela. Assessment of and Tools for Alternative Strategies of Fire Management in Pemón Indigenous Lands. *Human Ecology* 38: 663–673. doi:10.1007/s10745-010-9344-0.

Bilbao, B. A., A. Millán, H. Vessuri, J. Mistry, R. Salazar-Gascón, and R. Gómez. 2021. To Burn or not to Burn? The History behind the Construction of a New Paradigm of Fire Management in Venezuela through Interculturality: Local Actions of National and Regional Impact. *Biodiversidade Brasileira - BioBrasil*: 99–127. doi:10.37002/biobrasil.v11i2.1878.

Bodin, Ö. 2017. Collaborative environmental governance: Achieving collective action in social-ecological systems. *Science* 357. doi:10.1126/science.aan1114.

Böhlen, M., J. Liu, and R. Iryadi. 2022. Who Speaks for the Forest? Local Knowledge, Participatory Mapping and Collaborative evaluaation for GIS Analysis in the Tropics of Central Bali, Indonesia. *The International Archives of the Photogrammetry, Remote Sensing and Spatial Information Sciences*: 73–80. doi:10.5194/isprs-archives-XLVIII-4-W1-2022-73-2022.

Börner, J., K. Kis-Katos, J. Hargrave, and K. König. 2015. Post-Crackdown Effectiveness of Field-Based Forest Law Enforcement in the Brazilian Amazon. Edited by Ricardo Bomfim Machado. *PLOS ONE* 10. doi:10.1371/journal.pone.0121544.

Bossel, H. 1999. *Indicators for sustainable development: theory, method, applications ; a report to the Balaton group*. Winnipeg: IISD.

Boulton, C. A., T. M. Lenton, and N. Boers. 2022. Pronounced loss of Amazon rainforest resilience since the early 2000s. *Nature Climate Change* 12: 271–278. doi:10.1038/s41558-022-01287-8.

Bowman, D. M. J. S., and J. J. Sharples. 2023. Taming the flame, from local to global extreme wildfires. *Science* 381: 616–619. doi:10.1126/science.adi8066.

Bowman, D. M. J. S., J. Balch, P. Artaxo, W. J. Bond, M. A. Cochrane, C. M. D'Antonio, R. DeFries, F. H. Johnston, et al. 2011. The human dimension of fire regimes on Earth: The human dimension of fire regimes on Earth. *Journal of Biogeography* 38: 2223–2236. doi:10.1111/j.1365-2699.2011.02595.x.

Bowman, D. M. J. S., J. A. O'Brien, and J. G. Goldammer. 2013. Pyrogeography and the Global Quest for Sustainable Fire Management. *Annual Review of Environment and Resources* 38: 57–80. doi:10.1146/annurev-environ-082212-134049.

Bowman, M. S., G. S. Amacher, and F. D. Merry. 2008. Fire use and prevention by traditional households in the Brazilian Amazon. *Ecological Economics* 67: 117–130. doi:10.1016/j.ecolecon.2007.12.003.

Bowman, M. S., B. S. Soares-Filho, F. D. Merry, D. C. Nepstad, H. Rodrigues, and O. T. Almeida. 2012. Persistence of cattle ranching in the Brazilian Amazon: A spatial analysis of the rationale for beef production. *Land Use Policy* 29: 558–568. doi:10.1016/j.landusepol.2011.09.009.

Brancalion, P. H. S., D. R. A. de Almeida, E. Vidal, P. G. Molin, V. E. Sontag, S. E. X. F. Souza, and M. D. Schulze. 2018. Fake legal logging in the Brazilian Amazon. *Science Advances* 4. doi:10.1126/sciadv.aat1192.

Brando, P. M., D. C. Nepstad, J. K. Balch, B. Bolker, M. C. Christman, M. Coe, and F. E. Putz. 2012. Fire-induced tree mortality in a neotropical forest: the roles of bark traits, tree size, wood density and fire behavior. *Global Change Biology* 18: 630–641. doi:10.1111/j.1365-2486.2011.02533.x.

Brando, P. M., J. K. Balch, D. C. Nepstad, D. C. Morton, F. E. Putz, M. T. Coe, D. Silverio, M. N. Macedo, et al. 2014. Abrupt increases in Amazonian tree mortality due to drought-fire interactions. *Proceedings of the National Academy of Sciences* 111: 6347–6352. doi:10.1073/pnas.1305499111.

Brando, P. M., B. Soares-Filho, L. Rodrigues, A. Assunção, D. Morton, D. Tuchschneider, E. C. M. Fernandes, M. N. Macedo, et al. 2020. The gathering firestorm in southern Amazonia. *Science Advances* 6. doi:10.1126/sciadv.aay1632.

Brito, B., P. Barreto, A. Brandão, S. Baima, and P. H. Gomes. 2019. Stimulus for land grabbing and deforestation in the Brazilian Amazon. *Environmental Research Letters* 14: 064018. doi:10.1088/1748-9326/ab1e24.

Broadbent, E. N., G. P. Asner, M. Keller, D. E. Knapp, P. J. C. Oliveira, and J. N. Silva. 2008. Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. *Biological Conservation*: 13.

Brondízio, E. S., Y. Aumeeruddy-Thomas, P. Bates, J. Carino, Á. Fernández-Llamazares, M. F. Ferrari, K. Galvin, V. Reyes-García, et al. 2021. Locally Based, Regionally Manifested, and Globally Relevant: Indigenous and Local Knowledge, Values, and Practices for Nature. *Annual Review of Environment and Resources* 46: 481–509. doi:10.1146/annurev-environ-012220-012127.

Cammelli, F., and A. Angelsen. 2019. Amazonian farmers' response to fire policies and climate change. *Ecological Economics* 165. doi:10.1016/j.ecolecon.2019.106359.

Cammelli, F., E. Coudel, and L. de F. N. Alves. 2019. Smallholders' Perceptions of Fire in the Brazilian Amazon: Exploring Implications for Governance Arrangements. *Human Ecology* 47: 601–612. doi:10.1007/s10745-019-00096-6.

Cammelli, F., R. D. Garrett, J. Barlow, and L. Parry. 2020. Fire risk perpetuates poverty and fire use among Amazonian smallholders. *Global Environmental Change* 63. doi:10.1016/j.gloenvcha.2020.102096.

Campanharo, W. A., A. P. Lopes, L. O. Anderson, T. F. M. R. Da Silva, and L. E. O. C. Aragão. 2019. Translating Fire Impacts in Southwestern Amazonia into Economic Costs. *Remote Sensing* 11. doi:10.3390/rs11070764.

Cano-Crespo, A., P. J. C. Oliveira, A. Boit, M. Cardoso, and K. Thonicke. 2015. Forest edge burning in the Brazilian Amazon promoted by escaping fires from managed pastures. *Journal of Geophysical Research: Biogeosciences* 120: 2095–2107. doi:10.1002/2015JG002914.

Caretta, M. A. 2015. Situated knowledge in cross-cultural, cross-language research: a collaborative reflexive analysis of researcher, assistant and participant subjectivities. *Qualitative Research* 15: 489–505. doi:10.1177/1468794114543404.

Carmenta, R., L. Parry, A. Blackburn, S. Vermeylen, and J. Barlow. 2011. Understanding Human-Fire Interactions in Tropical Forest Regions: a Case for Interdisciplinary Research across the Natural and Social Sciences. *Ecology and Society* 16. doi:10.5751/ES-03950-160153.

Carmenta, R., S. Vermeylen, L. Parry, and J. Barlow. 2013. Shifting Cultivation and Fire Policy: Insights from the Brazilian Amazon. *Human Ecology* 41: 603–614. doi:10.1007/s10745-013-9600-1.

Carmenta, R., G. A. Blackburn, G. Davies, C. de Sassi, A. Lima, L. Parry, W. Tych, and J. Barlow. 2016. Does the Establishment of Sustainable Use Reserves Affect Fire Management in the Humid Tropics? Edited by Clinton N. Jenkins. *PLOS ONE* 11. doi:10.1371/journal.pone.0149292.

Carmenta, R., E. Coudel, and A. M. Steward. 2019. Forbidden fire: Does criminalising fire hinder conservation efforts in swidden landscapes of the Brazilian Amazon? *The Geographical Journal* 185: 23–37. doi:10.1111/geoj.12255.

Carmenta, R., F. Cammelli, W. Dressler, C. Verbicaro, and J. G. Zaehringer. 2021. Between a rock and a hard place: The burdens of uncontrolled fire for smallholders across the tropics. *World Development* 145. doi:10.1016/j.worlddev.2021.105521.

Carrero, G. C., R. T. Walker, C. S. Simmons, and P. M. Fearnside. 2022. Land grabbing in the Brazilian Amazon: Stealing public land with government approval. *Land Use Policy* 120. doi:10.1016/j.landusepol.2022.106133.

Carroll, M. S., C. M. Edgeley, and C. Nugent. 2021. Traditional use of field burning in Ireland: history, culture and contemporary practice in the uplands. *International Journal of Wildland Fire* 30: 399–409. doi:10.1071/WF20127.

Carroll, R., A. B. Lawson, C. Faes, R. S. Kirby, M. Aregay, and K. Watjou. 2015. Comparing INLA and OpenBUGS for hierarchical Poisson modeling in disease mapping. *Spatial and Spatio-temporal Epidemiology* 14–15: 45–54. doi:10.1016/j.sste.2015.08.001.

Carvalho, R., A. P. D. de Aguiar, and S. Amaral. 2020. Diversity of cattle raising systems and its effects over forest regrowth in a core region of cattle production in the Brazilian Amazon. *Regional Environmental Change* 20. doi:10.1007/s10113-020-01626-5.

Cavalcante, R. B. L., B. M. Souza, S. J. Ramos, M. Gastauer, W. R. Nascimento Junior, C. F. Caldeira, and P. W. M. Souza-Filho. 2021. Assessment of fire hazard weather indices in the eastern Amazon: a case study for different land uses. *Acta Amazonica* 51: 352–362. doi:10.1590/1809-4392202101172.

Celis, N., A. Casallas, E. A. Lopez-Barrera, M. Felician, M. De Marchi, and S. E. Pappalardo. 2023. Climate Change, Forest Fires, and Territorial Dynamics in the Amazon Rainforest: An Integrated Analysis for Mitigation Strategies. *ISPRS International Journal of Geo-Information* 12. doi:10.3390/ijgi12100436.

Chaffin, B. C., H. Gosnell, and B. A. Cosens. 2014. A decade of adaptive governance scholarship: synthesis and future directions. *Ecology and Society* 19. doi:10.5751/ES-06824-190356.

Chanza, N., and W. Musakwa. 2022. Indigenous local observations and experiences can give useful indicators of climate change in data-deficient regions. *Journal of Environmental Studies and Sciences* 12: 534–546. doi:10.1007/s13412-022-00757-x.

Chen, H., J. M. Samet, P. A. Bromberg, and H. Tong. 2021. Cardiovascular health impacts of wildfire smoke exposure. *Particle and Fibre Toxicology* 18. doi:10.1186/s12989-020-00394-8.

Copes-Gerbitz, K., S. M. Hagerman, and L. D. Daniels. 2021. Situating Indigenous knowledge for resilience in fire-dependent social-ecological systems. *Ecology and Society* 26. doi:10.5751/ES-12757-260425.

Copes-Gerbitz, K., S. M. Hagerman, and L. D. Daniels. 2022. Transforming fire governance in British Columbia, Canada: an emerging vision for coexisting with fire. *Regional Environmental Change* 22. doi:10.1007/s10113-022-01895-2.

Cosens, B. A., L. Gunderson, and B. C. Chaffin. 2018. Introduction to the Special Feature Practicing Panarchy: Assessing legal flexibility, ecological resilience, and adaptive governance in regional water systems experiencing rapid environmental change. *Ecology and Society* 23. doi:10.5751/ES-09524-230104.

Costa, J. G., P. M. Fearnside, I. Oliveira, L. O. Anderson, L. E. O. E. C. De Aragão, M. R. N. Almeida, F. S. Clemente, E. D. S. Nascimento, et al. 2023. Forest Degradation in the Southwest Brazilian Amazon: Impact on Tree Species of Economic Interest and Traditional Use. *Fire* 6. doi:10.3390/fire6060234.

Creswell, J. W., V. L. Plano Clark, M. L. Gutmann, and W. E. Hanson. 2003. *Advanced mixed methods research designs*. Thousand Oaks.

Csillik, O., M. Keller, M. Longo, A. Ferraz, E. Rangel Pinagé, E. B. Görgens, J. P. Ometto, V. Silgueiro, et al. 2024. A large net carbon loss attributed to anthropogenic and natural disturbances in the Amazon Arc of Deforestation. *Proceedings of the National Academy of Sciences* 121. doi:10.1073/pnas.2310157121.

Cumming, G. S., D. H. M. Cumming, and C. L. Redman. 2006. Scale Mismatches in Social-Ecological Systems: Causes, Consequences, and Solutions. *Ecology and Society* 11. doi:10.5751/ES-01569-110114.

Da Ponte, E., F. Alcasena, T. Bhagwat, Z. Hu, L. Eufemia, A. P. Dias Turetta, M. Bonatti, S. Sieber, et al. 2023. Assessing wildfire activity and forest loss in protected areas of the Amazon basin. *Applied Geography* 157. doi:10.1016/j.apgeog.2023.102970.

Da Silva Arruda, V. L., A. A. C. Alencar, O. A. De Carvalho Júnior, F. De Figueiredo Ribeiro, F. V. De Arruda, D. E. Conciani, W. V. Da Silva, and J. Z. Shimbo. 2024. Assessing four decades of fire behavior dynamics in the Cerrado biome (1985 to 2022). *Fire Ecology* 20. doi:10.1186/s42408-024-00298-4.

Davidson, E. A., A. C. de Araújo, P. Artaxo, J. K. Balch, I. F. Brown, M. M. C. Bustamante, M. T. Coe, R. S. DeFries, et al. 2012. The Amazon basin in transition. *Nature* 481: 321–328. doi:10.1038/nature10717.

De Almeida, A. S., I. C. G. Vieira, and S. F. B. Ferraz. 2020. Long-term assessment of oil palm expansion and landscape change in the eastern Brazilian Amazon. *Land Use Policy* 90. doi:10.1016/j.landusepol.2019.104321.

De Faria, B. L., P. M. Brando, M. N. Macedo, P. K. Panday, B. S. Soares-Filho, and M. T. Coe. 2017. Current and future patterns of fire-induced forest degradation in Amazonia. *Environmental Research Letters* 12. doi:10.1088/1748-9326/aa69ce.

De Faria, B. L., A. Staal, C. A. Silva, P. A. Martin, P. K. Panday, and V. L. Dantas. 2021. Climate change and deforestation increase the vulnerability of Amazonian forests to post-fire grass invasion. *Global Ecology and Biogeography* 30: 2368–2381. doi:10.1111/geb.13388.

De Oliveira, A. S., R. G. Rajão, B. S. Soares Filho, U. Oliveira, L. R. S. Santos, A. C. Assunção, R. Van Der Hoff, H. O. Rodrigues, et al. 2019. Economic losses to sustainable timber production by fire in the Brazilian Amazon. *The Geographical Journal* 185: 55–67. doi:10.1111/geoj.12276.

De Resende, A. F., B. W. Nelson, B. M. Flores, and D. R. De Almeida. 2014. Fire Damage in Seasonally Flooded and Upland Forests of the Central Amazon. *Biotropica* 46: 643–646. doi:10.1111/btp.12153.

De Santana, M. M. M., R. N. De Vasconcelos, E. Mariano Neto, and W. D. J. S. Da Franca Rocha. 2024. Machine Learning Model Reveals Land Use and Climate's Role in Amazon Wildfires: Present and Future Scenarios. *Fire* 7. doi:10.3390/fire7100338.

DeFries, R., M. Herold, L. Verchot, M. N. Macedo, and Y. Shimabukuro. 2013. Export-oriented deforestation in Mato Grosso: harbinger or exception for other tropical forests? *Philosophical Transactions of the Royal Society B: Biological Sciences* 368. doi:10.1098/rstb.2012.0173.

Denwood, T., J. J. Huck, and S. Lindley. 2022. Participatory Mapping: A Systematic Review and Open Science Framework for Future Research. *Annals of the American Association of Geographers* 112: 2324–2343. doi:10.1080/24694452.2022.2065964.

Deutsch, S., and R. Fletcher. 2022. The 'Bolsonaro bridge': Violence, visibility, and the 2019 Amazon fires. *Environmental Science & Policy* 132: 60–68. doi:10.1016/j.envsci.2022.02.012.

Dinno, A. 2015. Nonparametric Pairwise Multiple Comparisons in Independent Groups using Dunn's Test. *The Stata Journal: Promoting communications on statistics and Stata* 15: 292–300. doi:10.1177/1536867X1501500117.

Dobrovolski, R., R. Loyola, L. Rattis, S. F. Gouveia, D. Cardoso, R. Santos-Silva, D. Gonçalves-Souza, L. M. Bini, et al. 2018. Science and democracy must orientate Brazil's path to sustainability. *Perspectives in Ecology and Conservation* 16: 121–124. doi:10.1016/j.pecon.2018.06.005.

Dou, Y., R. F. B. da Silva, H. Yang, and J. Liu. 2018. Spillover effect offsets the conservation effort in the Amazon. *Journal of Geographical Sciences* 28: 1715–1732. doi:10.1007/s11442-018-1539-0.

Drüke, M., B. Sakschewski, W. Von Bloh, M. Billing, W. Lucht, and K. Thonicke. 2023. Fire may prevent future Amazon forest recovery after large-scale deforestation. *Communications Earth & Environment* 4. doi:10.1038/s43247-023-00911-5.

Dubreuil, V., N. Debortoli, B. Funatsu, V. Nédélec, and L. Durieux. 2012. Impact of land-cover change in the Southern Amazonia climate: a case study for the region of Alta Floresta, Mato Grosso, Brazil. *Environmental Monitoring and Assessment* 184: 877–891. doi:10.1007/s10661-011-2006-x.

East, A., A. Hansen, D. Armenteras, P. Jantz, and D. W. Roberts. 2023. Measuring Understory Fire Effects from Space: Canopy Change in Response to Tropical Understory Fire and What This Means for Applications of GEDI to Tropical Forest Fire. *Remote Sensing* 15. doi:10.3390/rs15030696.

Eilola, S., N. Käyhkö, and N. Fagerholm. 2021. Lessons learned from participatory land use planning with high-resolution remote sensing images in Tanzania: Practitioners' and participants' perspectives. *Land Use Policy* 109. doi:10.1016/j.landusepol.2021.105649.

Eloy, L., B. A. Bilbao, J. Mistry, and I. B. Schmidt. 2019. From fire suppression to fire management: Advances and resistances to changes in fire policy in the savannas of Brazil and Venezuela. *The Geographical Journal* 185: 10–22. doi:10.1111/geoj.12245.

Eloy, L., R. Ramos, M. Schmidt, K. y Ono, A. Steward, and J. Feirreira. 2021. Manejo do fogo por povos indígenas e comunidades tradicionais no Brasil. *Povos tradicionais e biodiversidade no Brasil : contribuições dos povos indígenas, quilombolas e comunidades tradicionais para a biodiversidade, políticas e ameaças*.

Eriksen, C. 2007. Why do they burn the bush? Fire, rural livelihoods, and conservation in Zambia. *The Geographical Journal* 173: 242–256. doi:10.1111/j.1475-4959.2007.00239.x.

Eriksen, C., and D. L. Hankins. 2014. The Retention, Revival, and Subjugation of Indigenous Fire Knowledge through Agency Fire Fighting in Eastern Australia and California. *Society & Natural Resources* 27: 1288–1303. doi:10.1080/08941920.2014.918226.

Esquivel-Muelbert, A., T. R. Baker, K. G. Dexter, S. L. Lewis, R. J. W. Brienen, T. R. Feldpausch, J. Lloyd, A. Monteagudo-Mendoza, et al. 2019. Compositional response of Amazon forests to climate change. *Global Change Biology* 25: 39–56. doi:10.1111/gcb.14413.

Eufemia, L., A. P. Dias Turetta, M. Bonatti, E. Da Ponte, and S. Sieber. 2022. Fires in the Amazon Region: Quick Policy Review. *Development Policy Review* 40. doi:10.1111/dpr.12620.

Evans, L. S., K. Brown, and E. H. Allison. 2011. Factors Influencing Adaptive Marine Governance in a Developing Country Context: a Case Study of Southern Kenya. *Ecology and Society* 16: art21. doi:10.5751/ES-04084-160221.

Falleiro, R. de M., L. Steil, M. S. de Oliveira, I. Lando, A. M. C. Cunha, and G. C. Zacharias. 2021. Histórico, Avaliação, Oportunidades e Desafios do Manejo Integrado do Fogo nas Terras Indígenas Brasileiras.

Fanin, T., and G. R. Van Der Werf. 2015. Relationships between burned area, forest cover loss, and land cover change in the Brazilian Amazon based on satellite data. *Biogeosciences* 12: 6033–6043. doi:10.5194/bg-12-6033-2015.

Fearnside, P. M. 2005. Deforestation in Brazilian Amazonia: History, Rates, and Consequences. *Conservation Biology* 19: 680–688. doi:10.1111/j.1523-1739.2005.00697.x.

Feeley, K. J., and M. R. Silman. 2016. Disappearing climates will limit the efficacy of Amazonian protected areas. Edited by Janet Franklin. *Diversity and Distributions* 22: 1081–1084. doi:10.1111/ddi.12475.

Feng, X., C. Merow, Z. Liu, D. S. Park, P. R. Roehrdanz, B. Maitner, E. A. Newman, B. L. Boyle, et al. 2021. How deregulation, drought and increasing fire impact Amazonian biodiversity. *Nature* 597: 516–521. doi:10.1038/s41586-021-03876-7.

Fernández-Llamazares, Á., I. Díaz-Reviriego, A. C. Luz, M. Cabeza, A. Pyhälä, and V. Reyes-García. 2015. Rapid ecosystem change challenges the adaptive capacity of Local Environmental Knowledge. *Global Environmental Change* 31: 272–284. doi:10.1016/j.gloenvcha.2015.02.001.

Fernández-Llamazares, Á., D. Lepofsky, K. Lertzman, C. G. Armstrong, E. S. Brondizio, M. C. Gavin, P. O. Lyver, G. P. Nicholas, et al. 2021. Scientists' Warning to Humanity on Threats to Indigenous and Local Knowledge Systems. *Journal of Ethnobiology* 41: 144–169. doi:10.2993/0278-0771-41.2.144.

Feron, S., R. R. Cordero, A. Damiani, S. MacDonell, J. Pizarro, K. Goubanova, R. Valenzuela, C. Wang, et al. 2024. South America is becoming warmer, drier, and more flammable. *Communications Earth & Environment* 5. doi:10.1038/s43247-024-01654-7.

Ferreira Barbosa, M. L., R. C. Delgado, C. Forsad de Andrade, P. E. Teodoro, C. A. Silva Junior, H. S. Wanderley, and G. F. Capristo-Silva. 2021. Recent trends in the fire dynamics in Brazilian Legal Amazon: Interaction between the ENSO phenomenon, climate and land use. *Environmental Development* 39. doi:10.1016/j.envdev.2021.100648.

Ferro, P. D., G. Mataveli, J. D. S. Arcanjo, D. J. Dutra, T. P. D. Medeiros, Y. E. Shimabukuro, A. C. M. Pessôa, G. De Oliveira, et al. 2024. Regional-Scale Assessment of Burn Scar Mapping in Southwestern Amazonia Using Burned Area Products and CBERS/WFI Data Cubes. *Fire* 7. MDPI AG: 67. doi:10.3390/fire7030067.

Fidelis, A., S. T. Alvarado, A. C. S. Barradas, and V. R. Pivello. 2018. The Year 2017: Megafires and Management in the Cerrado. *Fire* 1: 49. doi:10.3390/fire1030049.

de Filho, F. J. B. O., and J. P. Metzger. 2006. Thresholds in landscape structure for three common deforestation patterns in the Brazilian Amazon. *Landscape Ecology* 21: 1061–1073. doi:10.1007/s10980-006-6913-0.

Flores, B. M., R. Fagoaga, B. W. Nelson, and M. Holmgren. 2016. Repeated fires trap Amazonian blackwater floodplains in an open vegetation state. Edited by Jos Barlow. *Journal of Applied Ecology* 53: 1597–1603. doi:10.1111/1365-2664.12687.

Folke, C., J. Colding, and F. Berkes. 2009. Building resilience and adaptive capacity in social-ecological systems. In *Navigating Social-Ecological System*, Cambridge University Press.

Fonseca, M. G., L. O. Anderson, E. Arai, Y. E. Shimabukuro, H. A. M. Xaud, M. R. Xaud, N. Madani, F. H. Wagner, et al. 2017. Climatic and anthropogenic drivers of northern Amazon fires during the 2015-2016 El Niño event. *Ecological Applications* 27: 2514–2527. doi:10.1002/eap.1628.

Fonseca, M. G., L. M. Alves, A. P. D. Aguiar, E. Arai, L. O. Anderson, T. M. Rosan, Y. E. Shimabukuro, and L. E. O. E. C. De Aragão. 2019. Effects of climate and land-use change scenarios on fire probability during the 21st century in the Brazilian Amazon. *Global Change Biology* 25: 2931–2946. doi:10.1111/gcb.14709.

Fonseca Morello, T., R. Marchetti Ramos, L. O. Anderson, N. Owen, T. M. Rosan, and L. Steil. 2020. Predicting fires for policy making: Improving accuracy of fire brigade allocation in the Brazilian Amazon. *Ecological Economics* 169. doi:10.1016/j.ecolecon.2019.106501.

Fonseca-Morello, T., R. Ramos, L. Steil, L. Parry, J. Barlow, N. Markusson, and A. Ferreira. 2017. Fires in Brazilian Amazon: Why does policy have a limited impact? *Ambiente & Sociedade* 20: 19–38. doi:10.1590/1809-4422asoc0232r1v2042017.

Ford, A. E. S., S. P. Harrison, Y. Kountouris, J. D. A. Millington, J. Mistry, O. Perkins, S. S. Rabin, G. Rein, et al. 2021. Modelling Human-Fire Interactions: Combining Alternative Perspectives and Approaches. *Frontiers in Environmental Science* 9. doi:10.3389/fenvs.2021.649835.

Fuchs, R., J. Raymond, K. Winkler, and M. Rounsevell. 2024. The EU's new anti-deforestation law has severe loopholes that could be exploited by the forthcoming EU-MERCOSUR trade agreement. *Environmental Research Letters* 19: 091005. doi:10.1088/1748-9326/ad69ab.

Fuller, D. O. 2006. Tropical forest monitoring and remote sensing: A new era of transparency in forest governance? *Singapore Journal of Tropical Geography* 27: 15–29. doi:10.1111/j.1467-9493.2006.00237.x.

Funk, C., P. Peterson, M. Landsfeld, D. Pedreros, J. Verdin, S. Shukla, G. Husak, J. Rowland, et al. 2015. The climate hazards infrared precipitation with stations—a new environmental record for monitoring extremes. *Scientific Data* 2. doi:10.1038/sdata.2015.66.

Garcia, L. C., J. K. Szabo, F. De Oliveira Roque, A. De Matos Martins Pereira, C. Nunes Da Cunha, G. A. Damasceno-Júnior, R. G. Morato, W. M. Tomas, et al. 2021. Record-breaking wildfires in the world's largest continuous tropical wetland: Integrative fire management is urgently needed for both biodiversity and humans. *Journal of Environmental Management* 293. doi:10.1016/j.jenvman.2021.112870.

Garrett, R. D., I. Koh, E. F. Lambin, Y. le Polain de Waroux, J. H. Kastens, and J. C. Brown. 2018. Intensification in agriculture-forest frontiers: Land use responses to development and conservation policies in Brazil. *Global Environmental Change* 53: 233–243. doi:10.1016/j.gloenvcha.2018.09.011.

Gasparri, N. I., and Y. L. P. De Waroux. 2015. The Coupling of South American Soybean and Cattle Production Frontiers: New Challenges for Conservation Policy and Land Change Science: The coupling of soy and cattle frontiers. *Conservation Letters* 8: 290–298. doi:10.1111/conl.12121.

Gatti, L. V., L. S. Basso, J. B. Miller, M. Gloor, L. Gatti Domingues, H. L. G. Cassol, G. Tejada, L. E. O. C. Aragão, et al. 2021. Amazonia as a carbon source linked to deforestation and climate change. *Nature* 595: 388–393. doi:10.1038/s41586-021-03629-6.

Gibbs, H. K., L. Rausch, J. Munger, I. Schelly, D. C. Morton, P. Noojipady, B. Soares-Filho, P. Barreto, et al. 2015. Brazil's Soy Moratorium. *Science* 347: 377–378. doi:10.1126/science.aaa0181.

Gibbs, H. K., J. Munger, J. L'Roe, P. Barreto, R. Pereira, M. Christie, T. Amaral, and N. F. Walker. 2016. Did Ranchers and Slaughterhouses Respond to Zero-Deforestation Agreements in the Brazilian Amazon?: Brazil's zero-deforestation pacts. *Conservation Letters* 9: 32–42. doi:10.1111/conl.12175.

Giglio, L. 2007. Characterization of the tropical diurnal fire cycle using VIRS and MODIS observations. *Remote Sensing of Environment* 108: 407–421. doi:10.1016/j.rse.2006.11.018.

Giglio, L., W. Schroeder, and J. V. Hall. 2020. MODIS Collection 6 and Collection 6.1 Active Fire Product User's Guide.

Glaser, B. G., and A. L. Strauss. 1967. *The discovery of grounded theory: strategies for qualitative research*. London New York: Routledge.

Gollnow, F., L. de B. V. Hissa, P. Rufin, and T. Lakes. 2018. Property-level direct and indirect deforestation for soybean production in the Amazon region of Mato Grosso, Brazil. *Land Use Policy* 78: 377–385. doi:10.1016/j.landusepol.2018.07.010.

Gómez-Baggethun, E., and V. Reyes-García. 2013. Reinterpreting Change in Traditional Ecological Knowledge. *Human Ecology* 41: 643–647. doi:10.1007/s10745-013-9577-9.

Gómez-Baggethun, E., V. Reyes-García, P. Olsson, and C. Montes. 2012. Traditional ecological knowledge and community resilience to environmental extremes: A case study in Doñana, SW Spain. *Global Environmental Change* 22: 640–650. doi:10.1016/j.gloenvcha.2012.02.005.

Gómez-Baggethun, E., E. Corbera, and V. Reyes-García. 2013. Traditional Ecological Knowledge and Global Environmental Change: Research findings and policy implications. *Ecology and Society* 18. doi:10.5751/ES-06288-180472.

Goulart, F. F., M. J. Chappell, F. Mertens, and B. Soares-Filho. 2023. Sparing or expanding? The effects of agricultural yields on farm expansion and deforestation in the tropics. *Biodiversity and Conservation* 32: 1089–1104. doi:10.1007/s10531-022-02540-4.

Gutiérrez-Vélez, V. H., M. Uriarte, R. DeFries, M. Pinedo-Vásquez, K. Fernandes, P. Ceccato, W. Baethgen, and C. Padoch. 2014. Land cover change interacts with drought severity to change fire regimes in Western Amazonia. *Ecological Applications* 24: 1323–1340. doi:10.1890/13-2101.1.

Hamed, K. H., and A. Ramachandra Rao. 1998. A modified Mann-Kendall trend test for autocorrelated data. *Journal of Hydrology* 204: 182–196. doi:10.1016/S0022-1694(97)00125-X.

Harwell, E. 2000. Remote Sensibilities: Discourses of Technology and the Making of Indonesia's Natural Disaster. *Development and Change* 31: 307–340. doi:10.1111/1467-7660.00156.

Hayes, N., and R. Rajão. 2011. Competing institutional logics and sustainable development: the case of geographic information systems in Brazil's Amazon region. *Information Technology for Development* 17: 4–23. doi:10.1080/02681102.2010.511701.

Hecht, S. 2020. Why the Brazilian Amazon Burns. *Current History* 119: 60–65. doi:10.1525/curh.2020.119.814.60.

Heckenberger, M. J., A. Kuikuro, U. T. Kuikuro, J. C. Russell, M. Schmidt, C. Fausto, and B. Franchetto. 2003. Amazonia 1492: Pristine Forest or Cultural Parkland? *Science* 301.

Heckenberger, M. J., J. Christian Russell, J. R. Toney, and M. J. Schmidt. 2007. The legacy of cultural landscapes in the Brazilian Amazon: implications for biodiversity. *Philosophical Transactions of the Royal Society B: Biological Sciences* 362: 197–208. doi:10.1098/rstb.2006.1979.

Heilmayr, R., L. L. Rausch, J. Munger, and H. K. Gibbs. 2020. Brazil's Amazon Soy Moratorium reduced deforestation. *Nature Food* 1: 801–810. doi:10.1038/s43016-020-00194-5.

Heinrich, V. H. A., R. Dalagnol, H. L. G. Cassol, T. M. Rosan, C. T. De Almeida, C. H. L. Silva Junior, W. A. Campanharo, J. I. House, et al. 2021. Large carbon sink potential of secondary forests in the Brazilian Amazon to mitigate climate change. *Nature Communications* 12. doi:10.1038/s41467-021-22050-1.

Herrera, D., A. Pfaff, and J. Robalino. 2019. Impacts of protected areas vary with the level of government: Comparing avoided deforestation across agencies in the Brazilian Amazon. *Proceedings of the National Academy of Sciences* 116: 14916–14925. doi:10.1073/pnas.1802877116.

Hethcoat, M. G., J. M. B. Carreiras, D. Edwards, R. G. Bryant, C. A. Peres, and S. Quegan. 2020. Mapping pervasive selective logging in the south-west Brazilian Amazon 2000–2019. *Environmental Research Letters* 15. doi:10.1088/1748-9326/aba3a4.

Hopkins, P. E. 2015. Positionalities and Knowledge: Negotiating Ethics in Practice. *ACME: An International Journal for Critical Geographies*. ACME: An International Journal for Critical Geographies: 386–394. doi:10.14288/ACME.V6I3.787.

Hubbell, S. P., F. He, R. Condit, L. Borda-de-Água, J. Kellner, and H. ter Steege. 2008. How many tree species are there in the Amazon and how many of them will go extinct? *Proceedings of the National Academy of Sciences* 105: 11498–11504. doi:10.1073/pnas.0801915105.

Huffman, M. R. 2013. The Many Elements of Traditional Fire Knowledge: Synthesis, Classification, and Aids to Cross-cultural Problem Solving in Fire-dependent Systems Around the World. *Ecology and Society* 18. doi:10.5751/ES-05843-180403.

International Society of Ethnobiology. 2006. International Society of Ethnobiology Code of Ethics (with 2008 additions).

Jacobson, L. D. S. V., S. D. S. Hacon, H. A. D. Castro, E. Ignotti, P. Artaxo, P. H. N. Saldiva, and A. C. M. P. De Leon. 2014. Acute Effects of Particulate Matter and Black Carbon from Seasonal Fires on Peak Expiratory Flow of Schoolchildren in the Brazilian Amazon. Edited by Qinghua Sun. *PLoS ONE* 9. doi:10.1371/journal.pone.0104177.

Jakimow, B., M. Baumann, C. Salomão, H. Bendini, and P. Hostert. 2023. Deforestation and agricultural fires in South-West Pará, Brazil, under political changes from 2014 to 2020. *Journal of Land Use Science* 18: 176–195. doi:10.1080/1747423X.2023.2195420.

Jakovac, C. C., M. Peña-Claros, R. C. G. Mesquita, F. Bongers, and T. W. Kuyper. 2016. Swiddens under transition: Consequences of agricultural intensification in the Amazon. *Agriculture, Ecosystems & Environment* 218: 116–125. doi:10.1016/j.agee.2015.11.013.

Janke, T., and M. Sentina. 2018. Indigenous Knowledge: Issues for Protection and Management. IP Australia, Commonwealth of Australia.

