MINISTÉRIO DA CIÊNCIA, TECNOLOGIA E INOVAÇÃO
INSTITUTO NACIONAL DE PESQUISAS ESPACIAIS

sid.inpe.br/mtc-m21d/2023/03.30.18.13-TDI

TEMPORAL AND SPATIAL PATTERNS OF INDIVIDUAL FIRES IN THE BRAZILIAN BIOMES AND THEIR RELATIONSHIP WITH LANDSCAPE CHANGES

Thais Pereira Medeiros

Master's Dissertation of the Graduate Course in Remote Sensing, guided by Dra. Liana Oighenstein Anderson, approved in March 27, 2023.

URL of the original document:
<http://urlib.net/8JMKD3MGP3W34T/48QK7GS>

INPE
São José dos Campos
2023

PUBLISHED BY:

Instituto Nacional de Pesquisas Espaciais - INPE
Coordenação de Ensino, Pesquisa e Extensão (COEPE)
Divisão de Biblioteca (DIBIB)
CEP 12.227-010
São José dos Campos - SP - Brasil
Tel.:(012) 3208-6923/7348
E-mail: pubtc@inpe.br

DIGITAL LIBRARY:
Dr. Gerald Jean Francis Banon
Clayton Martins Pereira - Divisão de Biblioteca (DIBIB)
DOCUMENT REVIEW:
Simone Angélica Del Ducca Barbedo - Divisão de Biblioteca (DIBIB)
André Luis Dias Fernandes - Divisão de Biblioteca (DIBIB)
ELECTRONIC EDITING:
Ivone Martins - Divisão de Biblioteca (DIBIB)
André Luis Dias Fernandes - Divisão de Biblioteca (DIBIB)

MINISTÉRIO DA CIÊNCIA, TECNOLOGIA E INOVAÇÃO
INSTITUTO NACIONAL DE PESQUISAS ESPACIAIS

sid.inpe.br/mtc-m21d/2023/03.30.18.13-TDI

TEMPORAL AND SPATIAL PATTERNS OF INDIVIDUAL FIRES IN THE BRAZILIAN BIOMES AND THEIR RELATIONSHIP WITH LANDSCAPE CHANGES

Thais Pereira Medeiros

Master's Dissertation of the Graduate Course in Remote Sensing, guided by Dra. Liana Oighenstein Anderson, approved in March 27, 2023.

URL of the original document:
<http://urlib.net/8JMKD3MGP3W34T/48QK7GS>

INPE
São José dos Campos
2023

Cataloging in Publication Data

Medeiros, Thais Pereira.
M467t Temporal and spatial patterns of individual fires in the brazilian biomes and their relationship with landscape changes / Thais Pereira Medeiros. – São José dos Campos : INPE, 2023.
xxiv + 96 p. ; (sid.inpe.br/mtc-m21d/2023/03.30.18.13-TDI)

Dissertation (Master in Remote Sensing) – Instituto Nacional de Pesquisas Espaciais, São José dos Campos, 2023.
Guiding : Dra. Liana Oighenstein Anderson.

1. Burned area. 2. Landscape changes. 3. Brazilian biomes. 4. Fire-sensitivity. 4. Fire-dependent. I.Title.

CDU 528.8:504.2

MINISTÉRIO DA
CIÊNCIA, TECNOLOGIA
E INOVAÇÃO

INSTITUTO NACIONAL DE PESQUISAS ESPACIAIS
Serviço de Pós-Graduação - SEPGR

DEFESA FINAL DE DISSERTAÇÃO DE THAIS PEREIRA DE MEDEIROS
REG. 488878/2021, BANCA Nº 055/2023

No dia 27 de março de 2023, de forma online, o(a) aluno(a) mencionado(a) acima defendeu seu trabalho final (apresentação oral seguida de arguição) perante uma Banca Examinadora, cujos membros estão listados abaixo. O(A) aluno(a) foi APROVADO(A) pela Banca Examinadora, por unanimidade, em cumprimento ao requisito exigido para obtenção do Título de Mestra em Sensoriamento Remoto, com a exigência de que o trabalho final a ser publicado deverá incorporar as correções sugeridas pela Banca Examinadora, com revisão pelo(s) orientador(es).

Título: "Temporal and Spatial Patterns of Individual Fires in the Brazilian Biomes and their Relationship with Landscape Changes"

Membros da banca:

Dr. Luiz Eduardo Oliveira e Cruz de Aragão – Presidente – INPE
Dra. Liana Oighenstein Anderson – Orientadora – CEMADEN
Dra. Maria Isabel Sobral Escada – Membro Interno – INPE
Dra. Swanni Tatiana Alvarado Romero – Membro Externo – Universidad del Rosario
Dr. Celso Henrique Leite Silva Junior – Membro Externo – UCLA

Declaração de aprovação de Swanni Tatiana Alvarado Romero anexa ao processo.

Documento assinado eletronicamente por **Maria Isabel Sobral Escada**, **Pesquisadora**, em 06/04/2023, às 11:13 (horário oficial de Brasília), com fundamento no § 3º do art. 4º do Decreto nº 10.543, de 13 de novembro de 2020.

Documento assinado eletronicamente por **Liana Oighenstein Anderson**, **Pesquisador**, em 10/04/2023, às 11:03 (horário oficial de Brasília), com fundamento no § 3º do art. 4º do Decreto nº 10.543, de 13 de novembro de 2020.

Documento assinado eletronicamente por **Celso Henrique Leite Silva Junior (E)**, **Usuário Externo**, em 19/04/2023, às 13:01 (horário oficial de Brasília), com fundamento no § 3º do art. 4º do Decreto nº 10.543, de 13 de novembro de 2020.

Documento assinado eletronicamente por **Luiz Eduardo Oliveira E Cruz de Aragão**, **Chefe da Divisão de Observação da Terra e Geoinformática**, em 03/05/2023, às 15:53 (horário oficial de Brasília), com fundamento no § 3º do art. 4º do Decreto nº 10.543, de 13 de novembro de 2020.

Referência: Processo nº 01340.002291/2023-87 SEI nº 10945190

"Que nada nos limite, que nada nos defina, que nada nos sujeite, que a liberdade seja nossa própria substância, já que viver é ser livre".

Simone de Beauvoir

A meus pais ***Rita Pereira*** *e* ***Donizete Medeiros*** *e às minhas avós* ***Maria de Moura*** *e* ***Eunice Coutinho*** *(in memoriam)*

ACKNOWLEDGEMENTS

Agradeço à CAPES (Coordenação de Aperfeiçoamento de Pessoal de Nível Superior Brasil - Código de Financiamento 001) pela concessão da bolsa de estudos do mestrado. Ao Programa de Pós-Graduação em Sensoriamento Remoto do Instituto Nacional de Pesquisas Espaciais (INPE), por ceder as instalações e recursos necessários ao desenvolvimento da pesquisa. À minha orientadora, Liana Anderson, pelo suporte, amparo, e transferência de conhecimentos ao longo dessa trajetória. Aos membros da banca examinadora, Isabel Escada, Celso Silva Junior, Swanni Alvarado e Luiz Aragão, por todas as dicas e contribuições para a melhora do trabalho. À minha mãe (Rita), meu pai (José) e minha irmã (Ju) por todo amor, carinho e incentivo. Aos meus filhos e filhas (vulgo animais de estimação), Balu, Toco (in memoriam) e Sebastião (cachorros), Beto e Marelo (gatos) por todo suporte emocional. Aos meus queridos amigos do laboratório TREES, em especial Débora, Ana Larissa, Deila, Henrique Leão, Isadora, Marcus e Poliana. Sem o apoio e ajuda de todos eles essa dissertação não teria saído. À Becca, Milhouse e Pedro, família que a UNESP me presenteou, e mesmo que agora distantes foram essenciais e estiveram tão presentes durante essa trajetória. Não poderia deixar de agradecer às minhas queridas amigas, Maria e Evelin, que mesmo depois de 12 anos mantiveram-se presentes. Aos moradores da melhor república de Rio Claro, Santa no Soy, em especial, Jesus, Mari e Paixão. Também não poderia deixar de agradecer aos queridos, Belo, Karen e Tolima, grandes amigos da época na empresa júnior da geografia. À Manu, Jay, Gabi, Edu e Xu pela amizade e parceria. Aos meus queridos amigos da Espeleologia, do grupo QA, em especial Liz, Higor e Pinóquio. Todas as expedições realizadas em caverna durante essa trajetória foram, acima de tudo, momentos de terapia.

ABSTRACT

The development of human civilization has been consolidated as the main source of fire ignitions on the Earth, resulting in a significant alteration of the natural fire regimes. The landscape changes promoted by human activities are modifying its conditions of susceptibility to fire. The ecosystems have different characteristics on fire response, which are separated into three groups, here accessed based on the Brazilian biomes classification: Fire-sensitivity (Amazon and Atlantic Forest, Fire-dependent (Cerrado, Pantanal and Pampa), and Fire-independent (Caatinga). This research aimed to answer the following questions: How is the spatial and temporal dynamics of fire characterized in the different biomes? How has fire affected natural vegetation and how has it changed during the last decades? How land use and land cover changes can lead to a more flammable landscape? Two datasets were analyzed to answer these questions: Burned Area, derived from the Global Fire Atlas (GFA); and Land Use and Land Cover, acquired from the MapBiomas Collection 7. Our time series ranged from 2003 to 2018. The methodological procedures focused on the extraction of the following metrics: Total Burned Area (FIRE) from the GFA; and based on the forest class definition was extracted the landscape metrics: Percentage of Landscape (PLAND), Total Core Area (TCA), Edge Density (ED), Number of Patches (NP). Also, we applied some statistical methods, such as Linear Regression (OLS - Ordinary Least Squares) and Mann-Kendall trend (TAU coefficient and SEN slope). In general, 2010 and 2007 can be considered as fire peak years and the critical periods for fire occurrence is concentrated on August and September months. Regarding the analysis of what burned in the Brazil territory according land use and land cover classes, 60% of the burned area occurred in Natural Vegetation classes (Forest, Savanna and Grassland, Wetland), highlighting Cerrado, Pantanal and Caatinga, in which more than 80% of fires occurred in natural areas. Concerning the spatial configuration of fire and landscape metrics, we observed 38% of the Brazil territory with positive trends for burned area and, an intrinsic relationship with landscape changes. Our study indicated that natural fires regimes has been changed by human actions, and their spatial and temporal dynamics can be influenced by the relationship with vegetation history and landscape changes. In addition, wildfires are among the greatest forms of disturbance in tropical ecosystems, making the areas more favourable for landscape changes. In Fire-Sensitivity biomes we observed that a more and more open and degraded forest offers less suitable habitat and resources for humid forest-dependent animal species. In Fire-Dependent biomes, although the fire happens naturally, extensive and frequent fires can negatively impact trees and invertebrates. Finally, in the Fire-Independent biome, due to recent human activities, the Caatinga has been increasingly affected by fires that subject it to degradation and can turn it into a fire-sensitive system.

Keywords: Burned Area. Landscape Changes. Brazilian Biomes. Fire-Sensitivity. Fire-Dependent. Fire-Independent.

PADRÕES ESPACIAIS E TEMPORAIS DAS CICATRIZES INDIVIDUAIS DO FOGO NOS BIOMAS BRASILEIROS E SUA RELAÇÃO COM MUDANÇAS DA PAISAGEM

RESUMO

O desenvolvimento da civilização humana consolidou-se como a principal fonte de ignição do fogo na Terra, resultando em uma alteração significativa dos regimes naturais de fogo. As mudanças na paisagem promovidas pelas atividades humanas estão modificando suas condições de susceptibilidade ao fogo. Os ecossistemas apresentam diferentes características quanto às respostas ao fogo, os quais são separados em três grupos distintos: Sensíveis ao Fogo (Amazônia e Mata Atlântica, Dependentes do Fogo (Cerrado, Pantanal e Pampa) e Independentes do Fogo (Caatinga). Neste sentido, a pesquisa objetivou responder as seguintes perguntas: Como o fogo é caracterizado temporalmente e espacialmente? Como o fogo afetou a vegetação natural e mudou durante as últimas décadas? Como as mudanças no uso e cobertura da terra podem levar a uma paisagem mais flamável? Dois conjuntos de dados foram analisados para responder a essas perguntas: Área Queimada, derivada do Global Fire Atlas (GFA) e, Uso e cobertura da terra, adquiridos da Coleção 7 do MapBiomas. Nossa série temporal variou de 2003 a 2018. Os procedimentos metodológicos concentraram-se na extração das seguintes métricas: Área Total Queimada (FIRE); e com base na definição da classe florestal foram extraídas as seguintes métricas da paisagem: Porcentagem da Paisagem (PLAND), Área Core Total (TCA), Densidade de Borda (ED), Número de Manchas (NP). Além disso, aplicamos alguns métodos estatísticos, como Regressão Linear (OLS - Ordinary Least Squares) e Tendência de Mann-Kendall (coeficiente TAU e inclinação SEN). Em geral, 2010 e 2007 podem ser considerados os anos de pico de incêndios e os períodos críticos para ocorrência de incêndios concentram-se nos meses de agosto e setembro. Quanto à análise do que queimou no território brasileiro, 60% da área queimada ocorreu nas classes de Vegetação Natural (Florestas, Savanas, Área Úmidas), destacando-se Cerrado, Pantanal e Caatinga, onde mais de 80% dos incêndios ocorreram em áreas naturais. No que diz respeito à tendência espacial do fogo e das métricas da paisagem, observamos que 38% do território brasileiro apresentou tendências positivas para área queimada, e também, uma relação intrínseca com mudanças na paisagem. Nosso estudo indicou que os regimes de queimadas naturais estão sendo alterados pelas ações antrópicas. Além disso, sua dinâmica espacial e temporal pode ser influenciada pela relação com a história da vegetação e mudanças na paisagem. Os incêndios florestais estão entre as maiores formas de perturbação dos ecossistemas tropicais, tornando as áreas mais favoráveis para mudanças na paisagem. Em biomas sensíveis ao fogo, observamos que uma floresta cada vez mais aberta e degradada oferece habitat e recursos menos adequados para espécies animais dependentes de florestas úmidas. Em biomas dependentes do fogo, embora o fogo ocorra naturalmente, queimadas extensas e frequentes podem impactar negativamente árvores e invertebrados. Por fim, no bioma Independente do Fogo, devido às atividades humanas recentes, a Caatinga tem sido cada vez mais afetada por queimadas que a sujeitam à degradação e podem torná-la um sistema sensível ao fogo.

Palavras-chave: Área Queimada. Mudanças da Paisagem. Biomas Brasileiros. Sensíveis ao Fogo. Dependentes do Fogo. Independentes do Fogo.

LIST OF FIGURES

Page

LIST OF TABLES

CONTENTS

Page

1 INTRODUCTION

Fires are common disturbances, occurring naturally ignited or lightning, intentionally or accidentally by people. Everyday thousands of hectares of forests, woodlands, savannas, grasslands, shrublands, tundra, wetlands, and agricultural fields burn on the entire Earth (except Antarctica). For both people and ecosystems, fire can be harmful or beneficial depending on where, when, and how it burns. Fire can be harmful mainly in ecosystems composed of plants and animals with a lack of adaptations to withstand or take advantage of it. Most of the natural burned area occurs in grasslands and savannas where fires maintain open landscapes by reducing shrub and tree cover (SCHOLES; ARCHER, 1997; ABREU et al., 2017; SILVA-JUNIOR et al., 2020).

Paleoclimatic and paleobotanic records prove the presence of natural burning regimes in the evolution of terrestrial ecosystems long before the emergence of humans, with their origin linked to the emergence of plants, nearly 420 million years ago (SCOTT; GLASSPOOL, 2006). The emergence and expansion of human civilization consolidated as the main source of fire ignitions on the Earth, resulting in a significant alteration of the natural fire regimes (PAUSAS; KEELEY, 2009). In addition, the landscape changes promoted by human activities are increasing its conditions of fire susceptibility, materialized mainly by the construction of new road networks and the advance of agricultural frontiers, with the removal of natural vegetation, the introduction of invasive species and use of fire as a tool for agriculture practices (FOLEY et al., 2005).

In general, fires can affect the distribution of global ecosystems (BOND et al., 2005), reducing plant biomass and changing the structure of vegetation communities (VEENENDAAL et al., 2018). Also, fires are dynamic ecological forces that have evolutionary consequences and are fundamentally shaped by human actions (MCLAUCHLAN et al., 2020), causing disturbances on soil's chemical composition, carbon and water cycles, as well as, the climate system through the release of greenhouse gases (WERF et al., 2004; WERF et al., 2010; ARAGAO et al., 2018).

In some future scenarios, land use change, dry season, burned area and carbon emissions are projected to increase in all Shared Socioeconomic Pathways (SSP) scenarios, driven by an increase in temperature and a decrease in moisture availability, as well as, increase in food production and use of bio-fuels (LI et al., 2017; BURTON et al., 2020; BURTON et al., 2022).

The ecosystems differ in their response and susceptibility to fire events, where the impacts of changes in fire regimes reach different levels according to the characteristics of a particularly affected ecosystem. Conforming to ecological terms this is closely related to the way in which a given landscape has historically evolved with fire (ALVES; ALVARADO, 2019). Ecosystems characterized by the dominance of grasses, for instance, are co-evolved with fire, where their plants and animals show several adaptations and synergies with it. On the other hand, tropical forests are not fire-adapted and do not easily burn unless they suffer extreme drought or degradation/deforestation. When forests burn, fire can cause extremely negative effects on their biodiversity (PIVELLO et al., 2021). Fire impacts in a given ecosystem are determined by fire regime, the pattern of fire type, frequency, seasonality, intensity, and extent (JURVELIUS, 2004; KEANE, 2013).

In this context, Hardesty et al. (2005), characterized the ecosystems into three different groups: Fire-sensitivity, Fire-dependent, and Fire-independent. This classification considers only natural ecosystems and natural fire regimes and does not consider human impacts. Also, this classification was developed to estimate the degree to which ecologically uncharacteristic fire regimes may pose a threat to the conservation and sustainability of major habitat types.

Fire-sensitivity is, in general, tropical moist broadleaf forests, which are composed of plant and animal species that have no adaptations to tolerate fire events and cause them serious damage. Naturally, they are formed by vegetation and ecosystem structure that inhibits the start or spread of fire, and in these regions, fire events are often human-induced, influencing long-term ecosystem structure and relative abundance of species, and, also, limiting the ecosystem's size (HARDESTY et al., 2005; PIVELLO et al., 2021). (WERF et al., 2004)

On the other hand, Fire-dependent is characterized by seasonally dry environments, with a large fuel build-up dry plant material, where the biota has been shaped by a diverse form of adaptations to survive fire events (SIMON et al., 2009; MAURIN et al., 2014). In some cases, like in savannas, fire acts as a fundamental factor in sustaining some types of native plants and animals, and these fire events are characterized by frequent and present low-intensity, operating to maintain an open structure with dominant grasses and forbs. The main characteristic of these ecosystems is the resilience of their plants and animals in dealing with exposure to fires (HARDESTY et al., 2005; PIVELLO et al., 2021).

Finally, the Fire-independent group is defined as a habitat with unfavorable climatic conditions or a lack of fuel and ignition sources to provide fire events (HARDESTY et al., 2005; PIVELLO et al., 2021).

In the vast extension of the Brazilian territory, it is possible to identify ecosystems that have different characteristics in relation to responses to fire. Biomes such as Cerrado, Pampa, and Pantanal would be considered Fire-dependent. On the other hand, biomes with predominantly forest vegetation, such as the Atlantic Forest and the Amazon, would be considered Fire-sensitivity; and finally, the Caatinga, although with few studies that highlight its historical relationship with fire, could be considered as Fire-independent (HARDESTY et al., 2005; PIVELLO et al., 2021).

Brazil has experienced unprecedented wildfires in the last decades and natural fire regimes are being modified by human activities, usually related to land use practices or due to climate extremes linked to global warming (ARAGAO et al., 2008; BRANDO et al., 2014; SILVA et al., 2021; LIBONATI et al., 2021; WEES et al., 2021; KUMAR et al., 2022). Actually, fires are often exacerbated by climate conditions, such as warmer and drier conditions, drought and heatwaves, in which are responsible for increasing the vegetation flammability (KUMAR et al., 2022; DAMIAN et al., 2021).

In the search for a better understanding of the natural processes' dynamics and the anthropic influences on the earth's surface, comprehending the effects of fire events in the landscape changes becomes an important scientific challenge. Fire is one of the key elements in the dynamics of terrestrial ecosystems and can be considered as an ecosystem service (PAUSAS; KEELEY, 2019). So, in this perspective, the importance of an adequate characterization of the spatial and temporal patterns of fire incidence on different ecosystems is revealed. In order to analyze fire regimes at multiple spatial scales, the use of remote sensing products are consolidated as an important data source, providing information about vast areas with multi-temporal and multi-spectral coverage.

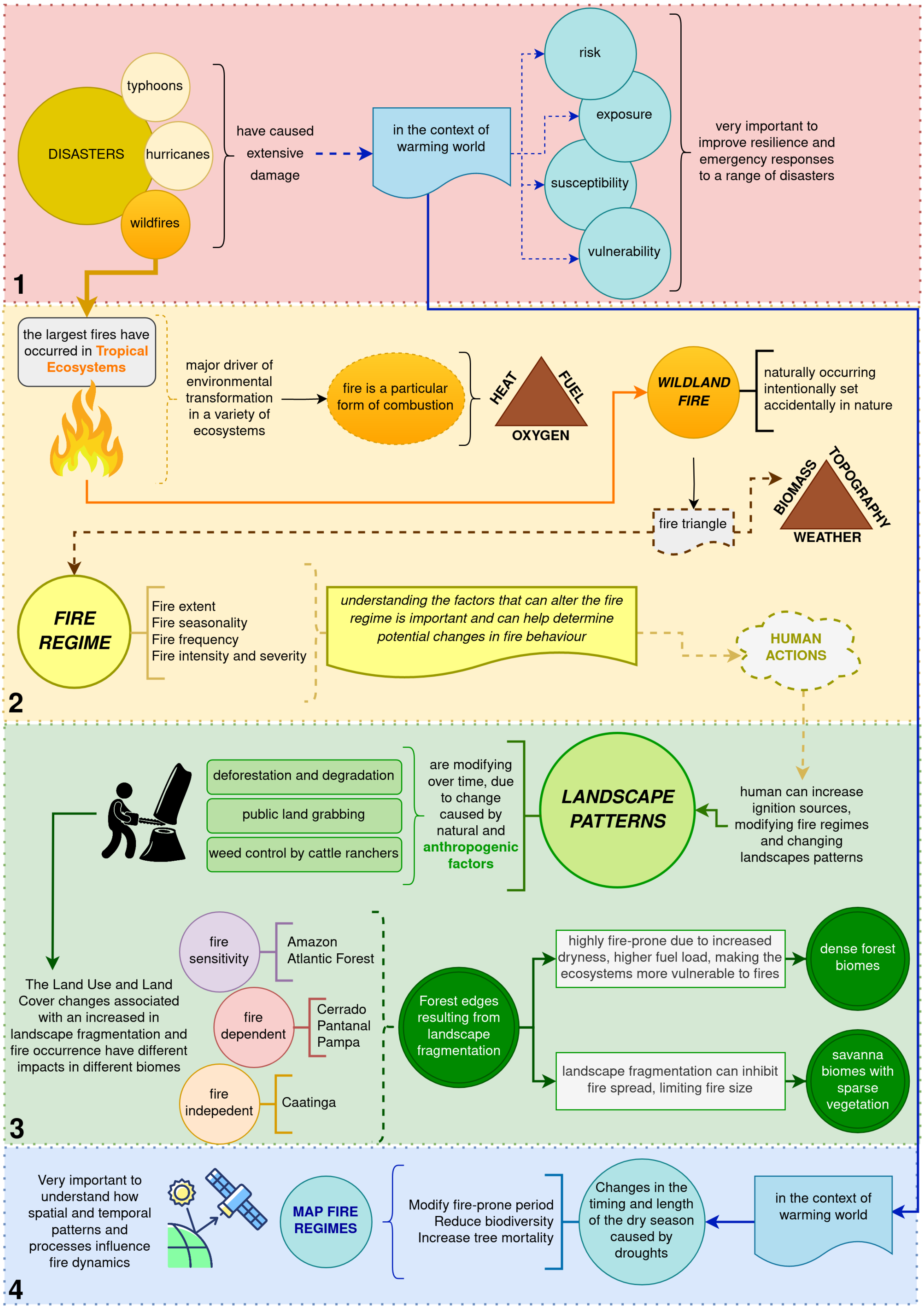
This study aimed to analyze the spatial and temporal dynamics of fires in the Brazilian Biomes and their relationship with landscape changes, to diagnose the areas most threatened by fire events. In this way, this research focused on comparing the fire and landscape patterns changes in the following categories, according to the vegetation history and their relationship with fires in the Brazilian Biomes: Fire-Sensitivity (Amazon and Atlantic Forest), Fire-Dependent (Pantanal, Pampa, and Cerrado), and Fire-Independent (Caatinga). Three main questions are proposed:

1. How is the spatial and temporal dynamics of fire characterized in the different biomes? 2. How has fire affected natural vegetation and how has it changed during the last decades? 3. How can land use and land cover changes lead to a more flammable landscape?

2 LITERATURE REVIEW

The structure of the Literature Review was organized into the following subsections, according to the conceptual model (Figure 2.1). A conceptual model is a representation of a system and explain, abstractly, the concepts involved in a given subject. In our case, we organized the concepts into four sections: [2.1] Conceptualization of fire in the tropics, highlighting how a major fire disasters can cause extensive impacts and consequences to the ecosystems and populations (part 1, in red color, of Figure 2.1); [2.2] Fire Ecology, this section was dedicated to the characterization of fires and to explain how fuel and flammability, weather conditions, topography and type of vegetation (fire triangle) influences the fire occurrence (part 2, in yellow color, of Figure 2.1); [2.3] Landscape Ecology, explaining how the fire patterns are modifying according to human-induced landscape changes (part 3, in green color, of Figure 2.1); [2.4] The use of remote sensing techniques as a strategy to understand fire dynamics, which represent a particularly useful tool for mapping and quantifying the patterns, sizes of burned areas (part 4, in blue color, of Figure 2.1). This conceptual model was important to define the structure of literature review.

Figure 2.1 - Conceptual model of the interactions between fire regime and landscape patterns.



Source: Produced by the author.

2.1 Wildfires risks and impacts

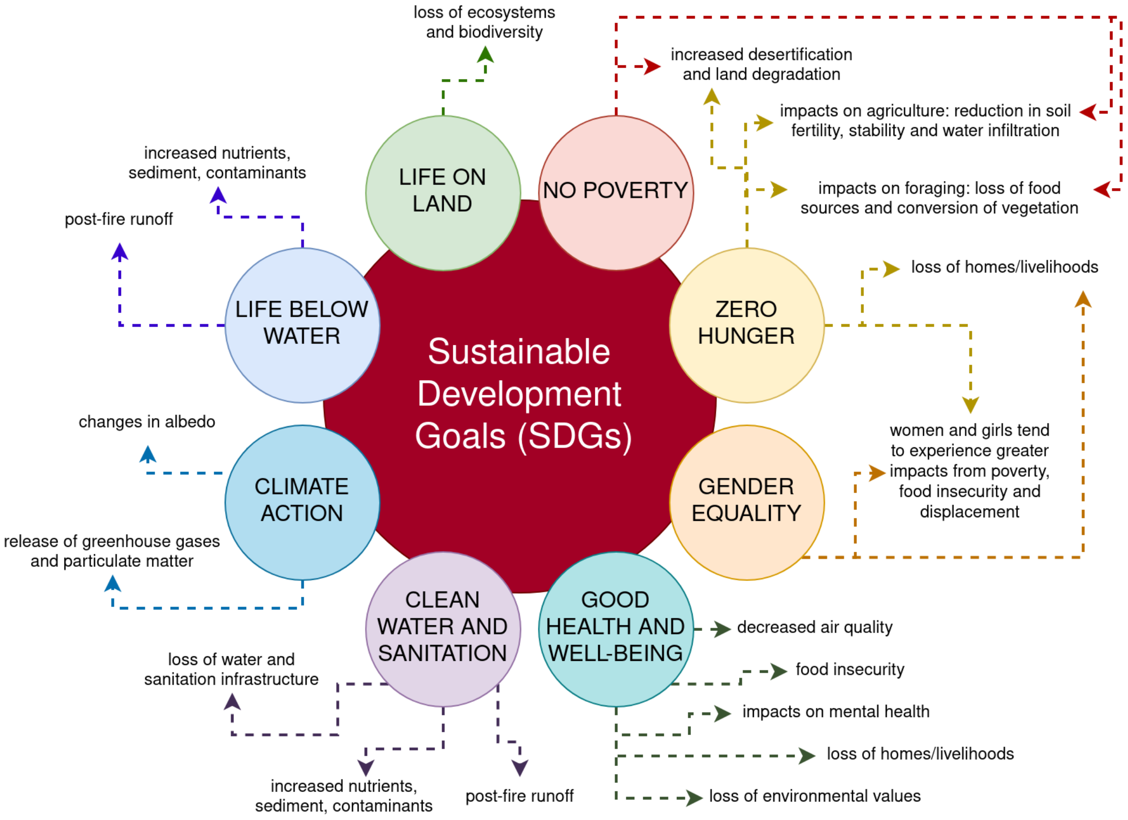
In the context of a warming world, where the changing nature of hazards is visible and influenced by rapid population growth and urbanization, and climate change impacts do not occur in isolation, studies of risk, exposure, susceptibility, and vulnerability are very important to improve resilience and emergency responses to a range of disasters.