Jesus, S. C. de, and A. M. Z. Catojo. 2020. Deforestation in Conservation Units of the Brazilian Amazon: the case of the Terra do Meio Mosaic. *Ciência e Natura* 42: 1–23. doi:10.5902/2179460X41390.

Johannes, R. E. 2002. Did indigenous conservation ethics exist? *Traditional Marine Resource Management and Knowledge Information Bulletin* 2.

Johansson, M. U., S. D. Senay, E. Creathorn, H. Kassa, and K. Hylander. 2019. Change in heathland fire sizes inside vs. outside the Bale Mountains National Park, Ethiopia, over 50 years of fire-exclusion policy: lessons for REDD+. *Ecology and Society* 24. doi:10.5751/ES-11260-240426.

Jolly, W. M., M. A. Cochrane, P. H. Freeborn, Z. A. Holden, T. J. Brown, G. J. Williamson, and D. M. J. S. Bowman. 2015. Climate-induced variations in global wildfire danger from 1979 to 2013. *Nature Communications* 6. doi:10.1038/ncomms8537.

Jones, M. W., J. T. Abatzoglou, S. Veraverbeke, N. Andela, G. Lasslop, M. Forkel, A. J. P. Smith, C. Burton, et al. 2022. Global and Regional Trends and Drivers of Fire Under Climate Change. *Reviews of Geophysics* 60. doi:10.1029/2020RG000726.

Jones, M. W., S. Veraverbeke, N. Andela, S. H. Doerr, C. Kolden, G. Mataveli, M. L. Pettinari, C. Le Quéré, et al. 2024. Global rise in forest fire emissions linked to climate change in the extratropics. *Science* 386. doi:10.1126/science.adl5889.

Juan Verdoy, P. 2021. Enhancing the SPDE modeling of spatial point processes with INLA, applied to wildfires. Choosing the best mesh for each database. *Communications in Statistics - Simulation and Computation* 50: 2990–3030. doi:10.1080/03610918.2019.1618473.

Jusys, T. 2018. Changing patterns in deforestation avoidance by different protection types in the Brazilian Amazon. Edited by Asim Zia. *PLOS ONE* 13. doi:10.1371/journal.pone.0195900.

Kalamandeen, M., E. Gloor, E. Mitchard, D. Quincey, G. Ziv, D. Spracklen, B. Spracklen, M. Adami, et al. 2018. Pervasive Rise of Small-scale Deforestation in Amazonia. *Scientific Reports* 8. doi:10.1038/s41598-018-19358-2.

Kastens, J. H., J. C. Brown, A. C. Coutinho, C. R. Bishop, and J. C. D. M. Esquerdo. 2017. Soy moratorium impacts on soybean and deforestation dynamics in Mato Grosso, Brazil. Edited by Krishna Prasad Vadrevu. *PLOS ONE* 12. doi:10.1371/journal.pone.0176168.

Keeley, J. E., J. G. Pausas, P. W. Rundel, W. J. Bond, and R. A. Bradstock. 2011. Fire as an evolutionary pressure shaping plant traits. *Trends in Plant Science* 16: 406–411. doi:10.1016/j.tplants.2011.04.002.

Keles, D., P. Delacote, A. Pfaff, S. Qin, and M. B. Mascia. 2020. What Drives the Erasure of Protected Areas? Evidence from across the Brazilian Amazon. *Ecological Economics* 176. doi:10.1016/j.ecolecon.2020.106733.

Kelly, L. T., K. M. Giljohann, A. Duane, N. Aquilué, S. Archibald, E. Batllori, A. F. Bennett, S. T. Buckland, et al. 2020. Fire and biodiversity in the Anthropocene. *Science* 370. doi:10.1126/science.abb0355.

Kelly, S. E. 2010. Qualitative Interviewing Techniques and Styles. In *The SAGE Handbook of Qualitative Methods in Health Research*, I. Bourgeault, R. Dingwall, and R. De Vries, 307–326. 1 Oliver's Yard, 55 City Road, London EC1Y 1SP United Kingdom: SAGE Publications Ltd. doi:10.4135/9781446268247.n17.

Klingler, M., P. D. Richards, and R. Ossner. 2018. Cattle vaccination records question the impact of recent zero-deforestation agreements in the Amazon. *Regional Environmental Change* 18: 33–46. doi:10.1007/s10113-017-1234-1.

Krebs, P., G. B. Pezzatti, S. Mazzoleni, L. M. Talbot, and M. Conedera. 2010. Fire regime: history and definition of a key concept in disturbance ecology. *Theory in Biosciences* 129: 53–69. doi:10.1007/s12064-010-0082-z.

Kuschnig, N., J. C. Cuaresma, T. Krisztin, and S. Giljum. 2021. Spatial spillover effects from agriculture drive deforestation in Mato Grosso, Brazil. *Scientific Reports* 11. doi:10.1038/s41598-021-00861-y.

Lapola, D. M., J. M. C. D. Silva, D. R. Braga, L. Carpigiani, F. Ogawa, R. R. Torres, L. C. F. Barbosa, J. P. H. B. Ometto, et al. 2020. A climate-change vulnerability and adaptation assessment for Brazil's protected areas. *Conservation Biology* 34: 427–437. doi:10.1111/cobi.13405.

Laris, P. 2002. Burning the Seasonal Mosaic: Preventative Burning Strategies in the Wooded Savanna of Southern Mali. *Human Ecology* 30: 155–186.

Laurance, W. F., A. K. M. Albernaz, G. Schroth, P. M. Fearnside, S. Bergen, E. M. Venticinque, and C. Da Costa. 2002. Predictors of deforestation in the Brazilian Amazon. *Journal of Biogeography* 29: 737–748. doi:10.1046/j.1365-2699.2002.00721.x.

Le Page, Y., D. Morton, C. Hartin, B. Bond-Lamberty, J. M. C. Pereira, G. Hurtt, and G. Asrar. 2017. Synergy between land use and climate change increases future fire risk in Amazon forests. *Earth System Dynamics* 8: 1237–1246. doi:10.5194/esd-8-1237-2017.

Levis, C., B. M. Flores, P. A. Moreira, B. G. Luize, R. P. Alves, J. Franco-Moraes, J. Lins, E. Konings, et al. 2018. How People Domesticated Amazonian Forests. *Frontiers in Ecology and Evolution* 5: 171. doi:10.3389/fevo.2017.00171.

Libonati, R., J. M. C. Pereira, C. C. Da Camara, L. F. Peres, D. Oom, J. A. Rodrigues, F. L. M. Santos, R. M. Trigo, et al. 2021. Twenty-first century droughts have not increasingly exacerbated fire season severity in the Brazilian Amazon. *Scientific Reports* 11. doi:10.1038/s41598-021-82158-8.

Lindgren, F., H. Rue, and J. Lindström. 2011. An explicit link between Gaussian fields and Gaussian Markov random fields: the stochastic partial differential equation approach: Link between Gaussian Fields and Gaussian Markov Random Fields. *Journal of the Royal Statistical Society: Series B (Statistical Methodology)* 73: 423–498. doi:10.1111/j.1467-9868.2011.00777.x.

Linley, G. D., C. J. Jolly, T. S. Doherty, W. L. Geary, D. Armenteras, C. M. Belcher, R. Bliege Bird, A. Duane, et al. 2022. What do you mean, 'megafire'? *Global Ecology and Biogeography* 31: 1906–1922. doi:10.1111/geb.13499.

Lopes, A., L. O. Demarchi, M. T. F. Piedade, J. Schöngart, F. Wittmann, C. B. R. Munhoz, C. S. Ferreira, and A. C. Franco. 2023. Predicting the range expansion of invasive alien grasses under climate change in the Neotropics. *Perspectives in Ecology and Conservation* 21: 128–135. doi:10.1016/j.pecon.2023.02.005.

Lyver, P. O., J. M. Wilmshurst, J. R. Wood, C. J. Jones, M. Fromont, P. J. Bellingham, C. Stone, M. Sheehan, et al. 2015. Looking Back for the Future: Local Knowledge and Palaeoecology Inform Biocultural Restoration of Coastal Ecosystems in New Zealand. *Human Ecology* 43: 681–695. doi:10.1007/s10745-015-9784-7.

Ma, J., J. Li, W. Wu, and J. Liu. 2023. Global forest fragmentation change from 2000 to 2020. *Nature Communications* 14. doi:10.1038/s41467-023-39221-x.

Macedo, M. N., R. S. DeFries, D. C. Morton, C. M. Stickler, G. L. Galford, and Y. E. Shimabukuro. 2012. Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. *Proceedings of the National Academy of Sciences* 109: 1341–1346. doi:10.1073/pnas.1111374109.

Machado, M. S., E. Berenguer, P. M. Brando, A. Alencar, I. Oliveras Menor, J. Barlow, and Y. Malhi. 2024. Emergency policies are not enough to resolve Amazonia's fire crises. *Communications Earth & Environment* 5. doi:10.1038/s43247-024-01344-4.

Machado-Silva, F., L. F. Peres, C. M. Gouveia, A. Enrich-Prast, R. B. Peixoto, J. M. C. Pereira, H. Marotta, P. J. F. Fernandes, et al. 2021. Drought Resilience Debt Drives NPP Decline in the Amazon Forest. *Global Biogeochemical Cycles* 35. doi:10.1029/2021GB007004.

Maclean, K., D. L. Hankins, A. C. Christianson, I. Oliveras, B. A. Bilbao, O. Costello, E. R. Langer, and C. J. Robinson. 2023. Revitalising Indigenous cultural fire practice: benefits and partnerships. *Trends in Ecology & Evolution* 38: 899–902. doi:10.1016/j.tree.2023.07.001.

Maezumi, S. Y., M. Robinson, J. de Souza, D. H. Urrego, D. Schaan, D. Alves, and J. Iriarte. 2018. New Insights From Pre-Columbian Land Use and Fire Management in Amazonian Dark Earth Forests. *Frontiers in Ecology and Evolution* 6. doi:10.3389/fevo.2018.00111.

Mano, G. B., A. Lopes, and M. T. F. Piedade. 2023. Will climate change favor exotic grasses over native ecosystem engineer species in the Amazon Basin? *Ecological Informatics* 75. doi:10.1016/j.ecoinf.2023.102102.

Marques, E. Q., B. H. Marimon-Junior, B. S. Marimon, E. A. T. Matricardi, H. A. Mews, and G. R. Colli. 2020. Redefining the Cerrado–Amazonia transition: implications for conservation. *Biodiversity and Conservation* 29: 1501–1517. doi:10.1007/s10531-019-01720-z.

Mataveli, G., M. W. Jones, R. Carmenta, A. Sanchez, D. J. Dutra, M. Chaves, G. de Oliveira, L. O. Anderson, et al. 2024. Deforestation falls but rise of wildfires continues degrading Brazilian Amazon forests. *Global Change Biology* 30. doi:10.1111/gcb.17202.

Matricardi, E. A. T., D. L. Skole, M. A. Pedlowski, and W. Chomentowski. 2013. Assessment of forest disturbances by selective logging and forest fires in the Brazilian Amazon using Landsat data. *International Journal of Remote Sensing* 34: 1057–1086. doi:10.1080/01431161.2012.717182.

McGarigal, K., S. A. Cushman, and E. Ene. 2023. FRAGSTATS v4: Spatial Pattern Analysis Program for Categorical Maps.

McKemey, M. B., Banbai Rangers, E. J. Ens, J. T. Hunter, M. Ridges, O. Costello, and N. C. H. Reid. 2021. Co-producing a fire and seasons calendar to support renewed Indigenous cultural fire management. *Austral Ecology* 46: 1011–1029. doi:10.1111/aec.13034.

McKemey, M. B., B. Rangers, Y. M. Rangers, O. Costello, J. T. Hunter, and E. J. Ens. 2022. 'Right-way' science: reflections on co-developing Indigenous and Western cross-cultural knowledge to support Indigenous cultural fire management. *Ecological Management & Restoration* 23: 75–82. doi:10.1111/emr.12532.

McKight, P. E., and J. Najab. 2010. Kruskal-Wallis Test. In *The Corsini Encyclopedia of Psychology*, ed. I. B. Weiner and W. E. Craighead, 1st ed., 1–1. Wiley. doi:10.1002/9780470479216.corpsy0491.

McWethy, D. B. 2019. Rethinking resilience to wildfire. *Nature Sustainability* 2: 797–804.

Mello-Théry, N. A. D., E. De Lima Caldas, B. M. Funatsu, D. Arvor, and V. Dubreuil. 2020. Climate Change and Public Policies in the Brazilian Amazon State of Mato Grosso: Perceptions and Challenges. *Sustainability* 12. doi:10.3390/su12125093.

Mendes-Oliveira, A. C., C. A. Peres, P. C. R. D. A. Maués, G. L. Oliveira, I. G. B. Mineiro, S. L. S. De Maria, and R. C. S. Lima. 2017. Oil palm monoculture induces drastic erosion of an Amazonian forest mammal fauna. Edited by Danilo Russo. *PLOS ONE* 12. doi:10.1371/journal.pone.0187650.

de Mendonça, M. J. C., M. del C. Vera Diaz, D. Nepstad, R. Seroa da Motta, A. Alencar, J. C. Gomes, and R. A. Ortiz. 2004. The economic cost of the use of fire in the Amazon. *Ecological Economics* 49: 89–105. doi:10.1016/j.ecolecon.2003.11.011.

Menezes, R. G., and R. Barbosa. 2021. Environmental governance under Bolsonaro: dismantling institutions, curtailing participation, delegitimising opposition. *Zeitschrift für Vergleichende Politikwissenschaft* 15: 229–247. doi:10.1007/s12286-021-00491-8.

Mistry, J., and M. Bizerril. 2011. Why It is Important to Understand the Relationship Between People, Fire and Protected Areas. *Biodiversidade Brasileira - BioBrasil* 2: 40–49.

Mistry, J., A. Berardi, V. Andrade, T. Krahô, P. Krahô, and O. Leonardos. 2005. Indigenous Fire Management in the cerrado of Brazil: The Case of the Krahô of Tocantíns. *Human Ecology* 33: 365–386. doi:10.1007/s10745-005-4143-8.

Mistry, J., A. Berardi, M. Simpson, O. Davis, and L. Haynes. 2010. Using a systems viability approach to evaluate integrated conservation and development projects: assessing the impact of the North Rupununi Adaptive Management Process, Guyana. *The Geographical Journal* 176: 241–252. doi:10.1111/j.1475-4959.2010.00357.x.

Mistry, J., A. Berardi, E. Bignante, and C. Tschirhart. 2015. Between a rock and a hard place: Ethical dilemmas of local community facilitators doing participatory research projects. *Geoforum* 61: 27–35. doi:10.1016/j.geoforum.2015.02.010.

Mistry, J., I. B. Schmidt, L. Eloy, and B. Bilbao. 2019. New perspectives in fire management in South American savannas: The importance of intercultural governance. *Ambio* 48: 172–179. doi:10.1007/s13280-018-1054-7.

Montibeller, B., A. Kmoch, H. Virro, Ü. Mander, and E. Uuemaa. 2020. Increasing fragmentation of forest cover in Brazil's Legal Amazon from 2001 to 2017. *Scientific Reports* 10. doi:10.1038/s41598-020-62591-x.

Moon, K., and D. Blackman. 2014. A Guide to Understanding Social Science Research for Natural Scientists: Social Science for Natural Scientists. *Conservation Biology* 28: 1167–1177. doi:10.1111/cobi.12326.

Moore, P. F. 2019. Global Wildland Fire Management Research Needs. *Current Forestry Reports* 5: 210–225. doi:10.1007/s40725-019-00099-y.

Morello, T., and L. Falcão. 2020. The Fire Management Dilemma in the Brazilian Amazon: Synthesizing Pathways of Causality across Five Case Studies in the State of Pará. *Human Ecology* 48: 397–409. doi:10.1007/s10745-020-00166-0.

Morello, T. F., L. Parry, N. Markusson, and J. Barlow. 2017. Policy instruments to control Amazon fires: A simulation approach. *Ecological Economics* 138: 199–222. doi:10.1016/j.ecolecon.2017.03.043.

Morgan, W. T., E. Darbyshire, D. V. Spracklen, P. Artaxo, and H. Coe. 2019. Non-deforestation drivers of fires are increasingly important sources of aerosol and carbon dioxide emissions across Amazonia. *Scientific Reports* 9. doi:10.1038/s41598-019-53112-6.

Morton, D. C., R. S. Defries, J. T. Randerson, L. Giglio, W. Schroeder, and G. R. Van Der Werf. 2008. Agricultural intensification increases deforestation fire activity in Amazonia: Deforestation Fires in Amazonia. *Global Change Biology* 14: 2262–2275. doi:10.1111/j.1365-2486.2008.01652.x.

Morton, D. C., Y. Le Page, R. DeFries, G. J. Collatz, and G. C. Hurtt. 2013. Understorey fire frequency and the fate of burned forests in southern Amazonia. *Philosophical Transactions of the Royal Society B: Biological Sciences* 368. doi:10.1098/rstb.2012.0163.

Myers, R. 2006. *Living with fire - sustaining ecosystems and livelihoods through integrated fire management*. The Nature Conservancy. Tallahassee, Florida.

Nelson, A., and K. M. Chomitz. 2011. Effectiveness of Strict vs. Multiple Use Protected Areas in Reducing Tropical Forest Fires: A Global Analysis Using Matching Methods. Edited by Hans Henrik Bruun. *PLoS ONE* 6. doi:10.1371/journal.pone.0022722.

Nepstad, D., S. Schwartzman, B. Bamberger, M. Santilli, D. Ray, P. Schlesinger, P. Lefebvre, A. Alencar, et al. 2006. Inhibition of Amazon Deforestation and Fire by Parks and Indigenous Lands: Inhibition of Amazon Deforestation and Fire. *Conservation Biology* 20: 65–73. doi:10.1111/j.1523-1739.2006.00351.x.

Nepstad, D., D. McGrath, C. Stickler, A. Alencar, A. Azevedo, B. Swette, T. Bezerra, M. DiGiano, et al. 2014. Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. *Science* 344: 1118–1123. doi:10.1126/science.1248525.

Nepstad, D. C., C. M. Stickler, B. S.- Filho, and F. Merry. 2008. Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363: 1737–1746. doi:10.1098/rstb.2007.0036.

Newing, H., C. Eagle, P. Rajindra, and C. Watson. 2011. *Conducting research in conservation: social science methods and practice*. New York: Routledge. doi:10.4324/9780203846452.

Newing, H., S. Brittain, A. Buchadas, O. Del Giorgio, C. F. Grasham, R. Ferritto, J. R. Garcia Marquez, M. Khanyari, et al. 2024. 'Participatory' conservation research involving indigenous peoples and local communities: Fourteen principles for good practice. *Biological Conservation* 296. doi:10.1016/j.biocon.2024.110708.

Nikolakis, W. D., and E. Roberts. 2020. Indigenous fire management: a conceptual model from literature. *Ecology and Society* 25. doi:10.5751/ES-11945-250411.

Nóbrega Spínola, J., M. J. Soares da Silva, J. R. Assis da Silva, J. Barlow, and J. Ferreira. 2020. A shared perspective on managing Amazonian sustainable-use reserves in an era of megafires. Edited by Alexandro B. Leverkus. *Journal of Applied Ecology* 57: 2132–2138. doi:10.1111/1365-2664.13690.

Nogueira, E. M., A. M. Yanai, S. S. de Vasconcelos, P. M. L. de Alencastro Graça, and P. M. Fearnside. 2018. Carbon stocks and losses to deforestation in protected areas in Brazilian Amazonia. *Regional Environmental Change* 18: 261–270. doi:10.1007/s10113-017-1198-1.

Nolte, C., A. Agrawal, K. M. Silvius, and B. S. Soares-Filho. 2013. Governance regime and location influence avoided deforestation success of protected areas in the Brazilian Amazon. *Proceedings of the National Academy of Sciences* 110: 4956–4961. doi:10.1073/pnas.1214786110.

Numata, I., S. S. Silva, M. A. Cochrane, and M. V. N. d'Oliveira. 2017. Fire and edge effects in a fragmented tropical forest landscape in the southwestern Amazon. *Forest Ecology and Management* 401: 135–146. doi:10.1016/j.foreco.2017.07.010.

Nunes, C. A., E. Berenguer, F. França, J. Ferreira, A. C. Lees, J. Louzada, E. J. Sayer, R. Solar, et al. 2022. Linking land-use and land-cover transitions to their ecological impact in the Amazon. *Proceedings of the National Academy of Sciences* 119. doi:10.1073/pnas.2202310119.

Oliveira, A. S., B. S. Soares-Filho, U. Oliveira, R. Van der Hoff, S. M. Carvalho-Ribeiro, A. R. Oliveira, L. C. Scheepers, B. A. Vargas, et al. 2021. Costs and effectiveness of public and

private fire management programs in the Brazilian Amazon and Cerrado. *Forest Policy and Economics* 127. doi:10.1016/j.forpol.2021.102447.

Oliveira, E. D. S. 2017. A terra (vivida) em movimento : nomeação de lugares e a luta Mẽtyktire-Mẽbêngôkre (Kayapó). Master, Universidade de Brasília. doi:10.26512/2017.09.D.31300.

de Oliveira, G., J. M. Chen, G. A. V. Mataveli, M. E. D. Chaves, H. T. Seixas, F. da S. Cardozo, Y. E. Shimabukuro, L. He, et al. 2020. Rapid Recent Deforestation Incursion in a Vulnerable Indigenous Land in the Brazilian Amazon and Fire-Driven Emissions of Fine Particulate Aerosol Pollutants. *Forests* 11. doi:10.3390/f11080829.

Oliveira, M. R., B. H. S. Ferreira, E. B. Souza, A. A. Lopes, F. P. Bolzan, F. O. Roque, A. Pott, A. M. M. Pereira, et al. 2022. Indigenous brigades change the spatial patterns of wildfires, and the influence of climate on fire regimes. *Journal of Applied Ecology* 59: 1279–1290. doi:10.1111/1365-2664.14139.

Oliveras Menor, I., N. Prat-Guitart, G. L. Spadoni, A. Hsu, P. M. Fernandes, R. Puig-Gironès, D. Ascoli, B. A. Bilbao, et al. 2025. Integrated fire management as an adaptation and mitigation strategy to altered fire regimes. *Communications Earth & Environment* 6. doi:10.1038/s43247-025-02165-9.

Opitz, T., F. Bonneu, and E. Gabriel. 2020. Point-process based Bayesian modeling of space–time structures of forest fire occurrences in Mediterranean France. *Spatial Statistics* 40. doi:10.1016/j.spasta.2020.100429.

PADDtracker. 2020. PADDDtracker. *PADDDtracker*.

Parry, I. M., P. D. L. Ritchie, and P. M. Cox. 2022. Evidence of localised Amazon rainforest dieback in CMIP6 models. *Earth System Dynamics* 13: 1667–1675. doi:10.5194/esd-13-1667-2022.

Pausas, J. G., and J. E. Keeley. 2014. Abrupt Climate-Independent Fire Regime Changes. *Ecosystems* 17: 1109–1120. doi:10.1007/s10021-014-9773-5.

Pausas, J. G., and J. E. Keeley. 2021. Wildfires and global change. *Frontiers in Ecology and the Environment* 19: 387–395. doi:10.1002/fee.2359.

Pelicice, F. M., and L. Castello. 2021. A political tsunami hits Amazon conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems* 31: 1221–1229. doi:10.1002/aqc.3565.

Pereira, A. S. A. de P., V. J. dos Santos, S. do C. Alves, A. Amaral e Silva, C. G. da Silva, and M. L. Calijuri. 2022. Contribution of rural settlements to the deforestation dynamics in the Legal Amazon. *Land Use Policy* 115. doi:10.1016/j.landusepol.2022.106039.

Perkins, O., M. Kasoar, A. Voulgarakis, T. Edwards, O. Haas, and J. D. A. Millington. 2025. The Spatial Distribution and Temporal Drivers of Changing Global Fire Regimes: A Coupled Socio-Ecological Modeling Approach. *Earth's Future* 13. American Geophysical Union (AGU). doi:10.1029/2024ef004770.

Perry, J. J., M. Sinclair, H. Wikmunea, S. Wolmby, D. Martin, and B. Martin. 2018. The divergence of traditional Aboriginal and contemporary fire management practices on Wik traditional lands, Cape York Peninsula, Northern Australia. *Ecological Management & Restoration* 19: 24–31. doi:10.1111/emr.12301.

Petty, A. M., V. deKoninck, and B. Orlove. 2015. Cleaning, Protecting, or Abating? Making Indigenous Fire Management "Work" in Northern Australia. *Journal of Ethnobiology* 35: 140–162. doi:10.2993/0278-0771-35.1.140.

Petzold, J., N. Andrews, J. D. Ford, C. Hedemann, and J. C. Postigo. 2020. Indigenous knowledge on climate change adaptation: a global evidence map of academic literature. *Environmental Research Letters* 15. doi:10.1088/1748-9326/abb330.

Pivello, V. R. 2011. The use of fire in the cerrado and Amazonian rainforests of Brazil: past and present. *Fire Ecology* 7: 16.

Pokorny, B., P. Pacheco, W. de Jong, and S. K. Entenmann. 2021. Forest frontiers out of control: The long-term effects of discourses, policies, and markets on conservation and development of the Brazilian Amazon. *Ambio* 50: 2199–2223. doi:10.1007/s13280-021-01637-4.

Posey, D. A. 1985. Indigenous management of tropical forest ecosystems: the case of the Kayap indians of the Brazilian Amazon. *Agroforestry Systems* 3: 139–158. doi:10.1007/BF00122640.

Pratzer, M., Á. Fernández-Llamazares, P. Meyfroidt, T. Krueger, M. Baumann, S. T. Garnett, and T. Kuemmerle. 2023. Agricultural intensification, Indigenous stewardship and land sparing in tropical dry forests. *Nature Sustainability*. doi:10.1038/s41893-023-01073-0.

Prestes, N. C. C. D. S., K. G. Massi, E. A. Silva, D. S. Nogueira, E. A. De Oliveira, R. Freitag, B. S. Marimon, B. H. Marimon-Junior, et al. 2020. Fire Effects on Understory Forest Regeneration in Southern Amazonia. *Frontiers in Forests and Global Change* 3. doi:10.3389/ffgc.2020.00010.

PRODES. 2022. Coordenação-Geral de Observação da Terra.

Purnomo, E. P., R. Ramdani, Agustiyara, Q. P. V. Tomaro, and G. S. Samidjo. 2019. Land ownership transformation before and after forest fires in Indonesian palm oil plantation areas. *Journal of Land Use Science* 14: 37–51. doi:10.1080/1747423X.2019.1614686.

R Core Team. 2022. R: A language and environment for statistical computing (version V4.1). Vienna, Austria: R Foundation for Statistical Computing.

Ray, D., D. Nepstad, and P. Moutinho. 2005. Micromeeteorological and Canopy Controls of Fire Susceptibility in a Forested Amazon Landscape. *Ecological Applications* 15: 1664–1678. doi:10.1890/05-0404.

Ray, L. A., C. A. Kolden, and F. S. Chapin Iii. 2012. A Case for Developing Place-Based Fire Management Strategies from Traditional Ecological Knowledge. *Ecology and Society* 17. doi:10.5751/ES-05070-170337.

Reid, C. E., M. Brauer, F. H. Johnston, M. Jerrett, J. R. Balmes, and C. T. Elliott. 2016. Critical Review of Health Impacts of Wildfire Smoke Exposure. *Environmental Health Perspectives* 124: 1334–1343. doi:10.1289/ehp.1409277.

dos Reis, M., P. M. L. de A. Graça, A. M. Yanai, C. J. P. Ramos, and P. M. Fearnside. 2021. Forest fires and deforestation in the central Amazon: Effects of landscape and climate on spatial and temporal dynamics. *Journal of Environmental Management* 288. doi:10.1016/j.jenvman.2021.112310.

Reydon, B. P., V. B. Fernandes, and T. S. Telles. 2020. Land governance as a precondition for decreasing deforestation in the Brazilian Amazon. *Land Use Policy* 94. doi:10.1016/j.landusepol.2019.104313.

Reyes-García, V., J. Paneque-Gálvez, A. C. Luz, M. Gueze, M. J. Macía, M. Orta-Martínez, and J. Pino. 2014. Cultural Change and Traditional Ecological Knowledge: An Empirical Analysis from the Tsimane' in the Bolivian Amazon. *Human Organization* 73: 162–173. doi:10.17730/humo.73.2.31nl363qgr30n017.

Reyes-García, V., M. Guèze, I. Díaz-Reviriego, R. Duda, Á. Fernández-Llamazares, S. Gallois, L. Napitupulu, M. Orta-Martínez, et al. 2016. The Adaptive Nature of Culture: A Cross-Cultural Analysis of the Returns of Local Environmental Knowledge in Three Indigenous Societies. *Current Anthropology* 57: 761–784. doi:10.1086/689307.

Reyes-García, V., D. García-del-Amo, S. Álvarez-Fernández, P. Benyei, L. Calvet-Mir, A. B. Junqueira, V. Labeyrie, X. Li, et al. 2024. Indigenous Peoples and local communities report ongoing and widespread climate change impacts on local social-ecological systems. *Communications Earth & Environment* 5. doi:10.1038/s43247-023-01164-y.

Ribeiro, A. F. S., L. Santos, J. T. Randerson, M. R. Uribe, A. A. C. Alencar, M. N. Macedo, D. C. Morton, J. Zscheischler, et al. 2024. The time since land-use transition drives changes in fire activity in the Amazon-Cerrado region. *Communications Earth & Environment* 5. doi:10.1038/s43247-024-01248-3.

Ribeiro, L. M., A. Rodrigues, D. Lucas, and D. X. Viegas. 2020. The Impact on Structures of the Pedrógão Grande Fire Complex in June 2017 (Portugal). *Fire* 3. doi:10.3390/fire3040057.

Ribeiro, M. R., M. V. M. Lima, R. C. Ilacqua, E. J. L. Savoia, R. Alvarenga, A. Y. Vittor, R. D. Raimundo, and G. Z. Laporta. 2024. Amazon Wildfires and Respiratory Health: Impacts during the Forest Fire Season from 2009 to 2019. *International Journal of Environmental Research and Public Health* 21. doi:10.3390/ijerph21060675.

Richards, P., E. Arima, L. VanWey, A. Cohn, and N. Bhattarai. 2017. Are Brazil's Deforesters Avoiding Detection? *Conservation Letters* 10: 470–476. doi:10.1111/conl.12310.

de Robert, P., C. L. Garcés, A.-E. Laques, and M. Coelho-Ferreira. 2012. A beleza das roças: agrobiodiversidade Mebêngôkre-Kayapó em tempos de globalização Beautiful gardens: agrobiodiversity Mebêngôkre-Kayapó in globalization times. *Boletim do Museu Paraense Emílio Goeldi. Ciências Humanas* 7.

Rocha, R., and A. A. Sant'Anna. 2022. Winds of fire and smoke: Air pollution and health in the Brazilian Amazon. *World Development* 151. doi:10.1016/j.worlddev.2021.105722.

Rodrigues, A. M. 2022. *O Kayapó (Mẽtyktire) e o fogo: narrativas e práticas observadas no tempo e no espaço*. 1st ed. Atena Editora. doi:10.22533/at.ed.395221609.

Rodríguez, I. 2007. Pemon Perspectives of Fire Management in Canaima National Park, Southeastern Venezuela. *Human Ecology* 35: 331–343. doi:10.1007/s10745-006-9064-7.

Rogers, B. M., J. K. Balch, S. J. Goetz, C. E. R. Lehmann, and M. Turetsky. 2020. Focus on changing fire regimes: interactions with climate, ecosystems, and society. *Environmental Research Letters* 15. doi:10.1088/1748-9326/ab6d3a.

Roos, C. I., A. C. Scott, C. M. Belcher, W. G. Chaloner, J. Aylen, R. B. Bird, M. R. Coughlan, B. R. Johnson, et al. 2016. Living on a flammable planet: interdisciplinary, cross-scalar and varied cultural lessons, prospects and challenges. *Philosophical Transactions of the Royal Society B: Biological Sciences* 371. doi:10.1098/rstb.2015.0469.

Rorato, A. C., M. C. A. Picoli, J. A. Verstegen, G. Camara, F. G. Silva Bezerra, and M. I. S. Escada. 2021. Environmental Threats over Amazonian Indigenous Lands. *Land* 10. doi:10.3390/land10030267.

Rosa, I. M. D., C. Souza, and R. M. Ewers. 2012. Changes in Size of Deforested Patches in the Brazilian Amazon: Dynamics of Amazonian Deforestation. *Conservation Biology* 26: 932–937. doi:10.1111/j.1523-1739.2012.01901.x.

Roue, M., and D. Nakashima. 2018. Indigenous and Local Knowledge and Science: From Validation to Knowledge Coproduction. In *The International Encyclopedia of*

Anthropology, ed. H. Callan, 1st ed., 1–11. Wiley. doi:10.1002/9781118924396.wbiea2215.

Ruiz-Mallén, I., and E. Corbera. 2013. Community-Based Conservation and Traditional Ecological Knowledge: Implications for Social-Ecological Resilience. *Ecology and Society* 18. doi:10.5751/ES-05867-180412.

Russell-Smith, J., G. D. Cook, P. M. Cooke, A. C. Edwards, M. Lendrum, C. (Mick) Meyer, and P. J. Whitehead. 2013. Managing fire regimes in north Australian savannas: applying Aboriginal approaches to contemporary global problems. *Frontiers in Ecology and the Environment* 11. doi:10.1890/120251.

Russell-Smith, J., C. Monagle, M. Jacobsohn, R. L. Beatty, B. Bilbao, A. Millán, H. Vessuri, and I. Sánchez-Rose. 2017. Can savanna burning projects deliver measurable greenhouse emissions reductions and sustainable livelihood opportunities in fire-prone settings? *Climatic Change* 140: 47–61. doi:10.1007/s10584-013-0910-5.

Rutherford, T. K., and C. A. Schultz. 2019. Adapting wildland fire governance to climate change in Alaska. *Ecology and Society* 24. doi:10.5751/ES-10810-240127.

Santana, M. M. M. D., R. N. D. Vasconcelos, and E. Mariano-Neto. 2022. Fire propensity in Amazon savannas and rainforest and effects under future climate change. *International Journal of Wildland Fire* 32: 149–163. doi:10.1071/WF21174.

Santos, A. M. D., C. F. A. D. Silva, P. M. D. Almeida Junior, A. P. Rudke, and S. N. D. Melo. 2021. Deforestation drivers in the Brazilian Amazon: assessing new spatial predictors. *Journal of Environmental Management* 294. doi:10.1016/j.jenvman.2021.113020.

Santos, A. M. dos, C. F. A. da Silva, A. P. Rudke, and D. de Oliveira Soares. 2021. Dynamics of active fire data and their relationship with fires in the areas of regularized indigenous lands in the Southern Amazon. *Remote Sensing Applications: Society and Environment* 23. doi:10.1016/j.rsase.2021.100570.

Sathler, D., S. B. Adamo, and E. E. C. Lima. 2018. Deforestation and local sustainable development in Brazilian Legal Amazonia: an exploratory analysis. *Ecology and Society* 23. doi:10.5751/ES-10062-230230.

Savo, V., D. Lepofsky, J. P. Benner, K. E. Kohfeld, J. Bailey, and K. Lertzman. 2016. Observations of climate change among subsistence-oriented communities around the world. *Nature Climate Change* 6: 462–473. doi:10.1038/nclimate2958.

Schelly, I. H., L. L. Rausch, and H. K. Gibbs. 2021. Brazil's Cattle Sector Played Large Role in Fires During 2020 Moratorium. *Frontiers in Forests and Global Change* 4. doi:10.3389/ffgc.2021.760853.

Schielein, J., and J. Börner. 2018. Recent transformations of land-use and land-cover dynamics across different deforestation frontiers in the Brazilian Amazon. *Land Use Policy* 76: 81–94. doi:10.1016/j.landusepol.2018.04.052.

Schinko, T., C. Berchtold, J. Handmer, T. Deubelli-Hwang, E. Preinfalk, J. Linnerooth-Bayer, A. Scolobig, M. Serra, et al. 2023. A framework for considering justice aspects in integrated wildfire risk management. *Nature Climate Change*. doi:10.1038/s41558-023-01726-0.

Schmidt, I. B., C. B. Fonseca, M. C. Ferreira, and M. N. Sato. 2016. Implementação do Programa Piloto de Manejo Integrado do Fogo em três Unidades de Conservação do Cerrado. *Biodiversidade Brasileira - BioBrasil* 6.

Schmidt, I. B., L. C. Moura, M. C. Ferreira, L. Eloy, A. B. Sampaio, P. A. Dias, and C. N. Berlinck. 2018. Fire management in the Brazilian savanna: First steps and the way forward. Edited

by Rafael Zenni. *Journal of Applied Ecology* 55: 2094–2101. doi:10.1111/1365-2664.13118.

Schmidt, M. V. C., Y. U. Ikpeng, T. Kayabi, R. A. Sanches, K. Y. Ono, and C. Adams. 2021. Indigenous Knowledge and Forest Succession Management in the Brazilian Amazon: Contributions to Reforestation of Degraded Areas. *Frontiers in Forests and Global Change* 4. doi:10.3389/ffgc.2021.605925.

Schmink, M., J. Hoelle, C. V. A. Gomes, and G. M. Thaler. 2019. From contested to 'green' frontiers in the Amazon? A long-term analysis of São Félix do Xingu, Brazil. *The Journal of Peasant Studies* 46: 377–399. doi:10.1080/03066150.2017.1381841.

Schneider, M., and C. A. Peres. 2015. Environmental Costs of Government-Sponsored Agrarian Settlements in Brazilian Amazonia. Edited by RunGuo Zang. *PLOS ONE* 10. doi:10.1371/journal.pone.0134016.

Schwartzman, S., and B. Zimmerman. 2005. Conservation Alliances with Indigenous Peoples of the Amazon. *Conservation Biology* 19: 721–727. doi:10.1111/j.1523-1739.2005.00695.x.

Schwartzman, S., A. Alencar, H. Zarin, and A. P. Santos Souza. 2010. Social Movements and Large-Scale Tropical Forest Protection on the Amazon Frontier: Conservation From Chaos. *The Journal of Environment & Development* 19: 274–299. doi:10.1177/1070496510367627.

Schwartzman, S., A. V. Boas, K. Y. Ono, M. G. Fonseca, J. Doblas, B. Zimmerman, P. Junqueira, A. Jerozolimski, et al. 2013. The natural and social history of the indigenous lands and protected areas corridor of the Xingu River basin. *Philosophical Transactions of the Royal Society B: Biological Sciences* 368. doi:10.1098/rstb.2012.0164.

Serra, L., M. Saez, J. Mateu, D. Varga, P. Juan, C. Díaz-Ávalos, and H. Rue. 2014. Spatio-temporal log-Gaussian Cox processes for modelling wildfire occurrence: the case of Catalonia, 1994–2008. *Environmental and Ecological Statistics* 21: 531–563. doi:10.1007/s10651-013-0267-y.

Sherry, J., T. Neale, T. K. McGee, and M. Sharpe. 2019. Rethinking the maps: A case study of knowledge incorporation in Canadian wildfire risk management and planning. *Journal of Environmental Management* 234: 494–502. doi:10.1016/j.jenvman.2018.12.116.

Shuman, J. K., J. K. Balch, R. T. Barnes, P. E. Higuera, C. I. Roos, D. W. Schwilk, E. N. Stavros, T. Banerjee, et al. 2022. Reimagine fire science for the anthropocene. Edited by Karen E Nelson. *PNAS Nexus* 1. doi:10.1093/pnasnexus/pgac115.

Silva, C. F. A., S. T. Alvarado, A. M. Santos, M. O. Andrade, and S. N. Melo. 2022. Highway Network and Fire Occurrence in Amazonian Indigenous Lands. *Sustainability* 14. doi:10.3390/su14159167.

Silva, C. V. J., L. E. O. C. Aragão, J. Barlow, F. Espirito-Santo, P. J. Young, L. O. Anderson, E. Berenguer, I. Brasil, et al. 2018. Drought-induced Amazonian wildfires instigate a decadal-scale disruption of forest carbon dynamics. *Philosophical Transactions of the Royal Society B: Biological Sciences* 373. doi:10.1098/rstb.2018.0043.

Silva Junior, C., L. Aragão, M. Fonseca, C. Almeida, L. Vedovato, and L. Anderson. 2018. Deforestation-Induced Fragmentation Increases Forest Fire Occurrence in Central Brazilian Amazonia. *Forests* 9. doi:10.3390/f9060305.

Silva-Junior, C. H. L., A. T. M. Buna, D. S. Bezerra, O. S. Costa, A. L. Santos, L. O. D. Basson, A. L. S. Santos, S. T. Alvarado, et al. 2022. Forest Fragmentation and Fires in the Eastern Brazilian Amazon–Maranhão State, Brazil. *Fire* 5. doi:10.3390/fire5030077.

Silveira, J. M., J. Louzada, J. Barlow, R. Andrade, L. Mestre, R. Solar, S. Lacau, and M. A. Cochrane. 2016. A Multi-Taxa Assessment of Biodiversity Change After Single and Recurrent Wildfires in a Brazilian Amazon Forest. *Biotropica* 48: 170–180. doi:10.1111/btp.12267.

Silveira, M. V. F., C. A. Petri, I. S. Broggio, G. O. Chagas, M. S. Macul, C. C. S. S. Leite, E. M. M. Ferrari, C. G. V. Amim, et al. 2020. Drivers of Fire Anomalies in the Brazilian Amazon: Lessons Learned from the 2019 Fire Crisis. *Land* 9: 516. doi:10.3390/land9120516.

Silveira, M. V. F., C. H. L. Silva-Junior, L. O. Anderson, and L. E. O. C. Aragão. 2022. Amazon fires in the 21st century: The year of 2020 in evidence. *Global Ecology and Biogeography* 31: 2026–2040. doi:10.1111/geb.13577.

Silvério, D. V., P. M. Brando, J. K. Balch, F. E. Putz, D. C. Nepstad, C. Oliveira-Santos, and M. M. C. Bustamante. 2013. Testing the Amazon savannization hypothesis: fire effects on invasion of a neotropical forest by native cerrado and exotic pasture grasses. *Philosophical Transactions of the Royal Society B: Biological Sciences* 368. doi:10.1098/rstb.2012.0427.

Silvério, D. V., R. S. Oliveira, B. M. Flores, P. M. Brando, H. K. Almada, M. T. Furtado, F. G. Moreira, M. Heckenberger, et al. 2022. Intensification of fire regimes and forest loss in the Território Indígena do Xingu. *Environmental Research Letters* 17. doi:10.1088/1748-9326/ac5713.

Silvestrini, R. A., B. S. Soares-Filho, D. Nepstad, M. Coe, H. Rodrigues, and R. Assunção. 2011. Simulating fire regimes in the Amazon in response to climate change and deforestation. *Ecological Applications* 21. Wiley: 1573–1590. doi:10.1890/10-0827.1.

Simpson, D., H. Rue, A. Riebler, T. G. Martins, and S. H. Sørbye. 2017. Penalising Model Component Complexity: A Principled, Practical Approach to Constructing Priors. *Statistical Science* 32. doi:10.1214/16-STS576.

Singh, H., and S. K. Srivastava. 2024. From Firestick to Satellites: Technological Advancement and Indigenous Cultural Practice in Managing Forest Fires in Australia. *The Historic Environment: Policy & Practice*: 1–24. doi:10.1080/17567505.2024.2425246.

Sletto, B., and I. Rodriguez. 2013. Burning, fire prevention and landscape productions among the Pemon, Gran Sabana, Venezuela: Toward an intercultural approach to wildland fire management in Neotropical Savannas. *Journal of Environmental Management* 115: 155–166. doi:10.1016/j.jenvman.2012.10.041.

Smith, C., J. Ainscough, R. Alare, A. Croker, K. De Freitas, J. Millington, J. Mistry, O. Perkins, et al. 2024. How policy interventions influence burning to meet cultural and small-scale livelihood objectives. *Ecology and Society* 29. doi:10.5751/ES-14850-290135.

Smith, C., O. Perkins, J. Mistry, B. A. Bilbao, R. Bliege Bird, A. Cardinal Christianson, K. M. De Freitas, W. Dressler, et al. 2025. A global expert elicitation on present-day human–fire interactions. *Philosophical Transactions of the Royal Society B: Biological Sciences* 380. doi:10.1098/rstb.2023.0463.

Soares-Filho, B., P. Moutinho, D. Nepstad, A. Anderson, H. Rodrigues, R. Garcia, L. Dietzsch, F. Merry, et al. 2010. Role of Brazilian Amazon protected areas in climate change mitigation. *Proceedings of the National Academy of Sciences* 107: 10821–10826. doi:10.1073/pnas.0913048107.

Soares-Filho, B., R. Silvestrini, D. Nepstad, P. Brando, H. Rodrigues, A. Alencar, M. Coe, C. Locks, et al. 2012. Forest fragmentation, climate change and understory fire regimes on the

Amazonian landscapes of the Xingu headwaters. *Landscape Ecology* 27: 585–598. doi:10.1007/s10980-012-9723-6.

Soares-Filho, B. S., D. C. Nepstad, L. M. Curran, G. C. Cerqueira, R. A. Garcia, C. A. Ramos, E. Voll, A. McDonald, et al. 2006. Modelling conservation in the Amazon basin. *Nature* 440: 520–523. doi:10.1038/nature04389.

Sorrensen, C. 2009. Potential hazards of land policy: Conservation, rural development and fire use in the Brazilian Amazon. *Land Use Policy* 26: 782–791. doi:10.1016/j.landusepol.2008.10.007.

Soterroni, A. C., F. M. Ramos, A. Mosnier, J. Fargione, P. R. Andrade, L. Baumgarten, J. Pirker, M. Obersteiner, et al. 2019. Expanding the Soy Moratorium to Brazil's Cerrado. *Science Advances* 5. doi:10.1126/sciadv.aav7336.

Souza, C. M., J. Z. Shimbo, M. R. Rosa, L. L. Parente, A. A. Alencar, B. F. T. Rudorff, H. Hasenack, M. Matsumoto, et al. 2020. Reconstructing Three Decades of Land Use and Land Cover Changes in Brazilian Biomes with Landsat Archive and Earth Engine. *Remote Sensing* 12. doi:10.3390/rs12172735.

Souza, L. E. V. de, M. Fetz, B. P. Zagatto, and N. S. Pinho. 2022. Violence and Illegal Deforestation: The Crimes of "Environmental Militias" in the Amazon Forest. *Capitalism Nature Socialism* 33: 5–25. doi:10.1080/10455752.2021.1980817.

Sparovek, G., G. Berndes, I. L. F. Klug, and A. G. O. P. Barretto. 2010. Brazilian Agriculture and Environmental Legislation: Status and Future Challenges. *Environmental Science & Technology* 44: 6046–6053. doi:10.1021/es1007824.

Stahl, C., S. Fontaine, K. Klumpp, C. Picon-Cochard, M. M. Grise, C. Dezécache, L. Ponchant, V. Freycon, et al. 2017. Continuous soil carbon storage of old permanent pastures in Amazonia. *Global Change Biology* 23: 3382–3392. doi:10.1111/gcb.13573.

Stoof, C. R., and N. Kettridge. 2022. Living With Fire and the Need for Diversity. *Earth's Future* 10. doi:10.1029/2021EF002528.

Tasker, K. A., and E. Y. Arima. 2016. Fire regimes in Amazonia: The relative roles of policy and precipitation. *Anthropocene* 14: 46–57. doi:10.1016/j.ancene.2016.06.001.

Tavares, P. A., J. Ferreira, C. V. J. Silva, E. Berenguer, and J. Barlow. 2022. Exploring the Role of Deforestation and Cropland Expansion in Driving a Fire-Transition in the Brazilian Amazon. *Land* 11. doi:10.3390/land11122274.

Tedim, F., V. Leone, and T. K. McGee. 2020. *Extreme wildfire events and disasters: root causes and new management strategies*. Amsterdam: Elsevier.

Tengö, M., E. S. Brondizio, T. Elmqvist, P. Malmer, and M. Spierenburg. 2014. Connecting Diverse Knowledge Systems for Enhanced Ecosystem Governance: The Multiple Evidence Base Approach. *AMBIO* 43: 579–591. doi:10.1007/s13280-014-0501-3.

Thaler, G. M., C. Viana, and F. Toni. 2019. From frontier governance to governance frontier: The political geography of Brazil's Amazon transition. *World Development*: 14.

Tobler, W. R. 1970. A Computer Movie Simulating Urban Growth in the Detroit Region. *Economic Geography* 46. doi:10.2307/143141.

Trauernicht, C., B. W. Brook, B. P. Murphy, G. J. Williamson, and D. M. J. S. Bowman. 2015. Local and global pyrogeographic evidence that indigenous fire management creates pyrodiversity. *Ecology and Evolution* 5: 1908–1918. doi:10.1002/ece3.1494.

Turner, N. J., and F. Berkes. 2006. Coming to Understanding: Developing Conservation through Incremental Learning in the Pacific Northwest. *Human Ecology* 34: 495–513. doi:10.1007/s10745-006-9042-0.

Tyukavina, A., M. C. Hansen, P. V. Potapov, S. V. Stehman, K. Smith-Rodriguez, C. Okpa, and R. Aguilar. 2017. Types and rates of forest disturbance in Brazilian Legal Amazon, 2000–2013. *Science Advances* 3. doi:10.1126/sciadv.1601047.

Vale, M. M., E. Berenguer, M. Argollo de Menezes, E. B. Viveiros de Castro, L. Pugliese de Siqueira, and R. de C. Q. Portela. 2021. The COVID-19 pandemic as an opportunity to weaken environmental protection in Brazil. *Biological Conservation* 255. doi:10.1016/j.biocon.2021.108994.

Van Wees, D., G. R. Van Der Werf, J. T. Randerson, N. Andela, Y. Chen, and D. C. Morton. 2021. The role of fire in global forest loss dynamics. *Global Change Biology* 27: 2377–2391. doi:10.1111/gcb.15591.

Vasconcelos, S. S. D., P. M. Fearnside, P. M. L. D. A. Graça, E. M. Nogueira, L. C. D. Oliveira, and E. O. Figueiredo. 2013. Forest fires in southwestern Brazilian Amazonia: Estimates of area and potential carbon emissions. *Forest Ecology and Management* 291: 199–208. doi:10.1016/j.foreco.2012.11.044.

Venturieri, A., R. R. S. de Oliveira, T. K. Igawa, K. D. A. Fernandes, M. Adami, M. Júnior, C. A. Almeida, L. G. T. Silva, et al. 2022. The Sustainable Expansion of the Cocoa Crop in the State of Para; and Its Contribution to Altered Areas Recovery and Fire Reduction. *Journal of Geographic Information System* 14: 294–313. doi:10.4236/jgis.2022.143016.

Victoria, D. de C., R. F. B. da Silva, J. D. A. Millington, V. Katerinchuk, and M. Batistella. 2021. Transport cost to port though Brazilian federal roads network: Dataset for years 2000, 2005, 2010 and 2017. *Data in Brief* 36. doi:10.1016/j.dib.2021.107070.

Vitolo, C., F. Di Giuseppe, C. Barnard, R. Coughlan, J. San-Miguel-Ayanz, G. Libertá, and B. Krzeminski. 2020. ERA5-based global meteorological wildfire danger maps. *Scientific Data* 7: 216. doi:10.1038/s41597-020-0554-z.

van Vliet, N., C. Adams, I. C. G. Vieira, and O. Mertz. 2013. "Slash and Burn" and "Shifting" Cultivation Systems in Forest Agriculture Frontiers from the Brazilian Amazon. *Society & Natural Resources* 26: 1454–1467. doi:10.1080/08941920.2013.820813.

Volpato, M., C. F. Andrade, E. L. Silva, M. L. Barbosa, M. D. Andrade, Pedro. V. Rocha, R. C. Delgado, P. E. Teodoro, et al. 2023. Fire foci and their spatiotemporal relations to weather variables and land uses in the state of Mato Grosso. *Environment, Development and Sustainability* 25: 12419–12438. doi:10.1007/s10668-022-02573-3.

Wagner, C. E. V. 1974. Structure of the Canadian Forest Fire Weather Index. Department of Environment, Canadian Forestry Services.

Walker, K., A. Flores-Anderson, L. Villa, R. Griffin, M. Finer, and K. Herndon. 2022. An analysis of fire dynamics in and around indigenous territories and protected areas in a Brazilian agricultural frontier. *Environmental Research Letters* 17. doi:10.1088/1748-9326/ac8237.

Walker, W. S., S. R. Gorelik, A. Baccini, J. L. Aragon-Osejo, C. Josse, C. Meyer, M. N. Macedo, C. Augusto, et al. 2020. The role of forest conversion, degradation, and disturbance in the carbon dynamics of Amazon indigenous territories and protected areas. *Proceedings of the National Academy of Sciences* 117: 3015–3025. doi:10.1073/pnas.1913321117.

Welch, J. R., E. S. Brondízio, S. S. Hetrick, and C. E. A. Coimbra. 2013. Indigenous Burning as Conservation Practice: Neotropical Savanna Recovery amid Agribusiness Deforestation in Central Brazil. Edited by Brock Fenton. *PLoS ONE* 8. doi:10.1371/journal.pone.0081226.

West, T. A. P., and P. M. Fearnside. 2021. Brazil's conservation reform and the reduction of deforestation in Amazonia. *Land Use Policy* 100. doi:10.1016/j.landusepol.2020.105072.

Xaud, H. A. M., F. D. S. R. V. Martins, and J. R. D. Santos. 2013. Tropical forest degradation by mega-fires in the northern Brazilian Amazon. *Forest Ecology and Management* 294: 97–106. doi:10.1016/j.foreco.2012.11.036.

Xerente, P. P. G. D. S., and R. C. S. D. Oliveira. 2021. Abordagem Indígena sobre Manejo Integrado do Fogo em Terras Indígenas no Estado do Tocantins – Brasil. *Biodiversidade Brasileira - BioBrasil*: 67–74. doi:10.37002/biobrasil.v11i2.1719.

Xu, W., Y. Liu, S. Veraverbeke, W. Wu, Y. Dong, and W. Lu. 2021. Active Fire Dynamics in the Amazon: New Perspectives From High-Resolution Satellite Observations. *Geophysical Research Letters* 48. doi:10.1029/2021GL093789.

Yanai, A. M., E. M. Nogueira, P. M. L. de Alencastro Graça, and P. M. Fearnside. 2017. Deforestation and Carbon Stock Loss in Brazil's Amazonian Settlements. *Environmental Management* 59: 393–409. doi:10.1007/s00267-016-0783-2.

Yanai, A. M., P. M. L. de A. Graça, M. I. S. Escada, L. G. Ziccardi, and P. M. Fearnside. 2020. Deforestation dynamics in Brazil's Amazonian settlements: Effects of land-tenure concentration. *Journal of Environmental Management* 268. doi:10.1016/j.jenvman.2020.110555.

Yudjá/Juruna - Povos Indigenas no Brasil. 2025. *ISA*. https://pib.socioambiental.org/en/Povo:Yudj%C3%A1/Juruna. Accessed May 12.

Zant, M., A. Schlingmann, V. Reyes-García, and D. García-del-Amo. 2023. Incremental and transformational adaptation to climate change among Indigenous Peoples and local communities: a global review. *Mitigation and Adaptation Strategies for Global Change* 28: 57. doi:10.1007/s11027-023-10095-0.

Zheng, B., P. Ciais, F. Chevallier, E. Chuvieco, Y. Chen, and H. Yang. 2021. Increasing forest fire emissions despite the decline in global burned area. *Science Advances* 7. doi:10.1126/sciadv.abh2646.

Appendices

Appendices from Chapter 2

Appendix 2.1: Framework of potential drivers of fires regimes

At the initial stage of this research, I conducted a non-systematic literature review to investigate the potential theoretical framework of drivers of fire regimes in the region and identify relevant data sources. This non-systematic literature review encompasses 51 publications ranging from 2004 to 2022 and focusing on the Brazilian Amazon region. Out of these publications, 24 present results from quantitative analysis of drivers and impacts of fires, 17 results from quantitative analysis of drivers and impacts of deforestation, 4 results from quantitative analysis looking at the driver and impact of other sources of forest degradation, as well as 6 other publications providing richer context on the link between drivers of deforestation, fires and other sources of forest degradation. The following paragraphs describe the main categories of drivers of fire regimes that were identified through the literature review, while table S1 details different links identified between potential variables and fires regimes in the Brazilian Amazon.

Climate

There is a strong association between annual precipitation and fire occurrence within the Brazilian Amazon (Aragão et al. 2007; Arima et al. 2007; Soares-Filho et al. 2012; Fonseca et al. 2016; Fonseca et al. 2017; Silveira et al. 2020). While most of the rainforests in the region are too humid to burn, El-Nino events, Pacific Decadal Oscillation and Atlantic Multidecadal oscillations are triggering periodic droughts increasing considerably the number of active fires detected across Amazonian landscapes (Aragão et al. 2007; Fonseca et al. 2017; Aragão et al. 2018). Prolonged droughts lead Amazonian trees to lose part of their branches and leaves, resulting in an accumulation of fuel, an opening of the canopy, an increased penetration of solar radiation and ultimately more intense fire and higher post-fire mortality than in normal climatic conditions (Nepstad et al. 2008; Brando et al. 2014) However, chronic water deficit limits the regrowth of the vegetation a contributing to fuel scarcity (Silveira et al. 2020).

Agriculture and pastoralism

Increasing the profitability of ranching or crop farming might incentivize landholders to clear more land, using fires in the process, especially when cleared land is intended for crop cultivation. Fires is used more intensely for land clearing associated with crop cultures than ranching (Morton et al. 2008; Aragão et al. 2018). After land clearing, fires continue to be used, especially in low-intensity farming systems and pastures, for getting rid of the regrowing vegetation, creating many ignition points that frequently escape into nearby

forests (Cano-Crespo et al. 2015). However, mechanization and intensification of agriculture reduce the need to use fires and increase the value of fire-vulnerable assets on agricultural land, sending incentives for better fire management (Morello and Falcão 2020).

Forest degradation

Before deforestation and conversion to agricultural land, Amazonian forests might face several types of disturbance (Tyukavina et al. 2017). In the early stage of frontier expansions, logging is an important source of pressure, leading to an accumulation of fuelwood due to vegetation disturbance, damage to the canopy increasing the penetration of solar radiation and fragmentation of the landscape making the forest more prone to fires (Asner et al. 2005; Asner et al. 2006; Broadbent et al. 2008). The road opened during the logging process fragment the forest cover, improve the accessibility of forested areas and profitability of ranching/farming venture: significant parts of logged forests are deforested within the next years (Asner et al. 2006).

Fragmentation of the forest cover has several impacts on the fire regime: edges are favoring drier microclimate, increase mortality rates and impact the vegetal communities and thus fuel structure (Balch et al. 2015). It also increases the interface between the agricultural landscape, on which fire is frequently used, and forests, thus increasing the possibility of escaped fires (Cano-Crespo et al. 2015). Understory fires also influence future fires: even low-intensity burn results in tree mortality, fuel accumulation, damage of the canopy and invasion of the forest by grass species, all processes that increase the intensity of future fires (Barlow and Peres 2008; Balch et al. 2011; Balch et al. 2015). Finally, deforestation is one of the most important drivers of fire regimes in the region: after felling the trees, they are left on the ground to dry before being lit on fires several times for getting rid of the biomass and allow agriculture (Morton et al. 2008).

Infrastructures and development

The Brazilian Amazon has a limited road network and many areas that are distant from densely populated areas, markets and governmental infrastructure. The distance from the road and port destined for exportation determine the potential profitability of deforestation and agricultural ventures, as well as access to labor and agricultural inputs. Most deforestation in the Brazilian Amazon and associated fires, occurred close to roads and rivers (Barber et al. 2014; Fonseca et al. 2017). However, areas close to major roads have better access to agricultural inputs and labour and could have a higher degree of mechanization and/or intensification of their agricultural system, which incentive landholders to invest more into fire-risk reduction and/or find alternative land management technics, while reducing the need to use fires for agricultural production due to better market prices (Bowman et al. 2008).

The relationship between fire and population density appears non-linear: while initially increase in population is accompanied by an increase in fire use for land clearing and agriculture, it seems that the relationship reverses after a threshold is reached (Silveira et

al. 2020). This could be explained by the consolidation of agricultural frontiers in densely populated areas and the increase of fire-vulnerable assets on the land, encouraging local stakeholders to reach better fire governance, as well as a higher degree of mechanization of agriculture (Morello and Falcão 2020).

The rural settlement, areas designated by the INCRA to be exploited by landless farmers and smallholders coming from other regions of Brazil, are of particular interest. Farmers can gain land titles from the INCRA, the governmental institution implementing the agrarian reform in Brazil, on the condition that they prove a "productive" use of the land. Thus rural settlements tend to have higher rates of deforestation and fire occurrence than other areas (Schneider and Peres 2015; Yanai et al. 2017). These areas, open for occupation, also concentrate tensions around land tenure: while part of the landholders wants to keep modest landholdings, part of settlers clear vegetation (thus increasing the value of the land plot) and sell their land to capitalized farmers (Carrero et al. 2020; Yanai et al. 2020).

Environmental policies

Over the 2005-2015 period, the Action Plan for the Prevention and Control of Deforestation in the Legal Amazon (PPCDam), coordinated the action of different Brazilian ministries aimed to decrease deforestation by more than 80%. While initially focusing on the improvement of satellite monitoring and law enforcement capacities as well as the demarcation of new protected areas, the latter phases emphasized the promotion of sustainable economic development and reducing deforestation on private lands (West and Fearnside 2021). Delimitation of new protected areas has succeeded in reducing deforestation rate and fire frequency, but their effectiveness depends on the type of protection system, deforestation pressures faced and managing authorities (Soares-Filho et al. 2010; Nolte et al. 2013; Carmenta et al. 2016; Herrera et al. 2019). Indigenous land, often located in high-pressure areas, tend to be the most efficient protection regime, followed by integral protection areas and then sustainable use area, allowing many types of human activities (Soares-Filho et al. 2010; Nolte et al. 2013). Many fires in the protected areas are occurring in places closes to road and border with unprotected land (Adeney et al. 2009; Santos et al. 2021; Walker et al. 2022).

The creation of a near real-time satellite monitoring system of deforestation to guide law enforcement on the ground was also a crucial point of the PPCDAm (Assunção et al. 2015; Börner et al. 2015). However, the size of the average deforestation patch has decreased over the 2005-2014 period to avoid detection and subsequent punishment by environmental authorities (Rosa et al. 2012; Richards et al. 2017; Kalamandeen et al. 2018). The dismantlement of IBAMA and INPE, the governmental agency responsible for respectively the law enforcement efforts and the satellite monitoring of deforestation, has led to a lower probability of punishment and an increase in deforestation patch size in recent years (de Area Leão Pereira et al. 2019; Carvalho et al. 2019; Ferrante and Fearnside 2019). Land conflicts, the creation of rural settlements and infrastructure projects also led to the

downgrading, downsizing or degazettement of around 90 000 km2 of protected areas in the Brazilian Amazon, even though there is mixed evidence of a short-term increase in deforestation rates in these areas (Pack et al. 2016; Keles et al. 2020).

In 2008, the critical county program (also called “priority list program” in certain instances) started to publish a “blacklist” of municipalities experiencing an increase in deforestation. The first list published included the 36 Brazilian municipalities responsible for 45% of the deforestation detected by PRODES in 2007 (Assunção and Rocha 2019). The blacklisted municipalities are subject to stricter administrative requirements for further forest clearing, suffer from a bad reputation, which could reduce business opportunities, and increase monitoring and enforcement actions by the IBAMA. Further restrictions can be adopted by state government such as restricted access to government-sponsored agricultural credits. However, they also benefit from increased support from state actors and NGOs to reduce their deforestation rate. The critical counties program has been efficient to reduce the deforestation rate of blacklisted counties and has a low cost of implementation (Cisneros et al. 2015).

Table S2.1. Potential drivers of the fire regimes identified through literature review and relationship with the fires regimes identified. To be included in the table, a publication should be analyzing fire regime using quantitative analysis, or deforestation if there are no data available on fire regimes, conduct an analysis in the Brazilian Amazon and include a spatial component

Drivers	Relationship
	Climate
Temperature	High temperatures favour fires (Morello 2020; Ferreira Barbosa et al. 2021)
Precipitations	Water deficit triggered by major drought increases the frequency of fires (Aragão et al. 2007; Morton et al. 2008; Adeney et al. 2009; Soares-Filho et al. 2012; Fonseca et al. 2017)
	Areas with higher precipitations tend to have less frequent fires (Arima et al. 2007; Fonseca et al. 2016; Morello 2020)
	Increasing water deficits are increasing and then decreasing the probability of having fires (Silveira et al. 2020)
	Agriculture and pastoralism
Agriculture	Crop production encourages the use of fires (Morton et al. 2008; Xu et al. 2021)
	Non-linear relationship between crop production and fire occurrences (Arima et al. 2007; Aragão and Shimabukuro 2010; Silveira et al. 2020)
	No significant effect (Morello 2020)
	Frequent agricultural fires escaping in nearby forest edges (Cano-Crespo et al. 2015)
Pastoralism	Beef production increases the use of fires (Arima et al. 2007; Fonseca et al. 2016)
	No significant effect (Morello 2020)
	Lower counts of fires when land clearing is related to ranching rather than crop production, but higher fire counts after land clearing (Morton et al. 2008; Aragão and Shimabukuro 2010)
	Increase in the number of fires until pasture covers more than 56% of the cell (Silveira et al. 2020)
	Frequent agricultural fires escaping in nearby forest edges (Cano-Crespo et al. 2015)

Drivers	Relationship
	Ecosystem integrity
Fragmentation	Forest fragmentation favours forest fires (Soares-Filho et al. 2012; Armenteras et al. 2013; Silveira et al. 2020)
Forest degradation	Marginal effect on fires (Fonseca et al. 2016) Past forest degradation favours fires (Morello 2020) Past fires favour fires (Barlow and Peres 2008; Balch et al. 2011; Brando et al. 2014; Balch et al. 2015; Silveira et al. 2020)
	Infrastructure and Development
Access to market	Proximity to roads and rivers favours fires (Adeney et al. 2009; Fonseca et al. 2017; Xu et al. 2021) Proximity to roads favours fire prevention activities and reduces the need to use fires (Bowman et al. 2008) Distance to road increases and then decreases the risk of fires (Arima et al. 2007; Silveira et al. 2020)
Rural settlements	Proportion of settlements raises the probability of fires (Fonseca et al. 2017)
Population	Increase and then decrease the probability of fires (Silveira et al. 2020)
	Environmental policies
Protected areas	Limit the number of fires, especially in areas with high deforestation pressure (Nepstad et al. 2006; Arima et al. 2007; Adeney et al. 2009) Fires within protected areas occurs mainly close to their border with unprotected land or close to roads (Adeney et al. 2009; Santos et al. 2021; Walker et al. 2022) High number of fires within municipalities with lots of protected areas/certain types of protected areas (Morello 2020; Silveira et al. 2020) No significant effect (Carmenta et al. 2016; Fonseca et al. 2016)
PADDD	No increase in deforestation in protected areas that were downsized or degazetted (Pack et al. 2016)
Blacklisting	Blacklisting program could reduce the deforestation rate of blacklisted counties (Cisneros et al. 2015).

Law enforcement	Field-based enforcement operations can reduce deforestation (Börner et al. 2015)

References

Adeney, J. M., N. L. Christensen, and S. L. Pimm. 2009. Reserves Protect against Deforestation Fires in the Amazon. Edited by Rob P. Freckleton. *PLoS ONE* 4. doi:10.1371/journal.pone.0005014.

Aragão, L. E. O. C., and Y. E. Shimabukuro. 2010. The Incidence of Fire in Amazonian Forests with Implications for REDD. *Science* 328: 1275–1278. doi:10.1126/science.1186925.

Aragão, L. E. O. C., Y. Malhi, R. M. Roman-Cuesta, S. Saatchi, L. O. Anderson, and Y. E. Shimabukuro. 2007. Spatial patterns and fire response of recent Amazonian droughts. *Geophysical Research Letters* 34. doi:10.1029/2006GL028946.

Aragão, L. E. O. C., L. O. Anderson, M. G. Fonseca, T. M. Rosan, L. B. Vedovato, F. H. Wagner, C. V. J. Silva, C. H. L. Silva Junior, et al. 2018. 21st Century drought-related fires counteract the decline of Amazon deforestation carbon emissions. *Nature Communications* 9: 536. doi:10.1038/s41467-017-02771-y.

de Area Leão Pereira, E. J., P. J. Silveira Ferreira, L. C. de Santana Ribeiro, T. Sabadini Carvalho, and H. B. de Barros Pereira. 2019. Policy in Brazil (2016–2019) threaten conservation of the Amazon rainforest. *Environmental Science & Policy* 100: 8–12. doi:10.1016/j.envsci.2019.06.001.

Arima, E. Y., C. S. Simmons, R. T. Walker, and M. A. Cochrane. 2007. Fire in the Brazilian Amazon: a Spatially Explicit Model for Policy Impact Analysis. *Journal of Regional Science* 47: 541–567. doi:10.1111/j.1467-9787.2007.00519.x.

Armenteras, D., T. M. González, and J. Retana. 2013. Forest fragmentation and edge influence on fire occurrence and intensity under different management types in Amazon forests. *Biological Conservation* 159: 73–79. doi:10.1016/j.biocon.2012.10.026.

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

Asner, G. P., E. N. Broadbent, P. J. C. Oliveira, M. Keller, D. E. Knapp, and J. N. M. Silva. 2006. Condition and fate of logged forests in the Brazilian Amazon. *Proceedings of the National Academy of Sciences* 103: 12947–12950. doi:10.1073/pnas.0604093103.

Assunção, J., and R. Rocha. 2019. Getting greener by going black: the effect of blacklisting municipalities on Amazon deforestation. *Environment and Development Economics* 24: 115–137. doi:10.1017/S1355770X18000499.

Assunção, J., C. Gandour, and R. Rocha. 2015. Deforestation slowdown in the Brazilian Amazon: prices or policies? *Environment and Development Economics* 20: 697–722. doi:10.1017/S1355770X15000078.

Balch, J. K., D. C. Nepstad, L. M. Curran, P. M. Brando, O. Portela, P. Guilherme, J. D. Reuning-Scherer, and O. de Carvalho. 2011. Size, species, and fire behavior predict

tree and liana mortality from experimental burns in the Brazilian Amazon. *Forest Ecology and Management* 261: 68–77. doi:10.1016/j.foreco.2010.09.029.

Balch, J. K., P. M. Brando, D. C. Nepstad, M. T. Coe, D. Silvério, T. J. Massad, E. A. Davidson, P. Lefebvre, et al. 2015. The Susceptibility of Southeastern Amazon Forests to Fire: Insights from a Large-Scale Burn Experiment. *BioScience* 65: 893–905. doi:10.1093/biosci/biv106.

Barber, C. P., M. A. Cochrane, C. M. Souza, and W. F. Laurance. 2014. Roads, deforestation, and the mitigating effect of protected areas in the Amazon. *Biological Conservation* 177: 203–209. doi:10.1016/j.biocon.2014.07.004.

Barlow, J., and C. A. Peres. 2008. Fire-mediated dieback and compositional cascade in an Amazonian forest. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363: 1787–1794. doi:10.1098/rstb.2007.0013.