Over the last decade, more wildfires are occurring, not only in regions where fires are common, such as savannas but also in areas not normally common, e. g., tropical forests, dry forests. The disturbance caused by wildfires can result in a large range of destructive ecological and social impacts, consequently causing tree mortality in fire-sensitive systems; changes in forest structure and composition; long-term reduction in carbon stocks; economic losses; harm the health of the local populations; impact air quality, clouds, precipitation, hydrological cycle, and moisture (LARSEN et al., 2017; CAMPANHARO et al., 2019; MARLIER et al., 2020; MATAVELI et al., 2019; ARGIBAY et al., 2020). These consequences of wildfires can in the long term represent a threat to the maintenance of United Nations Sustainable Development Goals (SDGs) (MARTIN, 2019) (Figure 2.2).

Also, fires can affect the climate and energy budget in two main ways: [1] Alterations to terrestrial ecosystem states and functioning; [2] Emissions of trace gases and aerosols (LI et al., 2017). Concerning the energy budget, fire is responsible for decreasing global average surface net radiation, sensible heat flux, and latent heat flux, and providing a little increase in-ground heat. The changes in surface air temperature caused by fires occur mainly due to a reduction in latent heat, which corresponds to the energy released or absorbed and associated with the physical state of water, which can change its form from solid to liquid, to vapour, and vice versa. This reduction in latent heat can be attributed to damage in vegetation canopy (Leaf Area Index — LAI), which decreases vegetation transpiration and canopy evaporation due to lower leaf area, fewer stomata, and less canopy interception and water storage, and increases soil evaporation by exposing more of the soil to the air and sunlight (LI et al., 2017).

As human-induced surface air temperature increases so do the frequency and intensity of the weather conditions to wildfires (JONES et al., 2020). When combined with increases in other factors such as the number of ignition sources and high levels of available fuel, the threat of wildfires becomes extreme, causing a positive trend in fire susceptibility during the 21st century (GOLDING; BETTS, 2008; JUSTINO et al., 2011).

Figure 2.2 - Impacts of wildfires and the consequences to United Nations Sustainable Development Goals (SDGs).



Source: Adapted from Sullivan et al. (2022).

2.2 Fire ecology

Fires are becoming a major driver of environmental transformation in several ecosystems, mainly in areas not previously adapted, for example, tropical rainforests, that have little adaptability to fire. Historically, fire has been central to terrestrial life ever since early anaerobic microorganisms poisoned the atmosphere with oxygen and multicellular plant life moved onto land, but, also, fire has been used as a tool for land-use management and can cause global impacts, affecting vegetation succession and sharing of greenhouse gas (GHG) emissions (CHUVIECO et al., 2014). During the Anthropocene and around the world fire has become more frequent, more intense, and more extended (COLLINS et al., 2021), which can imply a loss in life and structure, soil degradation, and change in vegetation and biodiversity (HARRISON et al., 2010). In this sense, the importance of developing better tools and approaches for fire prevention, assessment, and risk assessment is highlighted.

In general, fire is a particular form of combustion, an oxidation process, where the combination between fuels, oxygen and heat gives birth to fire on Earth. A fire begins with the combustion process, which requires a mixture of heat, fuel and oxygen, through breaking and reforming of chemical bonds (Figure 2.3). In wildland fires, fuels are primarily carbohydrates (cellulose and hemicellulose) derived from vegetative biomass, e.g. foliage, wood and humus (COCHRANE; RYAN, 2009).

The combustion can be divided into three main phases: preheating, gaseous, and smoldering. Preheating process is the preignition and dehydration phase of the combustion process, where endothermic process wherein fuel temperatures are raised, water and other volatiles are evaporated, and combustible gases are distilled from the fuels. The ignition temperatures for vegetative biomass are about 350º but cannot be reached until water in the fuels is driven out (WILLIAMS, 1982). This phase includes everything prior to actual ignitions of the fire.

In addition, gaseous phases begin when pyrolyzed fuels reach their heat of ignition. This is the temperature to which a fuel, in the presence of air, must be heated to start self-sustained combustion, wherein heat release is sufficient to maintain continued pyrolysis of the proximate solid fuels (COCHRANE; RYAN, 2009). The smoldering phase occurs when there is insufficient oxygen to support flaming combustion or when the easily pyrolyzed substances have been reduced to a level where flaming combustion is no longer possible. Smoldering fires spread very slowly along the surfaces of fuels and release very different types and amounts of volatiles and particulates than flaming combustion (CHRISTIAN et al., 2003).

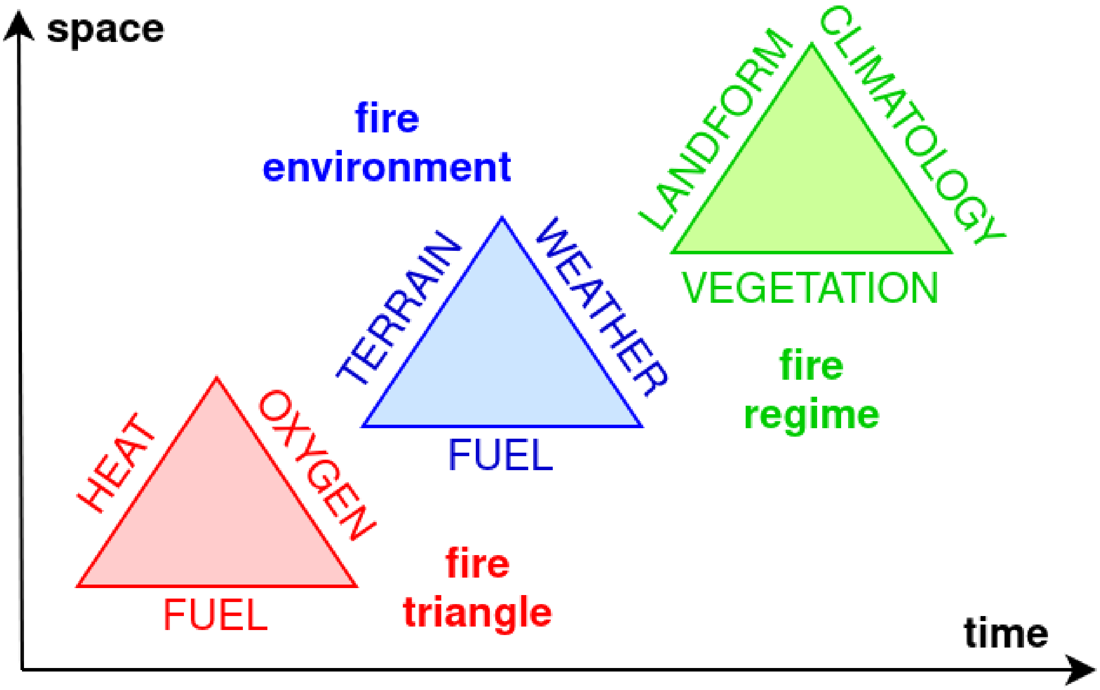
Furthermore, for fires to spread, it is necessary to transfer adequate amounts of heat to proximate fuels, which can be accomplished in three ways: conduction, convection and radiation. The first way, conduction, is a direct transfer of heat energy from one molecule to another, whereas, the second way, convection, refers to the transfer of heat through moving fluids. The third way, radiation, defined as the emission of energy as electromagnetic waves or as moving subatomic particles, is the main form of heat transfer responsible for preheating fuels and it controls fire spread rates for most wildland fires (COCHRANE; RYAN, 2009).

Another important aspect in the fire occurrence is the fuel, defined as the live and dead vegetative material available to combust, that is, any combustible substance:

anything that burns is “fuel”. In the case of wildland fires the biomass is literally what fuels wildfires and there are many factors related to fuels in wildland systems that govern whether and how fires can burn, such as, fuel composition, fuel loading and arrangement, fuel moisture, total fuels, live fuels, dead fuels, fuel continuity (COCHRANE; RYAN, 2009).

Oxygen is the third component of the fire triangle. Only heat and fuel are not enough to create the combustion process. Without sufficient oxygen there will be no fire. The hotter the environment, the less oxygen is required to maintain flaming combustion. Changes in wind direction can also rapidly change the rate and direction of fire spread (COCHRANE; RYAN, 2009).

Figure 2.3 - Fire concepts change across spatial and temporal scales.



Source: Adapted from Cochrane and Ryan (2009).

2.2.1 Wildland fire

The wildland fire can be naturally occurring, intentionally set, or accidentally in nature. The use of fire to manage food production has been changing increasingly the natural fire regimes, as a result of land management and climate change. The concept of fire regime alteration refers to the extent to which current patterns of fire have departed from the natural, historical, or ecological acceptable characteristics (HARDESTY et al., 2005). The alteration of key attributes of a fire regime can create long-term conditions that threaten the persistence of native plant and animal populations, degrading an ecosystem by changes in composition, structure and function. Understanding the direct effects of fires is necessary to know how their different characteristics interact with the surrounding environment.

A wildfire can be short and small in area, but more commonly occurs for an extended period, posing a significant risk to social, economic, and environmental values (SULLIVAN et al., 2022). According to the Sixth Intergovernmental Panel on Climate Change (IPCC) report, wildfires have become more frequent in some regions and will continue to increase with higher levels of global warming, mainly due to the continuing weather conditions (hot, dry, and windy) (PORTNER et al., 2022).

Variables such as fuel/flammability, weather conditions, and topography controls the fire behavior (speed, direction, and flame characteristics), the intensity of a fire and composes the fire triangle (Figure 2.4). Concerning the fuel/flammability, variables such as chemical composition, structure and arrangement, spatial continuity, density of fuel, type of vegetation controls the severity and extent of the fire (KEELEY, 2009; CHUVIECO et al., 2014; JUAREZ-OROZCO et al., 2017; SULLIVAN et al., 2022)).

Topography can directly influence the speed of fires (fires generally spread faster uphill than down) and the type and condition of the fuel by creating microclimates with localized moisture and growth conditions. The topography includes slope steepness, elevation, and aspect. Finally, weather conditions/climatology, the third aspect of the fire triangle, will influence the fire through changes in atmospheric stability, wind (speed and direction), air temperature, precipitation, and relative humidity, which can modify the combustibility of the fuel. The modifying forces of topography, fuel, and weather shape fire behaviour and comprise the fire environment (SULLIVAN et al., 2022).

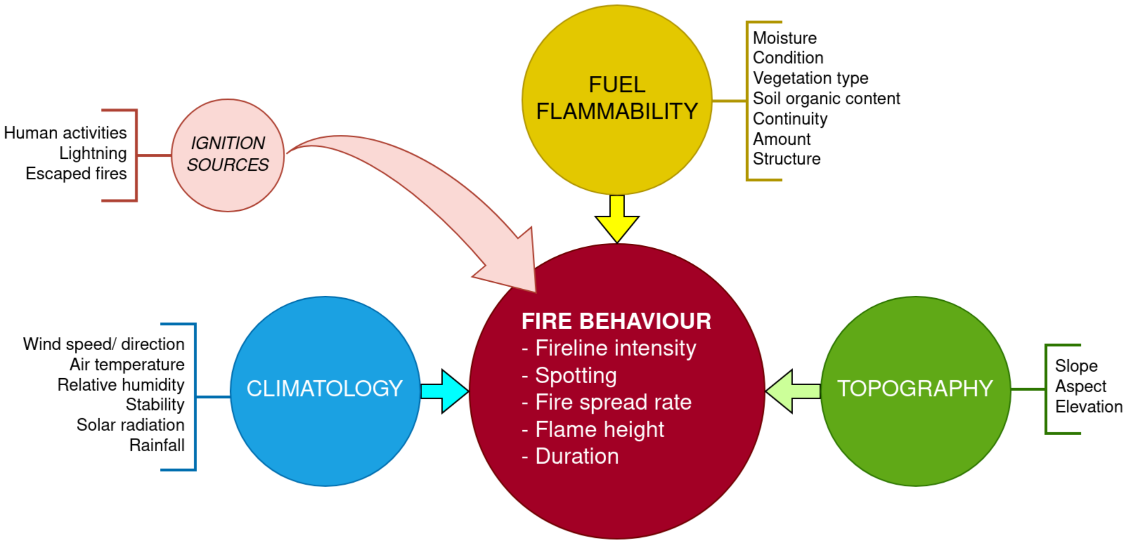
The understanding of the fire behavior triangle (fuel/flammability, weather conditions/climatology, and topography) is useful to achieve the drivers of the behavior of a single fire. Nonetheless, the fire regime is more embracing, including the types of ignition sources, intensity, severity, frequency, seasonality and extent of fire (COCHRANE; RYAN, 2009; KEELEY, 2009).

Fire intensity describes the rate of energy released from the combustion of biomass consumed in a fire per unit length of fireline (kW/m). The rate of fire intensity can varies between fuel types, fire spread modes and combustion conditions. Fire severity is defined as the immediate and direct impact of a fire on an ecosystem, depending of burn season, type of vegetation (and their adaptations to fire) and weather conditions. Fire frequency measures how common fires are in a given ecosystem and is defined as how many times an area burn in a specific period. The frequency of fire can have a strong effect on life cycle attributes, species composition, and community structure. In addition, fire seasonality describes the time of year, in which fires are most common. The impacts of fires can be greatly different depending on the phenological period when they occur. In the tropics, fires are usually constrained to the dry season or periods of unusual drought (e.g. El Ninõ). Finally, fire extent refers to the size of a given fire or an average fire size experienced by the ecosystem over long periods of time (COCHRANE; RYAN, 2009).

The fire regime can be affected by multiple factors, both natural and anthropogenic conditions. Natural conditions include vegetation structure and composition, amount, type, moisture content, climate and weather conditions, seasonal water deficit, and topography. In contrast, anthropogenic conditions are composed of land use, land management, crop and pasture techniques, religious traditions, and recreation practices (PIVELLO et al., 2021).

In summary, wildfires has great potential to cause increasing in landscape changes, fragmentation, flammability and ignition. On the other hand, drought, logging and deforestation can inhibit rainfall and increase even more the fire risk (ARAGAO et al., 2018; MARENGO et al., 2018; ASSIS et al., 2020; PIVELLO et al., 2021).

Figure 2.4 - Complex interactions between natural and anthropogenic factors that describe fire behavior.



Source: Adapted from Sullivan et al. (2022).

2.3 Landscape ecology

Regarding the human impacts on the natural ecosystems, they tend to alter disturbance regimes (BOWMAN et al., 2011). Along the history, human actions have altered the landscape, by modifying flow regimes, introducing livestock, changing grazing regimes, building infrastructure in inappropriate locations, altering soil erosion regimes, and ultimately, increasing ignition sources.

The term ‘Landscape’ was described by Alexander Von Humboldt, at the beginning of the XIX century, originated from the German term ‘Landchaft’, which means the totality of human living space (biosphere, geosphere, and noosphere) and considers a geographical-spatial connotation. Also, the Landscape can be defined as a heterogeneous area composed of a set of ecosystems that interact with each other and that are repeated in space (FORMAN; GODRON, 1986).

Still, the Landscape is a heterogeneous mosaic formed by interactive units and this heterogeneity is there for at least one factor, according to an observer and on a given scale. The sources of heterogeneity can come from different factors, as associated to physical phenomenons (topography, soils, moisture, hydro-geomorphological dynamic); anthropic perturbations (deforestation, fragmentation, roads and water reservoirs construction); or natural perturbations regimes (fire, tornado, hurricane, weeds) (METZGER, 2001).

Regarding Landscape Ecology, is the interdisciplinary science dedicated to study the interrelationship between human society and its living space (NAVEH; LIEBERMAN, 1994). In addition, Landscape Ecology refers to the study of structure, function and change in a heterogeneous land area composed of interacting ecosystems (FORMAN; GODRON, 1986). Also, Landscape Ecology considers the development and dynamics of spatial heterogeneity, spatial and temporal interactions and exchanges across heterogeneous landscapes (RISSER, 1987).

Patterns can be defined as the arrangement of shapes and behaviors, in which regularities are detected. In few words, refers to the recognition of common characteristics of the objects. The landscape patterns are modifying over time, due to changes caused by natural and anthropogenic factors (LAMBIN; STRAHLER, 1994) and can be characterized by two elements: spatial composition (What and how many elements?) and spatial configuration (How are the elements organized in the space? How are they characterized?). Some factors can be responsible for changing the landscape patterns, such as: landforms, climatology, soil, biotic interactions and land use (FORMAN, 1995).

Human occupation occurs in different ways according to different histories, actors, rural and urban producers, environmental, socioeconomic and infrastructure conditions, producing, consequently, different temporal and spatial patterns of land use and land cover (LAMBIN; STRAHLER, 1994; GODFREY; BROWDER, 1996; MERTENS; LAMBIN, 1999; GEIST; LAMBIN, 2002).

In general, anthropogenic land-use changes refer to a process of converting land, for agricultural use (crops or pasture), acting as a source of wildfire ignition (ARAGAO et al., 2008). These changes associated with a general lack of enforcement of environmental laws causes even more an increase in deforestation rates (NEPSTAD et al., 2014).

2.4 Remote sensing techniques: a strategy to understand fire dynamics

Remote sensing image and techniques represent a particularly useful tool for mapping and quantifying the patterns, sizes, and assessing the impacts of fires on a range of natural and social systems, e. g., allowing to simulate carbon emissions from biomass burning (SHIMABUKURO et al., 2020; WERF et al., 2009; LONG et al., 2019). Moreover, they are the most efficient source of information to understand the fire dynamics once they allow the acquisition of data from large areas of the surface daily and repeatedly (ANDELA et al., 2016; MATAVELI et al., 2018).

Analysis of the timing and distribution of fires is crucial for efficient fire management, helping governments and institutions to develop policies and emergency measures, providing information for fire prevention planning, assessing economic losses and ecological effects; monitoring changes in LULC, and developing atmospheric and climate impact models due to vegetation biomass burning (ICHOKU et al., 2012; SHIMABUKURO et al., 2020).

Earth observation data can be acquired by active fire detection, which registers the instantaneous temperature of objects using spectral bands in the interval of 4 to 11 micrometer (JUSTICE et al., 2002; PEREIRA et al., 2016) or by burned area estimation, which uses the spectral bands in the visible, near-infrared, and mid-infrared regions of the electromagnetic spectrum to detect changes in the spectral characteristics of land cover before and after the occurrence of fires (ROY et al., 2002; CHUVIECO et al., 2019).

The methods used to map and characterize fires are highly variable and burned areas can be detected by a variety of approaches, different purposes, and reach different scales (MOUILLOT et al., 2014; PESSOA et al., 2020). However, it is observed a lack of specific criteria in many methods, limiting their application and, also, a significant source of errors in burned area products, such as the presence of clouds, which reduces the ability to detect a fire hotspot; the lack of data on the moment of fire occurrence; incompatibility of the temporal and spatial resolutions of sensors (SHIMABUKURO et al., 2020). Currently, there are many global burned areas maps available to the scientific community, but they present different results due to sensor characteristics, image dates, spatial resolution, and mapping methods (HUMBER et al., 2019; PESSOA et al., 2020).

For instance, LANDSAT sensors, with 30 meters of spatial resolution, are adequate to map the burned areas, but due to their limited temporal resolution of only 16 days, this satellite is inefficient for researchers developed in savannas and pastureland regions, where scars from fire generally regenerate rapidly (HAWBAKER et al., 2017; CABRAL et al., 2018). On the other hand, the Moderate Resolution Imaging Spectroradiometer (MODIS) sensor, with a low spatial resolution (250 m) and daily acquisition, provides useful information for the global mapping of burned areas but underestimates small fire scars (CHEN et al., 2017; MELCHIORRE; BOSCHETTI, 2018).

The choice of which burned area product will be used in a study should consider the advantages and disadvantages in terms of the study objective, considering the regional performance of each one. The application of Remote Sensing techniques for reducing wildfire risk is a necessary component to achieve the United Nations 2030 Agenda for Sustainable Development, the objectives of the Sendai Framework for Disaster Risk Reduction 2015–2030, and the aims of the United Nations Decade on Ecosystem Restoration 2021–2030. Regardless of mitigation efforts, humans still need to learn to live with and manage the threats from wildfire.

3 MATERIALS AND METHODS

3.1 Study area

The study area corresponds to Brazil (Figure 3.1), located in South America, characterized by an extension of 8,516 mi km^2 and considered a megadiverse country divided into six biomes: Amazonia, Atlantic Forest, Cerrado, Pantanal, Pampa and Caatinga.

Figure 3.1 - Study Area. The green biomes (Amazon Basin and Atlantic Forest) are classified as Fire-sensitivity; the yellow biomes (Cerrado, Pampa and Pantanal) are classified as Fire-dependent/influenced; and the orange biome (Caatinga) is classified as Fire-independent.

Source: Adapted from Hardesty et al. (2005).

3.1.1 Amazonia

The Brazilian Amazon biome is the biggest Brazilian biome, covering an area of 4,196,943 km^2 and occupying 49.29% of the Brazil's territory. The Brazilian Amazon is one of the most studied and important biomes in the world, due to its capacity of carbon sequestration, containing one third of the Earth's species and representing half of the world's rainforest. Also, is an important exporter of several food and mineral commodities to global industry, including beef, milk, soy, corn, rice, bauxite, and iron ore. The Amazon River basin is the largest river basin on Earth, sustaining the largest continuous belt of floodplains and wetland on Earth. The vegetation is compose of a high species diversity and adaptation to different environments (JEZEQUEL et al., 2022).

3.1.2 Atlantic Forest

The Atlantic Forest stretches along Brazil's Atlantic Coast, occupying 1,110,182 km2 of the total area and representing about 13.04% of the national territory. The Atlantic Forest is the second largest forest in South America and one of the most biodiverse biomes, but is also one of the most threatened and impacted by encroachment of anthropogenic activities. Currently, pasture and agriculture are the two main land-use types in the Brazilian Atlantic Forest, impacting its conservation, increasing deforestation and landscape fragmentation and reducing landscape connectivity (RIBEIRO et al., 2011).

3.1.3 Caatinga

The Caatinga biome covers the northeast portion of Brazil, occupying 9.92% of the country's area (844,453 km^2). The Caatinga biome consists primarily of xeric shrubland coincides with the region called "Brazilian semi-arid", describing as the most biodiverse and the most populated semi-arid region in the world. The vegetation types ranges from the deciduous low scrub to small patches of tall dry forests, often fragmented (LEAL et al., 2003).

3.1.4 Cerrado

The Cerrado (Brazilian savanna) is the second biggest Brazilian biome (23.92%) and encompasses 2,036,448 km^2. The biome is a vast savanna ecoregion and described as the richest savanna of the world. The typical vegetation landscapes of the Cerrado biome consists of well-drained interfluves with gallery forests following the watercourses and ranging from dense grassland, usually with a sparse vegetation of shrubs and small trees. During the last three decades, the Cerrado have been extensively developed for agriculture and pasture with the active encouragement of the Brazilian government, producing a range of destruction, deforestation and degradation (LEHMANN et al., 2011).

3.1.5 Pantanal

The Brazilian Pantanal consists of a tropical wetland, representing 1.76% of the national territory (around 150,355 km^2). The landscape pattern consists of a mixture of floodable and non floodable grasslands, forests, open woodlands, and temporary or permanent aquatic habitats, supporting significant biodiversity. However, changes in the rainfall regime due to droughts intensification and increase in the landscape fragmentation may alter the dynamics of the wetlands. The flooding cycles are being changed, mainly due to deforestation for agriculture/pasture, and the construction of waterways, which contribute to the intensification of drier conditions favourable to fire spread (SCHULZ et al., 2019).

3.1.6 Pampa

Finally, the Brazilian Pampa biome occupies an area of 176,496 km^2, representing 2.07% of the entire country. The Brazilian Pampa are composed of vegetation coastal belt and the savanna environment that dominates the whole western part of the biome. The main forest formations are found at the northern limit of the biome, in the transition area to Atlantic Forest. The presence of natural grasslands made the region favorable to livestock and agriculture activities. This fragile biome does not support the intense use of the natural resources, where unsustainable production rate, mechanization, the introduction of exotic species and the cultivation of monocultures, have contributed to its degradation (ROESCH et al., 2009).

3.2 Data collection

Two datasets were used for the research analysis:

1. Burned Area, collected from Global Fire Atlas, a freely available dataset for the period between 2003 and 2018, that tracks the daily dynamics of individual fires to determine the timing and location of ignitions, fire size and duration, daily expansion, fire length, speed, and direction of spread (ANDELA et al., 2019).

2. Land Use and Land Cover (LULC), extracted from MapBiomas Collection 7, annual historical maps of LULC (1985-2021), based on random forest applied to

Landsat archives using Google Earth Engine (SOUZA et al., 2020).

The datasets are described in detail below.

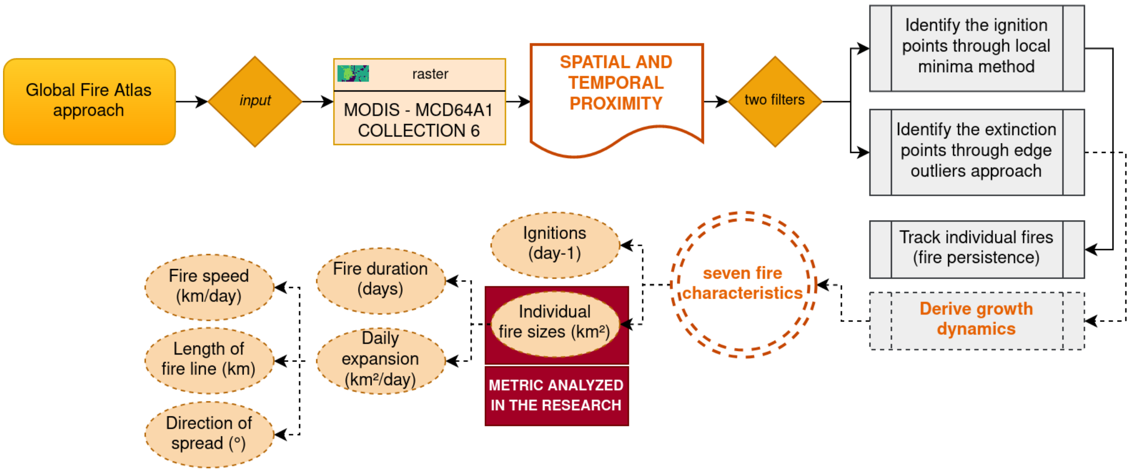
3.2.1 Global Fire Atlas

The Global Fire Atlas (GFA) is a global dataset that tracks the daily dynamics of individual fires based on a new methodology for identifying the location and timing of fire ignitions and estimating fire size and duration, and also, daily expansion, fire line length, speed, and direction of spread (ANDELA et al., 2019). GFA is derived from MODIS (Moderate Resolution Imaging Spectroradiometer), collection 6, which includes an estimated day-of-burn data layers at 500 m resolution (GIGLIO et al., 2018). Individual fire data were generated starting in 2003, when combined data from the Terra and Aqua satellite began to provide greater burn date certainty, and extended until November 2018.

The approach adopted by GFA (Figure 3.2) is based on the logic that fire-affected pixels can be separated into clusters according to spatial and temporal proximity, and this information can be used to study the number and size distributions of individual fires, fires shapes, and the location of ignitions points. One limitation of fire-clustering algorithms that rely on spatial and temporal proximity of fire pixels is the inability to separate individual fires within large burn patches that contain multiple ignitions points, most frequent phenomenon in grassy biomes (ANDELA et al., 2019).