Börner, J., K. Kis-Katos, J. Hargrave, and K. König. 2015. Post-Crackdown Effectiveness of Field-Based Forest Law Enforcement in the Brazilian Amazon. Edited by Ricardo Bomfim Machado. *PLOS ONE* 10: e0121544. doi:10.1371/journal.pone.0121544.

Bowman, M. S., G. S. Amacher, and F. D. Merry. 2008. Fire use and prevention by traditional households in the Brazilian Amazon. *Ecological Economics* 67: 117–130. doi:10.1016/j.ecolecon.2007.12.003.

Brando, P. M., J. K. Balch, D. C. Nepstad, D. C. Morton, F. E. Putz, M. T. Coe, D. Silverio, M. N. Macedo, et al. 2014. Abrupt increases in Amazonian tree mortality due to drought-fire interactions. *Proceedings of the National Academy of Sciences* 111: 6347–6352. doi:10.1073/pnas.1305499111.

Broadbent, E. N., G. P. Asner, M. Keller, D. E. Knapp, P. J. C. Oliveira, and J. N. Silva. 2008. Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. *Biological Conservation*: 13.

Cano-Crespo, A., P. J. C. Oliveira, A. Boit, M. Cardoso, and K. Thonicke. 2015. Forest edge burning in the Brazilian Amazon promoted by escaping fires from managed pastures. *Journal of Geophysical Research: Biogeosciences* 120: 2095–2107. doi:10.1002/2015JG002914.

Carmenta, R., G. A. Blackburn, G. Davies, C. de Sassi, A. Lima, L. Parry, W. Tych, and J. Barlow. 2016. Does the Establishment of Sustainable Use Reserves Affect Fire Management in the Humid Tropics? Edited by Clinton N. Jenkins. *PLOS ONE* 11. doi:10.1371/journal.pone.0149292.

Carrero, G. C., P. M. Fearnside, D. R. do Valle, and C. de Souza Alves. 2020. Deforestation Trajectories on a Development Frontier in the Brazilian Amazon: 35 Years of Settlement Colonization, Policy and Economic Shifts, and Land Accumulation. *Environmental Management* 66: 966–984. doi:10.1007/s00267-020-01354-w.

Carvalho, W. D., K. Mustin, R. R. Hilário, I. M. Vasconcelos, V. Eilers, and P. M. Fearnside. 2019. Deforestation control in the Brazilian Amazon: A conservation struggle being lost as agreements and regulations are subverted and bypassed. *Perspectives in Ecology and Conservation* 17: 122–130. doi:10.1016/j.pecon.2019.06.002.

Cisneros, E., S. L. Zhou, and J. Börner. 2015. Naming and Shaming for Conservation: Evidence from the Brazilian Amazon. Edited by Edward Webb. *PLOS ONE* 10. doi:10.1371/journal.pone.0136402.

Ferrante, L., and P. M. Fearnside. 2019. Brazil's new president and 'ruralists' threaten Amazonia's environment, traditional peoples and the global climate. *Environmental Conservation* 46: 261–263. doi:10.1017/S0376892919000213.

Ferreira Barbosa, M. L., R. C. Delgado, C. Forsad de Andrade, P. E. Teodoro, C. A. Silva Junior, H. S. Wanderley, and G. F. Capristo-Silva. 2021. Recent trends in the fire dynamics in Brazilian Legal Amazon: Interaction between the ENSO phenomenon, climate and land use. *Environmental Development* 39. doi:10.1016/j.envdev.2021.100648.

Fonseca, M. G., L. E. O. C. Aragão, A. Lima, Y. E. Shimabukuro, E. Arai, and L. O. Anderson. 2016. Modelling fire probability in the Brazilian Amazon using the maximum entropy method. *International Journal of Wildland Fire* 25: 955–969. doi:10.1071/WF15216.

Fonseca, M. G., L. O. Anderson, E. Arai, Y. E. Shimabukuro, H. A. M. Xaud, M. R. Xaud, N. Madani, F. H. Wagner, et al. 2017. Climatic and anthropogenic drivers of northern Amazon fires during the 2015-2016 El Niño event. *Ecological Applications* 27: 2514–2527. doi:10.1002/eap.1628.

Herrera, D., A. Pfaff, and J. Robalino. 2019. Impacts of protected areas vary with the level of government: Comparing avoided deforestation across agencies in the Brazilian Amazon. *Proceedings of the National Academy of Sciences* 116: 14916–14925. doi:10.1073/pnas.1802877116.

Kalamandeen, M., E. Gloor, E. Mitchard, D. Quincey, G. Ziv, D. Spracklen, B. Spracklen, M. Adami, et al. 2018. Pervasive Rise of Small-scale Deforestation in Amazonia. *Scientific Reports* 8: 1600. doi:10.1038/s41598-018-19358-2.

Keles, D., P. Delacote, A. Pfaff, S. Qin, and M. B. Mascia. 2020. What Drives the Erasure of Protected Areas? Evidence from across the Brazilian Amazon. *Ecological Economics* 176: 106733. doi:10.1016/j.ecolecon.2020.106733.

Morello, T., and L. Falcão. 2020. The Fire Management Dilemma in the Brazilian Amazon: Synthesizing Pathways of Causality across Five Case Studies in the State of Pará. *Human Ecology* 48: 397–409. doi:10.1007/s10745-020-00166-0.

Morello, T. F. 2020. Predicting fires for policy making: Improving accuracy of fire brigade allocation in the Brazilian Amazon. *Ecological Economics* 169: 14.

Morton, D. C., R. S. Defries, J. T. Randerson, L. Giglio, W. Schroeder, and G. R. Van Der Werf. 2008. Agricultural intensification increases deforestation fire activity in Amazonia: Deforestation Fires in Amazonia. *Global Change Biology* 14: 2262–2275. doi:10.1111/j.1365-2486.2008.01652.x.

Nepstad, D., S. Schwartzman, B. Bamberger, M. Santilli, D. Ray, P. Schlesinger, P. Lefebvre, A. Alencar, et al. 2006. Inhibition of Amazon Deforestation and Fire by Parks and Indigenous Lands: Inhibition of Amazon Deforestation and Fire. *Conservation Biology* 20: 65–73. doi:10.1111/j.1523-1739.2006.00351.x.

Nepstad, D. C., C. M. Stickler, B. S.- Filho, and F. Merry. 2008. Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point.

Philosophical Transactions of the Royal Society B: Biological Sciences 363: 1737–1746. doi:10.1098/rstb.2007.0036.

Nolte, C., A. Agrawal, K. M. Silvius, and B. S. Soares-Filho. 2013. Governance regime and location influence avoided deforestation success of protected areas in the Brazilian Amazon. *Proceedings of the National Academy of Sciences* 110: 4956–4961. doi:10.1073/pnas.1214786110.

Pack, S. M., M. N. Ferreira, R. Krithivasan, J. Murrow, E. Bernard, and M. B. Mascia. 2016. Protected area downgrading, downsizing, and degazettement (PADDD) in the Amazon. *Biological Conservation* 197: 32–39. doi:10.1016/j.biocon.2016.02.004.

Richards, P., E. Arima, L. VanWey, A. Cohn, and N. Bhattarai. 2017. Are Brazil's Deforesters Avoiding Detection? *Conservation Letters* 10: 470–476. doi:10.1111/conl.12310.

Rosa, I. M. D., C. Souza, and R. M. Ewers. 2012. Changes in Size of Deforested Patches in the Brazilian Amazon: Dynamics of Amazonian Deforestation. *Conservation Biology* 26: 932–937. doi:10.1111/j.1523-1739.2012.01901.x.

Santos, A. M. dos, C. F. A. da Silva, A. P. Rudke, and D. de Oliveira Soares. 2021. Dynamics of active fire data and their relationship with fires in the areas of regularized indigenous lands in the Southern Amazon. *Remote Sensing Applications: Society and Environment* 23. doi:10.1016/j.rsase.2021.100570.

Schneider, M., and C. A. Peres. 2015. Environmental Costs of Government-Sponsored Agrarian Settlements in Brazilian Amazonia. Edited by RunGuo Zang. *PLOS ONE* 10. doi:10.1371/journal.pone.0134016.

Silveira, M. V. F., C. A. Petri, I. S. Broggio, G. O. Chagas, M. S. Macul, C. C. S. S. Leite, E. M. M. Ferrari, C. G. V. Amim, et al. 2020. Drivers of Fire Anomalies in the Brazilian Amazon: Lessons Learned from the 2019 Fire Crisis. *Land* 9: 516. doi:10.3390/land9120516.

Soares-Filho, B., P. Moutinho, D. Nepstad, A. Anderson, H. Rodrigues, R. Garcia, L. Dietzsch, F. Merry, et al. 2010. Role of Brazilian Amazon protected areas in climate change mitigation. *Proceedings of the National Academy of Sciences* 107: 10821–10826. doi:10.1073/pnas.0913048107.

Soares-Filho, B., R. Silvestrini, D. Nepstad, P. Brando, H. Rodrigues, A. Alencar, M. Coe, C. Locks, et al. 2012. Forest fragmentation, climate change and understory fire regimes on the Amazonian landscapes of the Xingu headwaters. *Landscape Ecology* 27: 585–598. doi:10.1007/s10980-012-9723-6.

Tyukavina, A., M. C. Hansen, P. V. Potapov, S. V. Stehman, K. Smith-Rodriguez, C. Okpa, and R. Aguilar. 2017. Types and rates of forest disturbance in Brazilian Legal Amazon, 2000–2013. *Science Advances* 3. doi:10.1126/sciadv.1601047.

Walker, K., A. Flores-Anderson, L. Villa, R. Griffin, M. Finer, and K. Herndon. 2022. An analysis of fire dynamics in and around indigenous territories and protected areas in a Brazilian agricultural frontier. *Environmental Research Letters* 17: 084030. doi:10.1088/1748-9326/ac8237.

West, T. A. P., and P. M. Fearnside. 2021. Brazil's conservation reform and the reduction of deforestation in Amazonia. *Land Use Policy* 100. doi:10.1016/j.landusepol.2020.105072.

Xu, W., Y. Liu, S. Veraverbeke, W. Wu, Y. Dong, and W. Lu. 2021. Active Fire Dynamics in the Amazon: New Perspectives From High-Resolution Satellite Observations. *Geophysical Research Letters* 48. doi:10.1029/2021GL093789.

Yanai, A. M., E. M. Nogueira, P. M. L. de Alencastro Graça, and P. M. Fearnside. 2017. Deforestation and Carbon Stock Loss in Brazil's Amazonian Settlements. *Environmental Management* 59: 393–409. doi:10.1007/s00267-016-0783-2.

Yanai, A. M., P. M. L. de A. Graça, M. I. S. Escada, L. G. Ziccardi, and P. M. Fearnside. 2020. Deforestation dynamics in Brazil's Amazonian settlements: Effects of land-tenure concentration. *Journal of Environmental Management* 268. doi:10.1016/j.jenvman.2020.110555.

Appendix 2.2: Explanatory variables selection and preprocessing

Explanatory variables selection

Computation of large-scale models with many variables and interpretation of their results can prove challenging, especially if some of the variables rely on poor-quality data. Thus, after the identification of potential drivers of fire regimes and data sources that could be used to derive variables, I removed some variables based on:

- **Quality of the datasets:** some datasets were collected in ways that could make their interpretation challenging
- **Redundancy between variables:** several variables could be proxies for the same underlying drivers of fire regimes and/or are highly correlated to other variables
- **Distribution of the data:** some variable's distribution are skewed over few values and would bring little information in the large-scale models

The following variables, initially considered, have been removed from the models:

- **Temperature and precipitations:** these two climatic factors affect the flammability of the ecosystems by determining the balance between the input of water through precipitations and the output of water through evaporation and evapotranspiration. I used the Maximum Cumulated Water Deficit, a drought index that accounts for both phenomena (see next section for more details) and determined the amount of hydric stress vegetation is exposed to throughout a year.
- **Past fires:** while initially thought of as a potential proxy for past degradation of the forest, the interpretation of this variable could be quite challenging as fire tends to repeat over the same pixels and past fires could be a proxy for other phenomena driving fire occurrences. Moreover, it is challenging to know if a fire is burning over the same area as in the past, or other places across the pixel.
- **Law enforcement efforts** A list of embargos issued by the IBAMA for environmental infractions was available, but the data was aggregated at a municipality level.

Moreover, the distribution of the data was highly skewed, concerns mainly a few municipalities and is highly correlated to municipalities on the priority list.

- **Population** the data available were projections from the IBGE aggregated at a municipality level. The population is a proxy of human pressures, which is modelled by other explanatory variables in the model such as the transport cost or the presence of different agricultural land use. Moreover, the distribution was highly skewed with few small municipalities regrouping huge proportions of the populations, corresponding to the major urban centres.

Explanatory variables Preprocessing

Maximum cumulated water deficit: The algorithm used for deriving the maximum cumulated water deficit is similar to the one described in Aragão et al (2007) and provides an indication of the severity of drought reach over a year. For each pixel, a Cumulated Water Deficit (CWD) was calculated for each month (n) using these rules:

if $CWD_{n-1} - evapotranspiration_n + precipitation_n < 0$,

then $CWD_n = CWD_{n-1} - 100 + Precipitation_n$,

else $CWD_n = 0$

Then, for each pixel the lowest CWD value for each year was kept, representing the intensity of hydric stress over a year. A raster stack has been created with the Maximum cumulated water deficit for each year of the study period, before being divided into 6 categories. The data used for precipitations were from CHIRPS dataset (Funk et al. 2015).

Maximum Cumulated Water Deficit	period 1 (2009-2011)	period 2 (2012-2015)	period 3 (2016-2018)	period 4 (2019-2021)
category				
0 mm	9.1%	11%	7.4%	9.4%
1-100mm	23%	24%	22%	28%
101-200mm	21%	19%	29%	24%
201-300mm	24%	25%	22%	18%
301-400mm	15%	17%	13%	15%
401mm and more	8.4%	4.0%	6.1%	6.7%

Table S2.2. Summary of the percentage of pixels in each category over the 4 period of analysis.

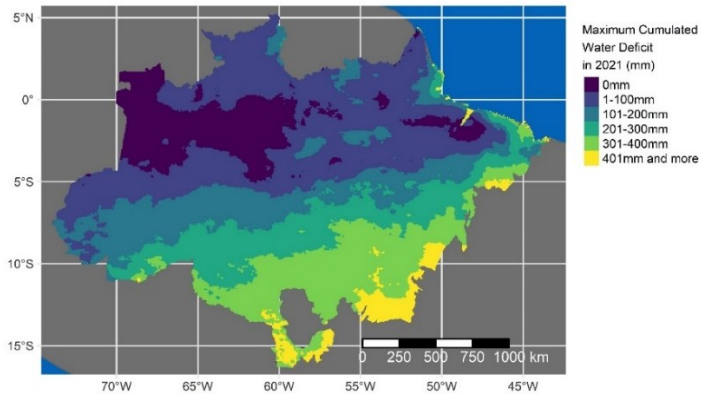
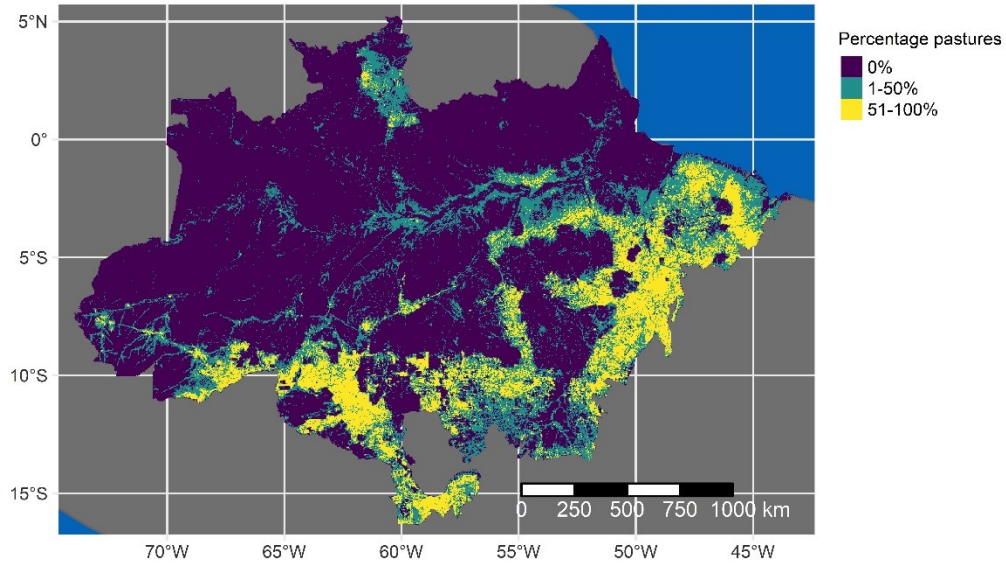


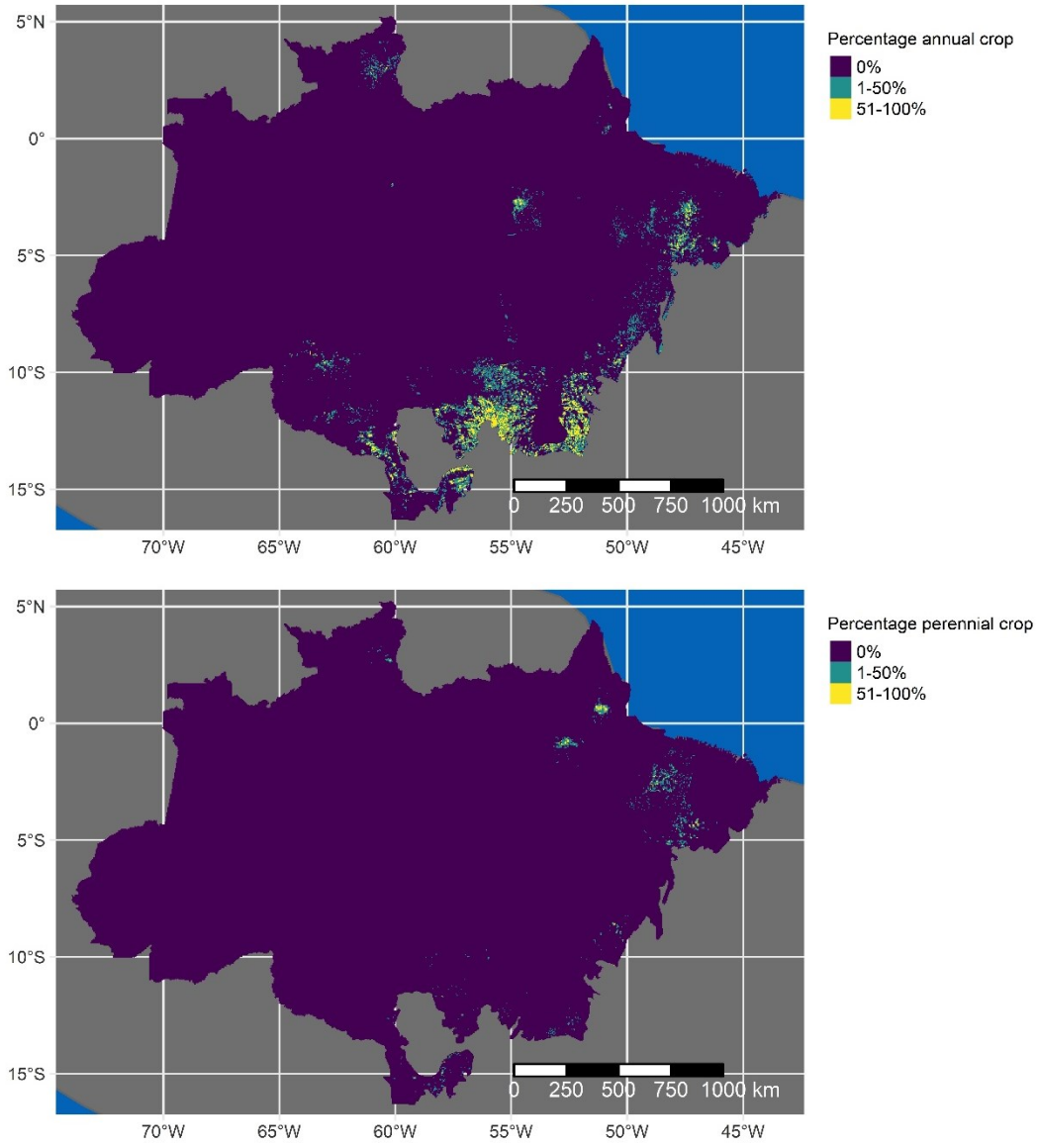
Figure S1. Map of the categorized MCWD in 2021.

Agricultural land use Mapbiomas collection 7 was used to look at land uses. The land use map was reclassified to create the following explanatory variables:

- Pasture: pasture and mosaic agriculture and pasture (ID 15+21)
- Annual crops: soybean, sugarcane, rice, cotton and other annual crops (ID 39+20+40+41+62)
- Perennial crops: Forest plantations, coffee, citrus and other perennial crops (ID 9, 46, 47, 48)

The coverage of each agricultural land was calculated for each 1km pixels and include into raster stacks, before being classified as covering less than 1%, 1 to 50% or 51 to 100% of the pixel. For the models on agricultural fires, I started by calculating areas with a coverage of >90% of agricultural land, and then derive the type of agricultural land covering the highest proportion of the pixel.




5°N
0°
5°S
10°S
15°S
70°W
65°W
60°W
55°W
50°W
45°W
0 250 500 750 1000 km
Percentage annual crop
0%
1-50%
51-100%
Percentage perennial crop
0%
1-50%
51-100%

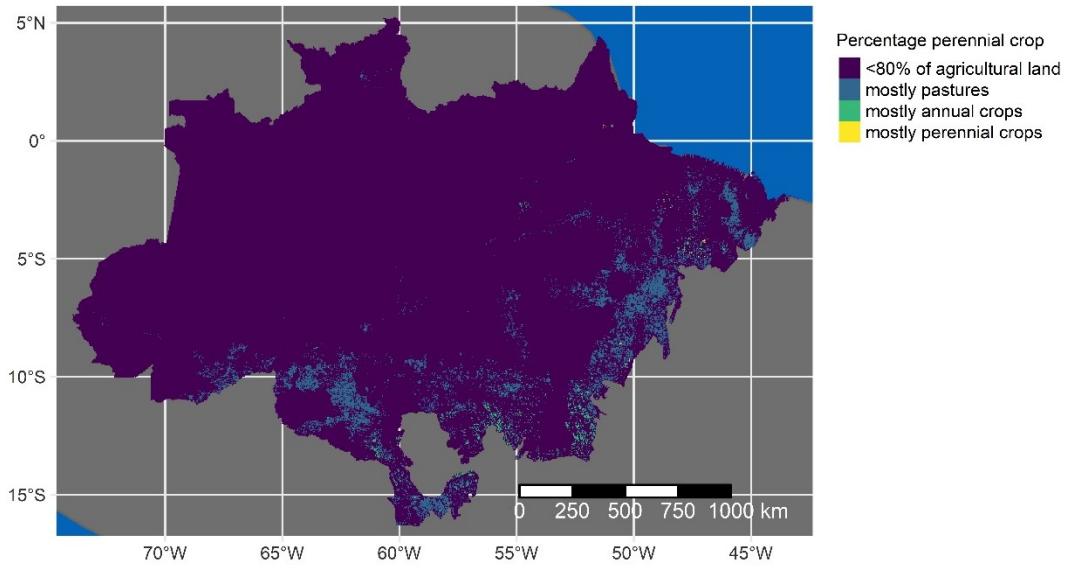
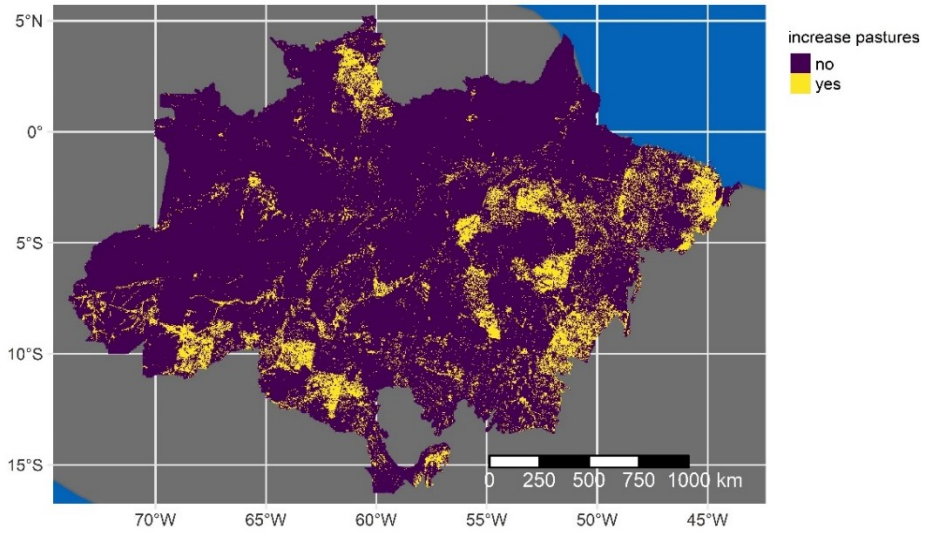


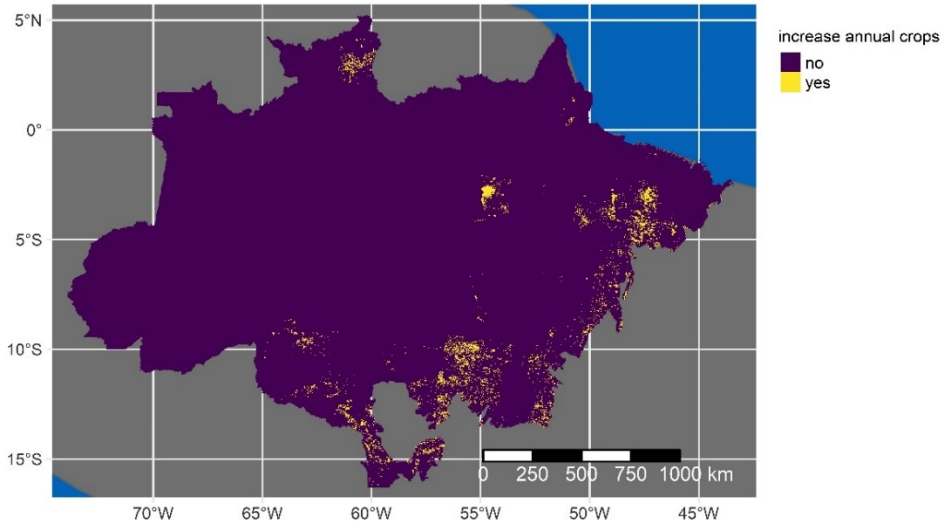
Figure S2. Map of the categorized agricultural land uses in 2021.

Agricultural land use	period 1 (2009-2011)	period 2 (2012-2015)	period 3 (2016-2018)	period 4 (2019-2021)
pasture				
0%	70%	70%	69%	68%
1-50%	19%	19%	20%	20%
51-100%	12%	12%	12%	12%
annual crop				
0%	98%	98%	97%	96%
1-50%	1.1%	1.6%	2.2%	2.8%
51-100%	0.4%	0.6%	0.9%	1.0%
perennial crop				
0%	99%	99%	99%	99%
1-50%	0.5%	0.5%	0.5%	0.6%
51-100%	<0.1%	<0.1%	<0.1%	<0.1%
dominant agricultural land use				
<80% of agricultural land	95%	95%	95%	95%
pasture	4.6%	4.5%	4.4%	4.6%
annual crop	0.2%	0.3%	0.5%	0.6%
perennial crop	<0.1%	<0.1%	<0.1%	<0.1%

Table S2.3. Summary of the percentage of pixels in each category over the 4 period of analysis

Agricultural land use increases MapBiomas collection 7 was used to look at the expansions of pastures, annual crops and perennial crops. The percentage of each agricultural land category was compared to the previous year, and pixels with an increase >1% of the identified land category was classified as having increasing cover.





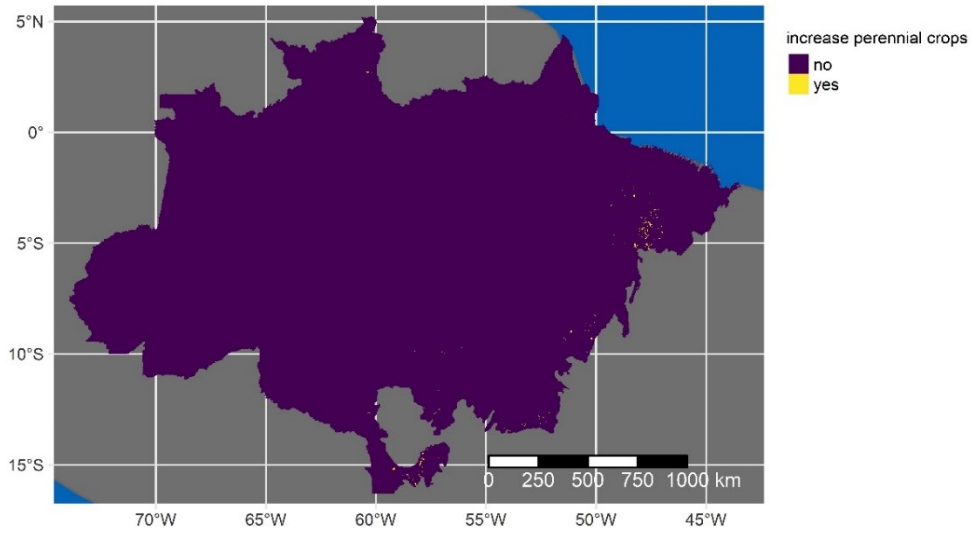


Figure S3. Map of the categorized agricultural land uses changes in 2021.

Increase agricultural land use	period 1 (2009-2011)	period 2 (2012-2015)	period 3 (2016-2018)	period 4 (2019-2021)
increase pasture				
increase	6.0%	6.3%	7.5%	8.5%
no increase	94%	94%	93%	92%
increase annual crop				
increase	0.6%	1.0%	1.1%	1.4%
no increase	99%	99%	99%	99%
increase perennial crop				
increase	<0.1%	0.1%	<0.1%	<0.1%
no increase	100%	100%	100%	100%

Table S2.4. Summary of the percentage of pixels in each category over the 4 period of analysis

Forest fragmentation Using the forest category of MapBiomas 7 (ID 3) and the *landscapemetrics* packages in R, edge density was calculated for every year at a 1 km resolution, a raster stack was created with the edge density values and was categorized.

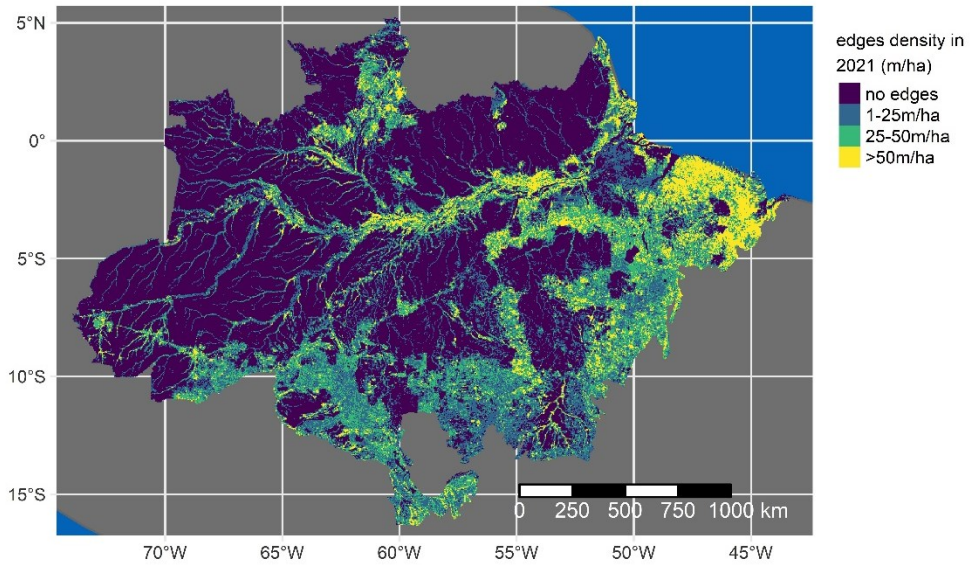


Figure S4. Map of the categorized edges density in 2021.

Fragmentation	period 1 (2009-2011)	period 2 (2012-2015)	period 3 (2016-2018)	period 4 (2019-2021)
edges density				
>50m/ha	6.1%	6.3%	6.9%	7.5%
1-25m/ha	22%	22%	22%	22%
25-50m/ha	14%	14%	15%	16%
no edges	58%	58%	57%	55%

Table S2.5. Summary of the percentage of pixels in each category over the 4 period of analysis

Distance agricultural edges MapBiomas collection 7 was used to identify 1 km pixels which contain any type of agricultural land use, before deriving between the centroid of these pixels and any 1 km pixels without any agricultural land. Then, the resulting raster stack was divided into 7 categories: pixels adjacent to agricultural edges, 0 to 1km, 1 to 2km, 2 to 5km, 5 to 10km, 10 to 25km and more than 25km.

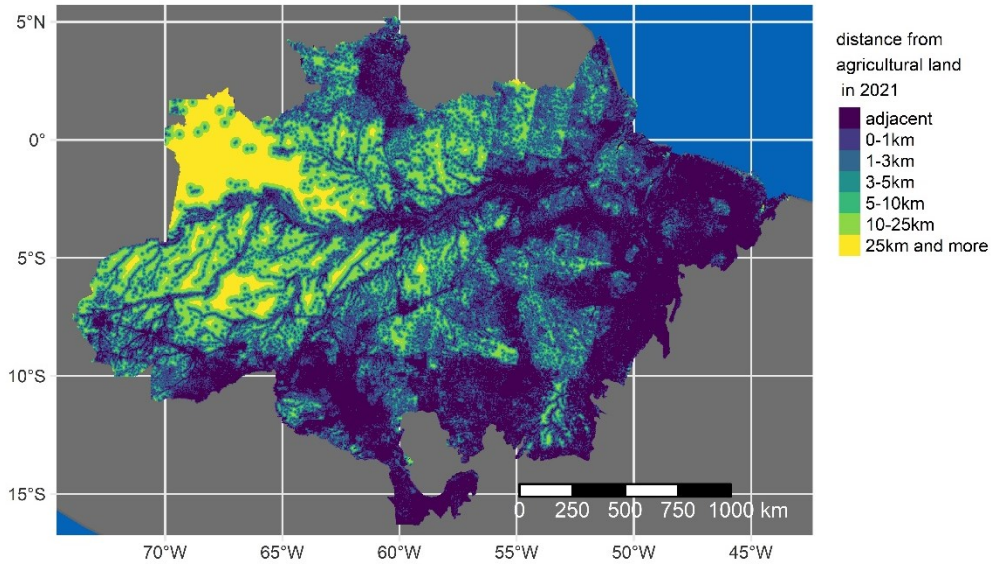


Figure S5. Map of the categorized distance to agricultural lands in 2021.

Distance from agricultural land	period 1 (2009-2011)	period 2 (2012-2015)	period 3 (2016-2018)	period 4 (2019-2021)
category				
Adjacent	28%	28%	29%	30%
0-1km	19%	19%	19%	20%
1-3km	16%	16%	15%	16%
3-5km	9.2%	9.2%	9.1%	8.9%
5-10km	12%	12%	12%	11%
10-25km	9.8%	9.6%	9.3%	9.5%
25km and more	6.7%	6.6%	6.4%	5.4%

Table S2.6. Summary of the percentage of pixels in each category over the 4 period of analysis

Remoteness The transport costs to port dataset developed by Victoria et al. 2021 (Victoria et al. 2021) was used, as it takes into account the evolution of the road network in the region, but also the presence of ports to export agricultural commodities. Since transport cost information was only available for 2005, 2010 and 2017, the transport cost of 2005 was used for 2009, the transport cost of 2010 was used for the 2011-2016 period and the transport

cost of 2017 was used for the 2017-2020 period. The values for transport costs to port were scaled by the mean standard deviations and compiled into a raster stack.

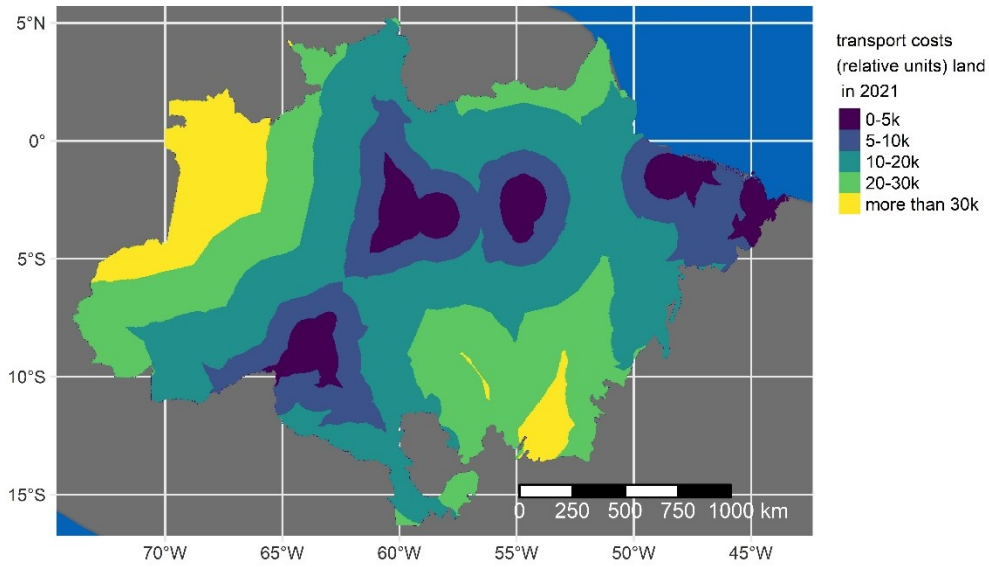


Figure S6. Map of the categorized transport costs in 2021.

Transport cost	period 1 (2009-2011)	period 2 (2012-2015)	period 3 (2016-2018)	period 4 (2019-2021)
Relative transport cost				
>5K RU	7.4%	7.9%	7.9%	7.9%
5-10K RU	17%	18%	18%	18%
10-20K RU	39%	39%	39%	39%
20-30K RU	26%	26%	26%	26%
more than 30K	9.9%	9.5%	9.4%	9.4%

Table S2.7. Summary of the percentage of pixels in each category over the 4 period of analysis

Governance protected areas data have been collected from the WDPA which includes both the spatial delimitation of protected areas, their categories according to the Brazilian classification system and the year of creation. The protected areas have been classified into the following categories:

- Sustainable use areas: include forests, environmental protection areas, sustainable development reserves, extractive reserves, areas of relevant ecological interest and natural heritage private reserves
- integral protection areas: include biological reserves, parks, ecological stations, wildlife refuges, and natural monuments.

- Indigenous lands: including only Indigenous land that has finished the delimitation process

These protected areas have been divided between periphery areas, corresponding to the first five kilometres between the protected areas and unprotected areas (thus not creating a buffer between two different protected areas), and core areas, more than 5km from the protected areas or Indigenous land border with unprotected areas.

Additionally, the database of PADDD events in the Brazilian Amazon was downloaded on padddtracker website (PADDtracker 2020), and the downgrading of protected areas was excluded as they might not necessarily represent a weaker protection effort in the region. A raster stack has been created with the proportions of pixels covered by PADDD events before or during each year of the period of study. Rural settlement polygons have been downloaded from the INCRA websites and then filtered to remove sustainable use areas that were included in the sustainable use areas.

For each year,Ie derived a raster with the dominant types of land governance on each pixel, corresponding to the 10 categories.

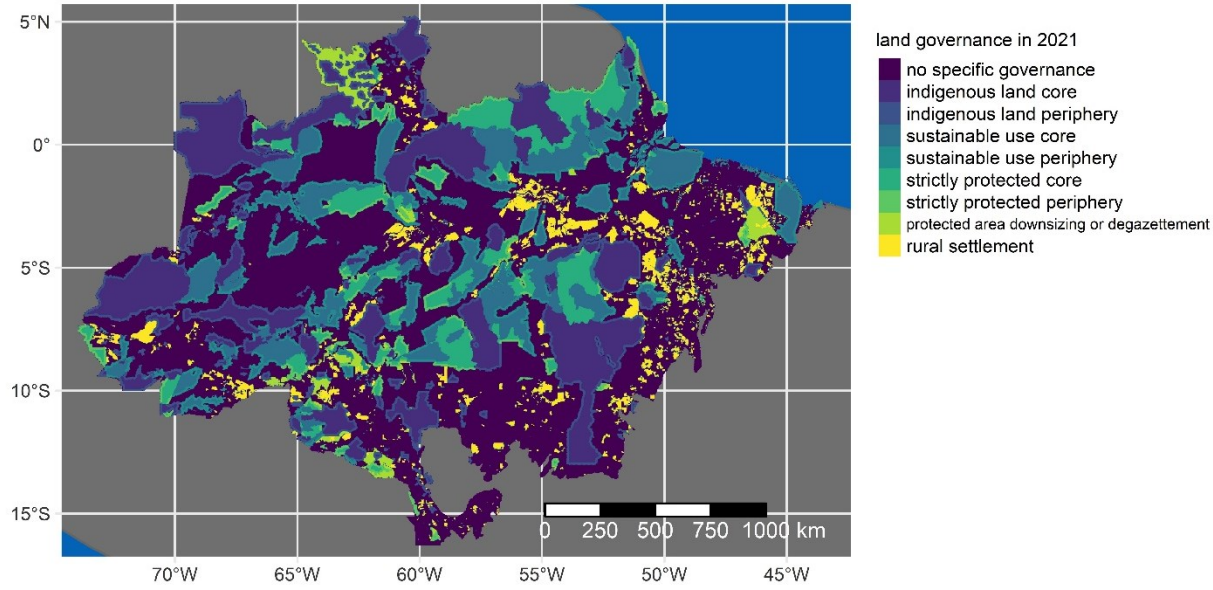


Figure S7. Map of the categorized land governance, including the rural settlements, protected areas and Indigenous lands in 2021.

Land governance	period 1 (2009-2011)	period 2 (2012-2015)	period 3 (2016-2018)	period 4 (2019-2021)
category				
no protection	47%	46%	45%	45%
indigenous land periphery	6.4%	6.5%	6.5%	6.5%
indigenous land core	15%	16%	16%	16%
sustainable use periphery	6.1%	6.0%	6.0%	6.0%
sustainable use core	10%	10%	11%	11%
strictly protected periphery	2.5%	2.5%	2.5%	2.5%
strictly protected core	6.0%	6.0%	6.4%	6.4%
protected area downsizing and degazettement	1.5%	1.7%	1.7%	1.7%
rural settlement	5.1%	5.2%	5.3%	5.3%

Table S2.8. Summary of the percentage of pixels in each category over the 4 period of analysis

Priority list

The list of priorities municipalities published by the ministry of the environment has been used to create a raster stack with values indicating if the municipality is currently on the blacklist or if it used to be on the blacklist but has been removed, indicating decreasing deforestation pressure and fulfilment of certain conditions such as the registration in the rural land registry.

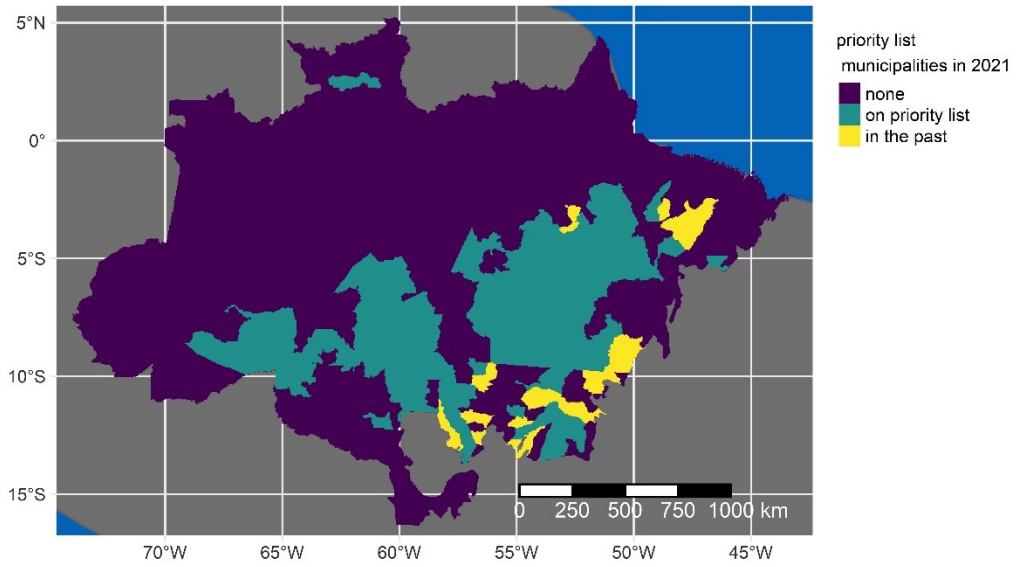


Figure S8. Map of the municipalities on the priority list in 2021.

Municipalities on priority list	period 1 (2009-2011)	period 2 (2012-2015)	period 3 (2016-2018)	period 4 (2019-2021)
category				
never on priority list	80%	79%	75%	73%
priority list	19%	20%	23%	24%
removed from priority list	0.2%	1.2%	2.1%	3.3%

Table S2.9. Summary of the percentage of pixels in each category over the 4 period of analysis

Appendix 2.3: Bayesian spatio-temporal modelling approach for understanding Active-Fires occurrence

Compared to previous work for understanding the drivers of fire regimes in the Brazilian Amazon, one major difference in my analysis was the inclusion of a spatio-temporal component. A careful design of the models attempts to include most of the important drivers of the fire regimes, but some drivers can hardly be captured by numerical variables (e.g. fine-scale governance process), while for other drivers no data sources could be identified (e.g. logging and forest degradation). According to Tobler's first law, "everything is related to everything else, but near things are more related than distant things" (Tobler 1970) and fires close to each other are more likely to be influenced by similar underlying processes than distant fires. Moreover, Actives-Fires detection is not completely independent: one large fire can lead to many active-fires detections clustered in space and time. In this annex, I provide a brief overview of the Bayesian statistical foundations of my modelling approach.

Log Gaussian Cox Process

Log Gaussian Cox Process is a class of models for modelling non-stationary point processes (Serra et al. 2014; Opitz et al. 2020). The Cox Process represents a Poisson process for the distribution of the points with an intensity function varying across the mathematical space, in this case across space and time. The intensity function of the Cox Process depends on a Gaussian Process that includes both the contribution of the explanatory variables and spatiotemporal dependence structure.

$$\text{Number Active Fires}_{(st)} \sim \text{Poisson}(\text{Intensity process}_{(st)})$$

$$\text{Intensity process}_{(st)} = \exp\left(\sum_{i=1}^{n} cov_{i(st)} * \beta_i + Y_{(st)}\right)$$

Considering that *st* represents a defined space and time for observation of the fire patterns, *n* represents the total number of covariates, *cov* the values of the covariate, β the coefficient

attributed to the covariate and Y the residual process explained by spatiotemporal correlations.

Bayesian inferences

In Bayesian statistics, the posterior distribution of a model parameter, in my case indicative of the impact of covariates on fire occurrence, is proportional to the density function of a model (likelihood) and a set of prior beliefs on the hyper-parameters. The objective of the approach is to estimate the posterior marginals of model effects and hyperparameters, that could be used to investigate both the impact of covariates on the response variables. Two approaches can be used to estimate the posterior joint distribution of the model parameters:

- Markov Chain Monte Carlo (MCMC)
- Integrated Nested Laplace Approximation (INLA)

The Integrated Nested Laplace Approximation, thanks to the use of computational properties of latent Gaussian models, reduce drastically the computation time compared to a classic MCMC algorithm with a moderate decline in precision (Carroll et al. 2015). We fitted my LGCP model using inlabru (Bachl et al. 2019), a wrapper R package for R-INLA.

Stochastic Partial Differential Equation (SPDE) approach

To represent the spatial correlation, I rely on the Matérn covariance function that determines the correlation between two predictors according to their distance. To embed this into INLA, the Stochastic Partial Differential Equation approach is used to represent the spatial autocorrelation into the model by simplifying a continuous Gaussian field into a more sober Gaussian Markov Random Field thanks to a discretization into non-intersecting triangles. A projector matrix is then created to associate each observation with three nodes of the mesh in which it is located, thus creating a sparse matrix with only three non-zero values per row. The spatial covariance function and the dense covariance matrix of a Gaussian Field are represented by a neighbourhood structure and a sparse precision matrix, graphically defined by a mesh (Juan Verdoy 2021). Briefly, the spatial process can be represented by the basic function:

$U_{(s)} = \sum k=1 m \psi k(s) wk$

where ψk are basis function, Wk are Gaussian distributed weight, m being the number of vertices in the mesh. The joint distribution for the weights determines the full distribution in the continuous domain.

Mesh creation

For each model, I generated a mesh based on the locations of the observation points. A minimal value of triangles edges of 1 kilometres has been set, to assure efficient computation of spatial autocorrelations even with a range value of around 5 kilometres. Other constraints on the angles of the triangles and the maximum number of triangles within the border have been imposed for having a fine mesh around active fires and a coarser mesh in areas with few active fires (Fig. S5). The border of the mesh has been simplified using the *inla.nonconvex.hull* function*:* to ensure all observed points are in triangles within the border of the mesh, and the mesh has been extended outside the border to compute spatial autocorrelations on the edges of the model.

Priors' distribution

We specified penalized complexity priors frameworks, a class of weakly informative priors (Simpson et al. 2017), for the spatial and temporal component and temporal components.

The penalized complexity priors of the Matérn-SPDE model can be controlled by two parameters:

Spatial range: The user defines a spatial range *p_0* and a lower tail quantile *p_p* for which spatial interactions will be smaller than the determined spatial range, such as P(*p*<*p_0*)=*p_p*.
Specification used: prior.range=c(10,0.5) correspond to a 50% chance that spatial interactions is less than 10 kilometers

Sigma: The user defines a standard deviation σ_0 and an upper tail quantile p_σ for which the effective standard deviation of the spatial field will be higher than the determined standard deviation, such as P(σ> σ_0)=p_ σ. Specification used: prior.sigma=c(15,0.05) correspond to a 5% chance that spatial interactions will have a deviation of more than 15 km.

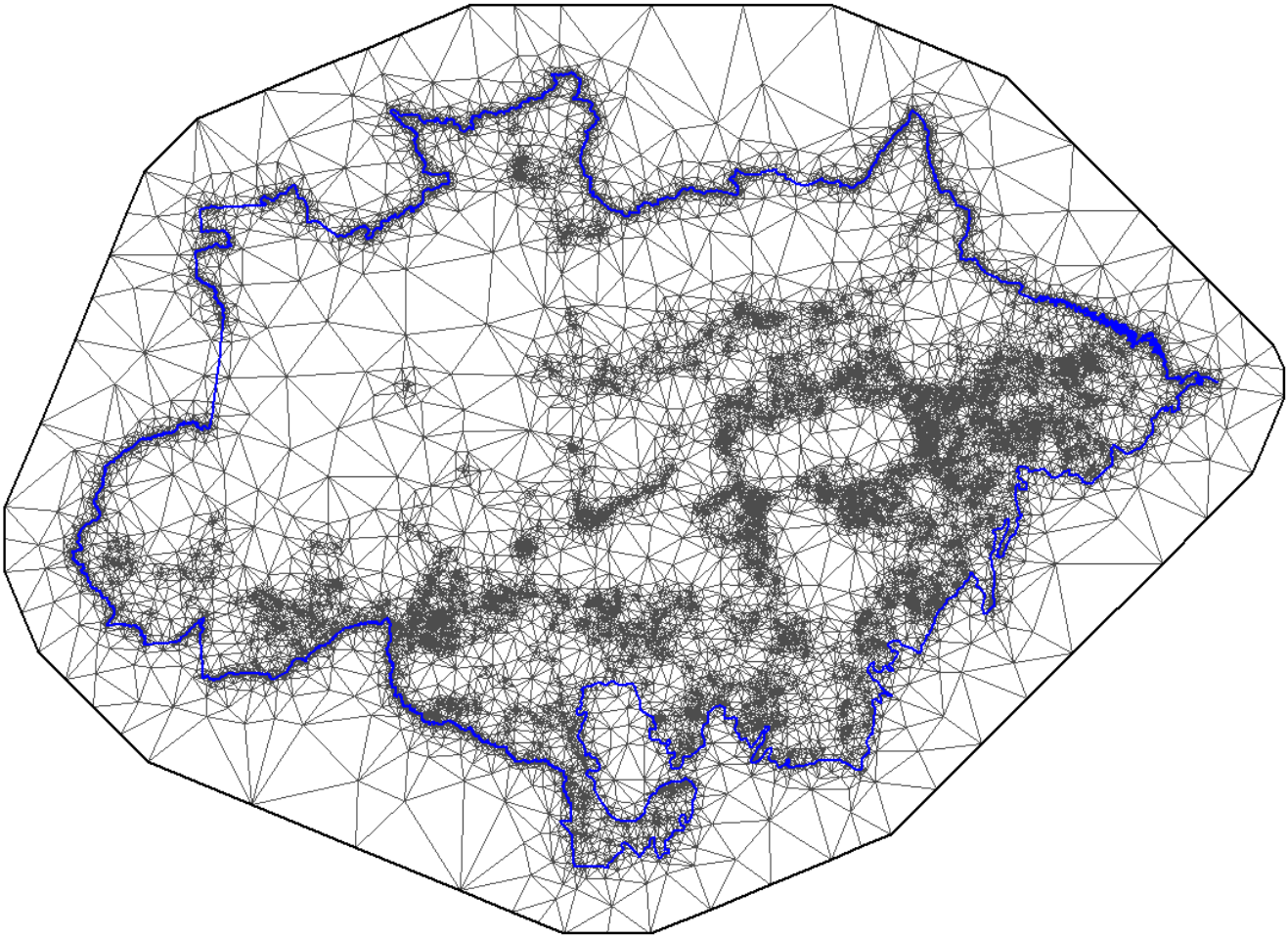


Figure S9. Mesh created for the 2009-201 deforestation fires model.

Appendix 2.4: Comparison of the residuals and observed fires for all of the models

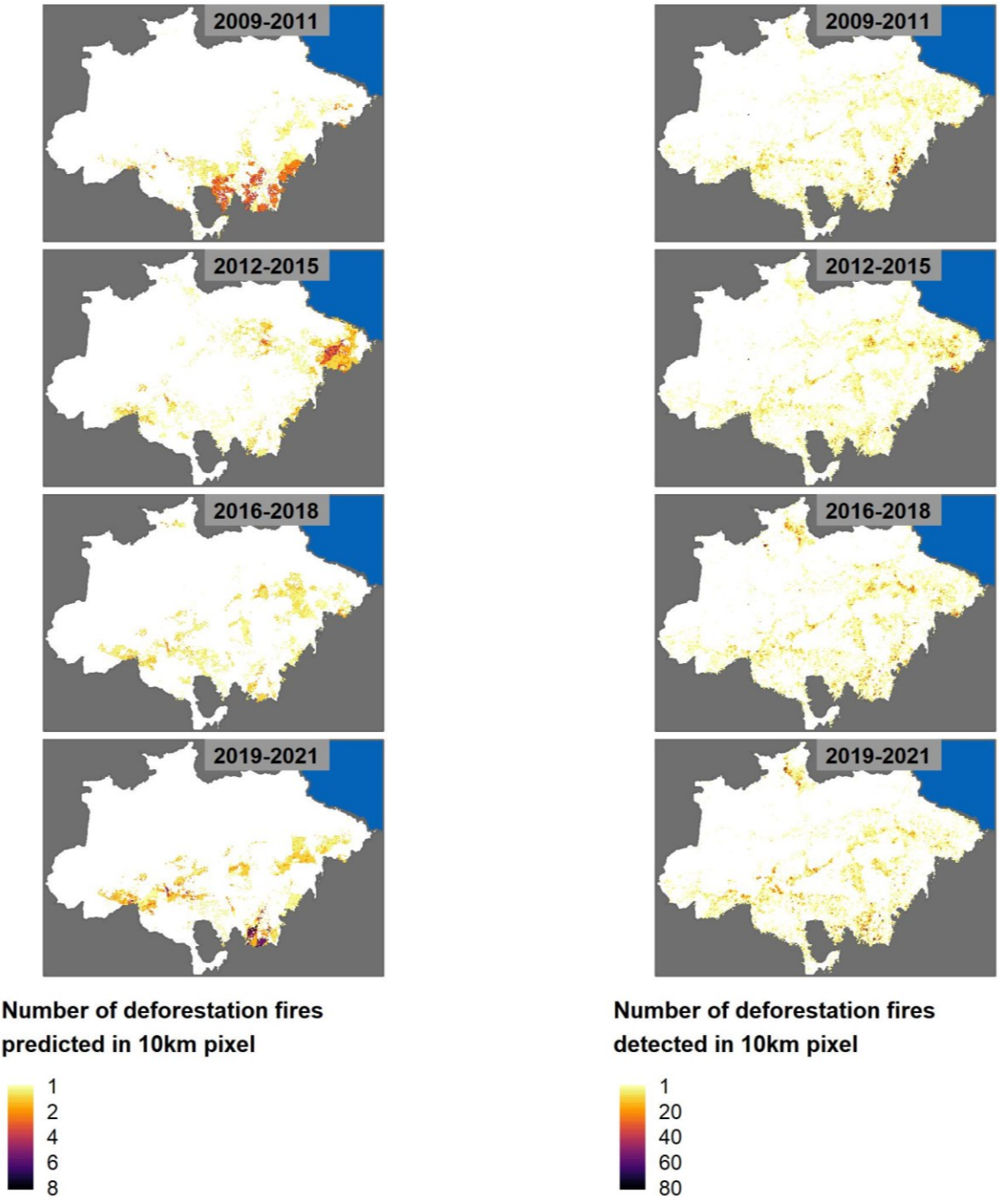


Figure S10. Map of the number of median numbers of deforestation fires predicted by the models (left column) and observed right column) in 10km2 pixels for the four periods.

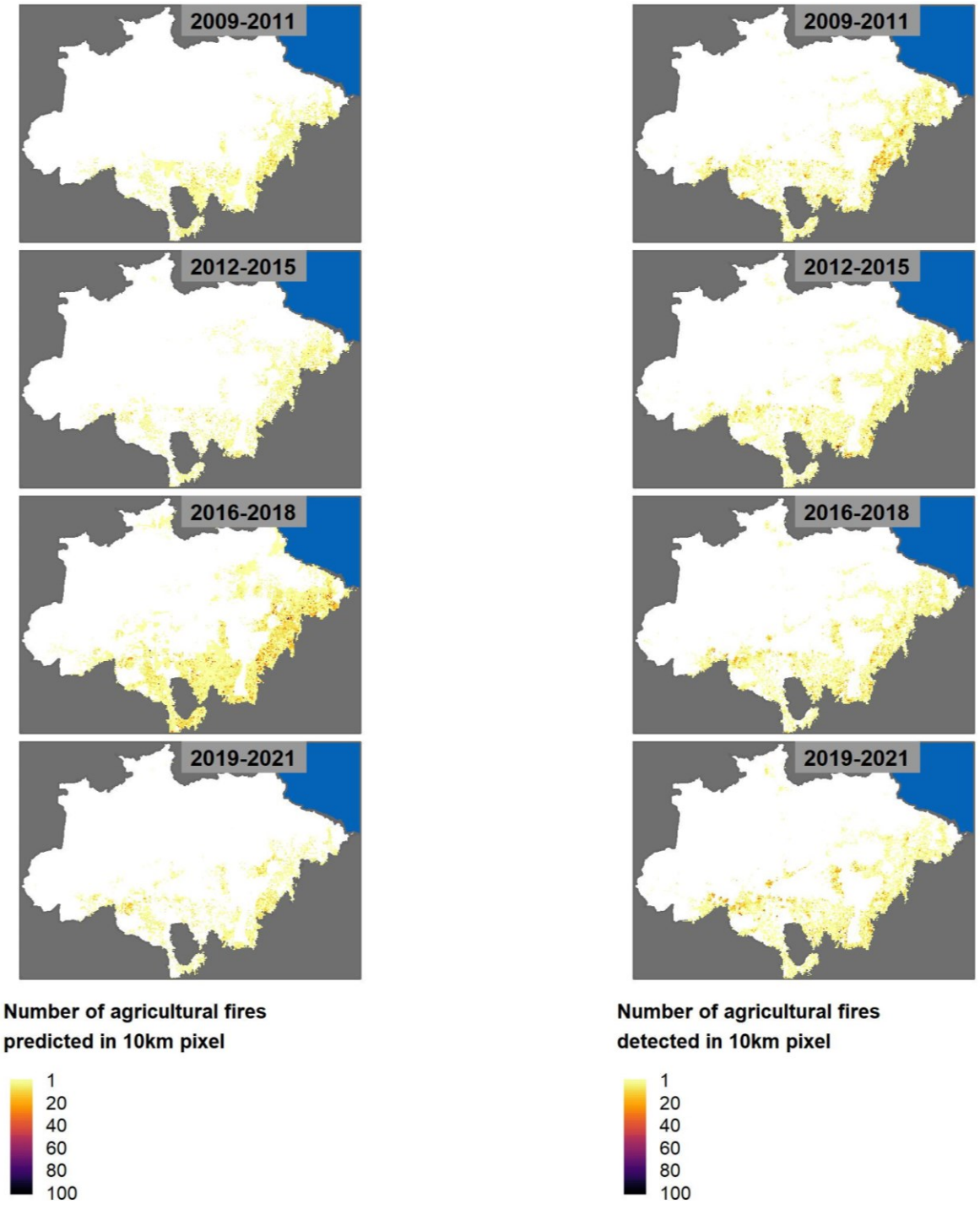


Figure S11. Map of the number of median numbers of agricultural fires predicted by the models (left column) and observed right column) in 10km2 pixels for the four periods.

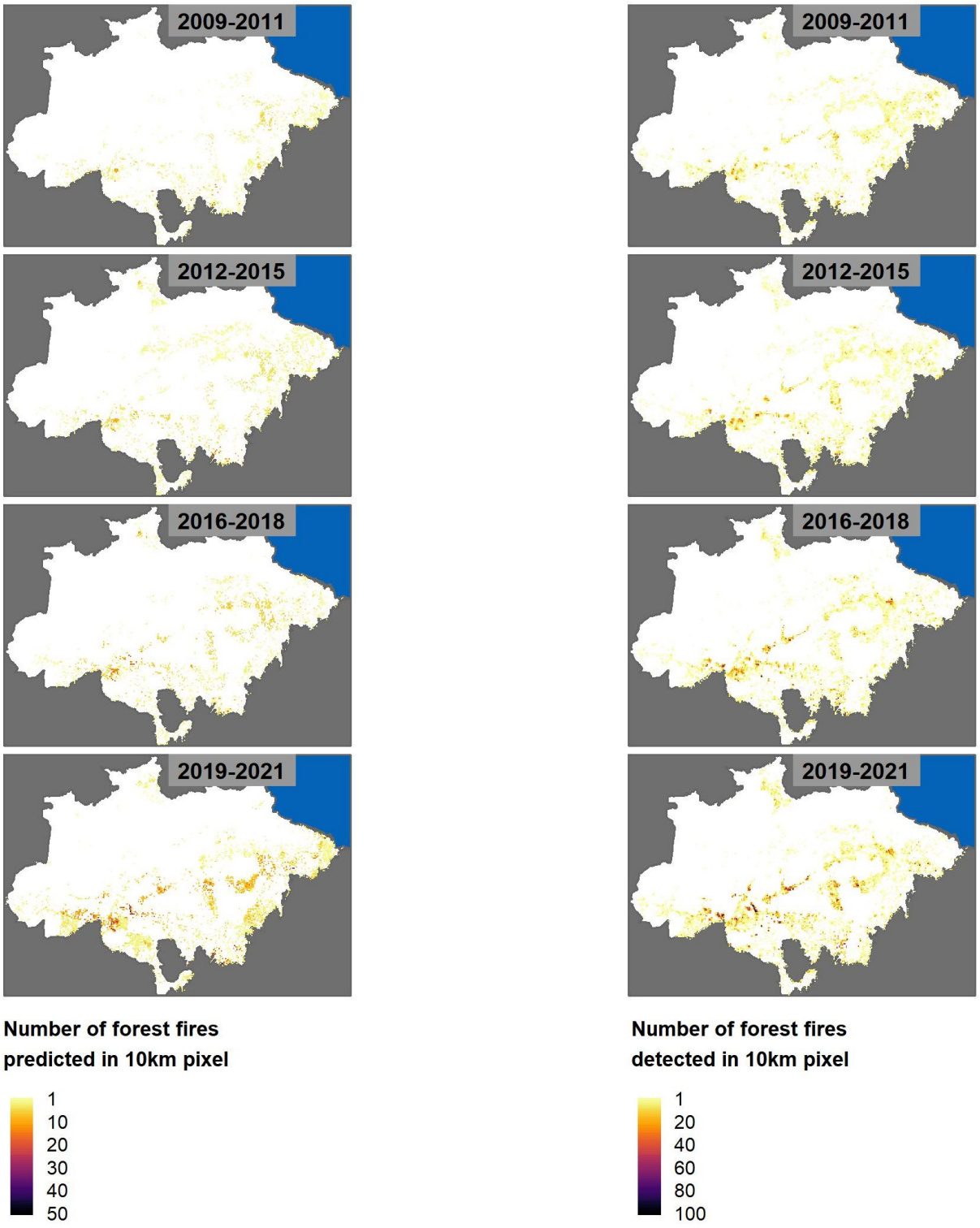


Figure S12. Map of the number of median numbers of forest fires predicted by the models (left column) and observed right column) in 10km2 pixels for the four periods.

Appendices from Chapter 3

Appendix 3.1. Results of the statistical test for analysis of land use and land use change

PERMANOVA results on land use composition					
	Df	Sum of Squares	R^2	Pseudo-F	p_value
Model	2.00	4.29	0.60	85.25	<0.005
Residual	114.00	2.87	0.40	NA	n.s.
Total	116.00	7.16	1.00	NA	n.s.

Table S3.1 Results from the PERMANOVA analysis of difference in land use across land governance type.

Pairwise Dunn's Tests for Land Use Composition Between Land Governance Types					
Governance type 1	**Governance type 2**	**Z Score**	**p-value**	**p-value (label)**	**Significance**
		forest			
indigenous land	rural settlement	8.876	0.000	<0.0001	***
indigenous land	undesignated land	6.356	0.000	<0.0001	***
rural settlement	undesignated land	−2.520	0.035	<0.05	*
		rocky outcrop			
indigenous land	rural settlement	10.348	0.000	<0.0001	***
indigenous land	undesignated land	5.174	0.000	<0.0001	***
rural settlement	undesignated land	−5.174	0.000	<0.0001	***
		savanna			
indigenous land	rural settlement	−6.449	0.000	<0.0001	***
indigenous land	undesignated land	2.333	0.059	n.s.	n.s.
rural settlement	undesignated land	8.783	0.000	<0.0001	***
		wetland			
indigenous land	rural settlement	5.077	0.000	<0.0001	***
indigenous land	undesignated land	10.155	0.000	<0.0001	***
rural settlement	undesignated land	5.077	0.000	<0.0001	***

Pairwise Dunn's Tests for Land Use Composition Between Land Governance Types					
Governance type 1	**Governance type 2**	**Z Score**	**p-value**	**p-value (label)**	**Significance**
		other			
indigenous land	rural settlement	−10.155	0.000	<0.0001	***
indigenous land	undesignated land	−5.077	0.000	<0.0001	***
rural settlement	undesignated land	5.077	0.000	<0.0001	***
		other temporary crops			
indigenous land	rural settlement	−6.760	0.000	<0.0001	***
indigenous land	undesignated land	−7.541	0.000	<0.0001	***
rural settlement	undesignated land	−0.781	1.000	n.s.	n.s.
		pasture			
indigenous land	rural settlement	−8.856	0.000	<0.0001	***
indigenous land	undesignated land	−6.376	0.000	<0.0001	***
rural settlement	undesignated land	2.480	0.039	<0.05	*
		river			
indigenous land	rural settlement	10.155	0.000	<0.0001	***
indigenous land	undesignated land	5.077	0.000	<0.0001	***
rural settlement	undesignated land	−5.077	0.000	<0.0001	***
		soybean			
indigenous land	rural settlement	−5.200	0.000	<0.0001	***
indigenous land	undesignated land	−7.141	0.000	<0.0001	***
rural settlement	undesignated land	−1.942	0.157	n.s.	n.s.
		forest plantation			
indigenous land	rural settlement	−5.617	0.000	<0.0001	***
indigenous land	undesignated land	−8.783	0.000	<0.0001	***
rural settlement	undesignated land	−3.166	0.005	<0.005	**

Table S3.2. Results from the post-hoc Dunn's test to evaluate land use differences across land governance type.

Mann-Kendall results on land use trends			
land use	**p-value**	**tau**	**TS slope (%.year^{-1})**
rural settlement			
other temporary crops	<0.0001	0.89	0.12
soybean	<0.0001	0.93	0.18
pasture	<0.0001	0.93	1.75
undesignated land			
pasture	<0.0001	0.59	0.60
soybean	<0.0001	0.98	0.37

Table S3.3. Results from the Man-Kendall test on trends of land use change.