The method used by GFA to isolate individual fires from daily moderate-resolution burned-area data is determined by two filters that account for uncertainties on the day of burn, in order to map the location and timing of fire ignitions and extent and duration of individual fires (Figure 3.2). The filter 1 is used to identify the ignitions points through local minima method, tracking individual fires (fire persistence) and the filter is used to identify the extinction points through edge outliers approach, deriving growth dynamics of individual fires. Through growth dynamics, the burned area was divided into seven fire characteristics: ignitions (day-1); individual fire sizes (km^2); fire duration (days); daily expansion (km^2 day-1); fire speed (km day-1); length of the fire line (km) and direction of spread (°) (ANDELA et al., 2019). Due to the spatial resolution of MODIS burned area product (500m), the minimum detected fire size is one MODIS pixel (approximately 25 ha). In this research we focused on exploring the fire size (km^2) metric, allowing the collection of total burned area per month and year (temporal patterns).

Figure 3.2 - The Global Fire Atlas approach.



Source: Adapted from Andela et al. (2019).

3.2.2 MapBiomas

The annual maps of MapBiomas Collection 7 were performed from the pixel-per-pixel classification method, through a machine learning approach within the Google Earth Engine (GEE) platform, based on a Random Forest classifier. This dataset was produced using the mosaic of images of Landsat satellites (5, 6 and 7), sensors Thematic Mapper (TM), Enhanced Thematic Mapper (ETM+) and Operational Land Imager/Thermal Infrared Sensor (OLI-TIRS), with 30 meters of spatial resolution, 16 days of temporal resolution and covering a range from 1985 to 2021. Originally, the LULC classes can be divided into 27 classes, grouped into 5 main categories: [1] Forest; [2] Non-Forest Natural Formation; [3] Farming; [4] Non-Vegetated Area; [5] Water (SOUZA et al., 2020).

3.3 Methodological procedures

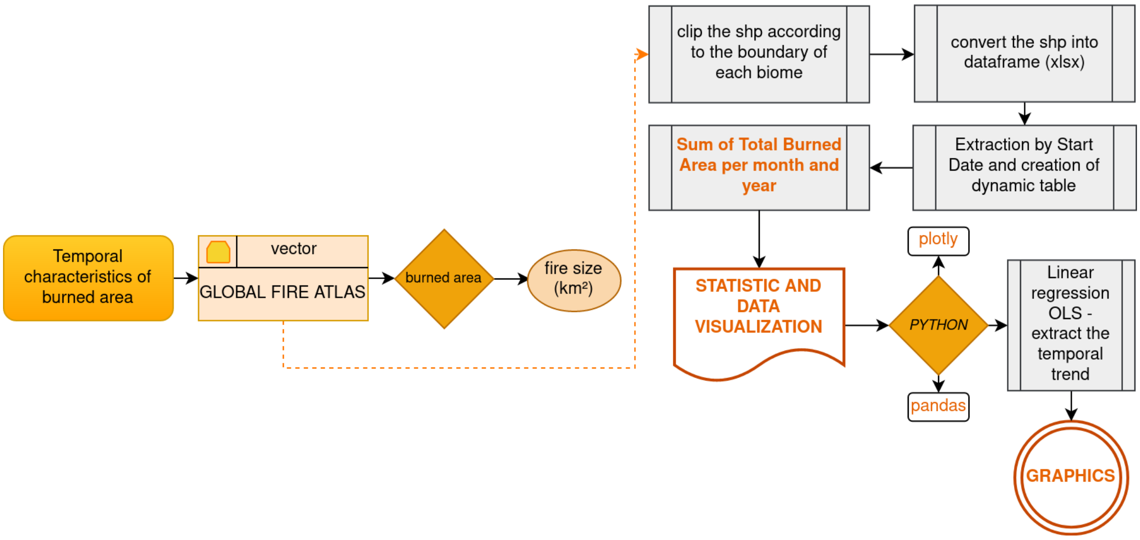
The research methods have been structured according to specific questions and can be summarized into the following subsections: Extraction of temporal and spatial fire patterns [3.3.1]; Extraction of what burned according to the land use and land cover classes [3.3.2]; Analysis with landscape metrics and integration between the variables [3.3.3].

3.3.1 Extraction of temporal and spatial fire patterns

Understanding the fire dynamics and separating large clusters of burned areas into individual fires is critical to any comprehension of the fire regime in human-dominated landscapes.

Aiming to analyze temporal characteristics of fire events in the Brazilian biomes we extracted the Total Burned Area, through the sum of total burned area per month and year. Also, we applied a linear regression (Ordinary Least Squares - OLS), in order to extract the temporal trend of fire for each biome. The OLS method can be defined as a linear regression technique used to estimate the unknown parameters in a model. This method relies on minimizing the sum of squared residuals between the actual (observed value of the dependent variable - burned area) and predicted values from the model (time). We applied a linear regression for each Brazilian biome according to different periods: 2003-2018 (general trend); 2003-2010; 2011-2018 (Figure 3.3). We divided into three different periods to understand the trend breaks of fire patterns along the time series.

Figure 3.3 - Step 1 of the methodological procedures: extraction of temporal characteristics of burned area.



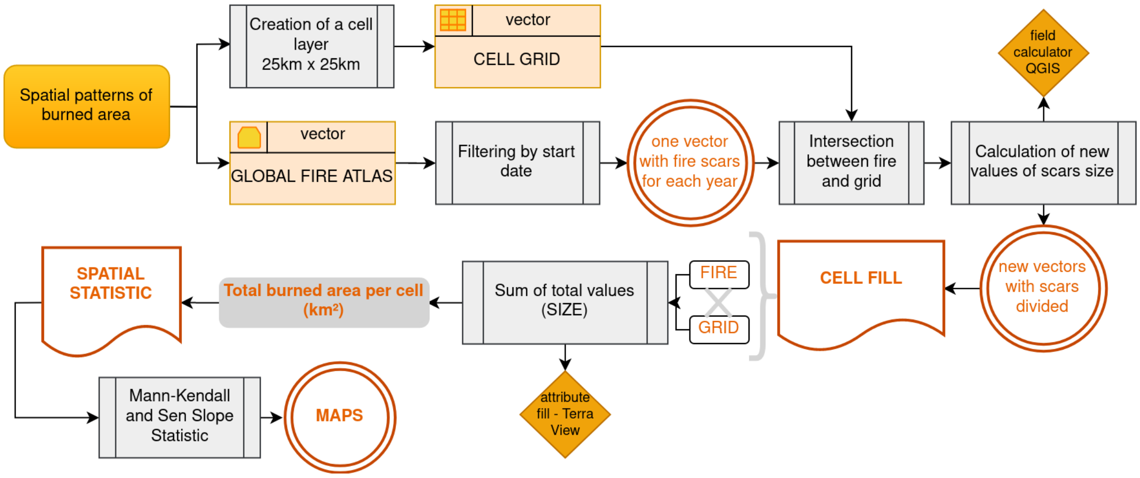
Source: Produced by the author.

The next step was dedicated to extract the spatial patterns of fires in the Brazilian biomes (Figure 3.4). First, a regular grid of size 25 km^2 was created to perform the cell filling. The main goal of this process is to homogenize information from different datasets sources, aggregating on a same spatial-temporal base. The fire data was filtered according to the start date, separating the fire polygons to each year and resulting in one shapefile for each year. Aiming to fill the information correctly, an intersection step between fire and grid was performed to divide the polygons according to each cell.

As the fire polygons were subdivided according to the intersection with grid cells, a new size (for each new polygon) was calculated, aiming to avoid errors in the filling processing. These new metrics were calculated using functions of the field calculator, exhibited in QGIS software. Finally, the cell fill was performed between the two vectors (fire and grid) using the function of 'attribute fill" available in TerraView software. To fill the fire size within the grid we used the 'sum of total values', aiming to calculate the total burned area per cell (km^2).

In order to identify the changes and variability of burned area across the time series we applied two robust non-parametric methods that are not particularly sensitive to discrepant data, the Mann–Kendall test (MANN, 1945) and the Sen's Slope estimator (SEN, 1968), using the packages available in Rstudio: 'wql' (https://cran.r-project.org/web/packages/wql/wql.pdf) and 'kandall' (https://cran.r-project.org/web/packages/Kendall/Kendall.pdf). The Mann-Kendall statistical test for trend is used to assess whether a dataset values is increasing over time or decreasing over time, and whether the trend in either direction is statistically significant. The Mann-Kendall test does not assess the magnitude of change, varying between +1 (positive changes over the time) to -1 (negative changes over the time) (MANN, 1945). The magnitude of change was assessed through the calculation of sen slope metric (SEN, 1968). A significance level of 0.05 (p-value<0.05) was adopted.

Figure 3.4 - Step 2 of the methodological procedures: extraction of spatial patterns of burned area.



Source: Produced by the author.

3.3.2 Extraction of what burned in the Brazilian biomes, according to land use land cover classes

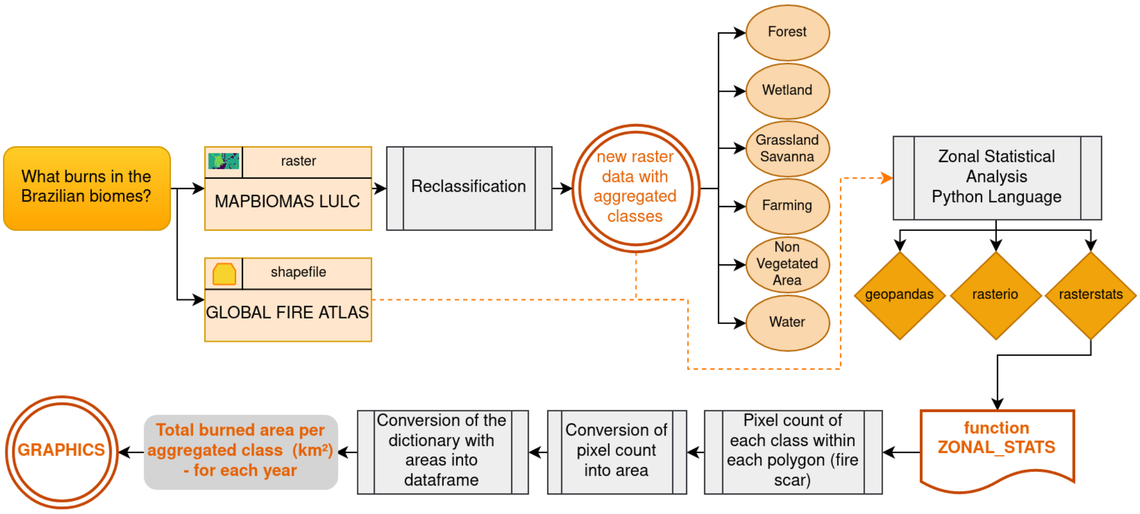
In order to produce the diagnosis of the burn extension that the natural vegetation's suffered, we first performed a reclassification process to aggregate the land use and land cover classes according: Forest (Forest Formation, Sandy Coastal Plain Vegetation), Wetland (Mangrove, Wetland, Hypersaline Tidal Flat), Grassland/Savanna (Savanna Formation, Grassland, Rocky Outcrop, Herbaceous Sandbank Vegetation, Other non Forest Formations), Farming (Pasture, Agriculture, Forest Plantation, Mosaic of Uses), Non Vegetated Area (Beach, Dune and Sand Spot, Urban Area, Mining, Other non Vegetated Areas) and Water (River, Lake and Ocean). Forest, Wetland, Grassland/Savanna and Water were considered as natural formations. On the other hand, Farming and Non Vegetated Area were considered as anthropic uses.

To quantify what burns in the Brazilian biomes we applied a zonal statistical analysis between fire polygons (a vector data) and LULC (a raster data), using an automatic script available in Python Programming Language (Figure 3.5). Zonal Statistics is a fundamental operation which requires the combination of raster and vector data to compute aggregate values for a zone (SINGLA; ELDAWY, 2018). This zone is defined by the vector data (in our case the zone is each fire polygon) using the values provided by the raster data (in our case the area of each aggregated class, arising from LULC data).

We used three main libraries available in Python Language to perform the analysis: geopandas (https://geopandas.org/en/stable/); rasterio (https://rasterio.readthedocs.io/en/latest/); and rasterstats (https://pythonhosted.org/rasterstats/). We applied the function 'zonal-stats' and treated the raster data as categorical, that means, the raster values represent discrete classes. Our goal was to summarize the area of each class by polygon (fire polygons).

The second step was converting the pixel count into area, in which we multiplied the pixel count by the spatial resolution of the original raster (30mx30m) and divided by one million to convert it into km^2. Finally, we converted the dictionary into a dataframe, summing the total burned area per aggregated class and converting the area values into a percentage.

Figure 3.5 - Step 3 of the methodological procedures: extraction of what burned in the Brazilian biomes per year.



Source: Produced by the author.

3.3.3 Analysis with landscape metrics and integration between variables

Landscape ecology involves the study of landscape patterns, the interactions among patches within a landscape mosaic, and how these patterns and interactions change over time. Aiming to understand the landscape patterns of Brazilian biomes we extracted some landscape metrics (Table 3.1), using the landscape metrics package, available in Rstudio (https://cran.r-project.org/web/packages/landscapemetrics/index.html).

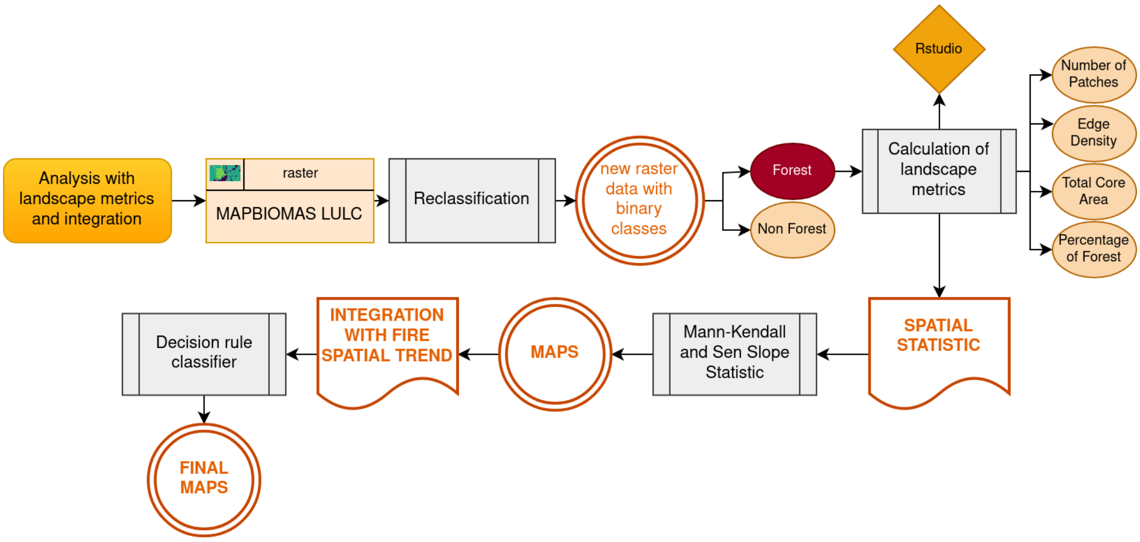
Table 3.1 - Description of the landscape metrics calculated.

Scale	**Acronym**	**Metric**	**Unit**
Class	PLAND	Percentage of Forest	%
Class/landscape	NP	Number of Forest Patches	count
Class/landscape	ED	Forest Edge Density	n/ha
Class/landscape	TCA	Total Forest Core Area	ha

We used a binary reclassification between Forest and Non Forest (Wetland, Grassland/Savanna, Farming, Non Vegetated Area, Water) and, calculated the landscape metrics for the Forest class. These metrics were calculated for all years of the time series and aggregated to the grid. Also, we applied the Mann–Kendall test (MANN, 1945) and the Sen's Slope estimator (SEN, 1968), to identify the changes and variabilities of landscape metrics across the time series.

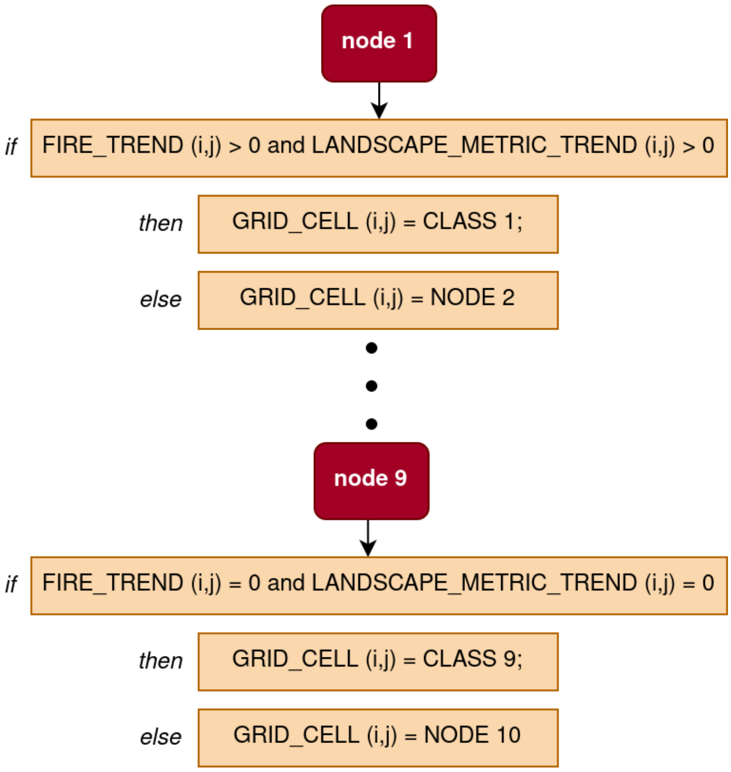
Finally, we performed a grid-based integration process between fire and landscape metrics using a decision rule classifier. This process was achieved by quantifying the number of grid cells with positive and negative fire trends overlapping with positive and negative grid cells of each landscape metric. The decision rule classifier was built with nodes, in which each node was developed with a Boolean operator (Figure 3.7).

Figure 3.6 - Step 4 of the methodological procedures: calculation of landscape metrics and integration process.



Source: Produced by the author.

Figure 3.7 - Schematic representation of the integration process obtained through decision rule classifier.



Source: Produced by the author.

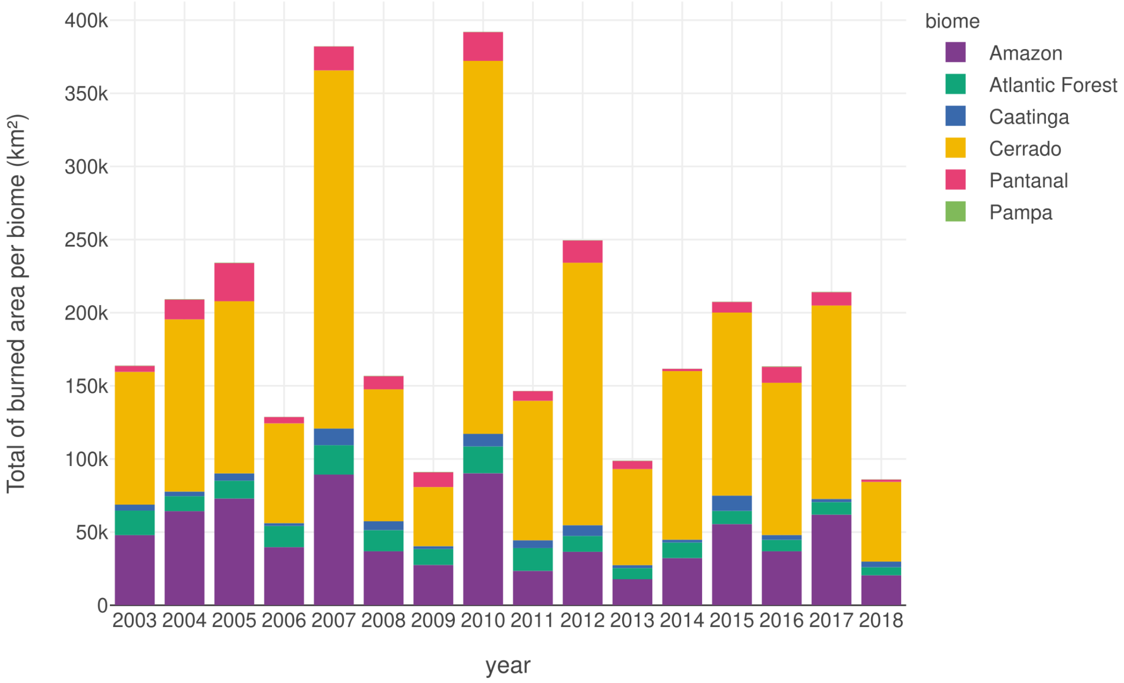
4 RESULTS

The results were organized into the following subsections: [4.1] General comparison of fire patterns in Brazilian biomes; then, a more detailed analysis structure on the intra-annual patterns of fires in each category: Fire-sensitivity, Fire-dependent and, Fire-independent biomes is presented [4.1.1]. The following section presents the analysis of what burns in the Brazilian biomes, according to land use and land cover aggregated classes, highlighting the natural vegetation fire-affected areas [4.2]. The section [4.3] shows the spatial patterns of burned area [4.3.1] and landscape metrics [4.3.2], and section [4.4] the integration between spatial fire patterns and landscape fragmentation.

4.1 Burned area extent and trend in the Brazilian biomes

In Brazil, over the time series (2003-2018), the burning peak years were 2010, 2007 and 2012, totaling 392,057 km², 382,163 km², and 249,596 km², respectively (Table 4.1 and Figure 4.1).

Figure 4.1 - Total burned area (km²) per Brazilian biome. Each color in the graphic represents one biome and each bar represents the total sum of the burned area in the entire country.



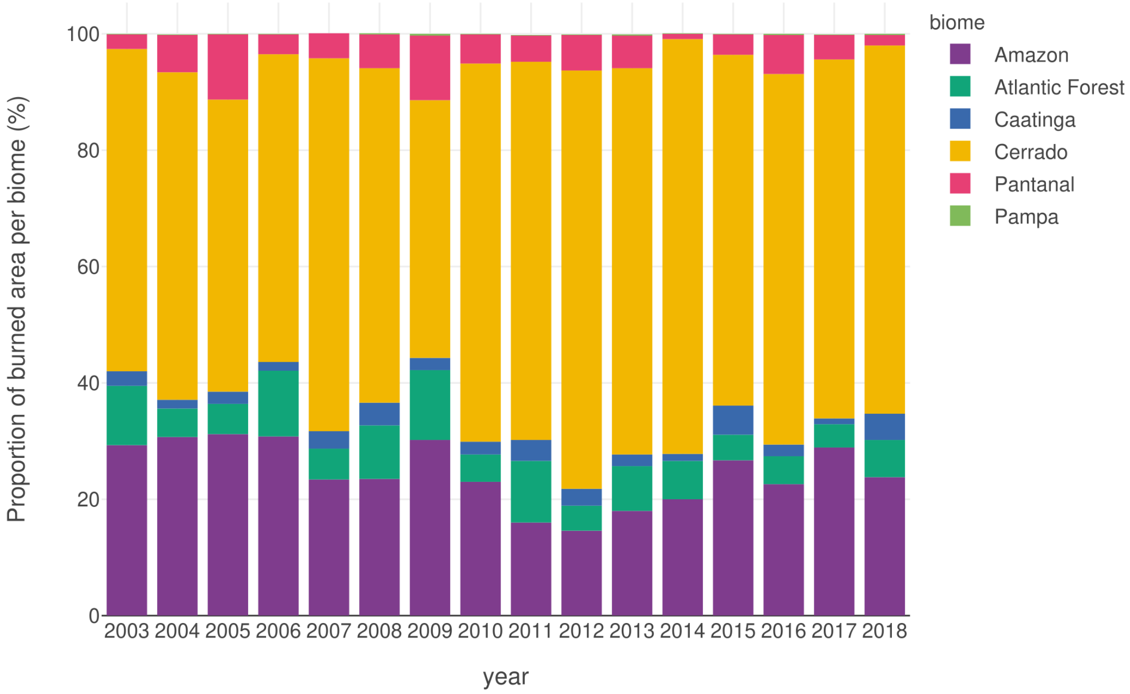
Source: Produced by the author.

Table 4.1 - Total burned area per year (km^2) in each Brazilian biome. AMZ represents the Amazon biome, ATF Atlantic Forest, CTG Caatinga, CRD Cerrado, PMP Pampa, PTN Pantanal. The last column - BR - represents the total sum of the burned area in the entire country.

Year	**AMZ**	**ATF**	**CTG**	**CRD**	**PMP**	**PTN**	**BR**
2003	47944.45	16787.69	4066.15	90748.05	208.76	4123.80	163878.9
2004	64326.25	10247.89	3151.17	117716.30	312.80	13473.27	209227.68
2005	72975.43	12219.69	4966.92	117683.46	266.54	26136.05	234248.09
2006	39723.42	14515.09	1911.36	68195.04	130.03	4320.16	128795.1
2007	89337.23	20095.00	11376.56	244873.20	157.27	16323.77	382163.03
2008	36903.93	14463.36	6062.02	90176.92	241.74	9026.62	156874.59
2009	27531.50	10922.12	1936.69	40416.91	254.90	10111.88	91174
2010	90247.60	18348.11	8583.45	254990.23	197.03	19690.82	392057.24
2011	23514.06	15587.35	5347.11	95329.36	115.80	6661.25	146554.93
2012	36543.88	10849.14	7332.07	179446.22	174.61	15250.38	249596.3
2013	17787.48	7616.98	2021.89	65674.00	235.51	5586.50	98922.36
2014	32269.87	10627.03	1931.78	115265.72	93.33	1508.88	161696.61
2015	55476.17	9040.39	10431.69	125163.63	195.25	7243.12	207550.25
2016	36898.23	7807.33	3318.68	104066.89	337.63	10894.36	163323.12
2017	61937.22	8574.34	2163.13	132255.84	268.74	9059.66	214258.93
2018	20463.14	5521.86	3874.38	54494.15	163.84	1567.02	86084.39

In the Amazon, we observed that 2010 was the peak year (90,000 km^2, responsible for 23% of the fire occurence in the Brazil territory), followed by 2007 (89,000 km^2, responsible for 23.4%) and 2005 (73,000 km^2, 31.2%). In the Atlantic Forest biome, we observed that the year which burned the most was 2007 (20,000 km^2, responsible only for 5.3% of the fire occurence in the Brazil territory), followed by 2010 (18,000 km^2, 4.7%) and 2005 (73,000 km^2, 5.2%). In Caatinga, 2007 was the year with the greater extent (11,000 km^2, responsible for 3% of the fire occurence), followed by 2015 (10,000 km^2, 5%), and 2010 (8,500 km^2, 2.2%). In Cerrado the years with the greatest extent of burned areas were 2010 (255,000 km^2, responsible for 65% of the fire ocurrence in the Brazil territory), followed by 2007 (245,000 km^2, responsible for 64.1%) and 2012 (179,000 km^2, responsible for 71.9%). In Pantanal was 2005 (26,000 km^2, responsible for 11.2%), followed by 2010 (20,000 km^2, 5%) and 2007 (16,000 km^2, 4.3%). Finally, in the Pampa biome, the patterns are very different when compared to the other biomes, where the year that burned the most was 2016 (338 km^2, responsible only for 0.2%), followed by 2004 (313 km^2, 0.1%) and 2017 (267 km^2, 0.1%). These percentages can be better visualized in Table 4.2 and Figure 4.2.

Figure 4.2 - Proportion of the burned area in Brazil, calculated by the division between total burned area in each year and each Brazilian biome and total burned area in Brazil, multiplied by 100. Each color in the graphic represents one biome.



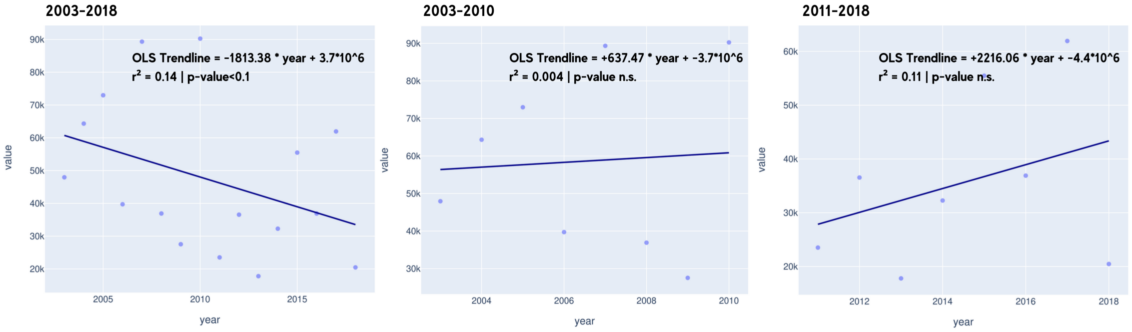
Source: Produced by the author.