Appendix 3.2. Results of the statistical test for analysis of fragmentation

Dunn's Test Results for Land Use Metrics			
Group Comparison	**Z-value**	**Adjusted p**	**Significance**
perimeter-area ratio			
indigenous land - rural settlement	−242.774	<0.0001	***
indigenous land - undesignated land	−211.745	<0.0001	***
rural settlement - undesignated land	133.792	<0.0001	***
contiguity index			
indigenous land - rural settlement	242.352	<0.0001	***
indigenous land - undesignated land	210.724	<0.0001	***
rural settlement - undesignated land	−133.980	<0.0001	***
nearest-neighbor distance			
indigenous land - rural settlement	−49.852	<0.0001	***
indigenous land - undesignated land	−42.211	<0.0001	***
rural settlement - undesignated land	21.829	<0.0001	***
large patch index			
indigenous land - rural settlement	265.243	<0.0001	***
indigenous land - undesignated land	234.773	<0.0001	***
rural settlement - undesignated land	−143.968	<0.0001	***

Table S3.4 Results of the post hoc Dunn's Test to evaluate the difference of forest fragmentation metrics in 2022 across land governance types

Dunn's Test Results for yearly change of Large Patch Index

Group Comparison	Z-value	Adjusted p	Significance
1992-2002			
indigenous land - rural settlement	−168.946	<0.0001	***
indigenous land - undesignated land	−130.786	<0.0001	***
rural settlement - undesignated land	102.127	<0.0001	***
2002-2012			
indigenous land - rural settlement	−94.684	<0.0001	***
indigenous land - undesignated land	−66.879	<0.0001	***
rural settlement - undesignated land	61.982	<0.0001	***
2012-2022			
indigenous land - rural settlement	−61.561	<0.0001	***
indigenous land - undesignated land	−46.016	<0.0001	***
rural settlement - undesignated land	38.968	<0.0001	***

Dunn's Test Results for yearly change of Perimeter-Area Ratio

Group Comparison	Z-value	Adjusted p	Significance
1992-2002			
indigenous land - rural settlement	−168.946	<0.0001	***
indigenous land - undesignated land	−130.786	<0.0001	***
rural settlement - undesignated land	102.127	<0.0001	***
2002-2012			
indigenous land - rural settlement	−94.684	<0.0001	***
indigenous land - undesignated land	−66.879	<0.0001	***
rural settlement - undesignated land	61.982	<0.0001	***
2012-2022			
indigenous land - rural settlement	−61.561	<0.0001	***
indigenous land - undesignated land	−46.016	<0.0001	***
rural settlement - undesignated land	38.968	<0.0001	***

Dunn's Test Results for yearly change of Nearest Neighbor Distance

Group Comparison	Z-value	Adjusted p	Significance
1992-2002			
indigenous land - rural settlement	−22.268	<0.0001	***
indigenous land - undesignated land	−20.966	<0.0001	***
rural settlement - undesignated land	8.464	<0.0001	***
2002-2012			
indigenous land - rural settlement	−17.154	<0.0001	***
indigenous land - undesignated land	−0.515	n.s.	ns
rural settlement - undesignated land	24.007	<0.0001	***
2012-2022			
indigenous land - rural settlement	−13.457	<0.0001	***
indigenous land - undesignated land	−1.171	n.s.	ns
rural settlement - undesignated land	19.285	<0.0001	***

Dunn's Test Results for yearly change of Contiguity Index

Group Comparison	Z-value	Adjusted p	Significance
1992-2002			
indigenous land - rural settlement	−168.443	<0.0001	***
indigenous land - undesignated land	−130.059	<0.0001	***
rural settlement - undesignated land	102.049	<0.0001	***
2002-2012			
indigenous land - rural settlement	−95.034	<0.0001	***
indigenous land - undesignated land	−65.776	<0.0001	***
rural settlement - undesignated land	63.088	<0.0001	***
2012-2022			
indigenous land - rural settlement	−61.708	<0.0001	***
indigenous land - undesignated land	−45.262	<0.0001	***
rural settlement - undesignated land	39.611	<0.0001	***

Table S3.5 Results of the post hoc Dunn's Test to evaluate difference in rate of change of forest fragmentation metrics in 2022 across land governance types

Dunn's Test Results for yearly change of Large Patch Index			
Group Comparison	Z-value	Adjusted p	Significance
indigenous land			
1992-2002/2002-2012	−16.353	<0.0001	***
2002-2012/2012-2022	−38.376	<0.0001	***
undesignated land			
1992-2002/2002-2012	91.026	<0.0001	***
2002-2012/2012-2022	24.771	<0.0001	***
rural settlment			
1992-2002/2002-2012	17.535	<0.0001	***
2002-2012/2012-2022	54.471	<0.0001	***

Dunn's Test Results for yearly change of Perimeter-Area Ratio			
Group Comparison	Z-value	Adjusted p	Significance
indigenous land			
1992-2002/2002-2012	−15.737	<0.0001	***
2002-2012/2012-2022	−37.367	<0.0001	***
undesignated land			
1992-2002/2002-2012	48.080	<0.0001	***
2002-2012/2012-2022	−11.858	<0.0001	***
rural settlment			
1992-2002/2002-2012	16.366	<0.0001	***
2002-2012/2012-2022	9.197	<0.0001	***

Dunn's Test Results for yearly change of Nearest Neighbor Distance			
Group Comparison	Z-value	Adjusted p	Significance
indigenous land			
1992-2002/2002-2012	−3.977	<0.0005	***
2002-2012/2012-2022	7.962	<0.0001	***
undesignated land			
1992-2002/2002-2012	30.378	<0.0001	***
2002-2012/2012-2022	5.931	<0.0001	***
rural settlment			
1992-2002/2002-2012	11.601	<0.0001	***
2002-2012/2012-2022	1.367	n.s.	ns

Dunn's Test Results for yearly change of Contiguity Index			
Group Comparison	Z-value	Adjusted p	Significance
indigenous land			
1992-2002/2002-2012	−15.585	<0.0001	***
2002-2012/2012-2022	−38.057	<0.0001	***
undesignated land			
1992-2002/2002-2012	49.007	<0.0001	***
2002-2012/2012-2022	−11.408	<0.0001	***
rural settlment			
1992-2002/2002-2012	16.559	<0.0001	***
2002-2012/2012-2022	10.552	<0.0001	***

Table S3.6 Results of the post hoc Dunn's Test to evaluate difference in rate of change of forest fragmentation metrics in 2022 across periods.

Appendix 3.3. Results of the statistical test for spatial distribution

Dunn's Test Results for fire occurence			
Group Comparison	Z-value	Adjusted p	Significance
Active-Fires.1000km^{-2}			
indigenous land/ rural settlement	−6.487	<0.0001	***
indigenous land / undesignated land	−4.586	<0.0001	***
rural settlement / undesignated land	1.901	n.s.	ns
Percentage of burned area			
indigenous land/ rural settlement	−3.738	<0.001	***
indigenous land / undesignated land	0.346	n.s.	ns
rural settlement / undesignated land	4.084	<0.0005	***

Table S3.7 Results of the post hoc Dunn's Test to evaluate the occurrence of fires across different types of land use categories

Dunn's Test Results for fire occurence			
Group Comparison	**Z-value**	**Adjusted p**	**Significance**
<10ha			
indigenous land - rural settlement	−6.644	<0.0001	***
indigenous land - undesignated land	−4.100	<0.0005	***
rural settlement - undesignated land	2.545	<0.05	***
10-100ha			
indigenous land - rural settlement	−6.809	<0.0001	***
indigenous land - undesignated land	−4.312	<0.0001	***
rural settlement - undesignated land	2.497	<0.05	***
100-1000ha			
indigenous land - rural settlement	−4.602	<0.0001	***
indigenous land - undesignated land	−5.552	<0.0001	***
rural settlement - undesignated land	−0.950	n.s.	ns
>10000ha			
indigenous land - rural settlement	5.068	<0.0001	***
indigenous land - undesignated land	4.782	<0.0001	***
rural settlement - undesignated land	−0.286	n.s.	ns

Table S3.8 Results of the post hoc Dunn's Test to evaluate the difference in the relative contribution of different fire size to total burned areas

Appendix 3.4. Results of the statistical test for fire characteristics

Mann-Kendall results on land use burned				
land use	**p-value**	**tau**	**TS slope (%.year^{-1})**	**significance**
rural settlement				
grassland	<0.05	0.36	0.08	*
pasture	<0.0005	−0.54	−1.17	***
rocky outcrop	<0.01	0.42	0.02	**
soybean	<0.01	0.41	0.03	**
wetland	<0.0005	0.58	0.46	***
indigenous land				
grassland	<0.005	−0.45	−0.02	**
wetland	<0.005	−0.48	−1.09	**
undesignated land				
forest	<0.01	0.42	1.11	**
other temporary crops	<0.005	−0.48	−0.23	**
pasture	<0.005	−0.49	−1.28	**
soybean	<0.01	0.41	0.07	**

Table S3.9 Results from the Man-Kendall test on trends of proportion of land use burned.

PERMANOVA results on land use composition					
	Df	**Sum of Squares**	**R^2**	**Pseudo-F**	**p_value**
Model	2.00	2.67	0.64	57.08	<0.005
Residual	63.00	1.48	0.36	NA	n.s.
Total	65.00	4.15	1.00	NA	n.s.

Table S3.10 Results from the PERMANOVA analysis of difference in land use burned across land governance type.

Pairwise Dunn's Tests for Land Use of Burned Area Between Land Governance Types					
Group 1	Group 2	Z Score	p-value	p-value (label)	Significance
forest					
indigenous land	rural settlement	1.273	0.610	n.s.	n.s.
indigenous land	undesignated land	1.273	0.610	n.s.	n.s.
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.
grassland					
indigenous land	rural settlement	−5.161	0.000	<0.0001	***
indigenous land	undesignated land	−5.161	0.000	<0.0001	***
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.
other					
indigenous land	rural settlement	3.370	0.002	<0.005	**
indigenous land	undesignated land	3.370	0.002	<0.005	**
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.
other temporary crops					
indigenous land	rural settlement	−5.279	0.000	<0.0001	***
indigenous land	undesignated land	−5.279	0.000	<0.0001	***
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.
pasture					
indigenous land	rural settlement	−5.656	0.000	<0.0001	***
indigenous land	undesignated land	−5.656	0.000	<0.0001	***
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.
rocky outcrop					
indigenous land	rural settlement	5.326	0.000	<0.0001	***
indigenous land	undesignated land	5.326	0.000	<0.0001	***
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.
savanna					
indigenous land	rural settlement	5.538	0.000	<0.0001	***
indigenous land	undesignated land	5.538	0.000	<0.0001	***
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.
wetland					
indigenous land	rural settlement	5.703	0.000	<0.0001	***
indigenous land	undesignated land	5.703	0.000	<0.0001	***
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.
forest plantation					
indigenous land	rural settlement	−5.773	0.000	<0.0001	***
indigenous land	undesignated land	−5.773	0.000	<0.0001	***
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.
soybean					
indigenous land	rural settlement	−5.811	0.000	<0.0001	***
indigenous land	undesignated land	−5.811	0.000	<0.0001	***
rural settlement	undesignated land	0.000	1.000	n.s.	n.s.

Table S3.11. Results from the post-hoc Dunn's test to evaluate difference in the proportions of land use of the burned area across land governance type.

Appendix 3.5. Results of the GAM model between FWI annual index and fire regimes

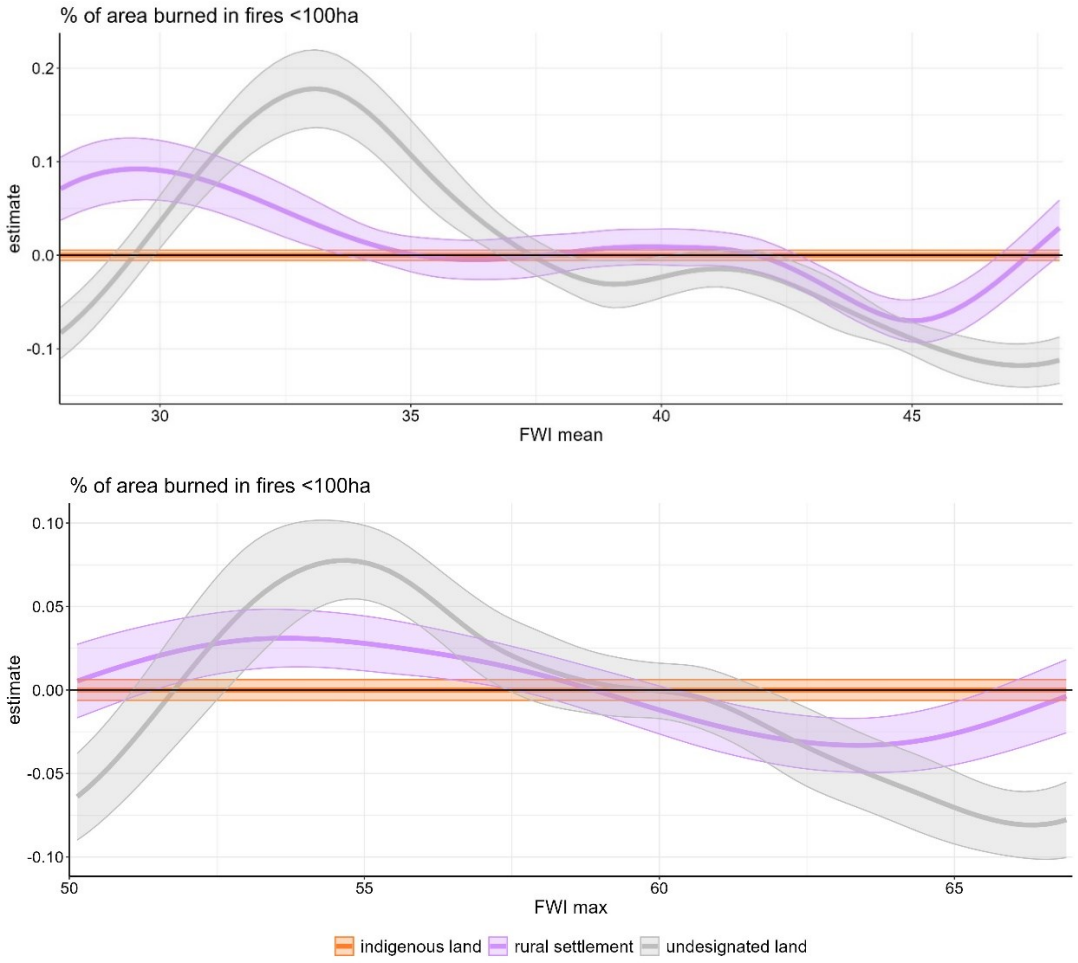


Figure S13. Smooth estimate of the effect of FWI mean and maximum FWI on the percentage of total area burned in fires of less than 100 hectares.

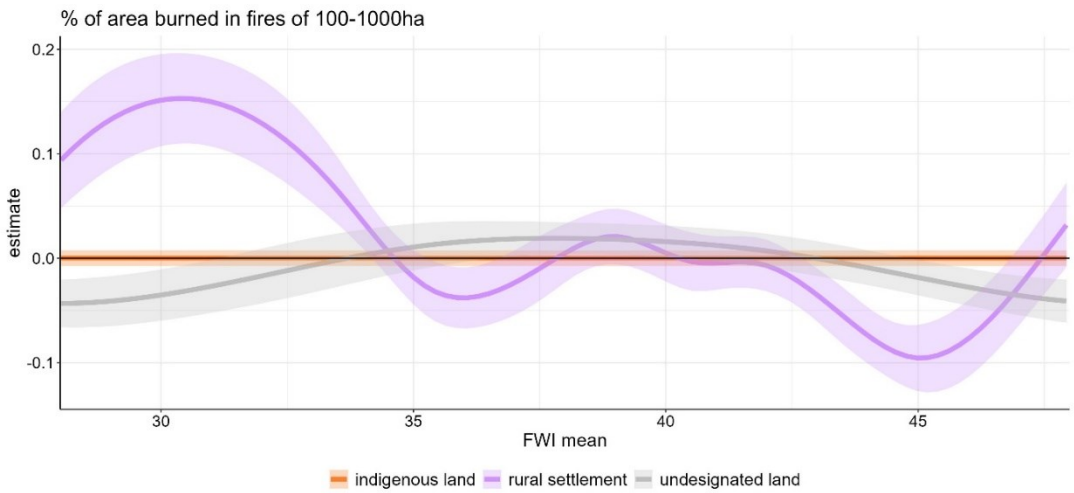


Figure S14. Smooth estimate of the effect of FWI mean on the percentage of total area burned in fires between 100 and 1000 hectares.

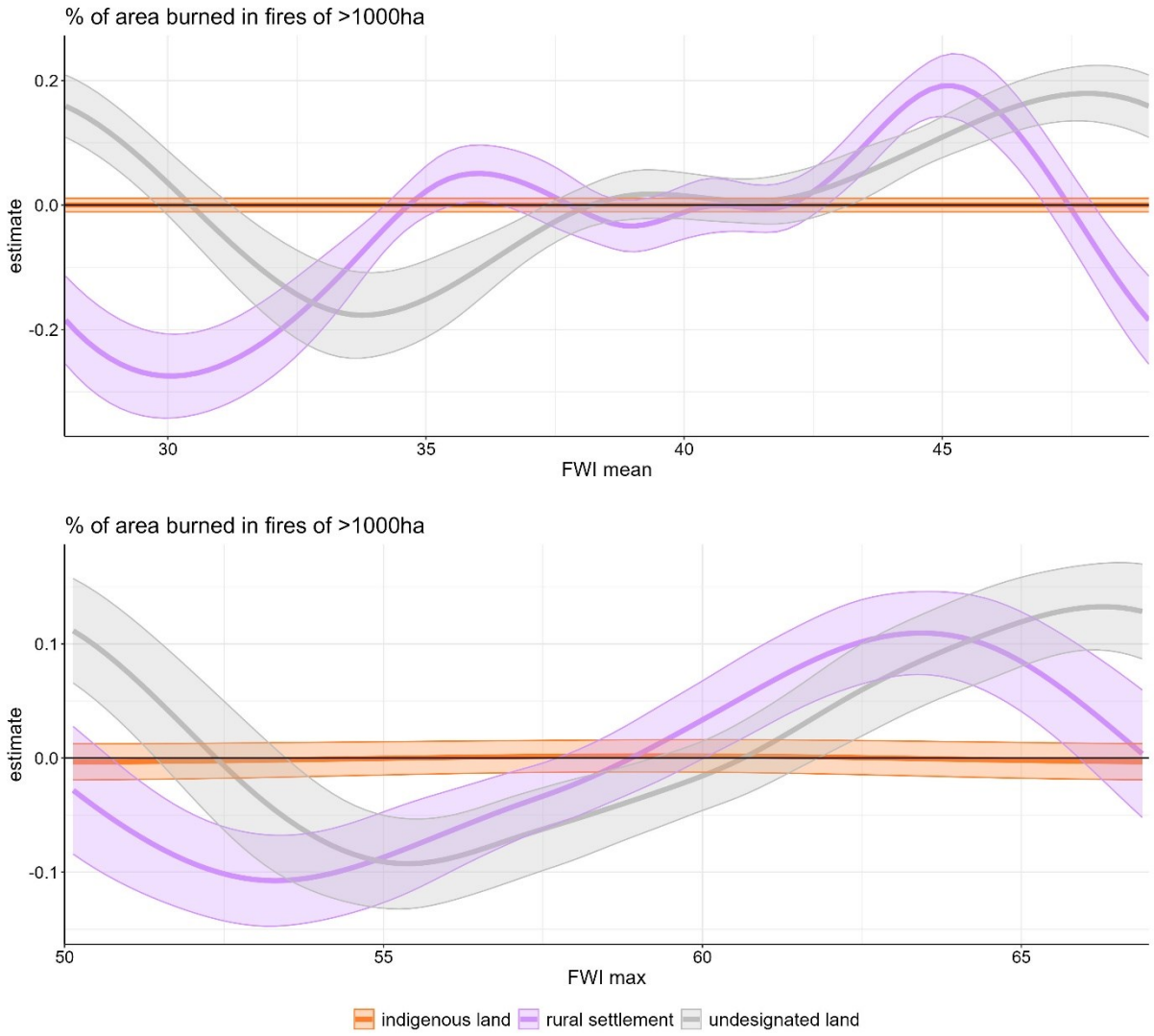


Figure S15. Smooth estimate of the effect of FWI mean and maximum FWI on the percentage of total area burned in fires of more than 1000 hectares

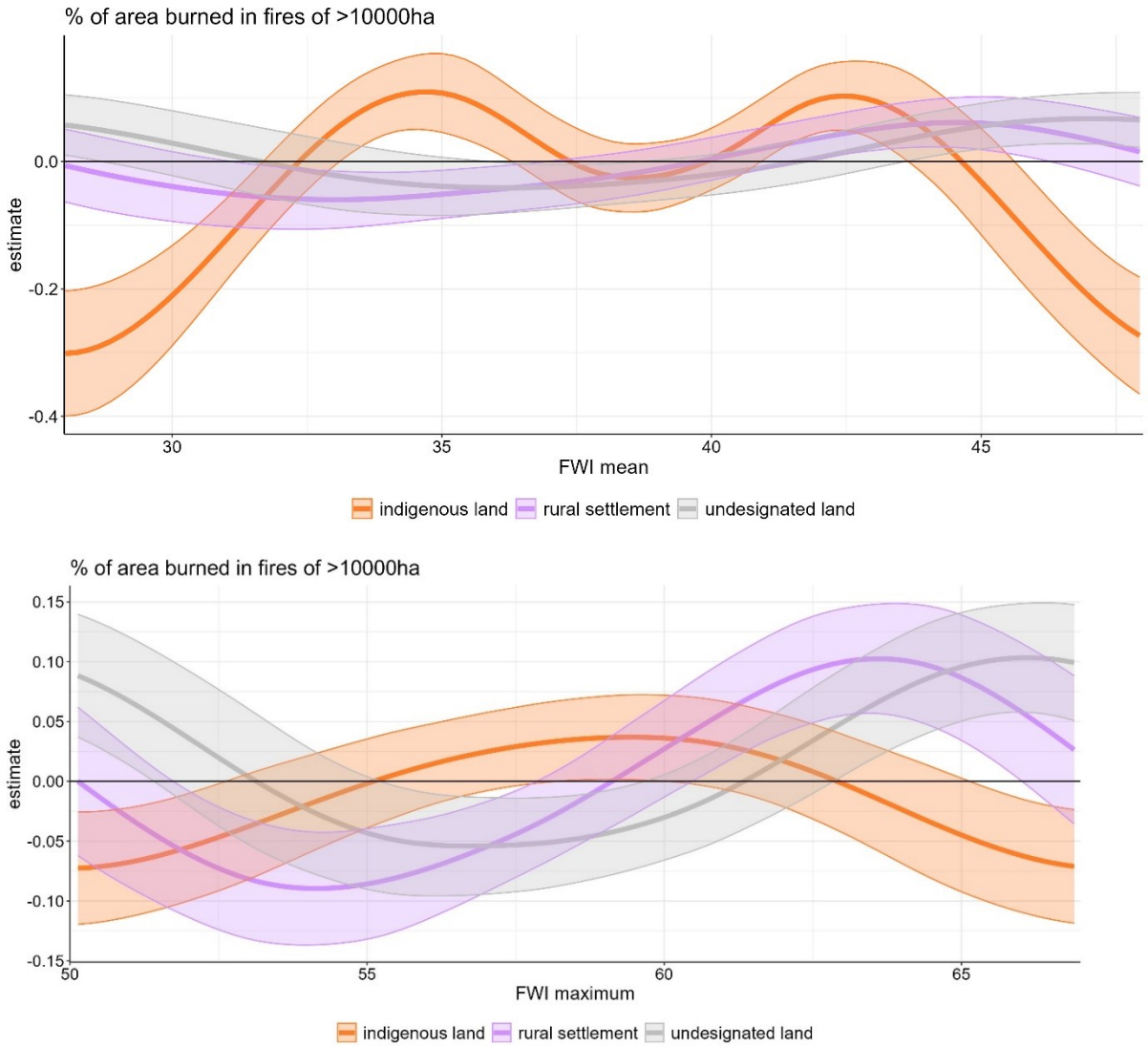


Figure S16. Smooth estimate of the effect of FWI mean and maximum FWI on the percentage of total area burned in fires of more than 1000 hectares

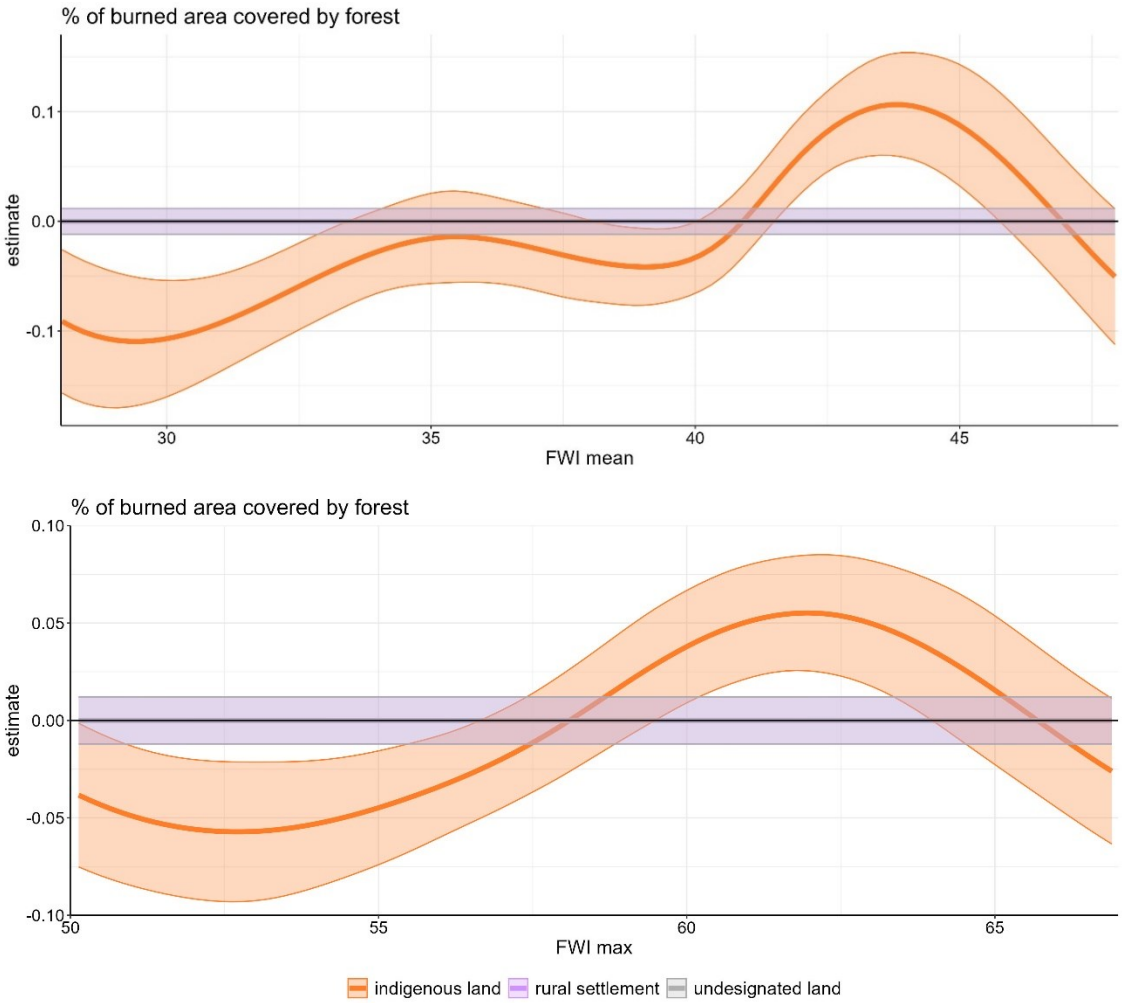


Figure S17. Smooth estimate of the effect of FWI mean and maximum FWI on the percentage of burned areas covered by forest.

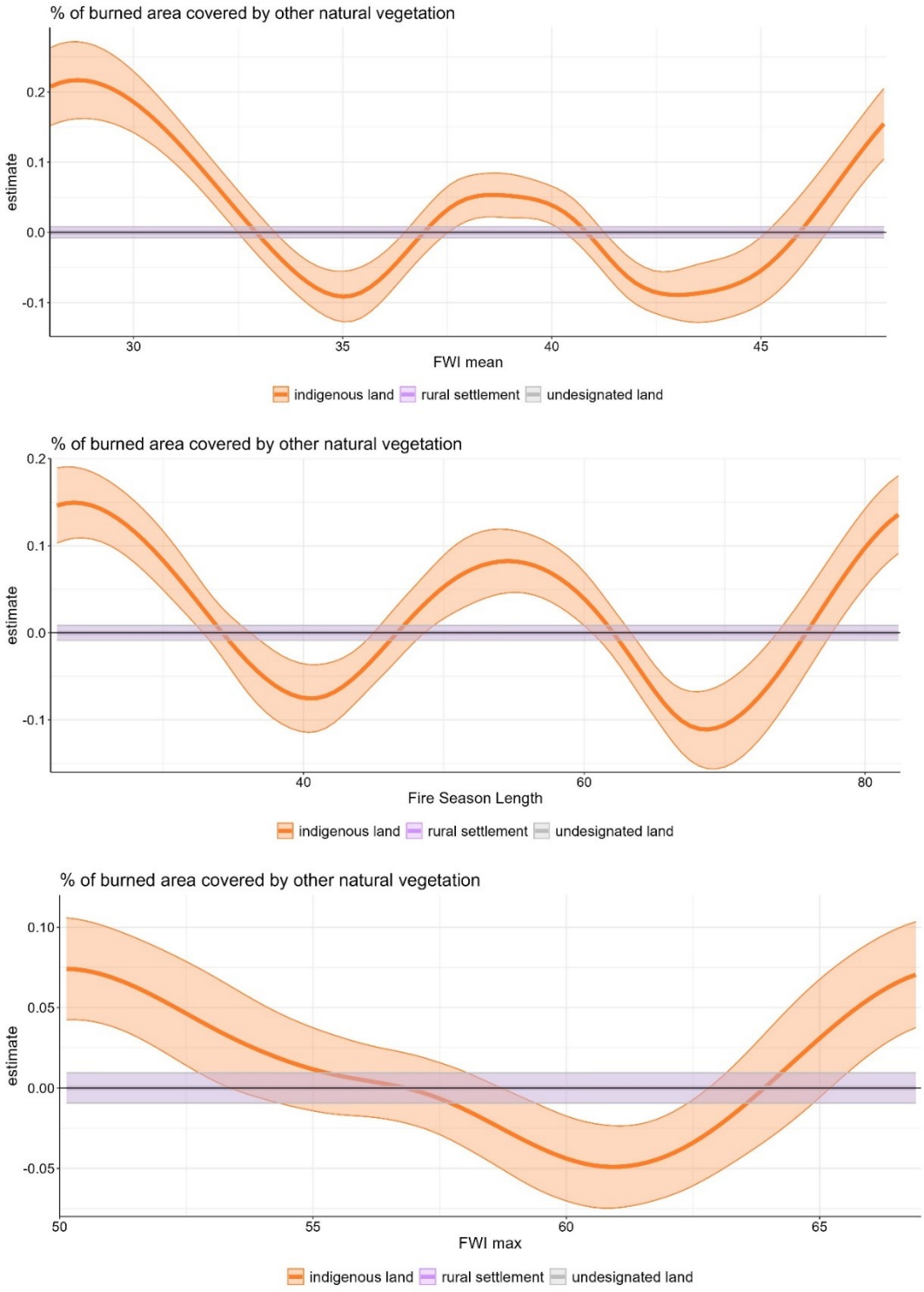


Figure S18. Smooth estimate of the effect of FWI mean and maximum FWI on the percentage of burned areas covered by other natural vegetation.

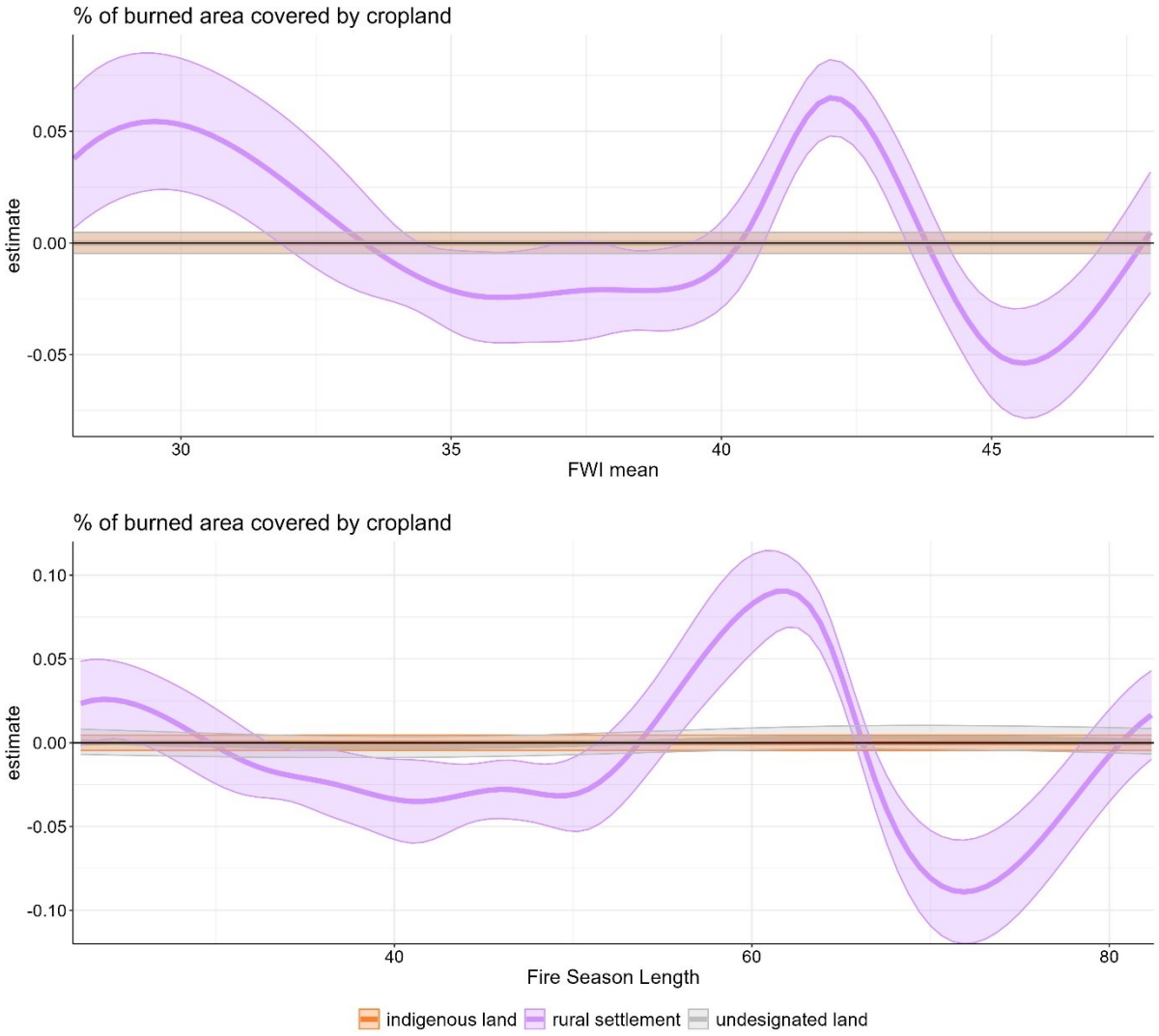
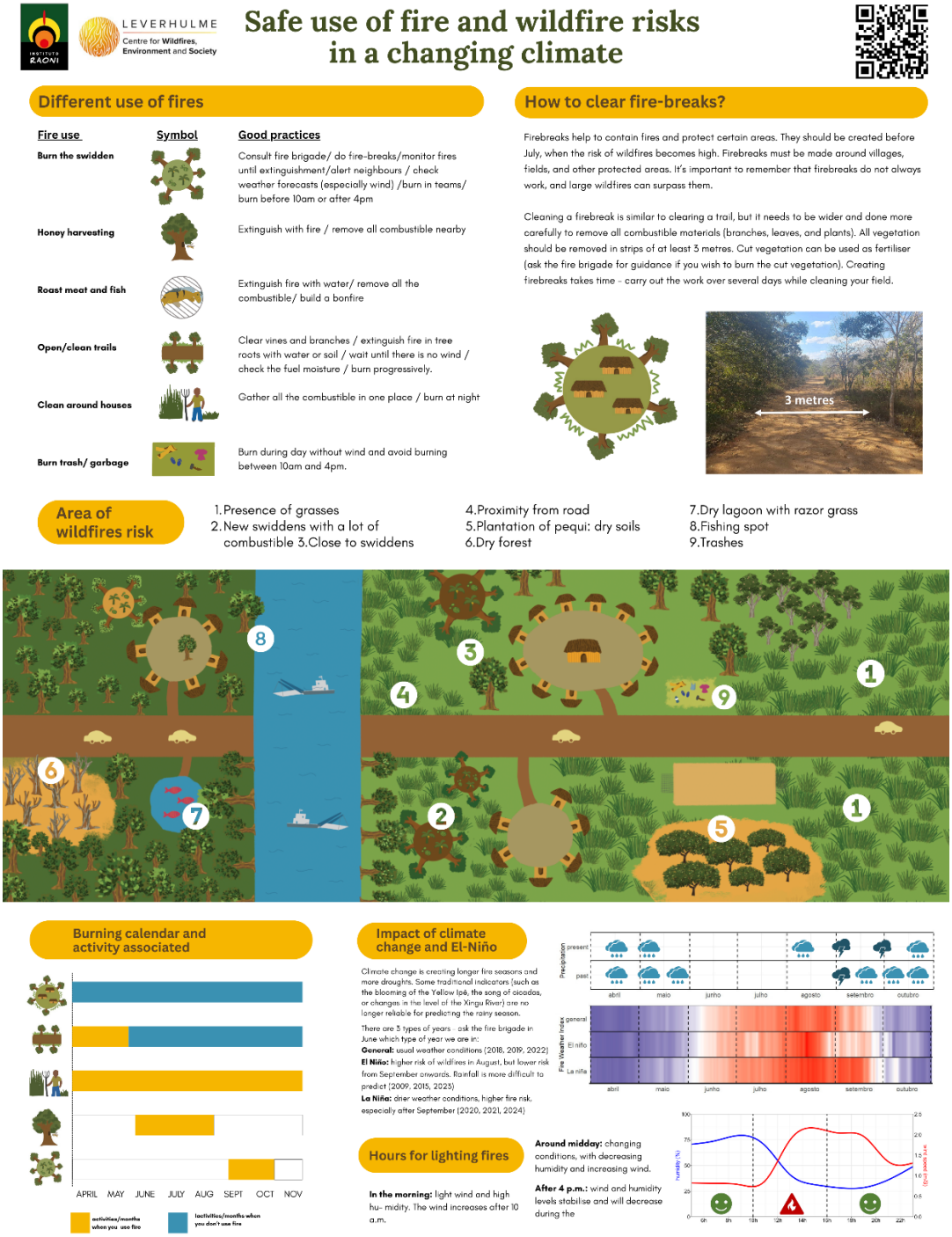


Figure S19. Smooth estimate of the effect of FWI mean and maximum FWI on the percentage of burned areas covered by cropland.