Concerning the trends of burned area, we observed, in Amazon biome, a decrease of burned area in the period 2003-2018, representing a variation of -1,813.38 km^2/year (p-value$<$0.1 and R^2=0.14), a increase in the period 2003-2010, representing a variation of +637.47 km^2/year (p-value n.s. and R^2=0.004) and, an increase in the period 2011-2018, representing a variation of +2216.06 km^2/year (p-value n.s. and R^2=0.11) (Figure 4.3).

Table 4.2 - Proportion of the burned area in Brazil, calculated by the division between total burned area in each year and each Brazilian biome and total burned area in Brazil, multiplied by 100. AMZ represents the Amazon biome, ATF Atlantic Forest, CTG Caatinga, CRD Cerrado, PMP Pampa, PTN Pantanal and BR Brazil.

Year	AMZ	ATF	CTG	CRD	PMP	PTN	BR
2003	29.3	10.2	2.5	55.4	0.1	2.5	100
2004	30.7	4.9	1.5	56.3	0.1	6.4	100
2005	31.2	5.2	2.1	50.2	0.1	11.2	100
2006	30.8	11.3	1.5	52.9	0.1	3.4	100
2007	23.4	5.3	3.0	64.1	0.0	4.3	100
2008	23.5	9.2	3.9	57.5	0.2	5.8	100
2009	30.2	12.0	2.1	44.3	0.3	11.1	100
2010	23.0	4.6	2.2	65.0	0.1	5.0	100
2011	16.0	10.6	3.6	65.0	0.1	4.5	100
2012	14.6	4.3	2.9	71.9	0.1	6.1	100
2013	18.0	7.7	2.0	66.4	0.2	5.6	100
2014	20.0	6.6	1.2	71.3	0.1	0.9	100
2015	26.7	4.4	5.0	60.3	0.1	3.5	100
2016	22.6	4.8	2.0	63.7	0.2	6.7	100
2017	28.9	4.0	1.0	61.7	0.1	4.2	100
2018	23.8	6.4	4.5	63.3	0.2	1.8	100

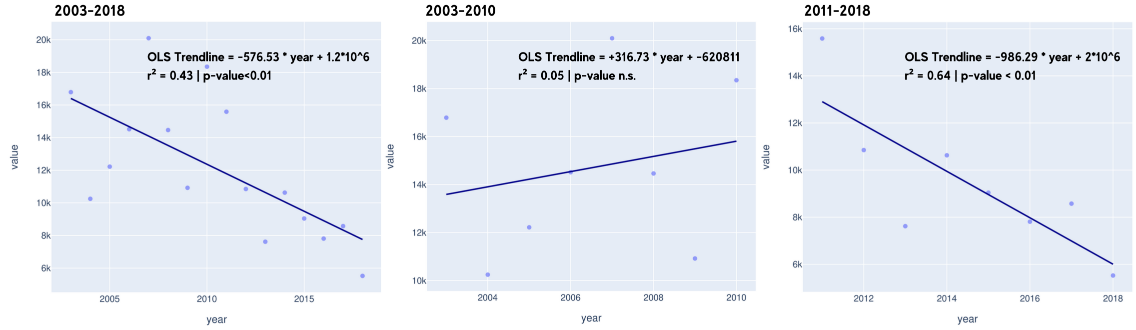
Figure 4.3 - Trend line of burned area over the time series (2003-2018; 2003-2010; 2011-2018) in the Amazon biome.



Source: Produced by the author.

In Atlantic Forest, we observed a decrease of burned area in the period 2003-2018, representing a variation of -576.53 km^2/year (p-value<0.01 and R^2=0.43); an increase in the period 2003-2010, representing a variation of +316.73 km^2/year (p-value n.s. and R^2=0.05); and a decrease in the period 2011-2018, representing a variation of -986.29 km²/year (p-value<0.01 and $R^2=0.64$) (Figure 4.4).

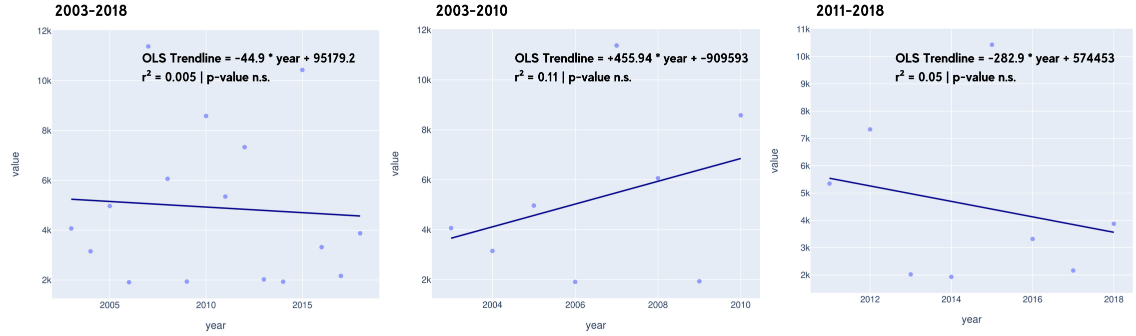
Figure 4.4 - Trend line of burned area over the time series (2003-2018; 2003-2010; 2011-2018) in the Atlantic Forest biome.



Source: Produced by the author.

In Caatinga biome, we observed a decrease of burned area in the period 2003-2018, representing a variation of -44.9 km²/year (p-value n.s. and $R^2=0.005$); an increase in the period 2003-2010, representing a variation of +455.94 km²/year (p-value n.s. and $R^2=0.11$); and a decrease in the period 2011-2018, representing a variation of -282.9 km²/year (p-value n.s. and $R^2=0.05$) (Figure 4.5).

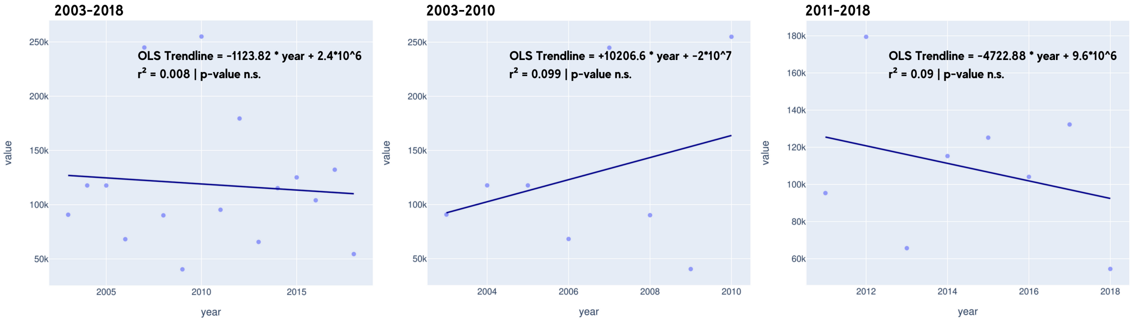
Figure 4.5 - Trend line of burned area over the time series (2003-2018; 2003-2010; 2011-2018) in the Caatinga biome.



Source: Produced by the author.

In Cerrado biome, we observed a decrease of burned area in the period 2003-2018, representing a variation of -1123.82 km²/year (p-value n.s. and $R^2=0.008$); an increase in the period 2003-2010, representing a variation of +10206.6 km^2/year (p-value n.s. and R^2=0.099); and a decrease in the period 2011-2018, representing a variation of -4722.88 km^2/year (p-value n.s. and R^2=0.09) (Figure 4.6).

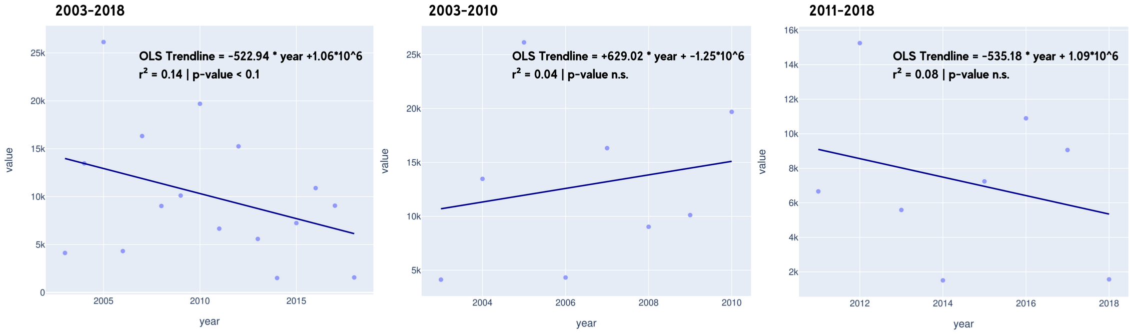
Figure 4.6 - Trend line of burned area over the time series (2003-2018; 2003-2010; 2011-2018) in the Cerrado biome.



Source: Produced by the author.

In Pantanal biome, we observed a decrease of burned area in the period 2003-2018, representing a variation of -522.94 km^2/year (p-value<0.1 and R^2=0.14); an increase in the period 2003-2010, representing a variation of +629.02 km^2/year (p-value n.s. and R^2=0.04); and a decrease in the period 2011-2018, representing a variation of -535.18 km^2/year (p-value n.s. and R^2=0.08) (Figure 4.7).

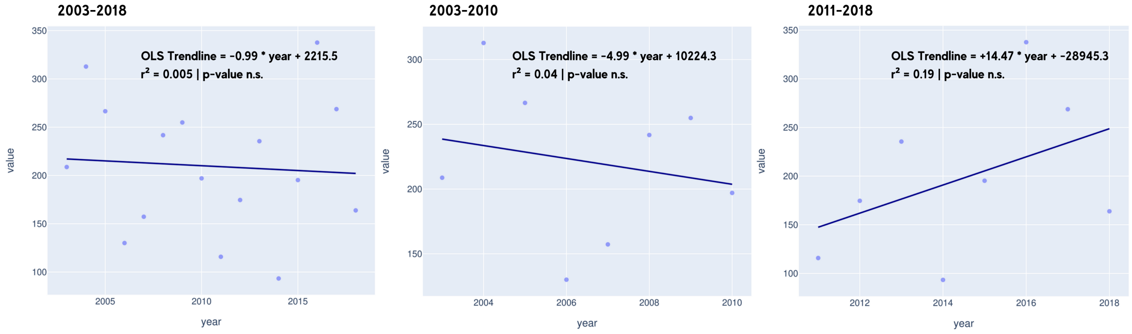
Figure 4.7 - Trend line of burned area over the time series (2003-2018; 2003-2010; 2011-2018) in the Pantanal biome.



Source: Produced by the author.

In Pampa biome, we observed a decrease of burned area in the period 2003-2018, representing a variation of -0.99 km^2/year (p-value n.s. and R^2=0.005); a decrease in the period 2003-2010, representing a variation of -4.99 km^2/year (p-value n.s. and R^2=0.04); and an increase in the period 2011-2018, representing a variation of +14.47 km^2/year (p-value n.s. and R^2=0.19) (Figure 4.8).

Figure 4.8 - Trend line of burned area over the time series (2003-2018; 2003-2010; 2011-2018) in the Pampa biome.



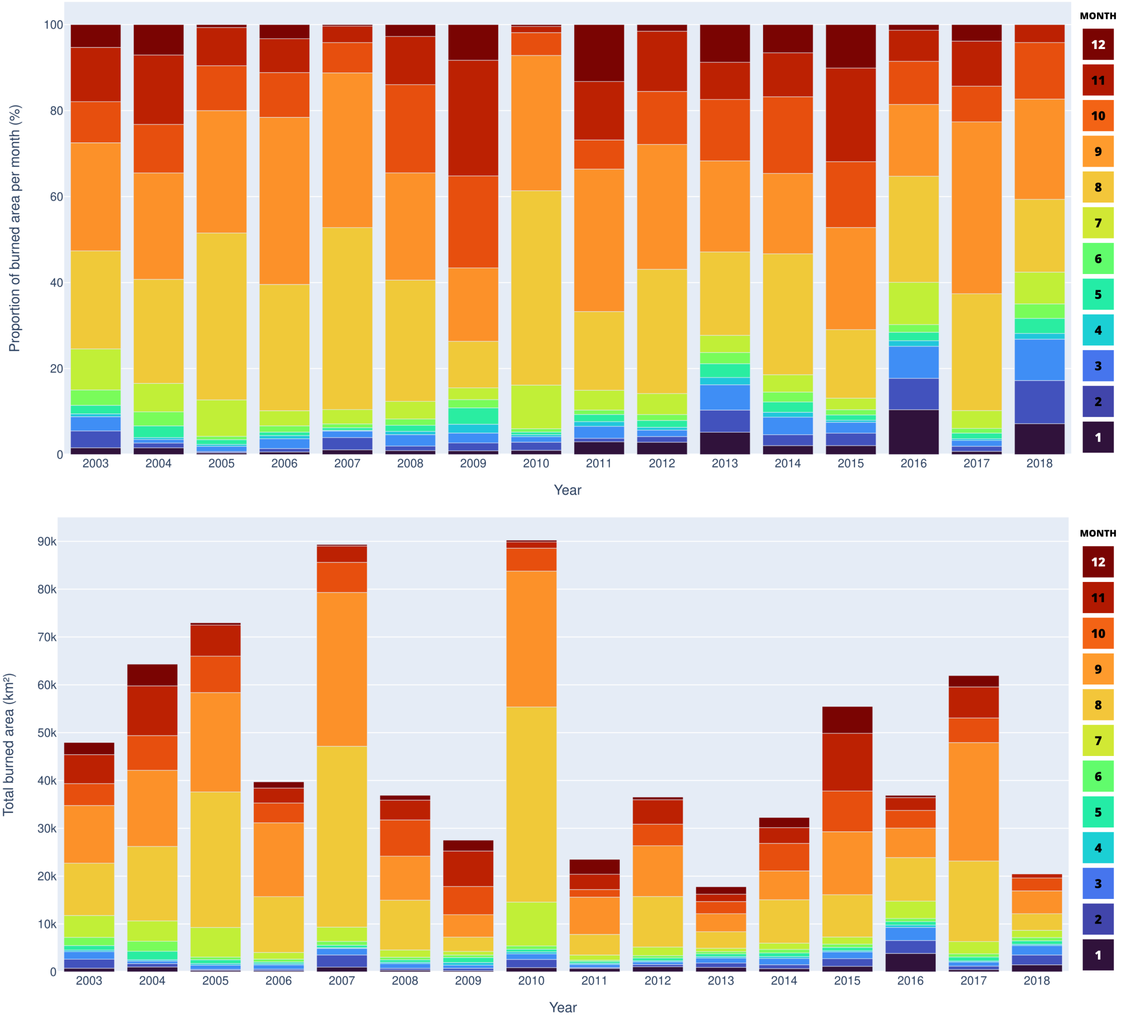
Source: Produced by the author.

4.1.1 Intra-annual patterns of fire in the Brazilian biomes

The analysis of monthly burned area patterns in the Brazilian biomes were grouped according to the division between: Fire-sensitivity, Fire-independent and, Fire-dependent. In this section we highlighted the intra-annual patterns of burned area over the time series (2003-2018) and the fire season, defined as the two months with the higher proportion of burned area in the year.

Concerning Fire-sensitivity biomes, we observed that, in Amazon, the fire season was well-marked in two months, August and September, except in 2009 (the fire season was concentrated on October and November) and 2015 (the fire season was concentrated on September and November). On average, the fire season, in Amazon, represented a proportion of 55% of the total burned area, where in 2007 August and September were responsible for a proportion of 78% of the total burned area in the year (burning 119,712.56 km^2) and, in 2018 August and September were responsible for a proportion of 40% (burning 25,981.12 km^2). In addition, August 2010 was the month that presented the greatest extent of burned area, representing a proportion of 45% (40,791 km^2) of the total burned of that year. In contrast, the month with the smaller burned area was April 2007 (totaling 156 km² burned and representing only 0.17% of the total burned of that year) (Figure 4.9).

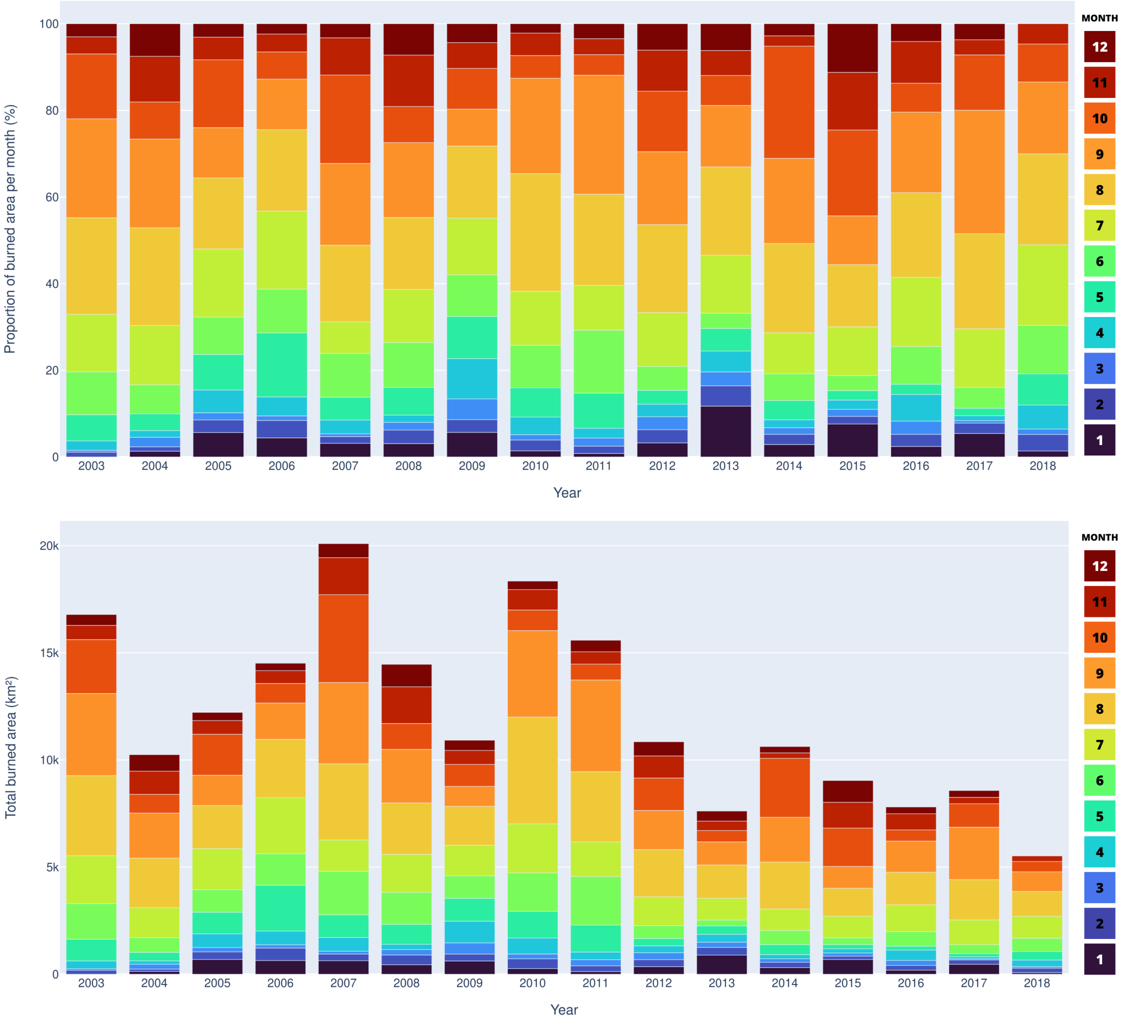
Figure 4.9 - Intra-annual patterns of fires in Amazon biome. On the top we observed the proportion of burned area per month and year and at bottom we observed the total burned area (km²) over the time series. Each month was represented by one color, where 1 represented January month and 12 represented December month.



Source: Produced by the author.

In Atlantic Forest biome, the fire season was concentrated, mainly, in August and September. Regarding the exceptions, in 2005, 2006, 2009 and 2018 the fire season was concentrated in July and August; in 2014 and 2015 was concentrated in August and September, and 2007 in September and October. On average, the fire season, in Atlantic Forest, represented a proportion of 40% of the total burned area, where in 2017 August and September were responsible for a proportion of 50% of the total burned area in the year (burning 4,325 km²) and, in 2009 July and August were responsible for a proportion of 30% (burning 3,249.44 km²). In addition, August 2010 was the month that presented the greatest extent of burned area, burning a proportion of 27% (4,982 km²). In contrast, the month with the smaller burned area was March 2017 (totaling 49 km² burned and representing only 0.57% of the total burned of the year) (Figure 4.10).

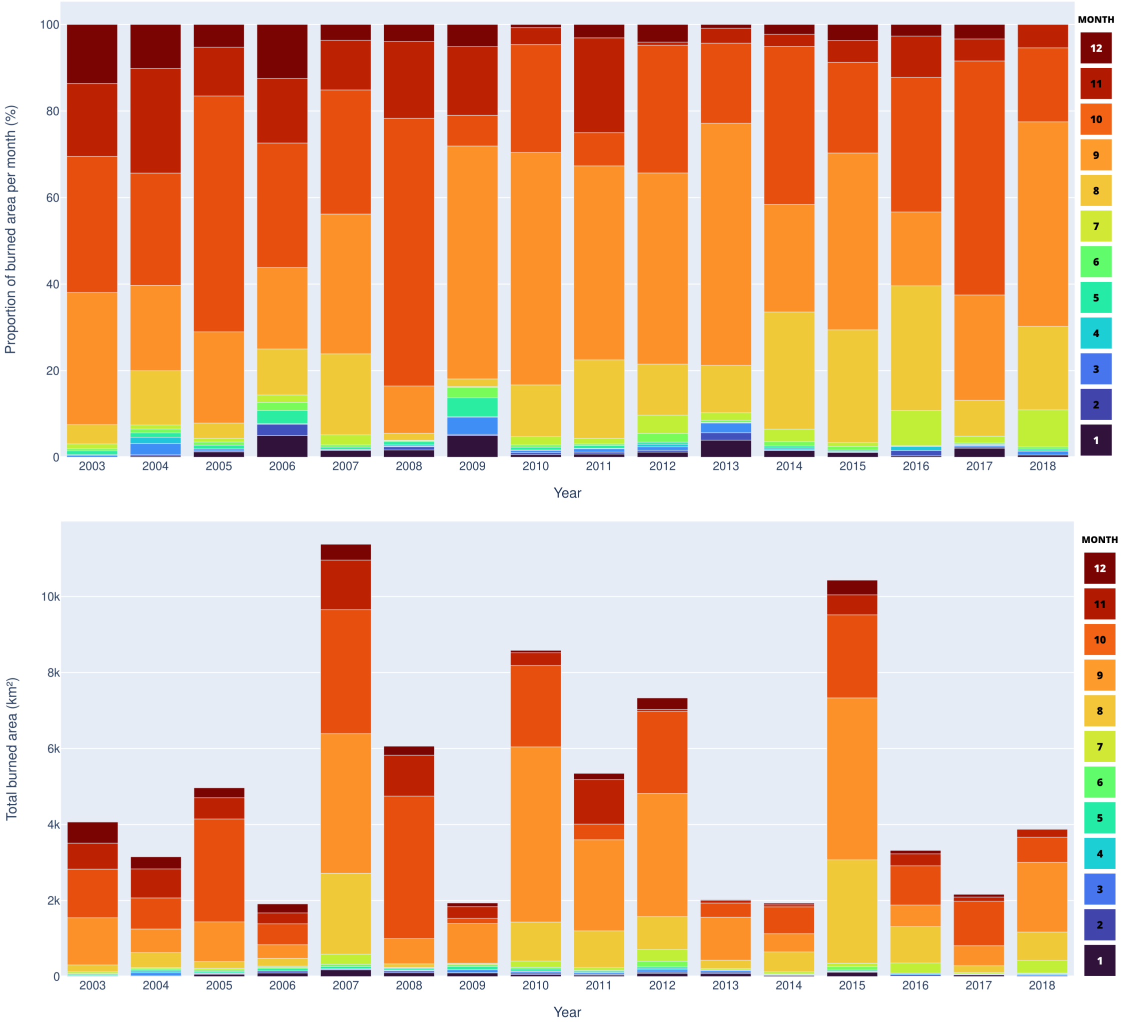
Figure 4.10 - Intra-annual patterns of fires in Atlantic Forest biome. On the top we observed the proportion of burned area per month and year and at bottom we observed the total burned area (km²) over the time series. Each month was represented by one color, where 1 represented January month and 12 represented December month.



Source: Produced by the author.

In Caatinga, a Fire-independent biome, the fire season was well-marked in two months, September and October, with some exceptions: in 2004 the fire season was concentrated in October and November; in 2009 and 2011 was concentrated in September and November; 2014 and 2016 in August and October; 2015 and 2018 in August and September. On average, the fire season, in Caatinga, represented a proportion of 67% of the total burned area, where in 2010 September and October were responsible for a proportion of 79% of the total burned area in the year (burning 6,749.77 km^2) and, in 2006 September and October were responsible for a proportion of 48% (burning 909.47 km^2). In addition, September 2010 was the month that presented the greatest extent of burned area, burning a proportion of 54% (4,609.22km^2). In contrast, the months with the smaller burned area were February 2014 and April 2013 (totaling 0.63 km^2 burned and representing only 0.17% of the total burned of the year, in both months) (Figure 4.11).

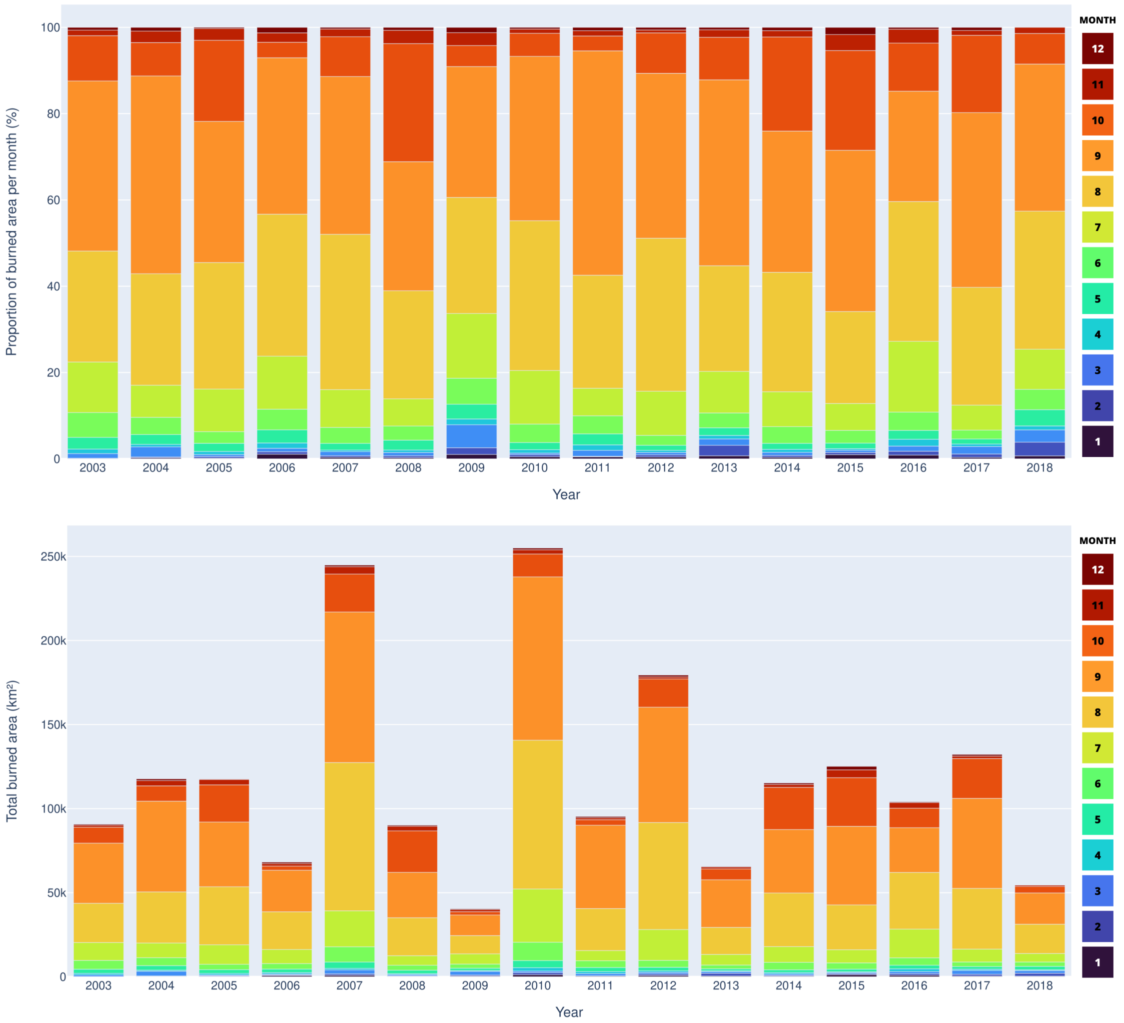
Figure 4.11 - Intra-annual patterns of fires in Caatinga biome. On the top we observed the proportion of burned area per month and year and at bottom we observed the total burned area (km^2) over the time series. Each month was represented by one color, where 1 represented January month and 12 represented December month.