Appendices from Chapter 4

Appendix 4.1. Poster design for the presentation of the results of the study and dissemination by the local fire brigades.



Appendix 4.2. Fire reports with early results from the research project

Riscos de incêndios florestais em Capoto/Jarina e arredores- região de Piaraçu

segunda versão do relatório preliminar sobre riscos de incêndios florestais em Capoto/Jarina e arredores (2024)

Autores: Michel Valette, Roiti Metuktire, Morena Mills

Projeto: Uso e o manejo do fogo dentro e no entorno de terras indígenas: um estudo no Norte de Mato Grosso, Brasil

Este relatório tem como objetivo fornecer informações úteis sobre o risco de incêndios florestais e suas mudanças ao longo dos últimos 30 anos. Esta segunda versão do relatório sugere estratégias que poderiam ser incluídas, mas que devem ser adaptadas e aprimoradas por meio de discussões com todos os brigadistas indígenas. Por exemplo, poderíamos considerar algumas modificações: :

- Inclusão de fatores de risco de incêndios florestais (topografia do local, presença de florestas sazonalmente inundadas)
- Aprimoramento da aparência dos mapas e figuras para facilitar a compreensão (por exemplo: mudança na cor usada),
- Alteração da área mapeada: concentrando-se em determinadas áreas com alto risco de incêndios florestais ou outras terras indígenas gerenciadas pelo Instituto Raoni.

focamos este relatório na área ao redor de Piaraçu e Pakaya. Existe uma versão mais antiga do relatório que focava nos riscos de incêndio em toda a área do Capoto/Jarina. Podemos determinar um conjunto de locais críticos para a criação de relatórios de risco de incêndio florestal com base nessas áreas. Um resumo das áreas identificadas como de alto risco está incluído na primeira página do relatório.

Em cada seção do relatório, explicamos as possíveis limitações. Os dados usados podem ter problemas e sua interpretação pode ser complexa. Assim, é essencial considerar e discutir essas limitações antes de usar os dados para orientar as atividades de campo.

Sumário

Parte 1: Riscos atuais de incêndios florestais 4

Detecção de fogo por satélites 4

Fragmentação florestal 6

Índice Meteorológico de Incêndio 7

Mudança diária na umidade e no vento 9

Parte 2: Mudança nos riscos de incêndios florestais 10

Índice Meteorológico de Incêndio 10

Cobertura da terra ao redor da Terra Indígena Capoto/Jarina 11

Degradação da vegetação 12

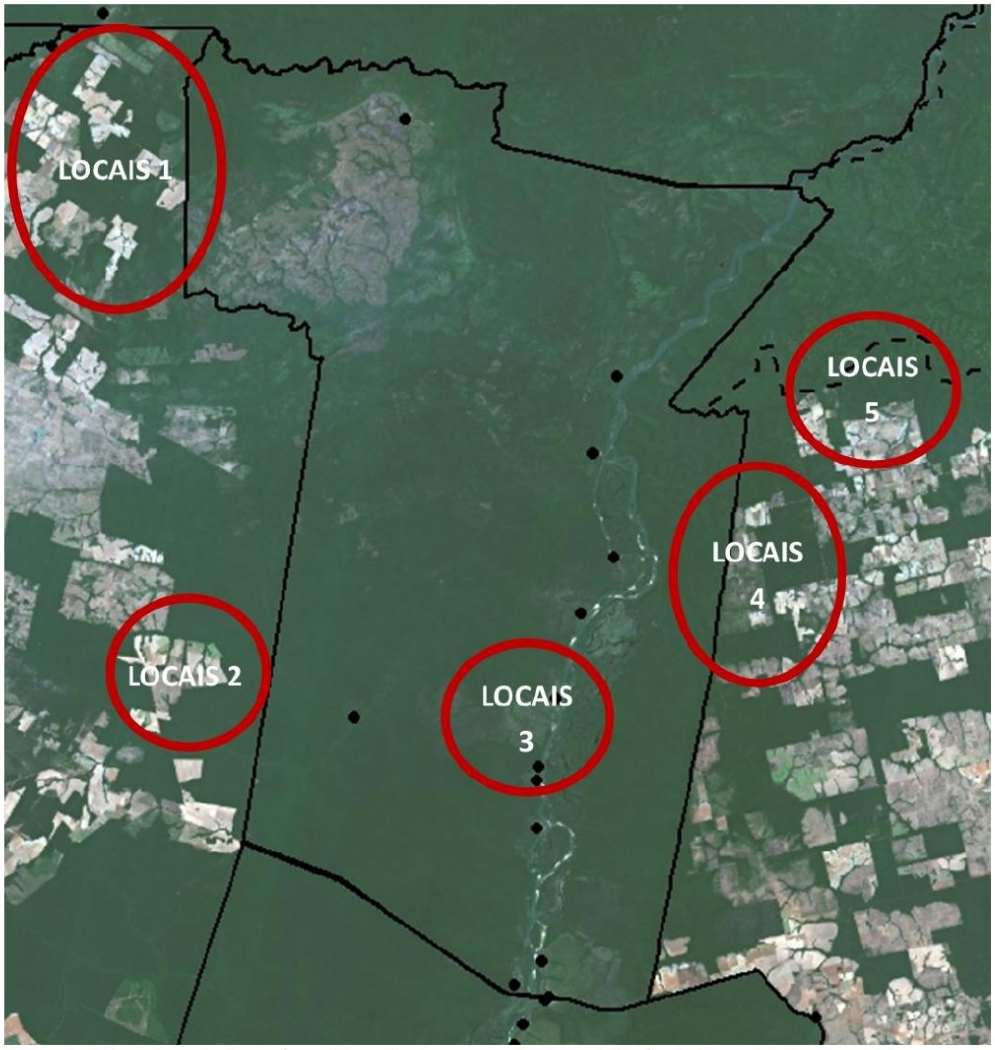


Figura 1. Mapa das principais áreas identificadas com alto risco de incêndio florestal. Pontos pretos representam as comunidades indigenas, dados da FUNAI.

Locais 1: expansões de monocultura e pastagens / fragmentação florestal / estradas / fogos passados

Locais 2: expansões de monocultura e pastagens / fragmentação florestal / estradas / fogos intensos passados

Locais 3: fragmentação flotestal / fogos passados

Locais 4: expansões de pastagens / fragmentação florestal / estradas / alguns fogos passados

Locais 5: expansões de monocultura e pastagens / fragmentação florestal / estradas / fogos intensos passados

Parte 1: Riscos atuais de incêndios florestais

Esta parte do relatório apresenta informações sobre os riscos atuais de incêndios florestais. O objetivo é mostrar as áreas e as épocas que podem ser afetadas por diferentes riscos de incêndios florestais, de modo a apoiar a ação da brigada indígena.

Detecção de fogo por satélites

Utilizando o Mapbiomas Fogo, mapeamos quantas vezes áreas ao redor de Piaraçu e Pakaya foram queimadas desde 2015 (Figura 2). Isso mostrou as áreas que queimaram com mais frequência, resultando potencialmente em maiores danos à vegetação devido a incêndios sucessivos, mudanças na composição da vegetação e áreas com maior probabilidade de queimar novamente.

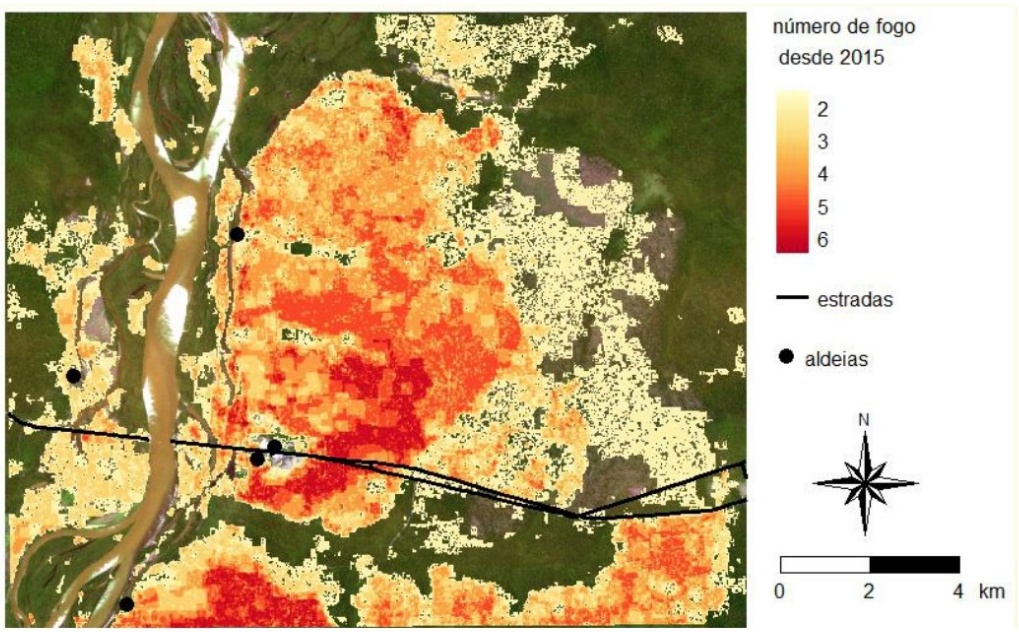


Figura 2. Mapa das áreas queimadas detectadas pelo Mapbiomas fogo entre 2015 e 2022, ao redor de Piaraçu.

As áreas que queimaram com mais frequência (pelo menos 4 vezes desde 2015) foram:

- Áreas invadidas por capim (braquiária) no nordeste, leste e sul de Piaraçu e Pakaya, com muitas áreas que queimaram de cinco a seis vezes em diferentes anos desde 2015
- Áreas 2 quilômetros ao sul de Pakaya, na margem sul do rio
- Áreas próximas a margem do rio Xingu, a oeste de Piaraçu
- Áreas 4 quilômetros a noroeste de Kretire

A vegetação até 4 quilômetros ao leste e 8 quilômetros ao norte de Piaraçu queimou com bastante frequência (até 3 vezes desde 2015). O mesmo aconteceu com a vegetação a sudeste do rio, perto de Pakaya, e com a área da margem oeste do rio Xingu, perto da aldeia Jatobá e ao sul da estrada. Algumas áreas com mais de 5 quilômetros ao norte e leste de Piaraçu queimaram periodicamente desde 2015 (até 2 vezes).

As áreas 1 km ao norte e nordeste de Piaraçu queimaram com menos frequência do que o entorno. A floresta ribeirinha ao sul de Pakaya não parece ter sido afetada por fogo e poderá constituir uma barreira aos incêndios vindos do lado sul.

Utilizando um produto de satélite diferente, analisamos o número de fogo ativos detectados e a sua intensidade máxima, indicando a quantidade de combustível consumido durante o fogo/incêndios florestais, os danos à vegetação e potencialmente a dificuldade de combatê-los e os danos resultantes à vegetação (Figura 3).

A frequência e as áreas queimadas são semelhantes aos mapas anteriores. Menos incêndios foram detectados nas imediações de Piaraçu e Pakaya. Isto pode ser devido a incêndios que queimaram rapidamente (ou porque foram controlados, ou porque houve uma resposta rápida para combater os incêndios), o que impediu a sua detecção. Este produto de satélite detectou fogo frequentes nas áreas leste e norte de Piaraçu e Kretire.

Os fogos de maior intensidade foram detectados 2 a 4 km a nordeste de Piaraçu. As áreas próximas a Kretire também parecem ter queimado com alta intensidade em comparação com outros locais. Houve também alguns incêndios de intensidade muito elevada detectados 9 quilômetros a leste de Kretire, e alguns incêndios intensos perto de Jatobá.

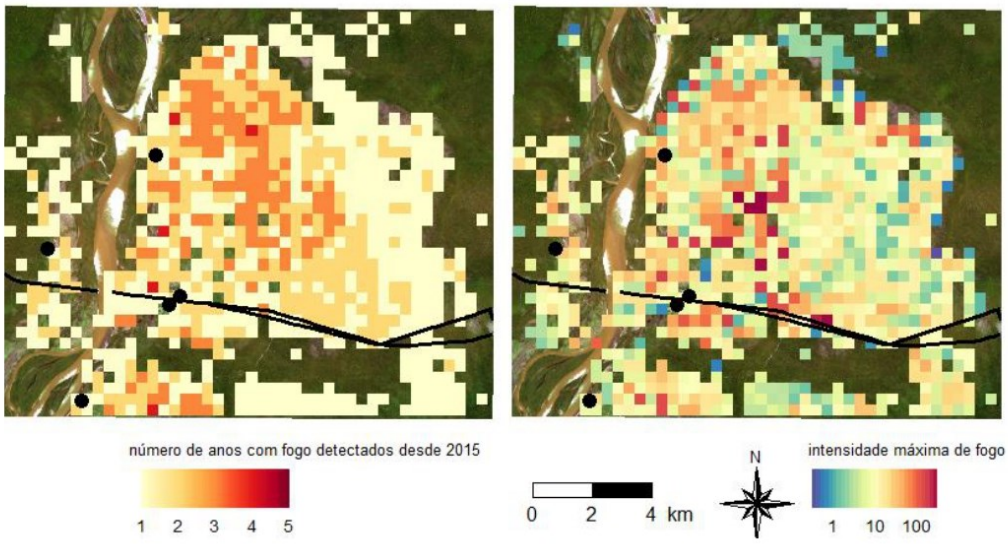


Figura 3. Número de anos em que os fogos foram detectados (esquerda) e a intensidade máxima dos fogos (direita) entre 2015 e 2023, ao redor de Piaraçu.

A percentagem de áreas queimada na paisagem varia muito de um ano para o outro (figura 4). A maior quantidade de terras queimadas ocorreu em 2020 e 2022, quando quase metade da paisagem foi queimada. Os dois anos com menores áreas queimadas foram 2016 e 2021.

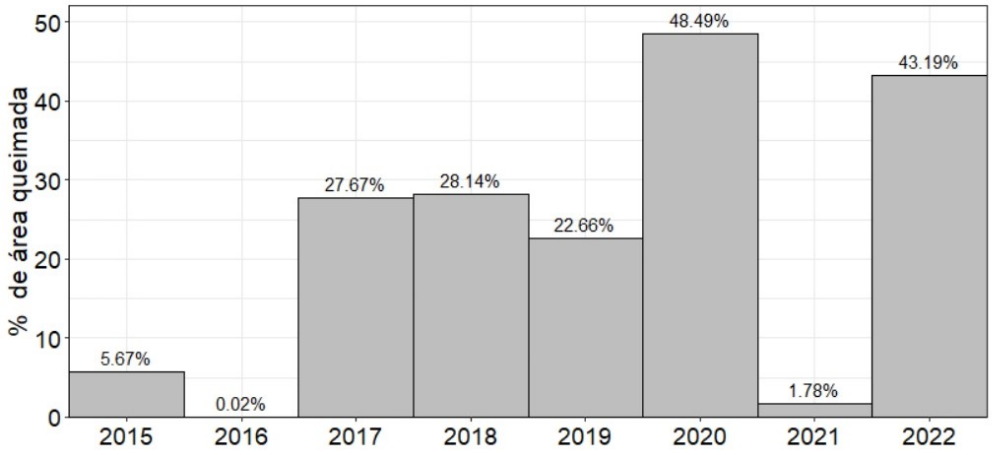


Figura 4. Proporção de terra representada na figura 2 que foi queimada todos os anos entre 2015 e 2022.

Limitações: A detecção de focos de fogo por satélite não é precisa: é possível detectá-las apenas por 2 horas a cada dia e somente quando não há cobertura de nuvens. Durante dias nublados, indivíduos podem optar por iniciar incêndios para evitar detecção, para fins como a grilagem de terras.

A detecção de áreas queimadas pode ser limitada devido ao rápido crescimento de certa vegetação, especialmente ervas.

Os focos de fogo também precisam atingir um certo tamanho para serem detectados, e focos de fogo florestais de baixa intensidade podem passar despercebidos. Estamos trabalhando no desenvolvimento de um método mais preciso para mapear incidência de fogo em toda a região.

Fragmentação florestal

Usamos uma medida que analisa grandes áreas de floresta, as quais podem ser mais resistentes a incêndios devido ao microclima local (Figura 5).

As florestas mais fragmentadas ficam próximas a margem do rio Xingu. Contudo, o impacto da fragmentação florestal nos riscos de incêndio poderia ser reduzido pela umidade gerada pelo rio.

As florestas até 4 quilômetros ao norte e leste de Kretire estão fragmentadas, assim como as florestas a 2 quilômetros ao norte e leste de Piaraçu, embora em um menor grau de fragmentação. As florestas no entorno imediato de Pakaya e Piaraçu também estão fragmentadas. Há também extensa fragmentação da floresta localizada na parte sudoeste da paisagem, a cerca de 8 quilômetros de Pakaya.

Limitações: O mapa de fragmentação florestal é baseado na diferença entre área classificada como floresta e não floresta pelo Mapbiomas. A cobertura do solo próxima à borda da floresta também é importante: florestas próximas a pastagens, roças em regeneração, cerrado ou rios enfrentam diferentes riscos de incêndio.

O processo de fragmentação florestal é complexo e vários aspectos poderiam ser considerados, como o comprimento das bordas, o tamanho dos fragmentos florestais e sua continuidade. A interpretação da fragmentação florestal deve ser baseada na experiência de brigadistas e usuários do fogo.

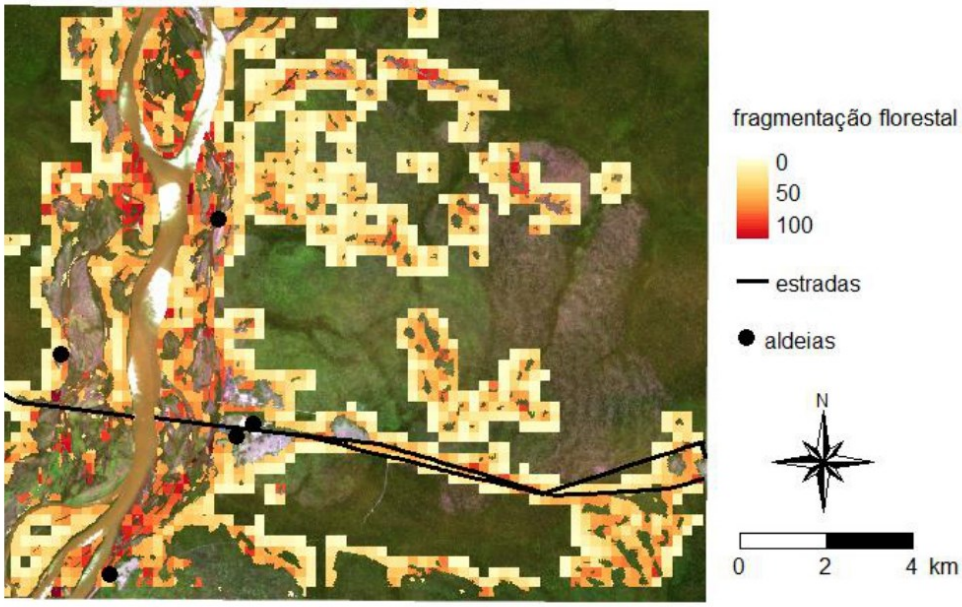


Figura 5. Mapa da fragmentação florestal ao redor de Piaraçu em 2022

Índice Meteorológico de Incêndio

Para entender o impacto das condições meteorológicas sobre os riscos de incêndios florestais, utilizamos o Índice Meteorológico de Incêndio (Figura 6). Esse índice utiliza temperatura, precipitação, umidade e vento para estimar a umidade de diferentes combustíveis e a probabilidade de propagação do fogo. O Índice Meteorológico de Incêndio é atualmente utilizado no Pantanal para avaliar os riscos de incêndios florestais nas terras indígenas.

Usamos dados de 2000 a 2022 para estimar o Índice Meteorológico de Incêndio durante a temporada de fogo, a fim de entender quando há maior probabilidade de queima intensa. Assim, é provável que estas se transformem em incêndios florestais, mas também produzam 'queimadas limpas', onde a maior parte do combustível será queimado. Separamos as condições climáticas durante os eventos El-Niñ (2002, 2004, 2009 e 2015), eventos La Ninã (2007, 2009, 2010, 2011, 2012, 2020 e 2021) e anos normais. Analisamos dois valores:

- Índice Meteorológico de Incêndio: probabilidade de que um fogo aceso nesse dia se torne intenso, resultando em combustão completa e, potencialmente, se espalhe para fora da área inicialmente atendida e se transforme em um incêndio florestal.

- Índice de Combustível Disponível: probabilidade de combustão de combustíveis grandes (galhos grandes, troncos...) e solo. Pode ser especialmente importante para queimadas realizadas em áreas sazonalmente inundadas com camadas profundas de materiais orgânicos ou áreas com muita vegetação morta.

Em um ano normal, temos valores máximos do Índice Meteorológico de Incêndio entre 1º e 20 de agosto, antes de diminuir. O Índice de Combustível Disponível atinge seu valor máximo no início de setembro, antes de diminuir.

Durante um ano de La Niña, temos valores máximos do Índice Meteorológico de Incêndio entre 1º de agosto e 15 de setembro, antes de um rápido declínio. O Índice de Combustível Disponível atinge seu valor máximo por volta de 15 de setembro e é maior do que nos anos normais e de El Niño. O Índice de Combustível Disponível é maior do que nos anos normais e de El Niño.

Durante o ano de El Niño, temos valores máximos do Índice Meteorológico de Incêndio de 1º a 15 de agosto, antes de uma rápida diminuição e um segundo aumento por volta de 15 de setembro. Isso é diferente de outros anos, pois pode haver dois picos no Índice Meteorológico de Incêndios, e os Riscos Meteorológicos de Incêndios são menos previsíveis. O índice de acúmulo atinge seu valor máximo por volta do dia 20 de agosto, antes de diminuir. O índice máximo de acúmulo é menor do que nos anos normais e de La Niña.

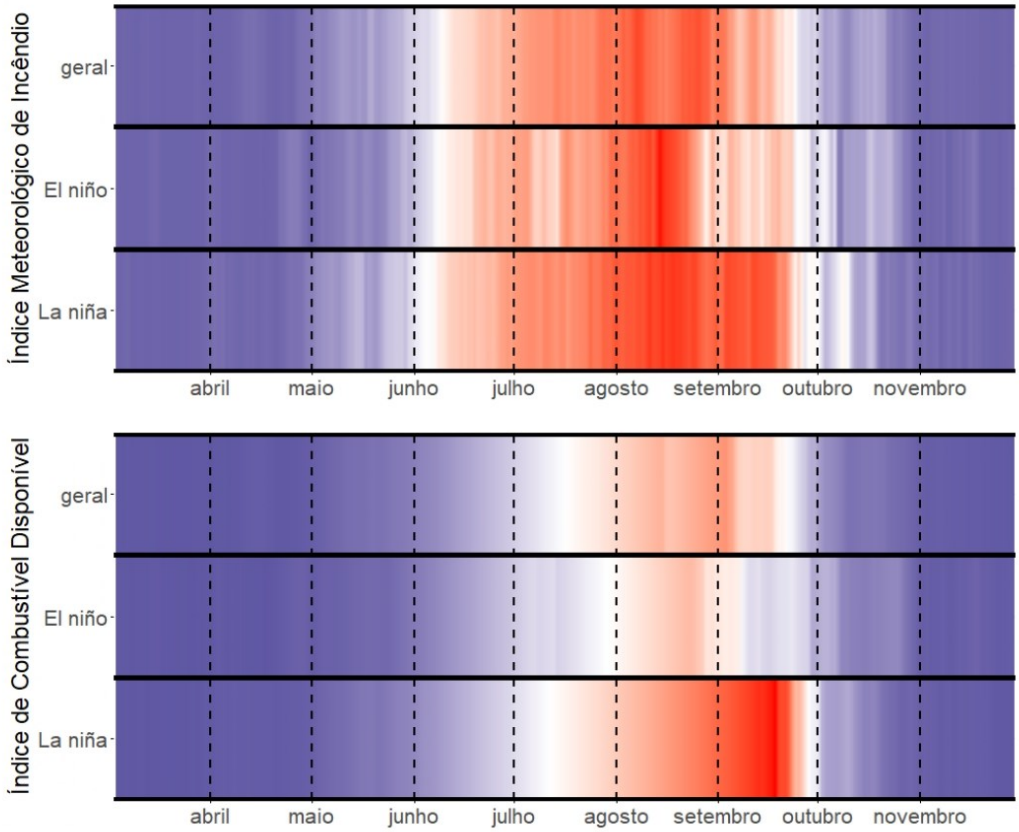


Figura 6. Médio diário de Índice Meteorológico de Incêndio (esquerda) e Índice de Combustível Disponível (direita) ao longo do ano de 2000/2022 na área de Capoto /Jarina. As linhas grossas de cores representam a média, e as áreas cinza representam a mudança de um ano para o outro.

Limitações: O Índice Meteorológico de Incêndio foi desenvolvido no Canadá e, embora funcione bem em muitas áreas do mundo, como a Amazônia e o Pantanal, pode não considerar totalmente o impacto local das condições climáticas nos riscos de incêndio. A discussão sobre a relação entre os riscos de incêndios florestais e as condições meteorológicas poderiam ajudar a compreender melhor essa questão.

Houve alguns anos marcados com eventos de El Niño, que apresentaram padrões de precipitação distintos. As previsões do Índice Meteorológico de Incêndio durante eventos do El Niño são mais incertas e devem ser consideradas com cautela. Além disso, outros fenômenos climáticos podem aumentar ou diminuir o impacto do ciclo El Niño/La Niña na região.

Um alto índice climático de incêndio permite a combustão rápida e completa de materiais combustíveis e pode facilitar o fogo controlado. Isto poderia reduzir a carga de trabalho e ajudar a implementar medidas adequadas de controle de focos de fogo.

Mudança diária na umidade e no vento

Utilizamos os dados coletados pela estação meteorológica de Guarantã do Norte para observar a mudança diária na umidade e na velocidade do vento durante o mês de setembro (Figura 7). Podemos perceber que um dia típico é dividido em 4 fases:

- Das 5h às 10h: a umidade é alta e a velocidade do vento baixa → é mais seguro queimar, mas pode ser difícil atear fogo à vegetação
- Das 10h às 16h: a umidade diminui e a velocidade do vento aumenta → os incêndios podem aumentar de intensidade neste período, mas é mais fácil atear fogo à vegetação
- Das 16h às 19h: umidade e velocidade do vento são altas e estáveis → facilidade de incendiar a vegetação e condições estáveis
- Das 19h às 24h: a umidade está aumentando e a velocidade do vento diminuindo rapidamente → mais seguro para queimar, mas mais difícil de atear fogo à vegetação + falta de visibilidade

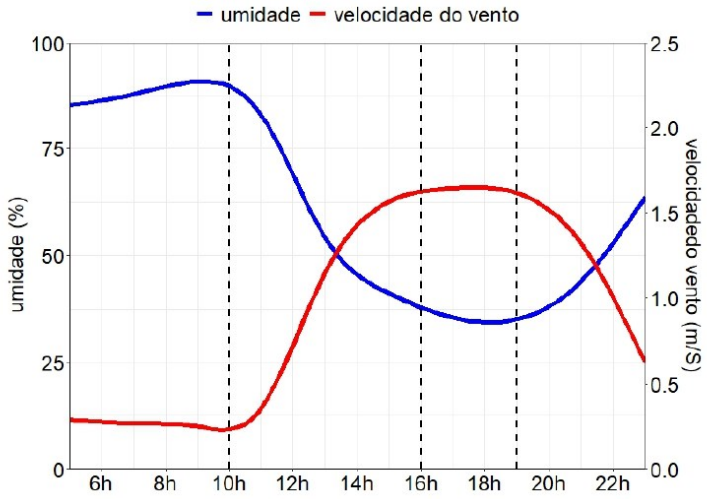


Figura 7. Evolução diária da umidade relativa e velocidade do vento registrada na estação meteorológica de Guaranta do Norte durante o mês de setembro.

Limitações: Os dados foram coletados na estação meteorológica de Guaranta do Norte, em uma paisagem agrícola. A floresta ao redor de Piaraçu pode ter impacto na velocidade do vento e na umidade relativa ao longo do dia, e os dados apresentados devem ser confirmados por observações ou medições locais.

Os dados aqui indicaram a velocidade do vento, mas não a mudança na direção do vento, o que pode ser especialmente complicado ao planejar uma queima.

Parte 2: Mudança nos riscos de incêndios florestais

Nesta parte do relatório, examinamos as mudanças nos riscos de incêndios florestais para entender quando e onde mudaram em comparação com o passado.

Índice Meteorológico de Incêndio

Analisamos as mudanças no Índice Meteorológico de Incêndio desde 2000/2022, em comparação com um período de referência de 1950 a 1999 (Figura 8). Utilizamos três métricas para resumir as mudanças na temporada de risco de incêndio:

- Número de dias com alto risco de incêndios florestais: aumentou em média 4 dias por ano em comparação com o período entre 1950 e 1999
- Duração da temporada de risco de incêndio: aumentou em média 20 dias por ano em comparação com o período de 1950 a 1999
- Índice Meteorológico de Incêndio médio: aumentou em média 5 dias em relação ao período entre 1950 e 1999, correspondendo, por exemplo, à diferença entre o Índice Meteorológico de Incêndio do início de julho a meados de julho

As aldeias ao longo do rio Xingu, entre Piaraçu e Kempo, registaram o maior aumento na duração da temporada de risco de incêndio e no número de dias com risco muito alto de incêndios florestais. Capoto teve o maior aumento nos riscos médios de incêndios florestais durante a temporada de risco de incêndio. As aldeias de Kremroro e Kempo registraram aumentos relativamente menores nos riscos de incêndios florestais.

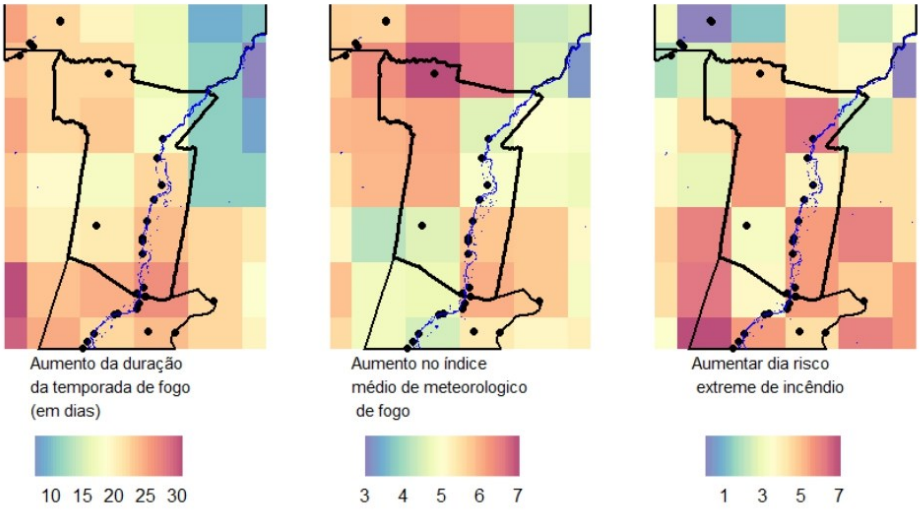


Figura 8. Alteração nas temporadas de risco de incêndio em toda a paisagem do Capoto/Jarina desde 2000, comparada ao período de 1950-1999

Também comparamos o Índice Meteorológico de Incêndio de cada dia do ano entre o período de referência (1950-1999) e os anos recentes (2000-2022), em Figura 9. O Índice Meteorológico de Incêndio aumenta um pouco nos meses de maio (especialmente no início de maio), junho e início de julho. Ele aumenta muito durante o mês de agosto e por volta do dia 15 de setembro. O Índice Meteorológico de Incêndio diminuiu um pouco na segunda metade de julho, no início de setembro e em novembro, e diminuiu entre o final de setembro e outubro.

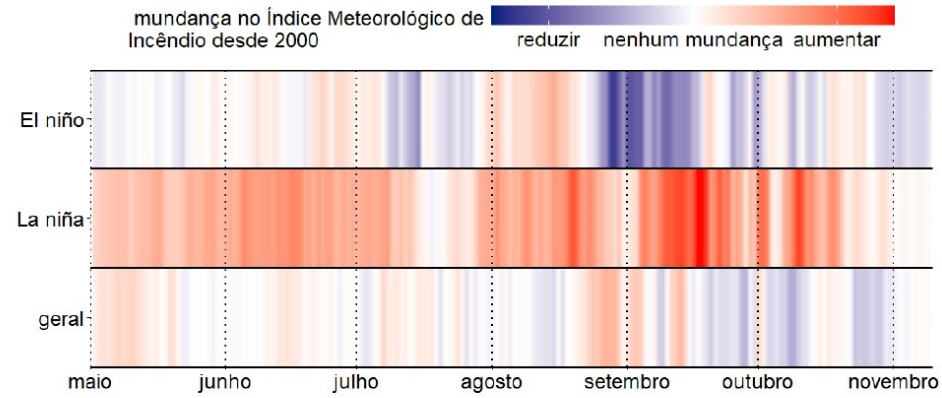


Figura 9. Mudança no Índice Meteorológico de Incêndio em todos os dias da temporada de risco de incêndio para o período de 2000 a 2022, quando comparado com o período de referência de 1950 a 1999.

Limitações: Os dados têm baixa resolução (25 km), falhando em capturar diferenças sutis na mudança climática local de cada comunidade. A figura 8 representa a mudança média nos últimos 20 anos, mas existem diferenças significativas entre anos afetados pelo El Niño, La Niña e condições normais. Seria útil discutir as mudanças observadas pelas brigadas indígenas e relatadas pela geração mais velha.

Cobertura da terra ao redor da Terra Indígena Capoto/Jarina

Utilizamos o Mapbiomas para observar a mudança na cobertura da terra no entorno da aldeia Piaraçu entre 1985 e 2022 (Figura 10).

As áreas ocupadas pelo rio Xingu diminuíram. O campo algado também diminuiu e foi substituído em alguns locais por savanas ou campestres. O Cerrado se expandiu, especialmente na margem oeste do Xingu. Grandes áreas de terra ao sul do rio perto de Pakaya e na parte sudeste da paisagem são hoje ocupadas por capim. Manchas menores de capim se espalharam nas imediações de Kapaia e Piaraçu, em uma mancha localizada próxima ao cruzamento das duas estradas (2 quilômetros a leste de Piaraçu), bem como próximo à floresta mais degradada, 4 a 8 km na leste e norte de Piaraçu.