Source: Produced by the author.

Finally, regarding Fire-dependent biomes, in Cerrado, the fire season was well-marked in two months, August and September, except in 2008 and 2015, where the fire season is concentrated in September and October. On average, the fire season, in Cerrado, represented a proportion of 66% of the total burned area, where in 2011 August and September were responsible for a proportion of 78% of the total burned area in the year (burning 74,540.6 km^2) and, in 2009 August and September were responsible for a proportion of 57% (burning 23,129.21 km^2). In addition, September 2010 was the month that presented the greatest extent of burned area, burning a proportion of 38% (97,119 km²). In contrast, the month with the smaller burned area was February 2011 (totaling 117 km² burned and representing only 0.12% of the total burned of the year) (Figure 4.12).

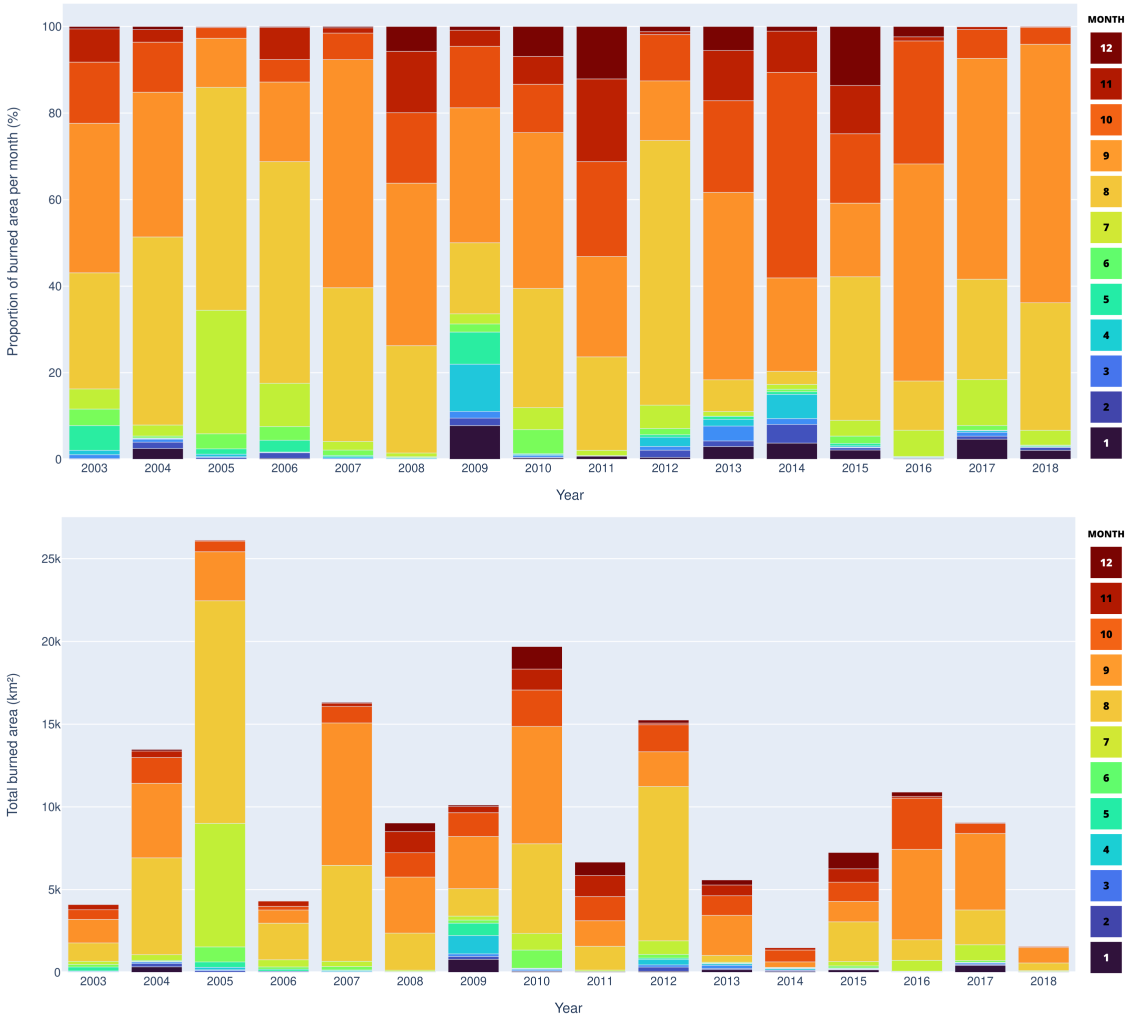
Figure 4.12 - Intra-annual patterns of fires in Cerrado biome. On the top we observed the proportion of burned area per month and year and at bottom we observed the total burned area (km²) over the time series. Each month was represented by one color, where 1 represented January month and 12 represented December month.



Source: Produced by the author.

In Pantanal, the fire season was concentrated, mainly, in August and September, except in 2012, 2015, 2017 and 2018, in which the fire season was concentrated in September and October and, 2005, in which the fire season was concentrated in July and August. On average, the fire season, in Pantanal, represented a proportion of 68% of the total burned area, where in 2007 August and September were responsible for a proportion of 88% of the total burned area in the year (burning 14,400.93 km^2) and, in 2011 September and October were responsible for a proportion of 45% (burning 3,005.34 km^2). In addition, August 2005 was the month that presented the greatest extent of burned area, burning a proportion of 51% (13,457.9 km^2). In contrast, the month with the smaller burned area was January 2007 (totaling 0.42 km^2 burned and representing only 0.002% of the total burned of the year) (Figure 4.13).

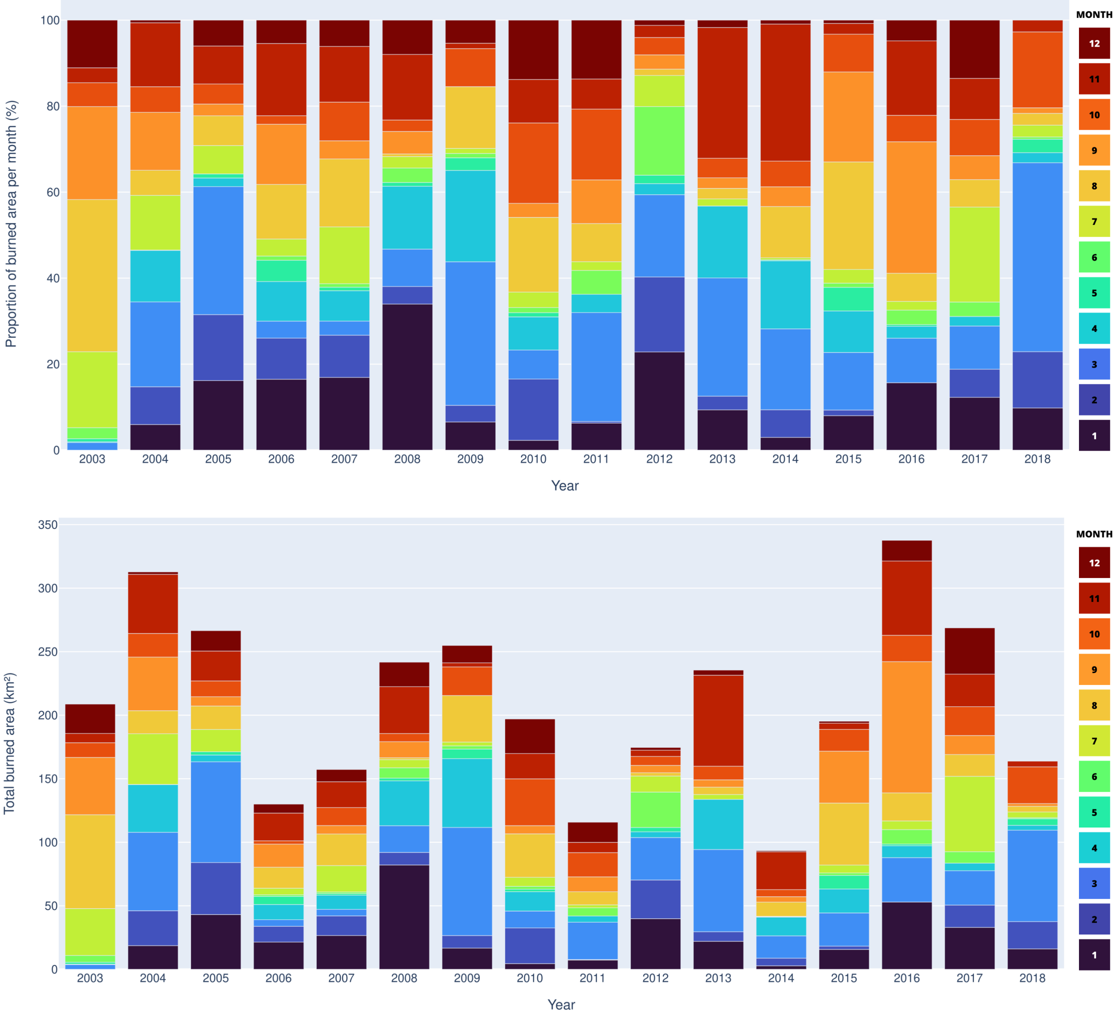
Figure 4.13 - Intra-annual patterns of fires in Pantanal biome. On the top we observed the proportion of burned area per month and year and at bottom we observed the total burned area (km^2) over the time series. Each month was represented by one color, where 1 represented January month and 12 represented December month.



Source: Produced by the author.

In Pampa, the fire season presented a heterogeneous configuration across the time, therefore, it was not possible to extract some general pattern. On average, the fire season, in Pampa, represented a proportion of 45% of the total burned area, where in 2018 March and October were responsible for a proportion of 62% of the total burned area in the year (burning 100.88 km^2) and, in 2007 January and August were responsible for a proportion of 33% (burning 51.46 km^2). In addition, September 2016 was the month that presented the greatest extent of burned area, burning a proportion of 31% (103.29 km^2). In contrast, the months with the smaller burned area were May 2014 and June 2004 (totaling 0.21 km^2 burned and representing only 0.22% of the total burned of the year, in both months)) (Figure 4.14).

Figure 4.14 - Intra-annual patterns of fires in Pampa biome. On the top we observed the proportion of burned area per month and year and at bottom we observed the total burned area (km^2) over the time series. Each month was represented by one color, where 1 represented January month and 12 represented December month.



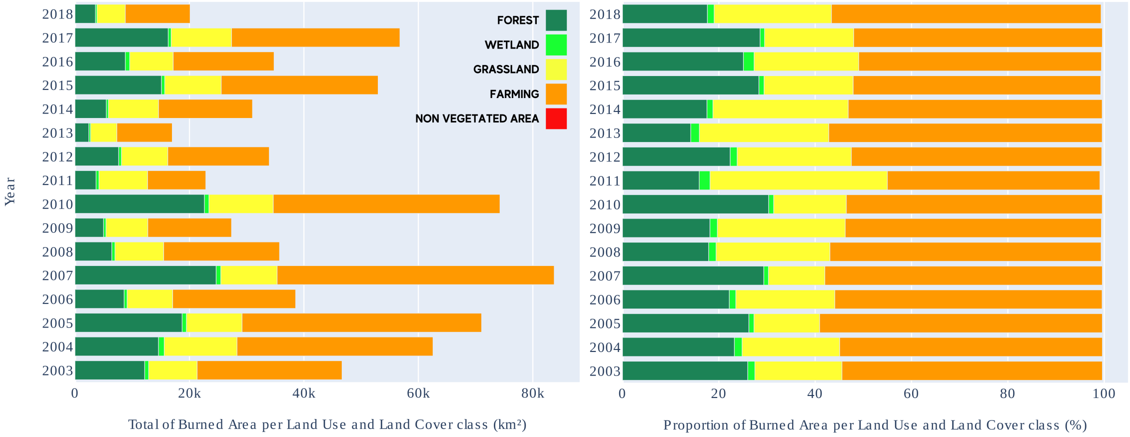
Source: Produced by the author.

4.2 What burns in the Brazilian biomes

In this topic we presented a summary of what burns in the Brazilian biomes, according to the land use and land cover aggregated classes, divided among Forest, Wetland, Savanna and Grassland (natural classes), Farming and Non Vegetated Area (anthropic classes). Also, we highlighted how the fire impacted natural vegetation. This analysis of what burns in the Brazilian biomes were grouped according to the division between: Fire-sensitivity, Fire-independent and, Fire-dependent.

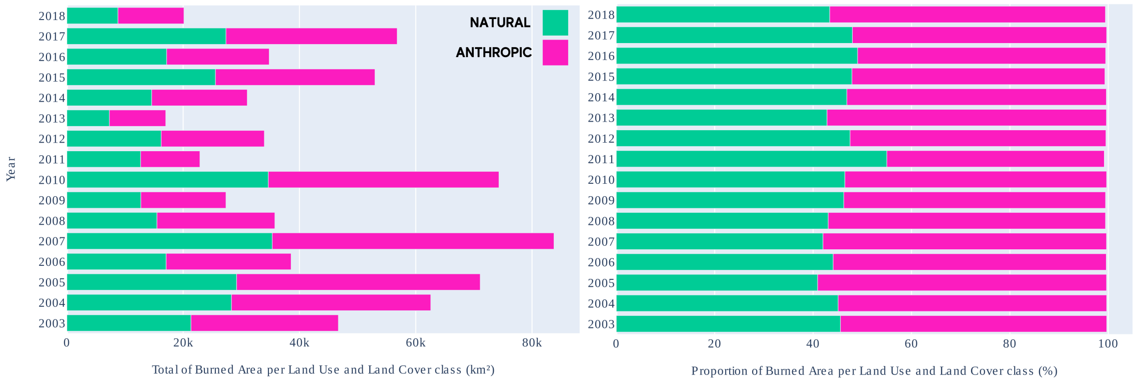
Regarding Fire-sensitivity biomes, in Amazon (Figure 4.15), the largest land cover class affected by fires was Farming, representing an average of 24,106.21 km² pasture and agricultural lands burned (54%). 2013 reached the minimum area of pasture and agricultural lands burned, totaling 9,689.15 km² (57% of the total burned in that year) and 2007 reached the maximum area, totaling 48,427.39 km² (58% of the total burned in that year). The second land cover more affected by fire events was Forest, representing an average of 10,982.01 km² Forest burned (23%). Also, 2013 reached the minimum area of burned Forest, totaling 2,418.48 km² (14% of the total burned in that year) and 2007 reached the maximum area, totaling 24,663.62 km² (29% of the total burned in that year). In summary, a mean of 20,233 km² (46%) natural vegetation and 24,128 km² (54%) anthropic land use were burned (Figure 4.16).

Figure 4.15 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Amazon biome. The aggregated classes were divided between Forest, Wetland, Savanna and Grassland, Farming and Non Vegetated Area.



Source: Produced by the author.

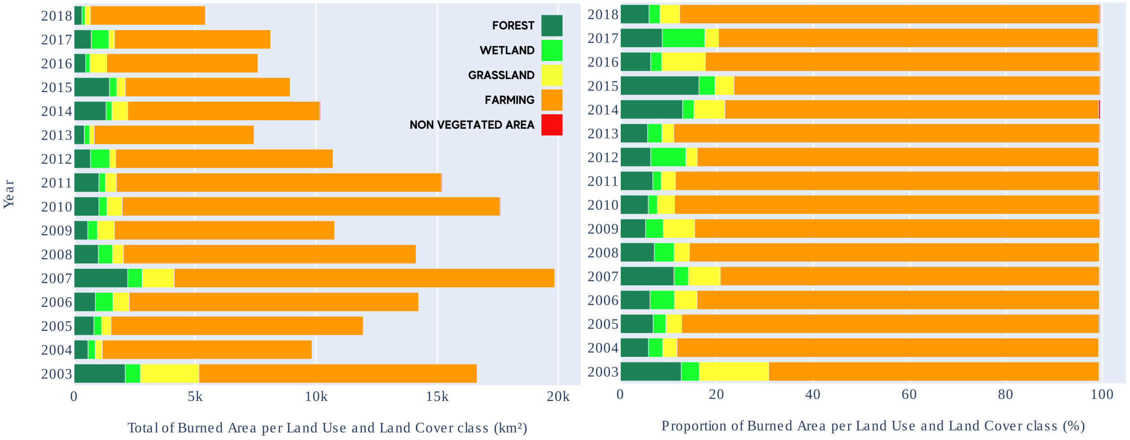
Figure 4.16 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Amazon biome. The classes were grouped into natural vegetation (Forest, Wetland, Savanna and Grassland) and anthropic classes (Farming and Non Vegetated Area).



Source: Produced by the author.

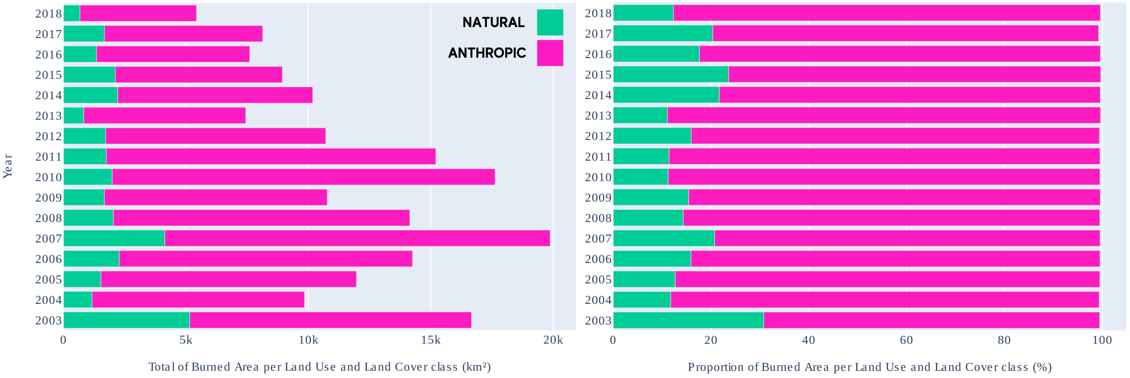
In Atlantic Forest (Figure 4.17), the largest land cover class affected by fires, also, was Farming, representing an average of 9,767.28 km² pasture and agricultural lands burned (83%). 2018 reached the minimum area of pasture and agricultural lands burned, totaling 4,752.61 km² (87% of the total burned in that year) and 2007 reached the maximum area, totaling 15,715.37 km² (79% of the total burned in that year). The second land cover more affected by fire events was Forest, representing an average of 974.75 km² Forest burned (8%). Also, 2018 reached the minimum area of burned Forest, totaling 323.59 km² (6% of the total burned in that year) and 2007 reached the maximum area, totaling 2,214.49 km² (11% of the total burned in that year). In summary, a mean of 2,016 km² (17%) natural vegetation and 9,786 km² (83%) anthropic land use were burned (Figure 4.18).

Figure 4.17 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Atlantic Forest biome. The aggregated classes were divided between Forest, Wetland, Savanna and Grassland, Farming and Non Vegetated Area.



Source: Produced by the author.

Figure 4.18 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Atlantic Forest biome. The classes were grouped into natural vegetation (Forest, Wetland, Savanna and Grassland) and anthropic classes (Farming and Non Vegetated Area).

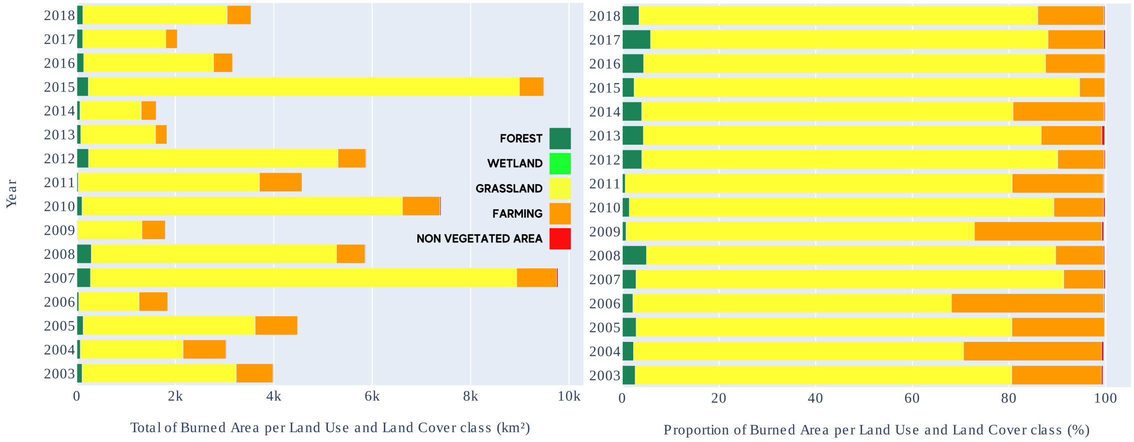


Source: Produced by the author.

In Caatinga (Figure 4.19), a Fire-independent biome, the largest land cover class affected by fires were Grassland and Savanna, representing an average of 3,681.58 km² Grassland and Savanna burned (80%). 2006 reached the minimum area of burned

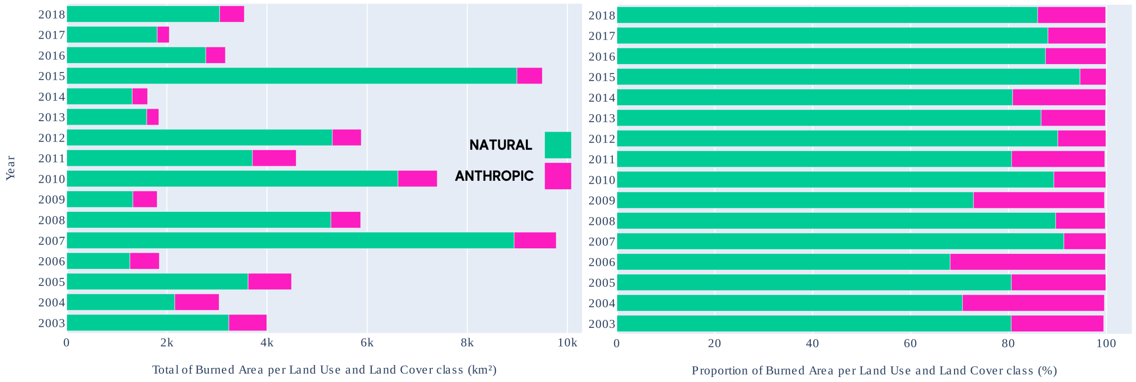
Grassland and Savanna, totaling 1220.99 km² (66% of the total burned in that year) and 2015 reached the maximum area, totaling 8,758.31 km² (92% of the total burned in that year). The second land cover more affected by fire events was Farming, representing an average of 579.27 km² pasture and agricultural lands burned (16%). 2013 reached the minimum area of burned Forest, totaling 231.05 km² (13% of the total burned in that year) and 2004 reached the maximum area, totaling 872.53 km² (29% of the total burned in that year). In summary, a mean of 3,811 km² (84%) natural vegetation and 591 km² (16%) anthropic land use were burned (Figure 4.20).

Figure 4.19 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Caatinga biome. The aggregated classes were divided between Forest, Wetland, Savanna and Grassland, Farming and Non Vegetated Area.



Source: Produced by the author.

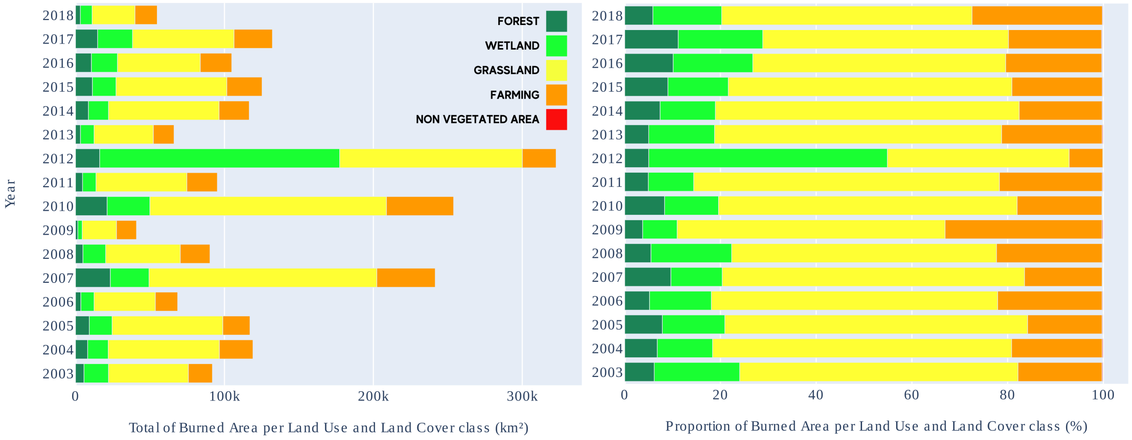
Figure 4.20 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Caatinga biome. The classes were grouped into natural vegetation (Forest, Wetland, Savanna and Grassland) and anthropic classes (Farming and Non Vegetated Area).



Source: Produced by the author.

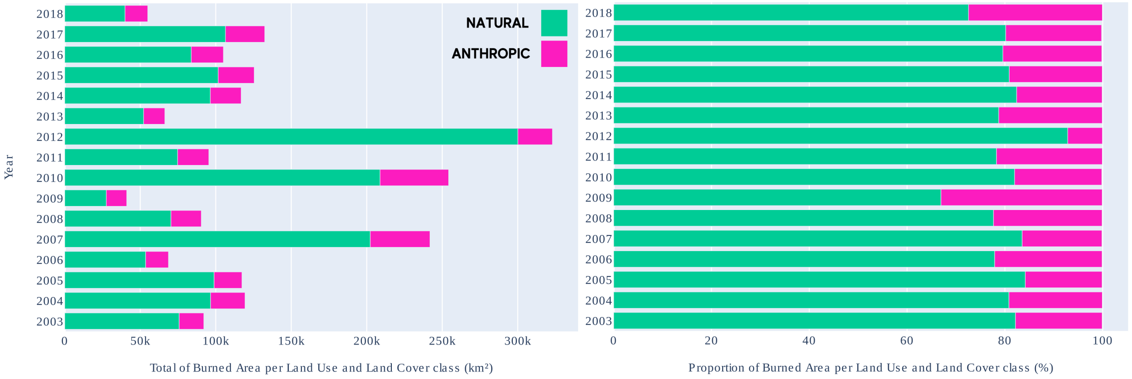
Concerning Fire-dependent biomes, in Cerrado (Figure 4.21) the largest land cover class affected by fires were Grassland and Savanna, representing an average of 72,076.58 km² Grassland and Savanna burned (58%). 2009 reached the minimum area of burned Grassland and Savanna, totaling 23,001.79 km² (56% of the total burned in that year) and 2010 reached the maximum area, totaling 158,848.27 km² (62% of the total burned in that year). The second land cover more affected by fire events was Wetland, representing an average of 24,064.01 km² Wetland burned (16%). 2009 reached the minimum area of burned Wetland, totaling 2,963.31 km² (7% of the total burned in that year) and 2010 reached the maximum area, totaling 28,726.48 km² (11% of the total burned in that year). In summary, a mean of 105,563 km² (80%) natural vegetation and 22,235 km² (20%) anthropic land use were burned (Figure 4.22).

Figure 4.21 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Cerrado biome. The classes were grouped into natural vegetation (Forest, Wetland, Savanna and Grassland) and anthropic classes (Farming and Non Vegetated Area).



Source: Produced by the author.

Figure 4.22 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Cerrado biome. The classes were grouped into natural vegetation (Forest, Wetland, Savanna and Grassland) and anthropic classes (Farming and Non Vegetated Area).

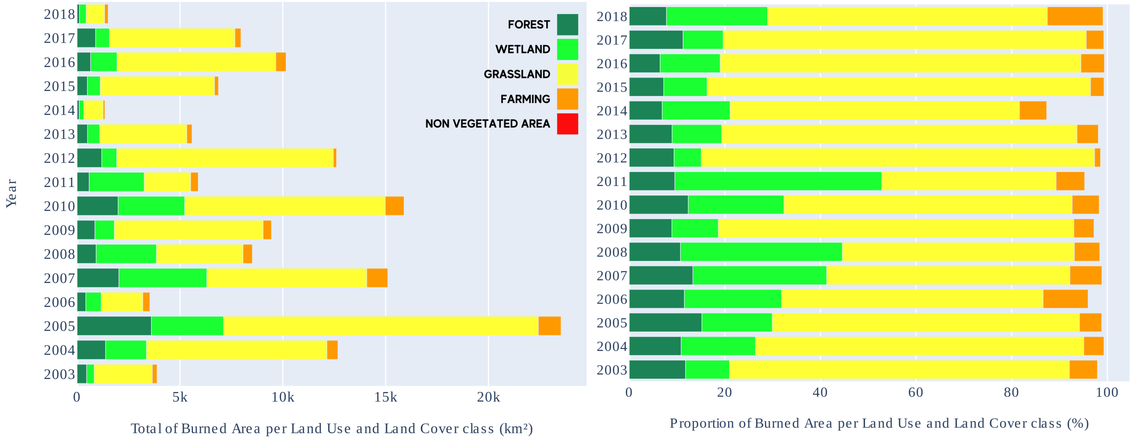


Source: Produced by the author.

In Pantanal (Figure 4.23) the largest land cover class affected by fires were Grassland and Savanna, representing an average of 6,004.06 km² Grassland and Savanna burned (65%). 2018 reached the minimum area of burned Grassland and Savanna, totaling

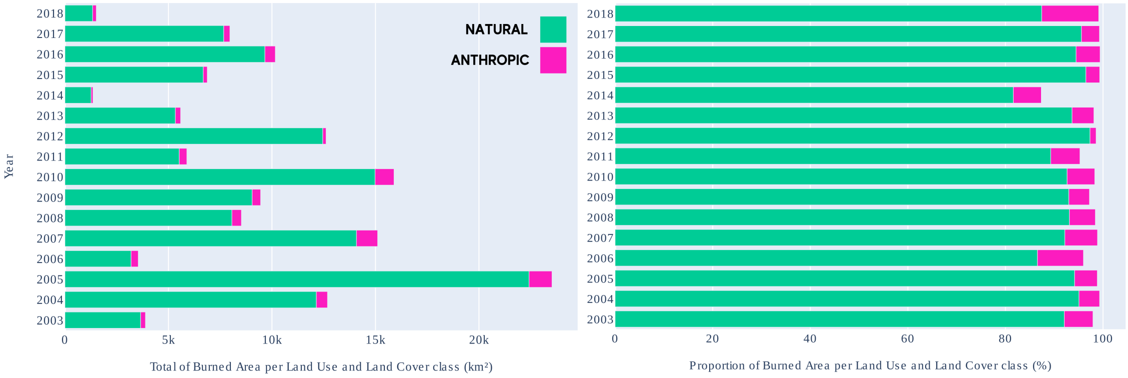
894.92 km² (59% of the total burned in that year) and 2005 reached the maximum area, totaling 15,293.77 km² (64% of the total burned in that year). The second land cover more affected by fire events was Wetland, representing an average of 1,570.29 km² Wetland burned (17%). 2014 reached the minimum area of burned Wetland, totaling 3507.26 km² (14% of the total burned in that year) and 2005 reached the maximum area, totaling 28,726.48 km² (15% of the total burned in that year). In summary, a mean of 8,594 km² (92%) natural vegetation and 443 km² (8%) anthropic land use were burned (Figure 4.24).

Figure 4.23 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Pantanal biome. The classes were grouped into natural vegetation (Forest, Wetland, Savanna and Grassland) and anthropic classes (Farming and Non Vegetated Area).



Source: Produced by the author.

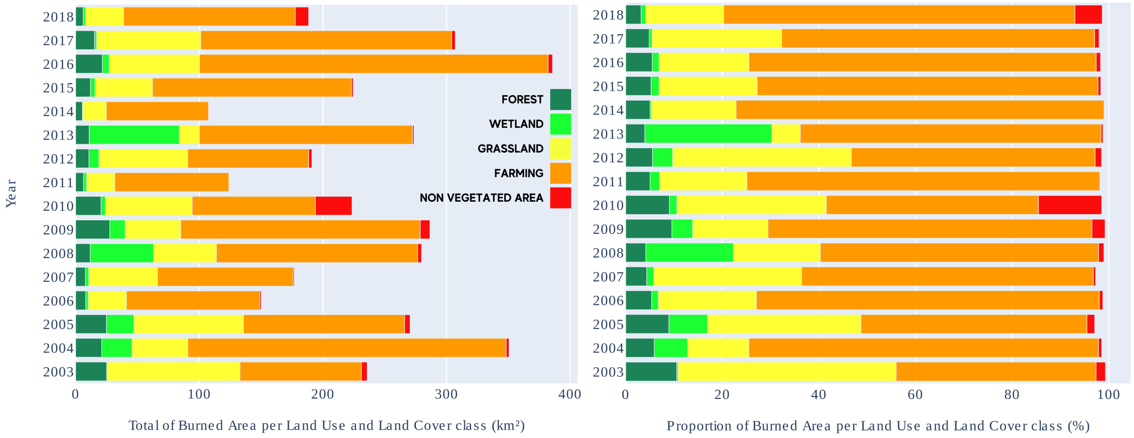
Figure 4.24 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Pantanal biome. The classes were grouped into natural vegetation (Forest, Wetland, Savanna and Grassland) and anthropic classes (Farming and Non Vegetated Area).



Source: Produced by the author.

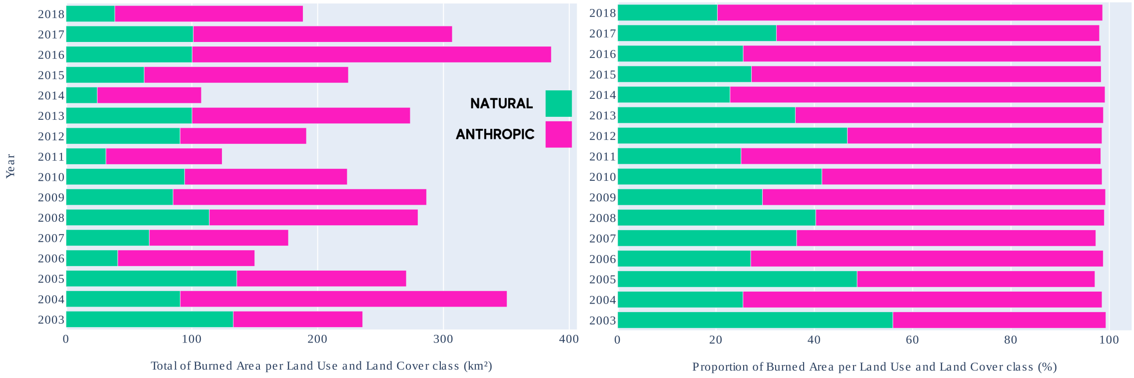
Finally, in Pampa (Figure 4.25) the largest land cover class affected by fires was Farming, representing an average of 149.36 km² pasture and agricultural lands burned (63%). 2014 reached the minimum area of pasture and agricultural lands burned, totaling 82.68 km² (76% of the total burned in that year) and 2016 reached the maximum area, totaling 281.93 km² (72% of the total burned in that year) of the total burned in that year. The second land cover more affected by fire events were Grassland and Savanna, representing an average of 53.59 km² Wetland burned (23%). 2013 reached the minimum area of burned Grassland and Savanna, totaling 16.28 km² (6% of the total burned in that year) and 2003 reached the maximum area, totaling 107.13 km² (45% of the total burned in that year). In summary, a mean of 154 km² (35%) natural vegetation and 82 km² (65%) anthropic land use were burned (Figure 4.26).

Figure 4.25 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Pampa biome. The classes were grouped into natural vegetation (Forest, Wetland, Savanna and Grassland) and anthropic classes (Farming and Non Vegetated Area).



Source: Produced by the author.

Figure 4.26 - Total burned area (km²) per Land Use and Land Cover class and Proportion of burned area (%) per Land Use and Land Cover class over the time series (2003-2018) in the Pampa biome. The classes were grouped into natural vegetation (Forest, Wetland, Savanna and Grassland) and anthropic classes (Farming and Non Vegetated Area).



Source: Produced by the author.

4.3 Spatial patterns of fires and landscape metrics in the Brazilian Biomes

In this section, we compiled the results obtained from the Mann-Kendall trend analysis (Tau Coefficient and Sen Slope), including burned area and all landscape metrics (Percentage of Forest - PLAND; Forest Edge Density - ED; Number of Forest Patches - NP; Total Forest Core Area - TCA).

4.3.1 Spatial patterns of fires in the Brazilian Biomes

We estimated a total of 2904 cells with significant negative values (60% cells < 0) and 1959 cells with significant positive values (40% cells > 0) in the Brazil territory (Figure 4.27 and Figure 4.28). Regarding the variation rate of burned area (sen slope - km^2/year), we observed 2677 cells (56%) with a rate of change of up to -5 km/year (p-value < 0.05 and Kendall's TAU < 0). In contrast, we observed 1730 cells (36%) with a rate of change of up to +5 km/year (p-value $< 0,05$ and Kendall's TAU > 0). Moreover, we visualized 140 cells (3%) with values higher than +5 km^2/year (p-value < 0.05 and Kendall's TAU > 0), reaching the maximum of +205.71 km^2/year, and 211 cells (5%) with values lower than -5 km^2/year (p-value < 0.05 and Kendall's TAU < 0), reaching the minimum of -48.76 km^2/year. These two both extremes values were located in the Pantanal biome, where the maximum positive trend (+205.71 km^2/year) and the minimum negative trend (-48.76 km^2/year) were located in the south-west portion of Mato Grosso do Sul (Figure 4.27 and Figure 4.28).

Regarding Fire-sensitivity biomes, in Amazon, maximum positive trends of burned area (sen slope $> +15$ km^2/year) were concentrated mainly in the southeast portion of Pará state and in the east portion of Mato Grosso. On the other hand, minimum negative trends of burned area (sen slope < -15 km^2/year) were concentrated also in the southeast portion of Pará state, south of Rondônia and south-west portion of Mato Grosso. In addition, was observed a cell with negative extreme value of sen slope (-43.83 km^2/year) in the Roraima state (Figure 4.27 and Figure 4.28).

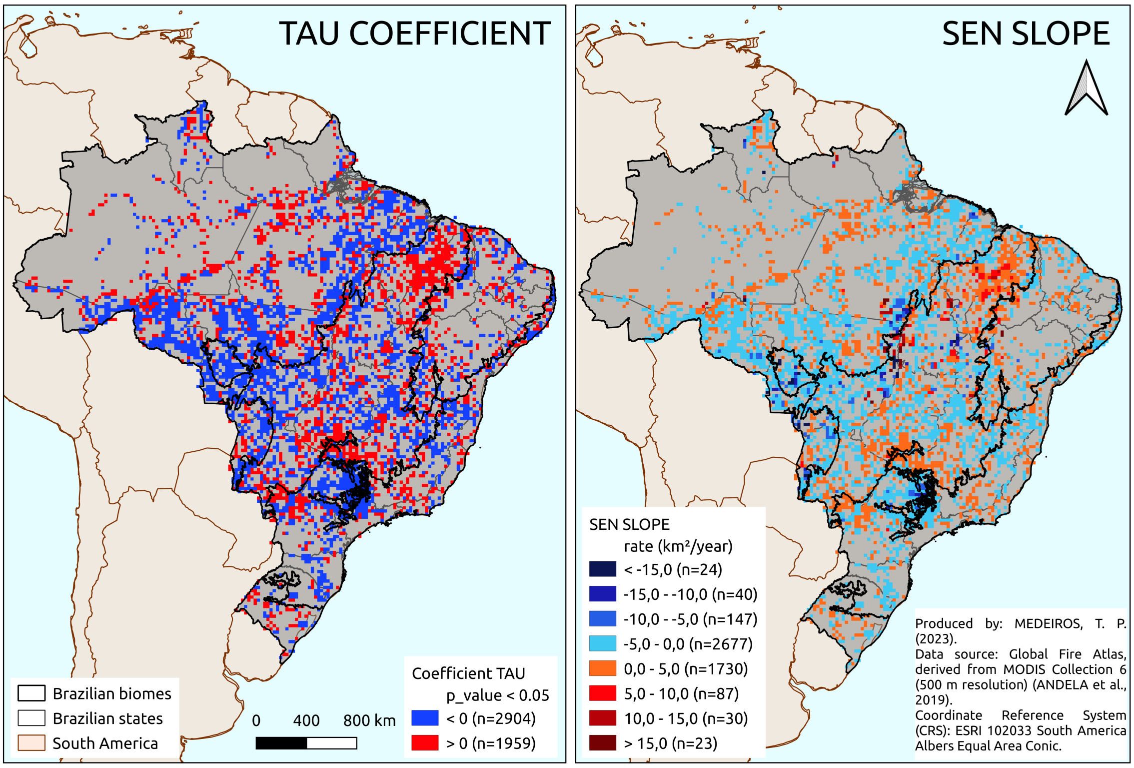
In Atlantic Forest biome, the variation in the burned area rate was more subtle when compared to Amazon biome (ranging from -5 to +5 km^2/year). We observed a patch of negative trends (sen slope < -5 km^2/year) in the center-northern portion of São Paulo state. In addition, a cell with a positive value of sen slope (+5.57 km^2/year) was observed in Paraná state (Figure 4.27 and Figure 4.28).

In Caatinga, a Fire-independent biome, we observed a patch of more expressive positive trends (sen slope $>$ +5 km^2/year) in the west tip Piauí and Bahia states. In the rest of Caatinga territory the variation in the burned area rate was more subtle, not reaching high values (ranging from -5 to +5 km^2/year) (Figure 4.27 and Figure 4.28).

In the case of Fire-dependent biomes, in Cerrado was observed high variation in the rate of burned area, both for positive and negative trends. Positive trends of burned area (sen slope $>$ +5 km^2/year) were located, mainly, in the east portion of Maranhão, north and south-west of Tocantins. Then, negative trends of burned area (sen slope $<$ -5 km^2/year) were located, mainly, in the southeast portion of Tocantins and south tip of Mato Grosso, highlighting a patch in the south-west (Figure 4.27 and Figure 4.28).

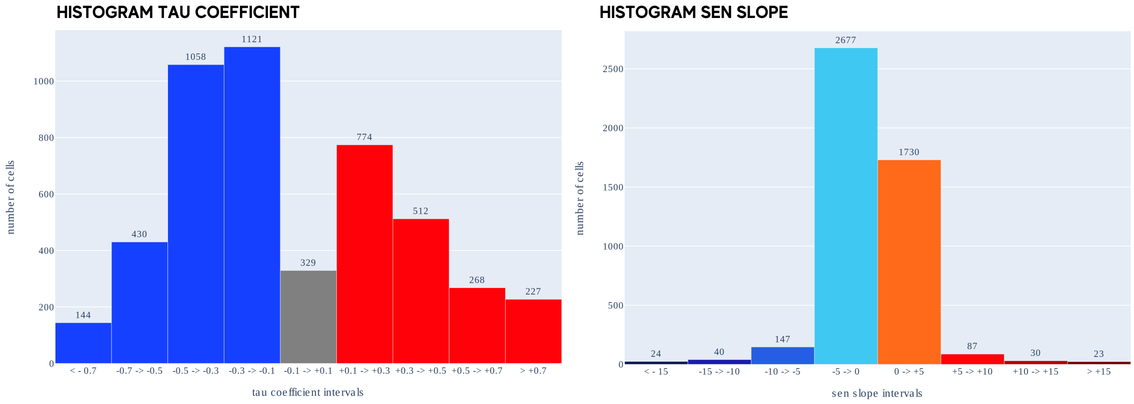
Finally, in Pampa biome, the variation in the burned area rate, also, was more subtle, not reaching high values (ranging from -5 to +5 km^2/year) (Figure 4.27 and Figure 4.28).

Figure 4.27 - Spatial trend of burned area in Brazil over the time series (2003-2018). Tau Coefficient - ranging from +1 to -1 (left) and Sen Slope - km²/year (right).



Source: Produced by the author.

Figure 4.28 - Histogram of Brazil Tau Coefficient (left) and Sen Slope (right).



Source: Produced by the author.

4.3.2 Spatial patterns of landscape metrics in the Brazilian Biomes

In this section, we analyzed the spatial patterns of landscape metrics in each Brazilian biome. Each figure represented the landscape patterns in each biome, summarized in four metrics: Percentage of Forest (PLAND), Forest Edge Density (ED); Number of Forest Patches (NP) and Total Forest Core Area (TCA).

Regarding the landscape patterns in Fire-sensitivity biome, in Amazon, we observed positive trends of Forest Edge Density (ED, sen slope > +0.5 m/ha/year) and Number of Forest Patches (NP, sen slope > +20 patches/year) in the east side of Pará state; east of Acré; along the Amazon River and close to highways; west of Mato Grosso and north portion of Rondônia (Figure 4.29). According to Table 4.3 we identified 7% of cells with positive trend of Forest Edge Density higher than +0.5 m/ha/year and 5.5% of cells with positive trend of Number of Forest Patches higher than +20 patches/year.

Nonetheless, we observed negative trends of ED (sen slope < -0.5 m/ha/year) and NP (sen slope < -40 patches/year) in the extreme northeast portion of Pará state and in the west side of Maranhão (Figure 4.29). Also, according to Table 4.3 we identified 2% of cells with negative trend of Forest Edge Density lower than -0.5 m/ha/year and 0.5% of cells with negative trend of Number of Forest Patches lower than -20 patches/year.

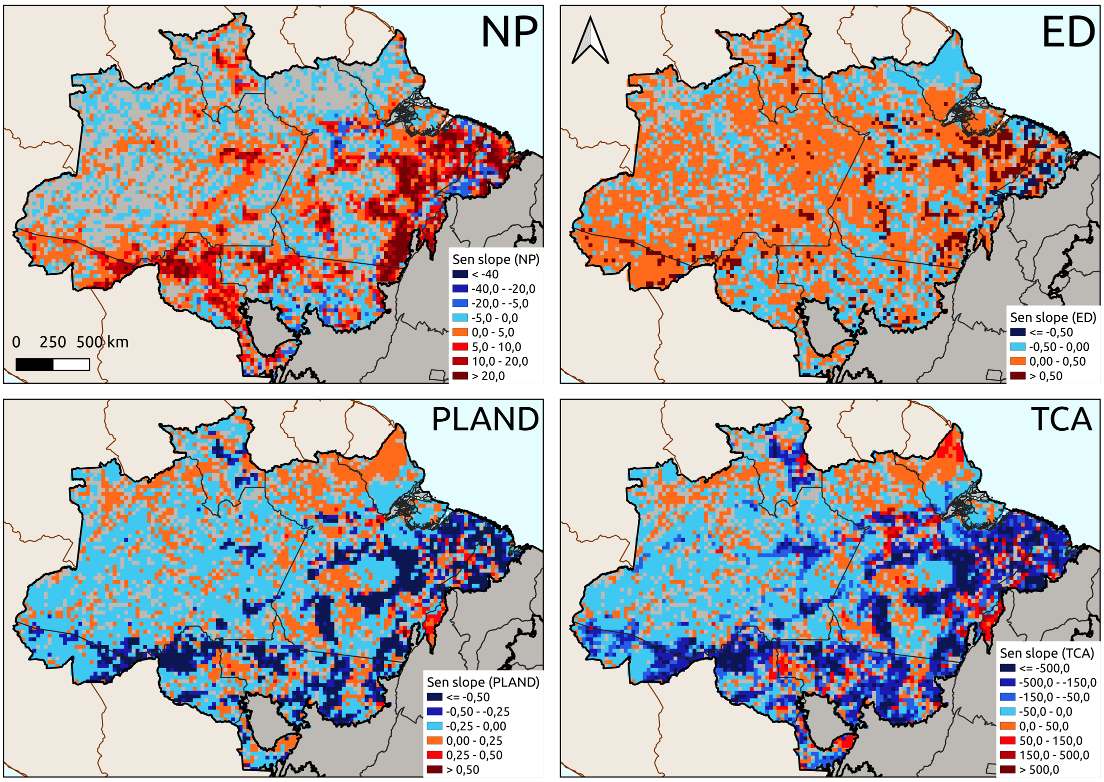
In the case of Percentage of Forest (PLAND) and Total Forest Core Area (TCA), we analyzed positive trends of PLAND (sen slope > +0.5 %/year) and TCA (sen slope > +500 ha/year) in the northwest tip of Tocantins and northwest portion of Pará state (Figure 4.29). In addition, we identified only 0.5% of cells with positive trend of Percentage of Forest higher than +0.5 %/year and only 0.03% of cells with positive trend of Total Forest Core Area higher than +500 ha/year (Table 4.3).

In contrast, we observed negative trends of PLAND (sen slope < -0.5 %/year) and TCA (sen slope < -500 ha/year) in the northeast and southwest portion of Pará state; east of Acré; north portion of Mato Grosso and Rondônia (Figure 4.29). Also, we identified 17% of cells with negative trend of Percentage of Forest lower than -0.5 %/year and 8% of cells with negative trend of Total Forest Core Area lower than -500 ha/year (Table 4.3).

Table 4.3 - Proportion of cells per class for each landscape metric in the Amazon biome. Forest Edge Density (ED); Number of Forest Patches (NP); Total Forest Core Area (TCA) and Percentage of Forest (PLAND).

NP	**TCA**	**PLAND**	**ED**
<-40 (0.1%)	<-500 (8%)	<-0.5 (17%)	<-0.5 (2%)
-40 to -20 (0.4%)	-500 to -150 (14%)	-0.5 to -0.25 (8%)	-0.5 to 0 (29%)
-20 to -5 (4%)	-150 to -50 (12%)	-0.25 to 0 (47%)	0 to +0.5 (62%)
-5 to 0 (18%)	-50 to 0 (39%)	0 to +0.25 (26%)	>+0.5 (7%)
0 to +5 (54%)	0 to +50 (21%)	+0.25 to +0.5 (1.5%)	
+5 to +10 (9%)	+50 to +150 (4%)	>+0.5 (0.5%)	
+10 to +20 (9%)	+150 to +500 (2%)		
>+20 (5.5%)	>+500 (0.03%)		

Figure 4.29 - Spatial trend of landscape metrics (Percentage of Forest - PLAND; Forest Edge Density - ED; Number of Forest Patches - NP; Total Forest Core Area - TCA) in the Amazon biome over the time series (2003-2018). Sen Slope - PLAND = %/year; ED = m/ha/year; NP = patches/year; TCA = ha/year.



Source: Produced by the author.

Concerning Atlantic Forest biome, we observed positive trends of Forest Edge Density (ED, sen slope $>$ +0.5 m/ha/year) and Number of Forest Patches (NP, sen slope $>$ +20 patches/year), mainly, in the southeast portion of Minas Gerais state; west side of São Paulo, extending until north of Paraná and in all extension of Santa Catarina (Figure 4.30). According to Table 4.4 we identified 27% of cells with positive trend of Forest Edge Density higher than +0.5 m/ha/year and 10% of cells with positive trend of Number of Forest Patches higher than +20 patches/year.

Nonetheless, we observed negative trends of ED (sen slope $<$ -0.5 m/ha/year) and NP (sen slope $<$ -20 patches/year), mainly, in the south portion of Atlantic Forest biome, with cells spaced across the territory (Figure 4.30). Also, according to Table 4.4 we identified only 0.2% of cells with negative trend of Forest Edge Density lower than -0.5 m/ha/year and only 0.5% of cells with negative trend of Number of Forest Patches lower than -20 patches/year.

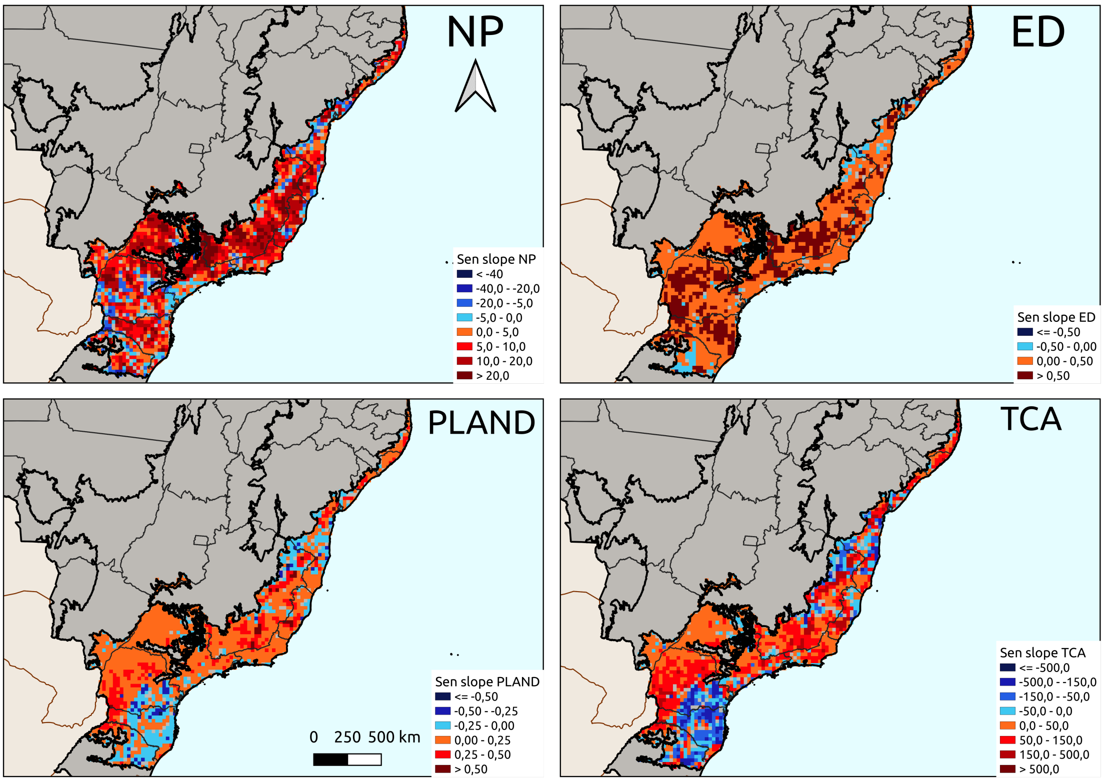
In the case of Percentage of Forest (PLAND) and Total Forest Core Area (TCA), we analyzed positive trends of PLAND (sen slope $>$ +0.25 %/year) and TCA (sen slope $>$ +150 ha/year) in the southeast portion of Minas Gerais state; south of Espírito Santo and west side of Paraná (Figure 4.30). In addition, we identified 16% of cells with positive trend of Percentage of Forest higher than +0.25 %/year and 5% of cells with positive trend of Total Forest Core Area higher than +150 ha/year (Table 4.4).

In contrast, we observed negative trends of PLAND (sen slope $<$ -0.25 %/year) and TCA (sen slope $<$ -150 ha/year) in the northeast portion of Minas Gerais state; southeast of Bahia; east side of Paraná, Santa Catarina and Rio Grande do Sul states (Figure 4.30). Also, we identified 3% of cells with negative trend of Percentage of Forest lower than -0.25 %/year and 5% of cells with negative trend of Total Forest Core Area lower than -150 ha/year (Table 4.4).

Table 4.4 - Proportion of cells per class for each landscape metric in the Atlantic Forest biome. Forest Edge Density (ED); Number of Forest Patches (NP); Total Forest Core Area (TCA) and Percentage of Forest (PLAND).

NP	TCA	PLAND	ED
<-40 (0%)	<-500 (0.1%)	<-0.5 (0.2%)	<-0.5 (0.2%)
-40 to -20 (0.5%)	-500 to -150 (4.9%)	-0.5 to -0.25 (2.8%)	-0.5 to 0 (7%)
-20 to -5 (7.5%)	-150 to -50 (10%)	-0.25 to 0 (20%)	0 to +0.5 (66%)
-5 to 0 (10%)	-50 to 0 (15%)	0 to +0.25 (61%)	>+0.5 (26.8%)
0 to +5 (22)	0 to +50 (38%)	+0.25 to +0.5 (15%)	
+5 to +10 (25%)	+50 to +150 (27%)	>+0.5 (1%)	
+10 to +20 (25%)	+150 to +500 (5%)		
>+20 (10%)	>+500 (0%)		

Figure 4.30 - Spatial trend of landscape metrics (Percentage of Forest - PLAND; Forest Edge Density - ED; Number of Forest Patches - NP; Total Forest Core Area - TCA) in the Atlantic Forest biome over the time series (2003-2018). Sen Slope - PLAND = %/year; ED = m/ha/year; NP = patches/year; TCA = ha/year.



Source: Produced by the author.