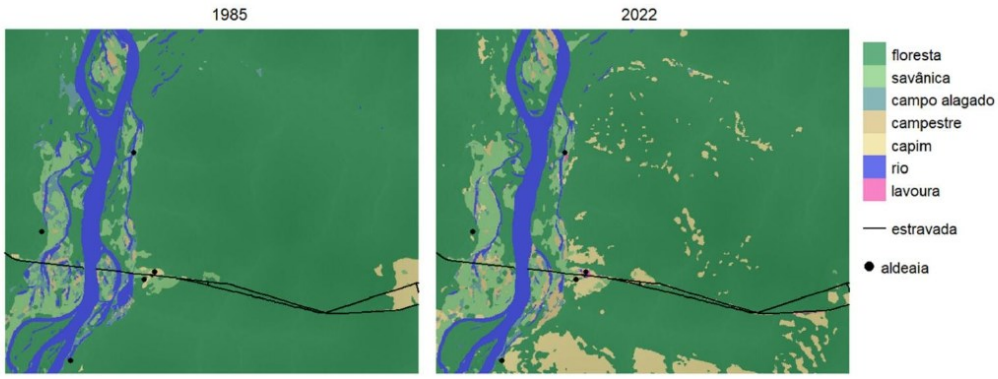


Figura 10. Mapa da mudança na cobertura do solo ao redor do Piaraçu entre 1985 e 2022.

Limitações: Parte da floresta pode ter experimentado degradação extensiva entre os dois períodos e ainda ser classificada como floresta, apesar de ser mais vulnerável a incêndios. Outras limitações são semelhantes à primeira parte do relatório.

Degradação da vegetação

Utilizamos imagens de satélite para identificar as áreas florestais onde a vegetação vem perdendo folhas desde 2017, o que poderia resultar em maior vulnerabilidade a incêndios florestais e maior exposição dos solos e da serapilheira à luz solar (Figura 11).

A maioria das áreas com floresta degradada está localizada na margem leste do rio Xingu. A maior degradação está situada em um pedaço de floresta localizado a cerca de 10 quilômetros a nordeste da vila de Piaraçu. Toda a floresta localizada entre 4 e 8 quilômetros a leste e nordeste de Piaraçu sofreu degradação significativa e, em menor grau, a floresta localizada entre 4 e 8 quilômetros ao norte de Piaraçu. Existem também pequenas manchas de floresta degradadas entre 1 e 4 quilômetros a leste de Pakaya, ao longo da estrada e principalmente a norte do rio. Há uma pequena mancha de floresta degradada a cerca de 6 quilômetros ao norte de Jatobá.

Limitações: Em alguns locais, a sazonalidade da floresta pode ter mudado e a degradação florestal pode ser menos extensa do que a estimada.

A diminuição da cobertura foliar também pode originar uma menor acumulação de serapilheira e, portanto, uma menor intensidade de incêndios florestais.

Parte da vegetação pode ter se adaptado e, embora estejam presentes algumas espécies de árvores e palmeiras com menos folhas, estas poderiam estar mais adaptadas às novas condições climáticas e às perturbações do fogo.

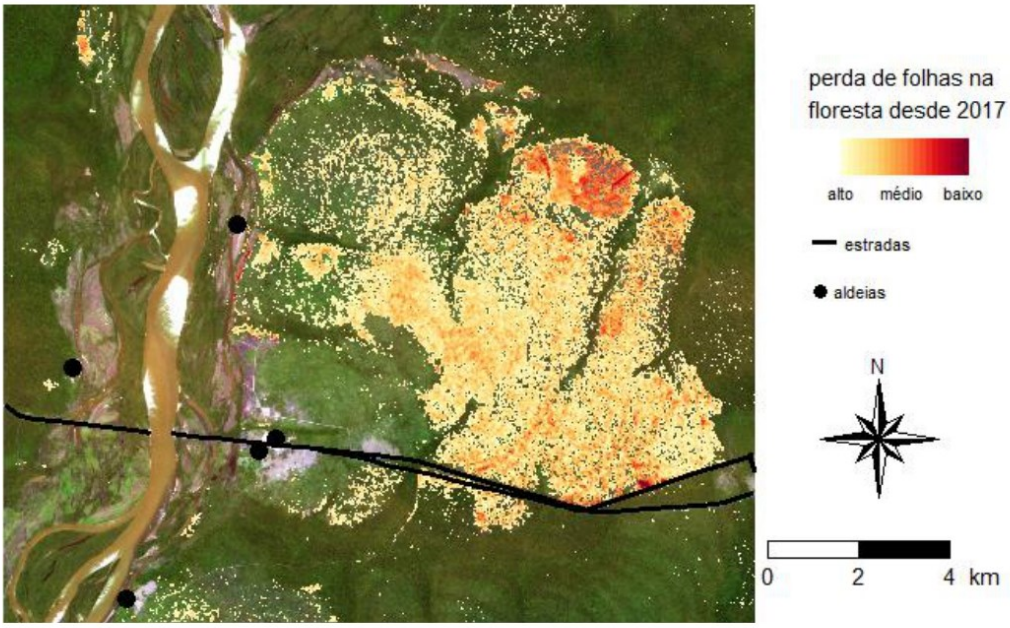


Figura 11. Mapa da degradação da vegetação ao redor do Piaraçu entre 2017 e 2023.

Appendix 4.3. Summary of research report for the Territorial and Environmental Management Plans for Capoto/Jarina

Climate change and wildfire risks

Participants in the research activities described changes in the local climate that affect the use of fires and increase the risks of wildfires: the start of the rain season is harder to predict, and have less rain, there are more frequent droughts and an overall rise in temperature. Analysis of the Fire Weather Index matches these observations: since 2000, the fire season has increased by about 22 days, and there have been 5 more days with very high wildfire risks. The Fire Weather Index changes mostly during September, when inhabitants usually set fires for their swiddens, the most intense use of fires in Capoto/Jarina.

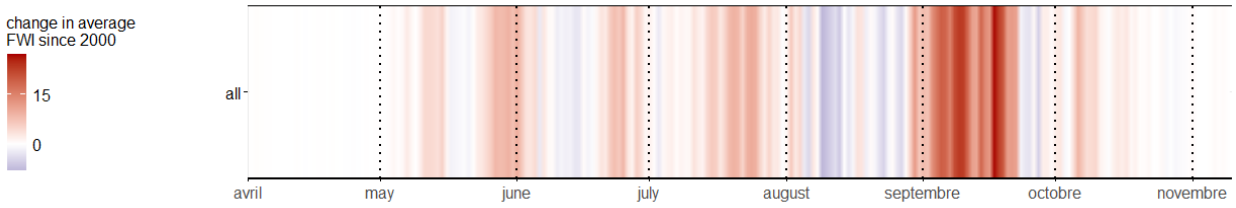


Figure 1. Figure showing the average change in Fire Weather Index since 2001

Use of fires supported by extensive Traditional Ecological Knowledge

A wide range of fires is commonly used by inhabitants of the Capoto/Jarina to support different livelihoods and cultural needs, such as preparing swiddens, collecting honey, or burning trash. These uses of fire are supported by detailed knowledge about when, where, and how to light these fires and reduce wildfire risks.

Unlike other villages, people in Capoto also use fires in large patches of grassland and savanna to meet multiple needs, including collecting resources, reducing the chance of encountering venomous animals, and improving access to different areas of the land. These fires also serve to control wildfire risks by reducing fuel load and preventing fires from spreading in nearby forests. Cessation of burning in the Capoto plateau could have unexpected effects on fire regimes due to the flammability of the landscape and risks of high-intensity wildfires in the absence of cultural burning.

Detailed knowledges of the local wildfire risks and location of resources to protect

Inhabitants have a good understanding of the local wildfire risks, based on frequent observation of the land and many years of experience. Participatory maps helped show the importance of wildfire risks in areas invaded by *Brachiaria spp.* in Piaraçu, and the ongoing problems caused by frequent fires in grassland patches along the Xingu River. This helps identify different areas at risk of wildfires, for example, because of changes in vegetation or the frequent use of small traditional fires, such as those used to grill fish. They also know about landscape features that could help fire brigades, such as small streams and riverine

forests that can help stop fires, and they know where precious, fire-vulnerable resources are located and need protection. This local knowledge can be useful to support the action of the fire brigade and ensure that they can fight wildfires efficiently while protecting important resources for the local communities.

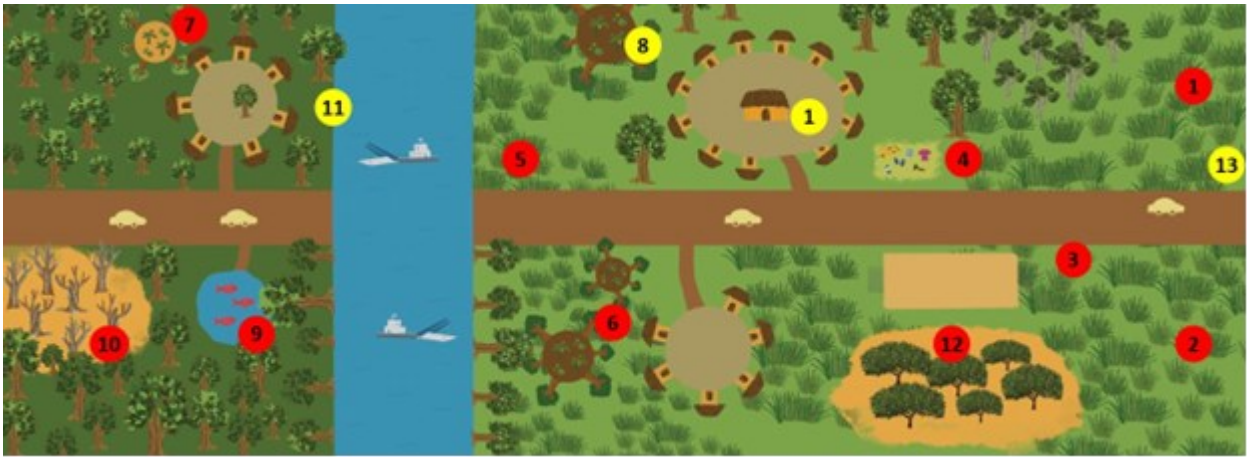


Figure 2. Illustration derived from participatory mapping exercise in the village of Piaraçu including all the sites discussed in the results. This illustration groups the insight from the participatory map of both women and men.

Recent changes in wildfires risks

Changes in the local climate have many effects on wildfire risks: fires have become more intense and harder to control. It has also affected traditional signs of fire seasons, like the flowering of yellow ipé. The soils along the Xingu bank are now drier and can catch fire, allowing fires to travel over long distances.

The spread of invasive *Brachiaria spp*. has greatly increased wildfire risks in the Piaraçu region. Because of their height and high flammability, they can easily catch embers from other fires and start intense wildfires. This has made some fires that were traditionally considered safe, like those for roasting fish, more dangerous. Another risk is areas of forest that have already burned, as they are more likely to catch fire again due to dead trees. These areas are especially common in the region of Piaraçu, but fires in recent years have affected large tracts of forest along the Xingu bank.

While all villages are affected by similar climate changes, the changes in vegetation differ in each village and could be captured through participatory mapping exercises.

Adaptation to wildfires

People in Capoto/Jarina use different strategies to adapt to the changing wildfire risks. The use of fire breaks for clearing swiddens has been widely adopted, and their width is usually adjusted depending on the wildfire risk. The timing of fire use has also changed, with more burning happening after one or more rainfall events, and often carried out in teams. Some communities have also started to place signs near fishing spots to remind people to properly

extinguish fires. The fire brigade has also played an important role in controlling wildfire risks by raising awareness, offering support for difficult burns, and, in the case of Piaraçu, replacing the traditional system for burning swiddens.

Appendix 4.4. Semi structured interview guide

Guide for Semi-structured interviews

Introduction:

Thank you for agreeing to meet with us today. We are xxx and xxx, working on a research project about the use of fires for landscape management in the North of Mato Grosso. We are interviewing landholders in and around the Capoto/Jarina indigenous lands to understand how fires are being used and how this use has changed over time.

You will find more information on the research project and your right as a participant in this study in the Participation Information sheet. If you agree to participate, you will need to sign a consent form. The research activity is expected to last for about one hour. You can withdraw from the research activity at any moment, without providing any reasons. If you decide to do so, we will delete the data that we collected through interviews. If you participate in the project, we will anonymize the data so your name will not appear in the final output of this research project. Before we start, I wanted to ask you if you are fine with me recording our discussion for further analysis.

Theme 1: Different use of fires and landscape management objectives
What do you use fire for? *Create a typology of different types of fires*

Do you use it for preparing agricultural fields? How do you use it then?

Do you use it to collect certain products in the forest or cerrado?

Do you use it to keep the surroundings of the house clean?

Are there other reasons for using fires?

(For each type of fire), where do you use these fires?

Which types of vegetation are burned?

Theme 2: Seasonality of fire use
When do you use each type of fire? *based on typology from theme 1*

Do you adapt the time of fire use to the meteorological conditions? If so, what are the most important meteorological factors (wind, temperature, precipitation…)?

Are there other factors influencing when you decide to light the fires (labour availability, regulation …)?

Did you change the way you decide when to burn over the last two decades?

When you are burning, are there important resources which are vulnerable to escape fires near burning sites?

Version 1.3
22/08/2023

Theme 3: Fires use and control measures

When and how do you prepare the burning site (fire breaks clearing, tree fell toward the centre of the plots, fuel curing, gathering of fuel in specific locations …)?

How do you light the fire (strip or ignitions pattern, tools used)?

Which fire control measures do you deploy?

- Do you use backfire or flank fire after initial ignitions?
- Do you carry fire control equipment to the burning sites?
- Do you monitor the fire until extinguishment?

How many people are usually present when you conduct the burning?

Are there practices at the community level to control fires?

- Do you warn your neighbours before burning?
- Are there compensation mechanisms in case a fire escapes from someone's plot?

Theme 4: Wildfires risk and fires regimes change

Which part of the landscape are particularly vulnerable to escape fires?

- Do areas which already burned became more vulnerable to wildfires? Why?
- Do other factors, such as logging or tree mortality, increase the fire danger in the landscape?

Do you think the landscape has become more vulnerable to wildfires over the last two decades?

- Does the climate become more conducive to fires? How?
- Is there other factors driving change in wildfires risks?
- Is the change in flammability widespread or concentrated in certain areas (for example specific vegetation types)?

Do the ways you used fires changed over the last two decades? How?

- What do you think has driven these changes?
- Does the change in climate conditions played a role in these changes? Environmental policies? Demand for agricultural products?

Appendix 4.5. Focus Group Discussion and field walks guides

Guide for Focus Group Discussions

The main idea behind the group's discussions is to discuss the use of fire, wildfire risks and how it is related to the territory around the villages. These discussions will serve to create a participatory map and seasonal burning calendar, which will allow us to record place-based fire knowledge yield through participatory techniques in a spatially and temporally explicit manner and extend their analysis. The Focus Group Discussion has been designed in a way to produce the following research outputs:

- **Fire typology:** resuming the main use of fires in the villages and information on where, how and when they are burned, fire control measures deployed, and other aspects deemed important by the researchers.
- **Seasonal burning calendar:** showing the seasonality of different fire uses and associated activities such as preparation of the burning sites.
- **Participatory maps:** showing the areas around a village that are managed by different fires or have a high wildfire risk, and complementary information noted during FGD.

Setting

Each of the 3 sessions should take place over a different day, leaving time for participants to think about each discussion and do complementary field walks during the rest of the day to triangulate information collected during the Focus Group Discussion.

Each Focus Group Discussion should last between 1 and 2 hours, including the time to obtain consent at the beginning of the Focus Group and check that participants are ok to share information collected at the end.

The Focus Group Discussion should happen in a sheltered place, where at least 15 people can meet comfortably. There should be a wall or board to display the participatory map and complementary paper canvas, and ideally a table for taking notes and putting the voice recorder.

The research team should account at least 3 researchers:

- **A facilitator:** introducing the research activity, guiding the groups through the different steps of each session, and animating the discussions.
- **A translator:** translating the questions and responses in a language that could be analyzed by the researchers afterwards, and noting part of the translation that could be ambiguous.
- **A note taker:** helping both with filling the participatory maps and blank canvas, and taking complementary notes about the responses of participants for restitution at the end of the FGD

Ideally, there should be between 5 and 10 people participating in the Focus Group Discussion, but the sampling needs to be discussed beforehand according to the possibility of the project and could be flexible.

Materials

For organizing these focus group discussions, we will need:

- Small printouts of zoomed-in portions of the map to help participants orient themselves and let them do drawings.
- Blank paper and craft paper that could added next to the map.
- Blue-tack, tap or pins to fix maps on the wall/blackboard.
- Markers.
- Colour pens.
- Notebook or paper for taking notes.
- Phone for pictures.
- Voice recorders.
- Camera for field walks.
- Stickers and sticky notes of various colours.

General Do's & Don'ts

Dos

- Familiarize the participants with the map by identifying landmarks and major features.
- Create a detailed map key and take good notes about the process, the symbols, and the information provided.
- Encourage participants to be respectful of everybody's thoughts as every participant has different knowledge and opinions of the area being mapped.
- Carefully question and make notes of information that seems uncertain or contentious.

Don'ts

- Do not let some members of the group move on mapping/describing the next type of fire before finishing the proceeding ones.
- Don't let a few members of the group dominate the mapping exercise.
- Insert your biases or "correct" participants with your knowledge of the area.

Session 1

Objectives of the session:

- make sure everyone can use the map and recognize important landmarks.
- first version of the fire typology
- map of areas managed by different types of fires

Step 1: Introduction (5 minutes)

1. Greet participants, thank them for their attendance at the discussion

2. Remind participants about the fact that:

- This discussion is part of a research project with Imperial College in London
- There is a document in Portuguese explaining in more detail how data could be used
- Imperial College could use the collected data anonymised.
- Participants are free to withdraw from the activity at any time
- Participants could ask us to keep some information private.

Ask if everyone agrees to record the session and start the recording.

3. Explain the main objectives of the day

Step 2. Enriching fires typology (30 minutes)

1. Ask which types of fires people use. For each type:

- Ask the participants for details about the objectives of the fire type
- Create a new column on a blank sheet of paper, displayed next to the map.
- Ask participants for a symbol we can use to represent this fire
- Add colour for stickers that will be used for the next steps.

2. After participants talk about all the types of fires they remember, you can ask follow-up questions about other fire uses based on results from semi-structured interviews.

Step 3: Base map creation (30 -40 minutes)

1. Ask participants to create a base map of craft paper to discuss the place where they use fire and have wildfire risk. Ask them to start drawing their villages, and then suggest drawing other places, such as trails, rivers, waterfalls, and swiddens

2. Ask each participant to locate their own house and swidden on the map, to ensure all of them are familiar with the map

Step 4: Identification of areas where fires are used (15-30 minutes)

1. For each type of fire identified in the previous step, ask participants to put stickers on the large map to represent areas managed by the corresponding fire types

2. After all groups get a chance to do it, ask if any groups want to add more stickers to the map

3. Ask if participants are happy with the map, remind them of the next session and let them know that they will have a chance to correct the map after

Session 2

Objectives of the session:

- Enrich the fire typology
- Understand the seasonality of different fire types

Step 1: Introduction (5 minutes)

1. Greet participants, thank them for their attendance at the discussion

2. Remind participants about the fact that this discussion is part of a research project with Imperial College in London.
Ask if there are any questions regarding ethics.
Ask if everyone agrees to record the session and start the recording.

3. Explain the main objectives of the day

Step 2: Reminder of the previous workshop (5 minutes)
1. Give names of important locations on the map, and go quickly through the types of fires used across different parts of the landscape.

2. Ask for corrections from the participants, either for the location of some fires or adding new types of fire

Step 3: Enriching the fire typology (40 minutes)
1. For each type of fire, ask questions about

- Who lights the fires (men, women, elders …)
- How and when are sites prepared?
- In which months are fires usually burning? Do people use other seasonal indicators to choose when to burn? (can prompt participants with seasonal indicators identified during semi-structured interviews)
- How do people decide when a good time is to light the fires (for example, wind conditions, state of the vegetation…)
- At what time of the day are the fires usually lit?

- Are there fire control measures deployed? (can prompt participants with fire control measures identified during semi-structured interviews)
- Is there a high risk of the fires escaping and which resources are likely to be damaged?
- Has the way to use the fire (location, times, technique) changed over time and why.

Step 4: Restitution (10 minutes)
1. Go through the map, resuming the fires which are burned across the fire season, starting from the beginning of the fire season. Resume changes in the way fires are done.

Session 3

Objectives of the session:
- Understand the wildfire risks in the landscape near the villages.
- Summarise the most interesting points of the previous Focus Group Discussions

Step 1: Introduction (5 minutes)

1. Greet participants, thank them for their attendance at the discussion

2. Remind participants about the fact that this discussion is part of a research project with Imperial College in London.
Ask if there are any questions regarding ethics.
Ask if everyone agrees to record the session and start the recording.

3. Explain the main objectives of the day

Step 2: Reminder of the previous workshop (5 minutes)
1. Go through a usual fire season, describing different types of fires used, the usual fire control measures deployed and changes over time

2. Ask for corrections from the participants

Step 3: Speak about past memorable fires (20 minutes)
1. Ask participants if some of them remember past wildfires

2. For each large wildfire event, ask if they remember:
- In which year it burned, and if possible, which month
- The origin of the fires (fire from a swidden, outside of the community etc…)
- What were the damages?
- If the community acted to control wildfires

Step 3: Speak about wildfire risks (30 minutes)

1. Ask participants of the subgroups to put a numbered sticker in the location with either significant wildfires risks or fire-vulnerable resources

2. Once all locations have been identified, ask participants to rank the wildfire risks in each place using a three-colour system (yellow for low risk, orange for medium risk and red for high risk)

3. Ask participants why each area (identified by the sticker number) is especially vulnerable to wildfires, or the resources of importance located there.

4. Ask participants if the wildfire risks have changed over time in these locations, and why.

Step 4: Restitution (10 minutes)
1. Resume quickly the main past wildfire events

2. Go through the areas with wildfire risks, giving an overview of the reason for wildfire risks, and areas with a changing wildfire risk

Step 5: Feedback (10 minutes)
1. Ask participants if they find some of the discussions useful and why

2. Ask what questions about fires and wildfires they would want to explore more

3. Ask participants what kind of outputs they would like to see from this exercise (a printed version of the map, seasonal burning calendar, something else)

Research Activity 3 – Field walks

After the group discussions, researchers could ask if participants are willing to bring the research team to some parts of the landscapes that have been identified during the group discussion. These field walks could be in parts of the landscapes that are managed by fires, have experienced past wildfires or have been identified as having a high wildfire risk.

The field walk is an opportunity for the research team to ask follow-up questions about different thematic that have been discussed during the day, such as the presence of different plants that could affect fire regimes, the way of preparing different sites before burning, andareas where fires are particularly easy or difficult to manage. Ideally, the research team could have a little time after the end of the activity to resume the most interesting points and think about a list of relevant questions to ask and feature in the landscape to identify. While non-exhaustive, here are some of the questions that could be asked to the participants during a walk:

- How are these sites prepared before burning?

- How was the fire ignited? Was there use of back and flank fires? Why?
- Which fire control measures were taken to control this burn?
- Are there trees or medicinal herbs that are vulnerable to fires?
- Which areas of the landscape are the hardest/easiest to burn properly? Why?
- Did the vegetation regenerate quickly after the fires?

During the field walks, the discussion with the guide should be recorded if possible (even so it could be harder than during a focus group discussion) and pictures of different features could be taken. At the end of the field walk, a small sketch could be drawn by the research teams and validated by the members of the community.

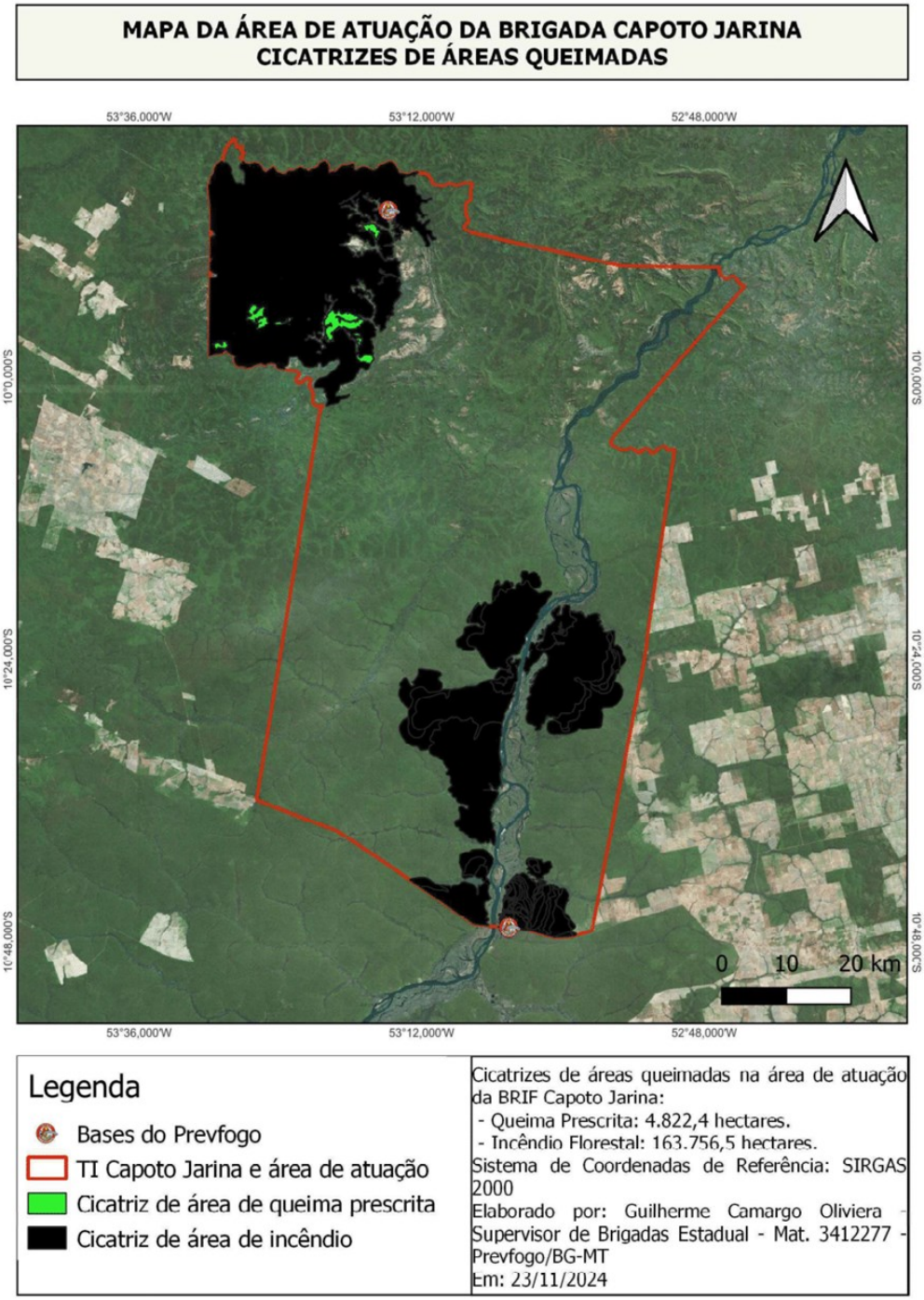
Appendix 4.6. Code structure

1st level of code	2nd level of code	Definition
Local change in climate	rainfall	Observations link to changes in rainfall patterns, regardless of the cause
	temperature	Observations link to change in temperature, regardless of cause
	Wind	Observations link to change in wind patterns, regardless of cause
	Impact of climate change	consequences of change in climate conditions
	Relevance of local ecological knowledge	perceived relevance of local ecological knowledge despite changes in climate
	Uncertainty about the future	speculation and uncertainties about future climatic conditions and their impact on local ecosystems
Environmental conditions	local ecological knowledge	Knowledge about the location, abundance and collection techniques of different natural resources
	Importance of ecosystems to support livelihood	Utility of natural resources, and the ecosystem services they provide, to support to livelihoods and cultural needs of the inhabitants
	Abundance and scarcity of natural resources	availability of different natural resources and their evolution
	External environmental degradation	sources of environmental degradation coming from outside of Capoto/Jarina, including deforestation, logging, pollution and other factors

1st level of code	2nd level of code	Definition
Fire use	Objectives of different fires	Objectives behind the use of different types of fires
	seasonality	time of the year when fires are usually lit, and relations with other seasonal activities and events in the landscape
	time of the day	preferences for the time of day for burning different types of fires
	decision to use fire or not	decisions to use fire or alternative land management tools
	preparation of the sites	All precautions were taken to prepare the site for burning before lighting a fire
	Fire control measures	Measures were deployed to prevent a fire from escaping and becoming a wildfire
	gender	Aspects of fire use that differ across genders
	specificity community	The difference in the way of using fires across different communities
Wildfires risks	Good and bad fires	Definition of desirable and undesirable fires and fire regimes
	Meteorological conditions	Meteorological conditions that are affecting the wildfire risks
	Combustible	Wildfire risks related to the presence of different types of fuels
	Ignitions sources	location and risks of different ignition sources that can start a wildfire
	neighbouring villages	risks of fires due to fires used by members of other villages from Capoto/Jarina
	damages	Potential damage and detrimental impact that could result from wildfires
	past wildfires	past wildfire events remembered by participants and associated damages
	Change in fire use and management	adaptation of fire use in response to the increasing wildfire risk

1st level of code	2nd level of code	Definition
Emergence of fire brigade	Response to growing wildfire risks	Emergence of the fire brigade in response to a growing wildfire risk
	Diffusion of fire brigade initiative	Spread of the fire brigade initiatives through different communities of Capoto/Jarina
	Introduction of firebreaks	Emergence, diffusion and use of firebreaks to contain fires
	Coordination of the fire brigades and the rest of the community	relationship between the fire brigades and different communities in Capoto/Jarina
	Delegation of wildfire risk management	Transfer of wildfire risk management responsibilities to the fire brigade
	Limits of firefighting	Situations where firefighting efforts and the fire brigade could be helpless
	Need of the firefighting brigade	Resources required by the fire brigade to increase its effectiveness.

Appendix 4.7 Map from the Prevfogo 2024 fire brigade report, classifying the savanna burning in Capoto as wildfires.



Appendices from Chapter 5

Appendix 5.1. Results of the statistical test for land use trends over the three villages of study and around Capoto/Jarina

Table S5.1. Results from the Man-Kendall test on trends of land use change in different buffer zones around Piaraçu.

Mann-Kendall results on land use trends in Piaraçu					
land use	**% in 1985**	**% in 2023**	**p-value**	**tau**	**TS slope (%.year^{-1})**
3 kilometers					
wetland	13.93	7.98	<0.0001	0.53	0.10
pasture	0.52	5.35	<0.0001	0.72	0.08
other	0.00	1.09	<0.0001	0.87	0.01
savanna	1.03	0.30	<0.0001	−0.65	−0.02
river	13.08	9.75	<0.0001	−0.69	−0.14
5 kilometers					
wetland	13.39	6.87	<0.0001	0.54	0.06
pasture	0.19	2.29	<0.0001	0.80	0.04
other	0.00	0.75	<0.0001	0.89	0.01
savanna	0.37	0.11	<0.0001	−0.65	−0.01
river	12.05	9.28	<0.0001	−0.74	−0.10
10 kilometers					
forest	64.02	67.20	<0.0001	0.60	0.03
wetland	6.92	3.21	<0.0001	0.57	0.03
floodable forest	20.99	22.60	<0.01	0.29	0.01
other	0.61	1.11	<0.05	0.28	0.01
river	7.47	5.89	<0.0001	−0.77	−0.06

Table S5.2. Results from the Man-Kendall test on trends of land use change in different buffer zones around Mupa.

Mann-Kendall results on land use trends in Mupa					
land use	**% in 1985**	**% in 2023**	**p-value**	**tau**	**TS slope (%.year^{-1})**
3 kilometers					
floodable forest	37.74	47.02	<0.0001	0.76	0.08
wetland	7.25	0.25	<0.0001	0.55	0.02
forest	36.30	37.16	<0.0001	0.85	0.01
other	0.00	0.12	<0.0001	0.55	0.00
soy	0.00	1.15	<0.005	0.38	0.00
river	18.71	14.30	<0.0001	−0.83	−0.12
5 kilometers					
floodable forest	45.06	51.36	<0.0001	0.72	0.06
wetland	7.23	2.09	<0.0001	0.55	0.03
forest	33.44	34.88	<0.0001	0.79	0.01
other	0.00	0.86	<0.0001	0.57	0.00
river	14.27	10.81	<0.0001	−0.82	−0.10
10 kilometers					
floodable forest	30.54	33.56	<0.0001	0.67	0.03
wetland	3.99	1.11	<0.0001	0.53	0.01
forest	57.37	58.43	<0.0001	0.73	0.01
other	0.05	0.48	<0.0001	0.61	0.00
river	8.05	6.41	<0.0001	−0.80	−0.05

Table S5.3. Results from the Man-Kendall test on trends of land use change in different buffer zones around Capoto.

Mann-Kendall results on land use trends in Capoto					
land use	**% in 1985**	**% in 2023**	**p-value**	**tau**	**TS slope (%.year^{-1})**
3 kilometers					
savanna	19.15	29.02	<0.0005	0.40	0.02
other	0.00	0.95	<0.0001	0.71	0.01
pasture	0.00	1.16	<0.0001	0.61	0.00
floodable forest	1.30	2.50	<0.05	−0.29	0.00
wetland	26.19	17.66	<0.0001	−0.66	−0.01
forest	45.95	41.32	<0.0001	−0.51	−0.03
5 kilometers					
other	0.00	1.12	<0.0001	0.84	0.01
savanna	16.85	23.69	<0.01	0.31	0.01
rocky outcrop	6.23	6.23	<0.0001	0.62	0.00
wetland	22.19	16.28	<0.0001	−0.74	0.00
forest	52.51	49.54	<0.0001	−0.58	−0.02
10 kilometers					
other	0.00	0.28	<0.0001	0.81	0.00
rocky outcrop	4.55	4.52	<0.0005	0.43	0.00
wetland	15.24	10.91	<0.0001	−0.71	0.00
forest	68.19	66.48	<0.0001	−0.58	0.00

Table S5.4. Results from the Man-Kendall test on trends of land use change in different buffer zones around Capoto.

Mann-Kendall results on land use trends around Capoto/Jarina					
land use	**% in 1985**	**% in 2023**	**p-value**	**tau**	**TS slope (%.year^{-1})**
20 kilometers					
pasture	10.43	27.15	<0.0001	0.74	0.46
other temporary crops	0.00	1.10	<0.0001	0.85	0.03
soybean	0.00	5.33	<0.0001	0.85	0.03
other	1.15	0.55	<0.0001	−0.66	0.00
savanna	0.31	0.14	<0.0001	−0.68	0.00
floodable forest	11.32	9.46	<0.0001	−0.65	−0.05
forest	76.79	56.28	<0.0001	−1.00	−0.51
10 kilometers					
pasture	6.38	18.05	<0.0001	0.79	0.27
soybean	0.00	2.20	<0.0001	0.81	0.01
floodable forest	10.22	9.42	<0.005	−0.34	−0.01
forest	82.08	69.01	<0.0001	−0.96	−0.28
5 kilometers					
pasture	6.98	17.83	<0.0001	0.56	0.17
soybean	0.00	1.21	<0.0001	0.65	0.00
savanna	0.52	0.31	<0.0001	−0.55	−0.01
forest	79.83	69.89	<0.0001	−0.73	−0.16