In the Caatinga, a Fire-independent biome, we observed positive trends of Forest Edge Density (ED, sen slope > +0.5 m/ha/year) and Number of Forest Patches (NP, sen slope > +20 patches/year), mainly, in the Sergipe, Pernambuco e Paraíba states (Figure 4.31). According to Table 4.5 we identified only 0.2% of cells with positive trend of Forest Edge Density higher than +0.5 m/ha/year and 1% of cells with positive trend of Number of Forest Patches higher than +20 patches/year.

Nonetheless, we observed negative trends of ED (sen slope < -0.5 m/ha/year) and NP (sen slope < -20 patches/year) more pronounced in the extreme north of Ceará state (Figure 4.31). Also, according to Table 4.5 we identified only 1.2% of cells with negative trend of Forest Edge Density lower than -0.5 m/ha/year and only 0.8% of cells with negative trend of Number of Forest Patches lower than -20 patches/year.

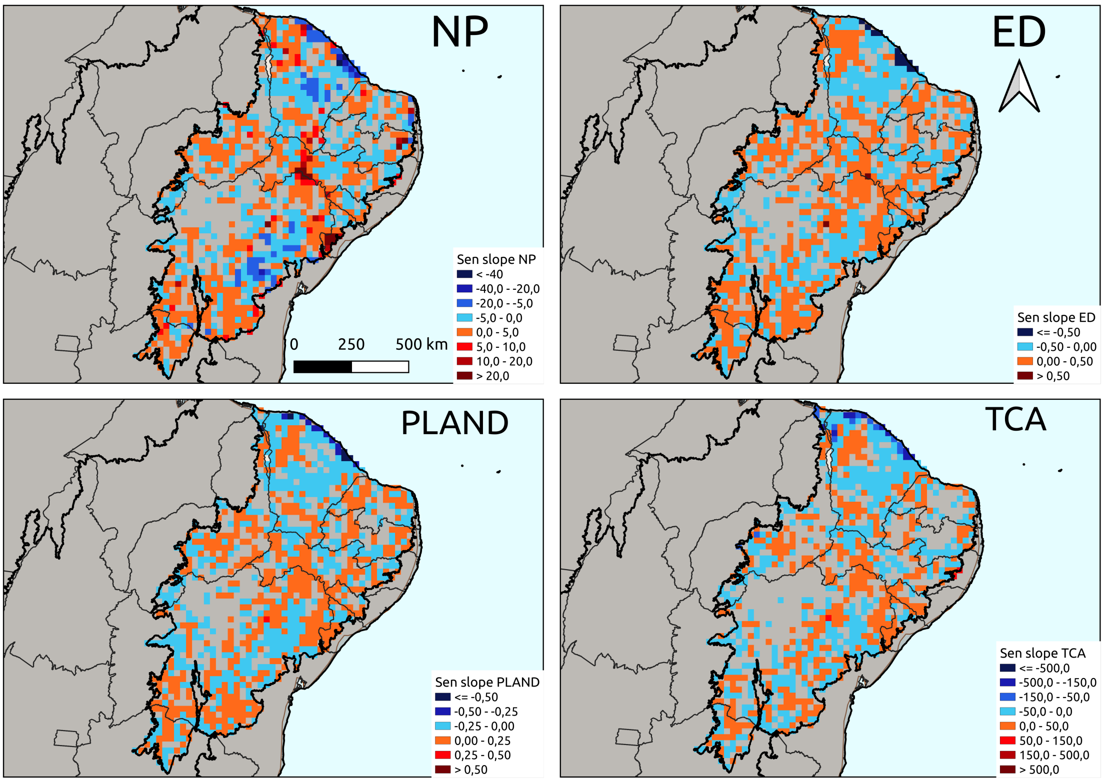
In the case of Percentage of Forest (PLAND) and Total Forest Core Area (TCA), we analyzed positive trends of PLAND (sen slope > 0.25 %/year) and TCA (sen slope > +50 ha/year) in the Pernambuco and Bahia states (Figure 4.31). In addition, we identified 0.1% of cells with positive trend of Percentage of Forest higher than +0.25 %/year and 0.3% of cells with positive trend of Total Forest Core Area higher than +50 ha/year (Table 4.5).

In contrast, we observed negative trends of PLAND (sen slope < -0.25 %/year) and TCA (sen slope < -150 ha/year) more pronounced in the extreme north of Ceará state (Figure 4.31). Also, we identified 1.5% of cells with negative trend of Percentage of Forest lower than -0.25 %/year and 1.2% of cells with negative trend of Total Forest Core Area lower than -150 ha/year (Table 4.5).

Table 4.5 - Proportion of cells per class for each landscape metric in the Caatinga biome. Forest Edge Density (ED); Number of Forest Patches (NP); Total Forest Core Area (TCA) and Percentage of Forest (PLAND).

NP	**TCA**	**PLAND**	**ED**
<-40 (0.2%)	<-500 (0%)	<-0.5 (0.5%)	<-0.5 (1.2%)
-40 to -20 (0.6%)	-500 to -150 (1.2%)	-0.5 to -0.25 (1%)	-0.5 to 0 (38.6%)
-20 to -5 (6%)	-150 to -50 (1.5%)	-0.25 to 0 (43%)	0 to +0.5 (60%)
-5 to 0 (30%)	-50 to 0 (44%)	0 to +0.25 (55.5%)	>+0.5 (0.2%)
0 to +5 (57%)	0 to +50 (53%)	+0.25 to +0.5 (0%)	
+5 to +10 (4%)	+50 to +150 (0.3%)	>+0.5 (0%)	
+10 to +20 (1.2%)	+150 to +500 (0%)		
>+20 (1%)	>+500 (0%)		

Figure 4.31 - Spatial trend of landscape metrics (Percentage of Forest - PLAND; Forest Edge Density - ED; Number of Forest Patches - NP; Total Forest Core Area - TCA) in the Caatinga biome over the time series (2003-2018). Sen Slope - PLAND = %/year; ED = m/ha/year; NP = patches/year; TCA = ha/year.



Source: Produced by the author.

Finally, in the category of Fire-dependent biomes, in the Cerrado, we observed positive trends of Forest Edge Density (ED, sen slope $>$ +0.5 m/ha/year) and Number of Forest Patches (NP, sen slope $>$ +20 patches/year) in the northeast of Mato Grosso do Sul, southwest of Minas Gerais, central portion of Maranhão and São Paulo (Figure 4.32). According to Table 4.6 we identified 2.2% of cells with positive trend of Forest Edge Density higher than +0.5 m/ha/year and 2.7% of cells with positive trend of Number of Forest Patches higher than +20 patches/year.

Nonetheless, we observed negative trends of ED (sen slope $<$ -0.5 m/ha/year) and NP (sen slope $<$ -20 patches/year), mainly, in the south of Tocantins and central portion of Minas Gerais (Figure 4.32). Also, according to Table 4.6 we identified only 0.4% of cells with negative trend of Forest Edge Density lower than -0.5 m/ha/year and only 0.15% of cells with negative trend of Number of Forest Patches lower than

-20 patches/year.

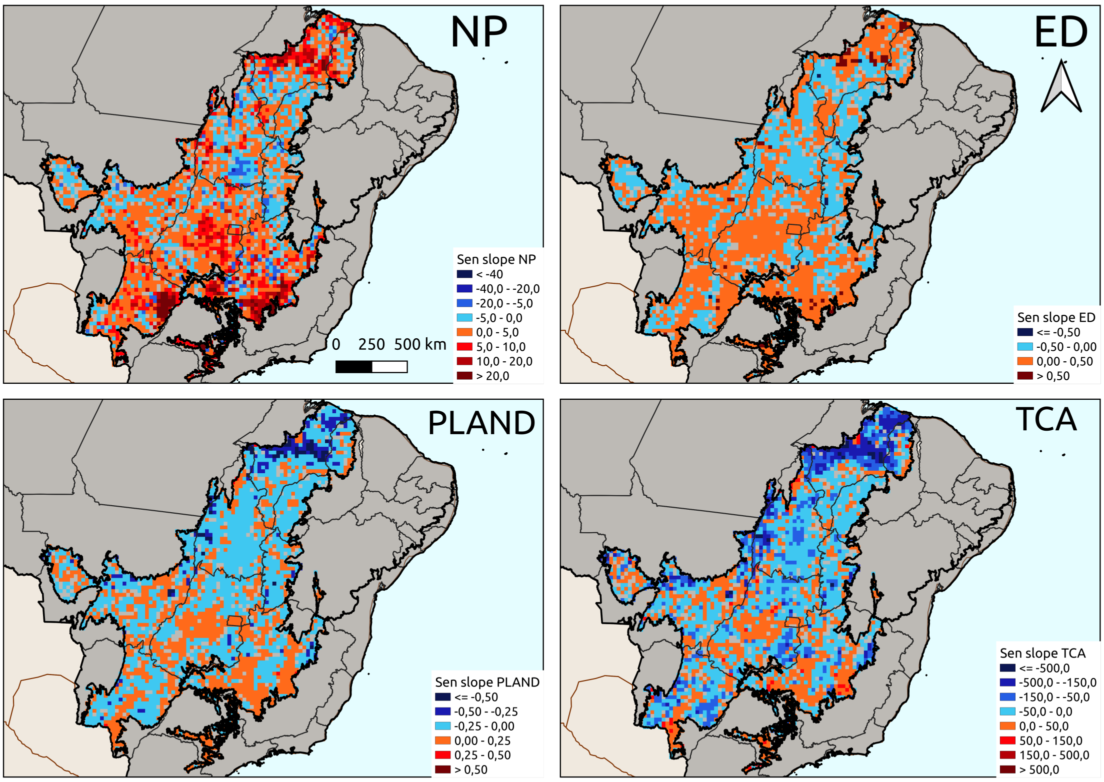
In the case of Percentage of Forest (PLAND) and Total Forest Core Area (TCA), we analyzed positive trends of PLAND (sen slope > 0.25 %/year) and TCA (sen slope $> +50$ ha/year) in the south of Mato Grosso do Sul, central-west of Minas Gerais and south of São Paulo (Figure 4.32). In addition, we identified only 0.07% of cells with positive trend of Percentage of Forest higher than +0.25 %/year and 2% of cells with positive trend of Total Forest Core Area higher than +50 ha/year (Table 4.6).

In contrast, we observed negative trends of PLAND (sen slope < -0.5 %/year) and TCA (sen slope < -150 ha/year) in the Maranhão state, west of Tocantins and central-east of Mato Grosso (Figure 4.32). Also, we identified 2% of cells with negative trend of Percentage of Forest lower than -0.5 %/year and 8% of cells with negative trend of Total Forest Core Area lower than -150 ha/year (Table 4.6).

Table 4.6 - Proportion of cells per class for each landscape metric in the Cerrado biome. Forest Edge Density (ED); Number of Forest Patches (NP); Total Forest Core Area (TCA) and Percentage of Forest (PLAND).

NP	**TCA**	**PLAND**	**ED**
<-40 (0%)	<-500 (1%)	<-0.5 (2%)	<-0.5 (0.4%)
-40 to -20 (0.15%)	-500 to -150 (7%)	-0.5 to -0.25 (5%)	-0.5 to 0 (38%)
-20 to -5 (4%)	-150 to -50 (15%)	-0.25 to 0 (56%)	0 to +0.5 (59.4%)
-5 to 0 (27.5%)	-50 to 0 (45%)	0 to +0.25 (37%)	>+0.5 (2.2%)
0 to +5 (46%)	0 to +50 (30.5%)	+0.25 to +0.5 (0.07%)	
+5 to +10 (14%)	+50 to +150 (2%)	>+0.5 (0%)	
+10 to +20 (6%)	+150 to +500 (0.03%)		
>+20 (2.7%)	>+500 (0%)		

Figure 4.32 - Spatial trend of landscape metrics (Percentage of Forest - PLAND; Forest Edge Density - ED; Number of Forest Patches - NP; Total Forest Core Area - TCA) in the Cerrado biome over the time series (2003-2018). Sen Slope - PLAND = %/year; ED = m/ha/year; NP = patches/year; TCA = ha/year.



Source: Produced by the author.

In the Pantanal biome, we observed positive trends of Forest Edge Density (ED, sen slope > +0 m/ha/year) and Number of Forest Patches (NP, sen slope > +5 patches/year), mainly, in the central-west portion of Pantanal biome (Figure 4.33). According to Table 4.7 we identified 39% of cells with positive trend of Forest Edge Density higher than +0 m/ha/year and 34% of cells with positive trend of Number of Forest Patches higher than +5 patches/year.

Nonetheless, we observed negative trends of ED (sen slope < -0.5 m/ha/year) and NP (sen slope < -5 patches/year) more intense in the Mato Grosso do Sul state, concentrated on east portion of Pantanal biome (Figure 4.33). Also, according to Table 4.7 we identified only 3% of cells with negative trend of Forest Edge Density lower than -0.5 m/ha/year and 19% of cells with negative trend of Number of Forest Patches lower than -5 patches/year.

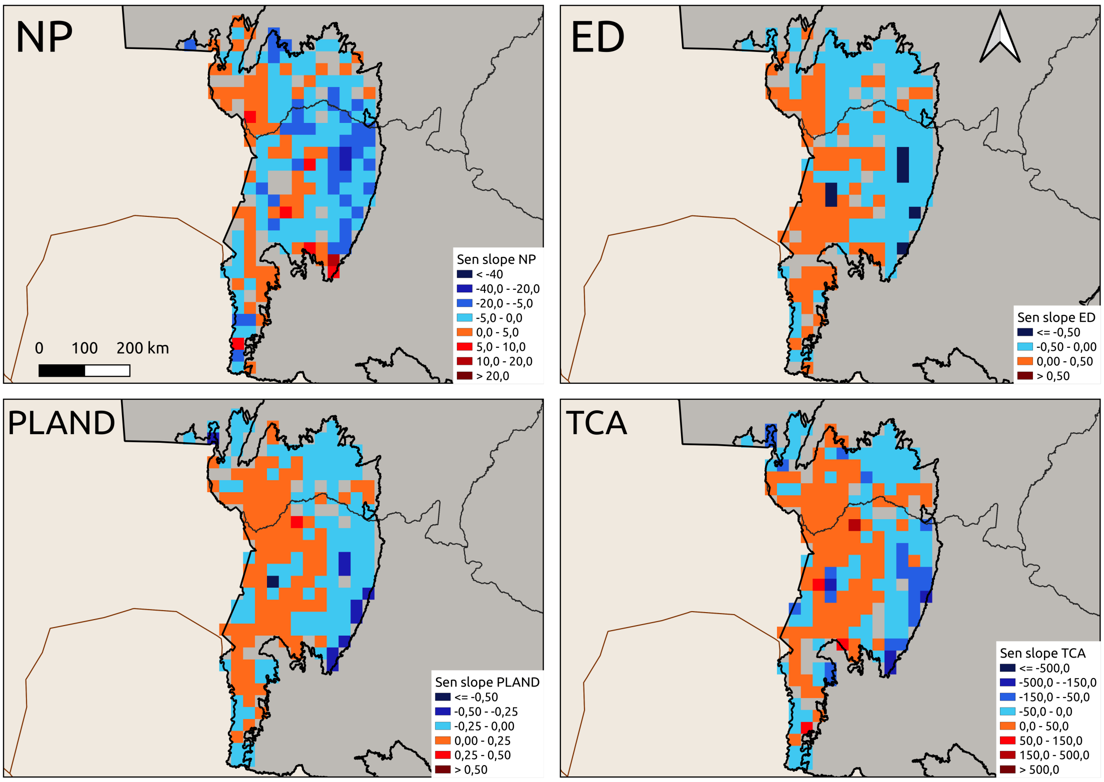
In the case of Percentage of Forest (PLAND) and Total Forest Core Area (TCA), we analyzed positive trends of PLAND (sen slope > 0.25 %/year) and TCA (sen slope $> +50$ ha/year), mainly, in the west portion of Pantanal biome (Figure 4.33). In addition, we identified only 0.5% of cells with positive trend of Percentage of Forest higher than +0.25 %/year and 13% of cells with positive trend of Total Forest Core Area higher than +50 ha/year (Table 4.7).

In contrast, we observed negative trends of PLAND (sen slope < -0.25 %/year) and TCA (sen slope < -50 ha/year) more intense in the Mato Grosso do Sul state, concentrated on east portion of Pantanal biome (Figure 4.33). Also, we identified 5.5% of cells with negative trend of Percentage of Forest lower than -0.25 %/year and only 2% of cells with negative trend of Total Forest Core Area lower than -50 ha/year (Table 4.7).

Table 4.7 - Proportion of cells per class for each landscape metric in the Pantanal biome. Forest Edge Density (ED); Number of Forest Patches (NP); Total Forest Core Area (TCA) and Percentage of Forest (PLAND).

NP	**TCA**	**PLAND**	**ED**
<-40 (0%)	<-500 (0%)	<-0.5 (0.5%)	<-0.5 (3%)
-40 to -20 (1%)	-500 to -150 (2%)	-0.5 to -0.25 (5%)	-0.5 to 0 (58%)
-20 to -5 (18%)	-150 to -50 (11%)	-0.25 to 0 (49%)	0 to +0.5 (39%)
-5 to 0 (47%)	-50 to 0 (39%)	0 to +0.25 (45%)	>+0.5 (0%)
0 to +5 (30%)	0 to +50 (46%)	+0.25 to +0.5 (1%)	
+5 to +10 (3%)	+50 to +150 (1%)	>+0.5 (0%)	
+10 to +20 (1%)	+150 to +500 (1%)		
>+20 (0%)	>+500 (0%)		

Figure 4.33 - Spatial trend of landscape metrics (Percentage of Forest - PLAND; Forest Edge Density - ED; Number of Forest Patches - NP; Total Forest Core Area - TCA) in the Pantanal biome over the time series (2003-2018). Sen Slope - PLAND = %/year; ED = m/ha/year; NP = patches/year; TCA = ha/year.



Source: Produced by the author.

In the Pampa biome, we observed positive trends of Forest Edge Density (ED, sen slope > +0.5 m/ha/year) and Number of Forest Patches (NP, sen slope > +20 patches/year) more intense in the central-east portion of the Pampa biome (Figure 4.34). According to Table 4.8 we identified 6% of cells with positive trend of Forest Edge Density higher than +0.5 m/ha/year and 21% of cells with positive trend of Number of Forest Patches higher than +20 patches/year.

Nonetheless, we observed negative trends of ED (sen slope < -0.5 m/ha/year) and NP (sen slope < -5 patches/year), mainly, in the extreme east and west portions of the Pampa biome (Figure 4.34). Also, according to Table 4.8 we identified only 1% of cells with negative trend of Forest Edge Density lower than -0.5 m/ha/year and 4% of cells with negative trend of Number of Forest Patches lower than -5 patches/year.

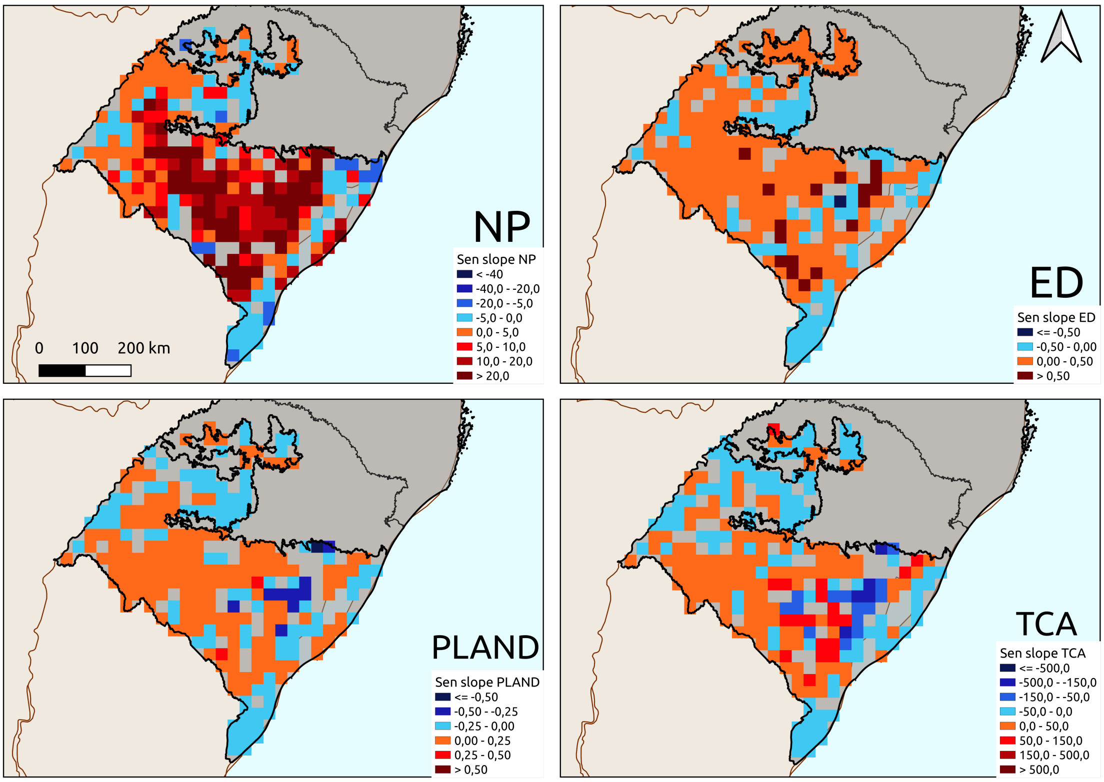
In the case of Percentage of Forest (PLAND) and Total Forest Core Area (TCA), we analyzed positive trends of PLAND (sen slope > +0.25 %/year) and TCA (sen slope > +50 ha/year) more intense in the east portion of the Pampa biome (Figure 4.34). In addition, we identified only 1% of cells with positive trend of Percentage of Forest higher than +0.25 %/year and 7% of cells with positive trend of Total Forest Core Area higher than +50 ha/year (Table 4.8).

In contrast, we observed negative trends of PLAND (sen slope < -0.25 %/year) and TCA (sen slope < -50 ha/year) more intense in the east portion of the Pampa biome (Figure 4.34). Also, we identified 4% of cells with negative trend of Percentage of Forest lower than -0.25 %/year and 8% of cells with negative trend of Total Forest Core Area lower than -50 ha/year (Table 4.8).

Table 4.8 - Proportion of cells per class for each landscape metric in the Pampa biome. Forest Edge Density (ED); Number of Forest Patches (NP); Total Forest Core Area (TCA) and Percentage of Forest (PLAND).

NP	**TCA**	**PLAND**	**ED**
<-40 (0%)	<-500 (0%)	<-0.5 (0.5%)	<-0.5 (1%)
-40 to -20 (0%)	-500 to -150 (2%)	-0.5 to -0.25 (3.5%)	-0.5 to 0 (24%)
-20 to -5 (4%)	-150 to -50 (6%)	-0.25 to 0 (34%)	0 to +0.5 (69%)
-5 to 0 (22%)	-50 to 0 (38%)	0 to +0.25 (61%)	>+0.5 (6%)
0 to +5 (25%)	0 to +50 (47%)	+0.25 to +0.5 (1%)	
+5 to +10 (12%)	+50 to +150 (7%)	>+0.5 (0%)	
+10 to +20 (16%)	+150 to +500 (0%)		
>+20 (21%)	>+500 (0%)		

Figure 4.34 - Spatial trend of landscape metrics (Percentage of Forest - PLAND; Forest Edge Density - ED; Number of Forest Patches - NP; Total Forest Core Area - TCA) in the Pampa biome over the time series (2003-2018). Sen Slope - PLAND = %/year; ED = m/ha/year; NP = patches/year; TCA = ha/year.



Source: Produced by the author.

4.4 Integration between spatial fire patterns and landscape metrics

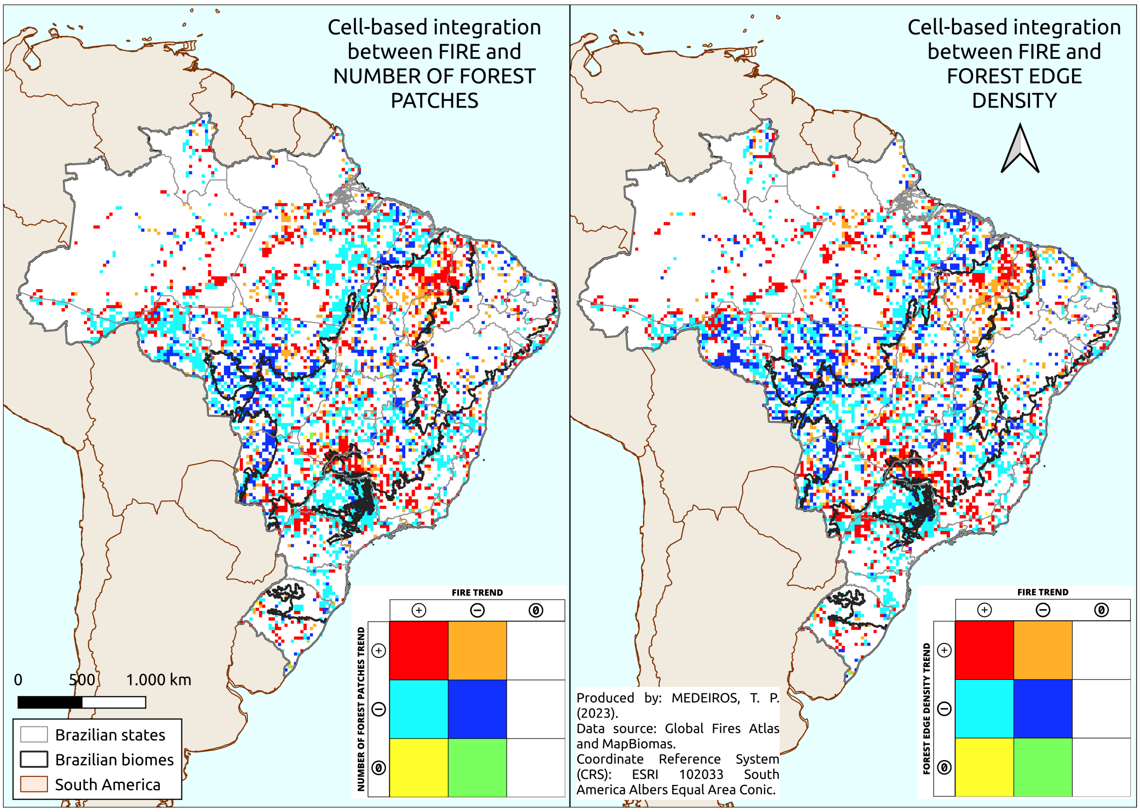
By combining fire and landscape metrics trend results, we were able to analyze the behavior of fire incidence in areas with increased or decreased landscape metrics rates (Figure 4.35 and Figure 4.36).

Concerning fire and Number of Forest Patches (NP) trend results in Brazil, we observed 28% of grid cells with increased fire rates overlapped by grid cells with positive Number of Forest Patches trends (Figure 4.35). These grid cells were concentrated, mainly, in the north (Maranhão state) and south (Mato Grosso do Sul, Goiás and Minas Gerais states) portion of Cerrado biome; central portion of Amazon biome (Pará, Amazonas and Rondônia states); central portion of Atlantic Forest (Paraná and Minas Gerais states).

In addition, we estimated 43% of grid cells with increased fire rates overlapped by grid cells with negative Number of Forest Patches trends (Figure 4.35). These grid cells were located, mainly, in the southeast portion of Amazon biome (Pará, Rondônia and Mato Grosso states); central portion of Cerrado biome (Mato Grosso and Goiás states); central portion of Atlantic Forest (São Paulo state).

Regarding fire and Forest Edge Density (ED) trend results in Brazil, we observed 27% of grid cells with increased fire rates overlapped by grid cells with positive Forest Edge Density trends (Figure 4.35). Also, we described 39% of grid cells with increased fire rates overlapped by grid cells with negative Forest Edge Density trends. The spatial configuration was the same obtained between fire and NP trends.

Figure 4.35 - Integration between fire and landscape metrics (Number of Forest Patches and Forest Edge Density).



Produced by: MEDEIROS, T. P. (2023).
Data source: Global Fires Atlas and MapBiomas.
Coordinate Reference System (CRS): ESRI 102033 South America Albers Equal Area Conic.

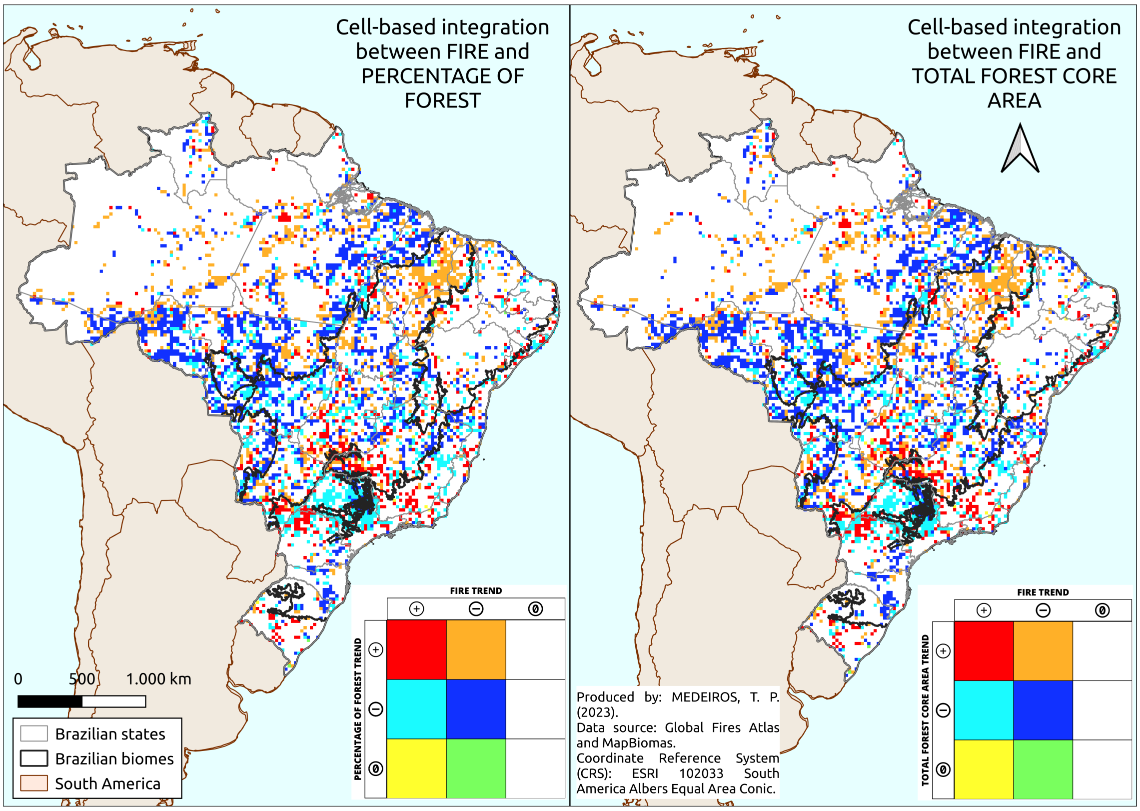
Source: Produced by the author.

Concerning fire and Percentage of Forest (PLAND) trend results in Brazil, we observed 24% of grid cells with decreased fire rates overlapped by grid cells with positive Percentage of Forest trends (Figure 4.36). These grid cells were concentrated, mainly, in the north portion of Cerrado biome (Maranhão and Piauí states); central portion of Amazon biome (Pará and Amazonas states).

In addition, we described 39% of grid cells with decreased fire rates overlapped by grid cells with negative Percentage of Forest trends (Figure 4.36). These grid cells were located, mainly, in the southeast portion of Amazon biome (Pará, Rondônia and Mato Grosso states); east portion of Pantanal biome (Mato Grosso do Sul state); central portion of Atlantic Forest biome (São Paulo state); central portion of Cerrado biome (Mato Grosso and Goiás states).

Regarding fire and Total Forest Core Area (TCA) trend results in Brazil, we observed 25% of grid cells with decreased fire rates overlapped by grid cells with positive Total Forest Core Area trends (Figure 4.36). Also, we described 37% of grid cells with decreased fire rates overlapped by grid cells with negative Total Forest Core Area trends. The spatial configuration was the same obtained between fire and PLAND trends.

Figure 4.36 - Integration between fire and landscape metrics (Percentage of Forest and Total Forest Core Area).



Produced by: MEDEIROS, T. P. (2023).
Data source: Global Fires Atlas and MapBiomas.
Coordinate Reference System (CRS): ESRI 102033 South America Albers Equal Area Conic.

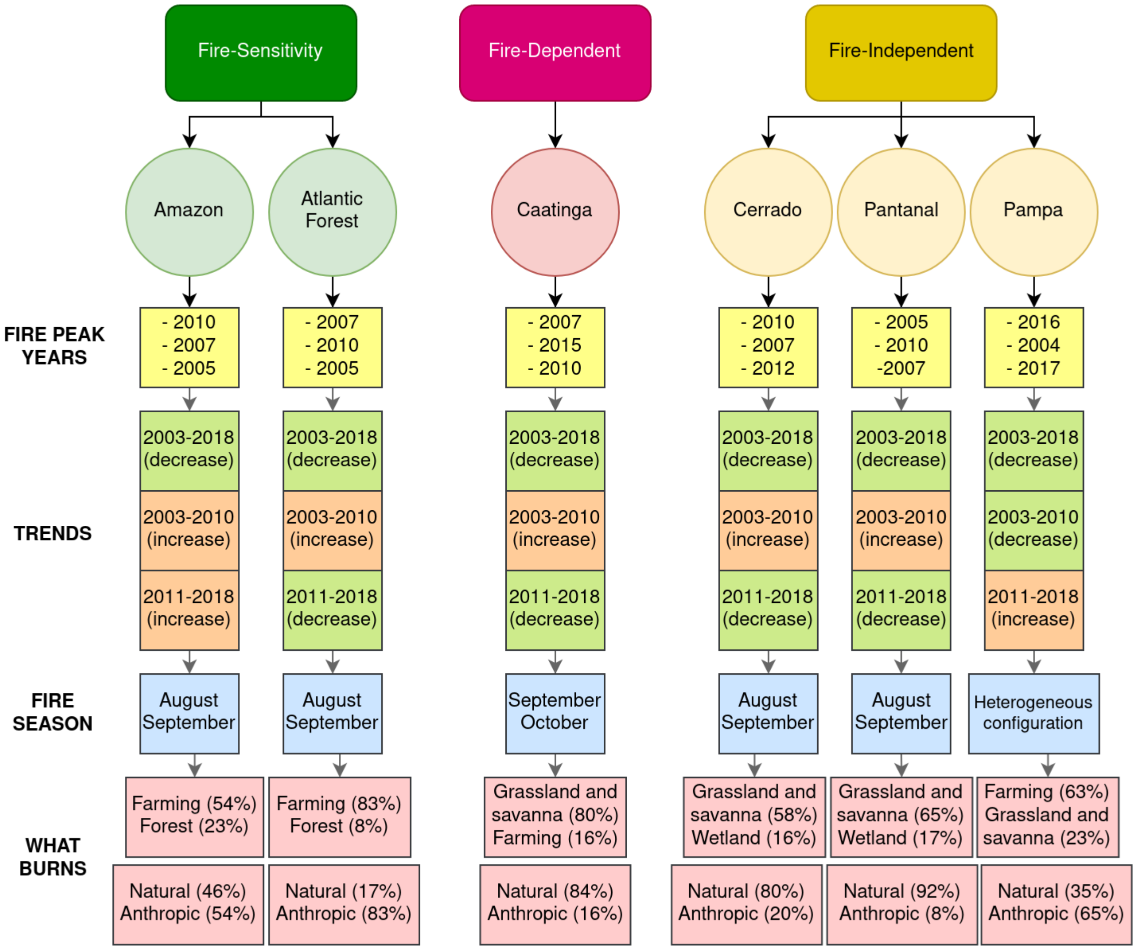
Source: Produced by the author.

5 DISCUSSION

The increase in the fire events and the extent of burned areas is not a problem that only afflicts Brazil. During the Anthropocene, fire has become more frequent (SINGLETON et al., 2019), more intense (CHERGUI et al., 2018) and more extended (COLLINS et al., 2021), causing consequences on the sustainability of socio-ecological systems (COLLINS et al., 2011).

In Figure5.1 we summarized our main results obtained by analysis of temporal patterns of fire in the Brazilian biomes, highlighting fire peak years, burned area trends, fire critical periods (fire season) and what burns according to the land use and land cover classes.

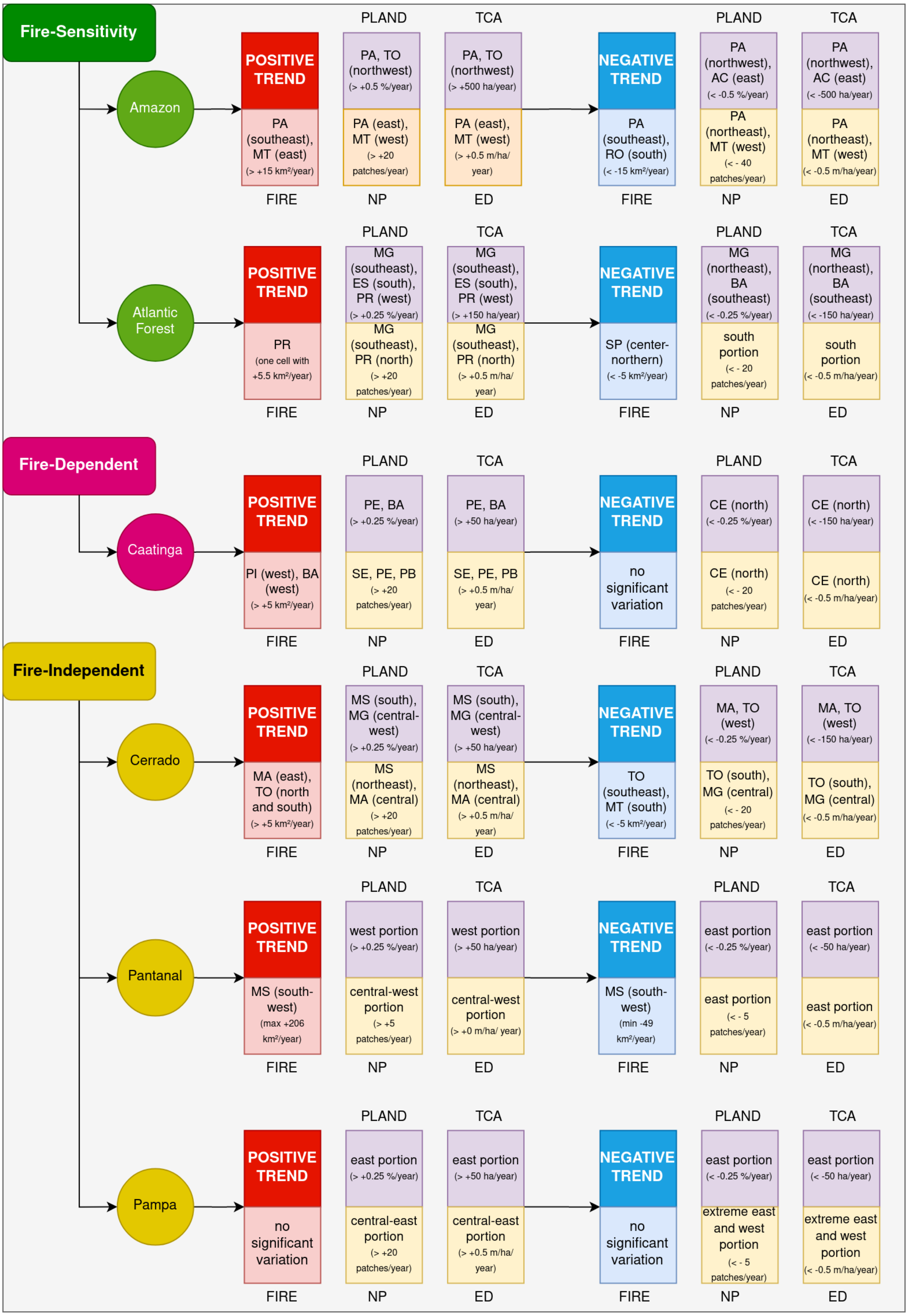
Figure 5.1 - Summarizing of main results obtained by analysis of temporal patterns of fire in the Brazilian biomes, highlighting fire peak years, burned area trends, fire critical periods (fire season) and what burns according to the land use and land cover classes.



Source: Produced by the author.

In addition, in the Figure 5.2, we observed the summarizing of results obtained by the analysis of spatial patterns of fire and landscape metrics in the Brazilian biomes. We highlighted the regions where the greatest trends of change, both positive and negative, were observed.

Figure 5.2 - Summarizing of main results obtained by analysis of spatial patterns of fire and landscape metrics in the Brazilian biomes.



Source: Produced by the author.

5.1 Fire occurrence and landscape changes in Brazil

Wildfires are among the greatest forms of disturbance in tropical ecosystems, making the areas more favourable for landscape changes like pasture transformation and advancement of the agricultural frontier. Land use and land cover changes (LULCC) can affect climate through changes in moisture and energy budgets, which is directly associated with deforestation of the Amazon forests. Its clear that non-Amazonian Brazil vegetation has received less attention, despite experiencing the highest transformation rate in the tropics (SALAZAR et al., 2015).

Precolonial pressures in ecosystems were expressed through settlement, cultivation, grazing, hunting and burning by indigenous people, which means temporary changes and, therefore, rapidly were reverted by ecological succession (KNAPP, 2007). Since 1900, due to European exploitation of natural resources, the biomes has been suffered significantly alterations, resulting in widespread transformations (ARMESTO et al., 2010). Global demand for food commodities (e.g. soybeans and beef) has pushed the expansion of agricultural frontier into former natural areas (RICHARDS et al., 2012).

In addition, the use of fire has always been present in human civilizations. The use of fire is culturally framed and transmitted, and it continues to undergo rapid changes in expression (BOWMAN et al., 2011). Understanding the fire regime alteration is crucial, because the survival of many species and ecosystems depends of the historical range of variability in fire activity.

In Brazil, we observed that the peak years of fires were 2010, 2007 and 2012. Concerning the Brazilian biomes, almost of them presented 2010 and 2007 as record years of fire occurrence, except the Pampa. The occurrence of fires, often caused by human actions, also can be associated with mega-droughts and ocean circulations such as the El Niño-Southern Oscillation (ENSO) event, among others. For example, the widespread occurrence of fires in 2010 can be attributed to the severe drought that occurred as a consequence of the El Niño-Southern Oscillation (ENSO), affecting mainly Cerrado and Amazon (biomes that presented 2010 as the year of fire peak) (MARENGO et al., 2011; LEWIS et al., 2011; LI et al., 2021). It is visible the influence of climate on the occurrence of fires, where changes in rainfall rates associated with an increase in temperature can impact the reduction of natural vegetation cover in Brazil. Changes in rainfall patterns and air temperature implied changes in land use and land cover and promoted changes in the original cover of natural vegetation (SALAZAR et al., 2015).

Despite fire occurring naturally in some ecosystems (e.g. savannas and grasslands), over the last two decades, an intense process of degradation and losses of original coverage was observed due to the expansion of agricultural and pasture areas, contributing to an increase in fire occurrence. Naturally, the spatial and temporal patterns of fires can be influenced by ignitions sources like lightning and season. Although, human activities can affect the fire regime in many ways, by changing fuel types, modifying fuel structure and continuity; igniting few or many fires in different seasons under various weather conditions (BOWMAN et al., 2011). One of the human activities is the conversion of natural areas to agricultural or urban use, which is a big problem and intensifies biodiversity loss and climate changes in these regions. In addition, the lack of public policies to combat deforestation can aggravate the biodiversity losses, especially in tropical areas such as the Amazon, Cerrado, and Pantanal in Brazil (BARBOSA et al., 2021; GARCIA et al., 2021; MARTINS et al., 2022; DELGADO et al., 2022).

Furthermore, the expansion of agricultural frontiers associated with the increase of fire source ignitions, even in fire-dependent environments, can affect the ecosystems and compromise the survival of many unique species (HERRERA et al., 2021). Crop areas such as sugarcane in Brazil, for example, also have a linear correlation with fire events, where cleaning of the area with burning is still done, mainly in regions of the Atlantic Forest, characterised by high ethanol production (TEODORO et al., 2022). Uncertainties in future landscape changes may result in dynamics very different from those experienced so far, creating consequences for ecosystem services based on land use (POPP et al., 2017).

5.1.1 Fire-sensitivity biomes

Humid tropical forests, dominant in the Amazon basin and in the Atlantic Forest, present no indication of evolutionary history influenced by fire, and their species do not have adaptations that favour their resistance to and resilience after fire events. Outside extreme climate events, rainforests are rarely affected by fire with natural ignitions. High humidity levels and lack of dry fuel from these forests prevent fire from starting and propagating. In general, natural fires usually occurs as a low-intensity surface fire and return intervals are estimated to be of hundreds or even thousands of years.

The impacts of fires in humid and semi deciduous tropical forests, in general, are very detrimental. The immediate impacts of fire include consuming the litter layer that protects the soil from erosion and recycles nutrients, killing most small trees and seedlings. Severe fires can kill most thin-barked large trees (UHL et al., 1990; BARLOW et al., 2002; STAVER et al., 2020); fire damage to roots may also lead to tree death (FLORES et al., 2016). The open canopy caused by tree death makes the forest susceptible to further fire events, inducing seriously harm local biodiversity and habitat integrity on a regional scale.

A more and more open and degraded forest offers less suitable habitat and resources for humid forest-dependent animal species. Fire mostly affects them at their edges, in contact with the fire prone ecosystem in which they are embedded, and recurrent fires can gradually reduce these forests (HEBERT-DUFRESNE et al., 2018).

The use of fire to manage the environment for food production, hunting, housing, rituals humans have changed the natural fire regimes in many ways, including ignition, suppression, and alteration of fuel type and amount. These phenomenons can, in long term, change forest structure, composition and flammability.

In Atlantic Forest, originally a fire-sensitive environment, the use of fire started early, where colonists used fire for land clearing and in warfare against indigenous groups. The use for deforestation and land management has continued over centuries, and currently, such fires are still frequently registered (SANTANA et al., 2020). Nevertheless, in the Amazon biome, the process of deforestation and land conversion started much later, largely associated with rubber tapping (MAEZUMI et al., 2018). Despite this, recurrent anthropogenic fires reached record levels over the last 20 years.

5.1.2 Fire-dependent biomes

Natural fires in the savannas and grasslands (in Brazil, Cerrado, Pantanal and Pampa) are typically surface fires, which pass rapidly and affect ground layer, through the consumption of fuel deposited on the top of soil (litter). Usually, are characterized by low intensity and return relatively frequently (3-6 years) (RAMOS-NETO; PIVELLO, 2000; PEREIRA et al., 2014). These fires occur, more often, in the transitional months between seasons, in general, in the beginning, or occasionally, at end of wet season, associated with lightning strikes that ignite the accumulated dry vegetation mass. Under these conditions, fire usually does not spread over large areas because it is extinguished by rainfall (RAMOS-NETO; PIVELLO, 2000; MEDEIROS; FIEDLER, 2004).

The fire consumes the fine and flammable tissue quickly, does not killing any plants or vertebrates and does not penetrating below topsoil layer. As a result, fires plants resprout, turning into a green flowery landscape (FIDELIS et al., 2018; FIDELIS, 2020; OVERBECK et al., 2005; OVERBECK; PFADENHAUER, 2007; OVERBECK et al., 2007). Also, many woody species have adaptations to resist fire, like thick bark that protect buds, allowing fast recovery of the canopy after a fire. In general, recurring fires maintain the characteristic structure of fire-dependent ecosystems (ROSAN et al., 2019).

Although, extensive high-temperature wildfires or very frequent fires, even in fire-dependent ecosystems, can negatively impact invertebrates (VASCONCELOS et al., 2017), less mobile vertebrates (ABOM; SCHWARZKOPF, 2016) and large mammals (SILVEIRA et al., 1999).

The fire-history in Cerrado (Brazilian savanna) started around 5 million years ago, with the spread o C4 grasses in the tropical and subtropical world. The C4 grasses are known as warm season grasses, where, during dry season or during extensive periods of droughts, provided plentiful fuel for wildfire (SIMON et al., 2009). This fact helped the establishment of a regime of recurrent fires.

The increase of fire frequency was marked in the Holocene period, linked to a highly seasonal climates and presence of human populations (CASSINO et al., 2020). Similar fire-history for Pampa grasslands was deduced, where herbaceous plant species show similar adaptations to fire (underground structures that protect the buds; regrowth and flowering after fire) (BEHLING et al., 2004).

Finally, through paleo-pollen analysis, also, was verified that fire affected Pantanal before human occupation. However, fire-history in the Pantanal biome was marked by a peculiar dynamic. The complex mosaic of vegetation (savannas, seasonal forests and grasslands) are controlled by a natural flooding regime, which was developed in the Pleistocene and was established with the drainage reorganization (ASSINE; SOARES, 2004). The rainfall patterns controls the seasonal flood pulse, that slowly moves from north (rainy season) to south (peak of the dry season) (ARRUDA et al., 2016). The intrinsic relationship between floods and fire that shape the vegetation communities (TOMAS et al., 2019).

During drought years, the floodplains are exposed a large amount of fuel that originates from dry vegetation and peat. In this case are a large risk of wildfires and the absence of waterlogged facilitates the fire spread (ARRUDA et al., 2016; TOMAS et al., 2019). This is what happened in the 2020 wildfires in the Pantanal, when river levels reached extremely low value and did not flood the wetlands (BARBOSA et al., 2022).

Currently, the fire regime have been changed by humans, where the frequency of fire are increasing and the timing of burning are changing. Anthropogenic fires, in general, occurred at middle or towards the end of the dry season, and, are more intense and spread over much greater extensions. In drought years this is more intensified because do not have rain to extinguish them. As a consequence, changes in spatial pattern and intensity are observed.

5.1.3 Fire-independent biome

Natural fire events in the Caatinga are rare (ALTHOFF et al., 2016), both because its vegetation does not provide continuous and easily flammable fuel and because there is a low incidence of lightning events in the region. In consequence, flora and fauna of the Caatinga lack adaptations to frequent fires. The Caatinga is thus classified as fire-independent under the natural fire regime and conditions. However, due to recent human activities, the Caatinga has been increasingly affected by fires that subject it to degradation (ALTHOFF et al., 2016) and can turn it into a fire-sensitive system.

6 CONCLUSIONS

Our study indicated that natural fire regimes have been changed by human actions, meaning a threat for ecosystems conservation and regeneration. Also, spatial and temporal dynamics can be influenced by its relationship with vegetation history and by landscape changes.

In general, 2010 and 2007 can be considered as fire peak years and the critical periods for fire occurrence is concentrated on August and September months. Regarding the analysis of what burned in the Brazil territory according land use and land cover classes, 60% of the burned area occurred in Natural Vegetation classes (Forest, Savanna and Grassland, Wetland), highlighting Cerrado, Pantanal and Caatinga, in which more than 80% of fires occurred in natural areas. Concerning the spatial configuration of fire and landscape metrics, we observed 38% of the Brazil territory with positive trends for burned area and, an intrinsic relationship with landscape changes.

In future works other analysis can be applied, such as: test other grid sizes; explore other fire metrics (speed, expansion, fire line, duration); explore other landscape metrics (connectivity, mean patch size, edge contrast, composition); expand the time series (2019 to 2023); test trends with other binary reclassification (pasture, agricultural classes, savanna, grassland, wetland); explore more the relationship between fire and landscape fragmentation.

In summary, command-and-control approaches are important and illegal fires need to be prosecuted, but it is also important to actively incentives land users to adopt alternative techniques to fires, such as agroforestry, crop livestock-forest integration, rotation between crop and pasture, no-tillage cultivation, shredding of cut vegetation, and thus allow for a transition to more sustainable and fire-free types of land use. Second, fire management as such can only be efficient if agencies are properly equipped, supplied, and trained; capacity building on the ground and the development of monitoring systems are thus important tasks. Third, research on fire should integrate different knowledge areas from biological to human science in a national research agenda that will create a better basis for developing landscapes that are more resilient to fire. In addition to this, education and outreach are necessary to introduce a deeper understanding on the role of fire to all professionals dealing with natural resources conservation and fire management.

REFERENCES

ABOM, R.; SCHWARZKOPF, L. Short-term responses of reptile assemblages to fire in native and weedy tropical savannah. **Global Ecology and Conservation**, v. 6, p. 58–66, 4 2016. ISSN 2351-9894. 77

ABREU, R. C.; HOFFMANN, W. A.; VASCONCELOS, H. L.; PILON, N. A.; ROSSATTO, D. R.; DURIGAN, G. The biodiversity cost of carbon sequestration in tropical savanna. **Science Advances**, v. 3, 2017. ISSN 23752548. 1

ALTHOFF, T. D.; MENEZES, R. S. C.; CARVALHO, A. L. de; PINTO, A. de S.; SANTIAGO, G. A. C. F.; OMETTO, J. P. H. B.; RANDOW, C. von; SAMPAIO, E. V. de S. B. Climate change impacts on the sustainability of the firewood harvest and vegetation and soil carbon stocks in a tropical dry forest in Santa Teresinha municipality, northeast Brazil. **Forest Ecology and Management**, v. 360, p. 367–375, 1 2016. ISSN 0378-1127. 78

ALVES, D. B.; ALVARADO, S. T. Variação espaço-temporal da ocorrência do fogo nos biomas brasileiros com base na análise de produtos de sensoriamento remoto. **Geografia**, v. 44, p. 321–345, 2019. 2

ANDELA, N.; MORTON, D. C.; GIGLIO, L.; PAUGAM, R.; CHEN, Y.; HANTSON, S.; WERF, G. R. V. D.; ANDERSON, J. T. The Global Fire Atlas of individual fire size, duration, speed and direction. **Earth System Science Data**, v. 11, p. 529–552, 4 2019. ISSN 18663516. 19, 20, 21

ANDELA, N.; WERF, G. R. V. D.; KAISER, J. W.; LEEUWEN, T. T. V.; WOOSTER, M. J.; LEHMANN, C. E. Biomass burning fuel consumption dynamics in the tropics and subtropics assessed from satellite. **Biogeosciences**, v. 13, p. 3717–3734, 6 2016. ISSN 17264189. 15

ARAGAO, L. E.; ANDERSON, L. O.; FONSECA, M. G.; ROSAN, T. M.; VEDOVATO, L. B.; WAGNER, F. H.; SILVA, C. V.; SILVA-JUNIOR, C. H.; ARAI, E.; AGUIAR, A. P.; BARLOW, J.; BERENGUER, E.; DEETER, M. N.; DOMINGUES, L. G.; GATTI, L.; GLOOR, M.; MALHI, Y.; MARENGO, J. A.; MILLER, J. B.; PHILLIPS, O. L.; SAATCHI, S. 21st century drought-related fires counteract the decline of Amazon deforestation carbon emissions. **Nature Communications**, v. 9, p. 1–12, 2 2018. ISSN 2041-1723. 1, 12

ARAGAO, L. E. O.; MALHI, Y.; BARBIER, N.; LIMA, A.; SHIMABUKURO, Y.; ANDERSON, L.; SAATCHI, S. Interactions between rainfall, deforestation and fires during recent years in the brazilian Amazonia. **Philosophical Transactions of the Royal Society B: Biological Sciences**, v. 363, p. 1779–1785, 2008. ISSN 09628452. 3, 14

ARGIBAY, D. S.; SPARACINO, J.; ESPINDOLA, G. M. A long-term assessment of fire regimes in a brazilian ecotone between seasonally dry tropical forests and savannah. **Ecological Indicators**, v. 113, 6 2020. ISSN 1470160X. 7

ARMESTO, J. J.; MANUSCHEVICH, D.; MORA, A.; SMITH-RAMIREZ, C.; ROZZI, R.; ABARZúA, A. M.; MARQUET, P. A. From the holocene to the anthropocene: a historical framework for land cover change in southwestern South America in the past 15,000 years. **Land Use Policy**, v. 27, p. 148–160, 4 2010. ISSN 02648377. 74

ARRUDA, W. D. S.; OLDELAND, J.; FILHO, A. C. P.; POTT, A.; CUNHA, N. L.; ISHII, I. H.; DAMASCENO, G. A. Inundation and fire shape the structure of riparian forests in the Pantanal, Brazil. **PLOS ONE**, v. 11, p. e0156825, 6 2016. ISSN 1932-6203. Available from: <<https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0156825>>. 77

ASSINE, M. L.; SOARES, P. C. Quaternary of the Pantanal, west-central Brazil. **Quaternary International**, v. 114, p. 23–34, 1 2004. ISSN 1040-6182. 77

ASSIS, T. O.; AGUIAR, A. P. D. D.; RANDOW, C. V.; GOMES, D. M. D. P.; KURY, J. N.; OMETTO, J. P. H.; NOBRE, C. A. CO2 emissions from forest degradation in brazilian amazon. **Environmental Research Letters**, v. 15, p. 104035, 9 2020. ISSN 1748-9326. Available from: <<https://iopscience.iop.org/article/10.1088/1748-9326/ab9cfchttps://iopscience.iop.org/article/10.1088/1748-9326/ab9cfc/meta>>. 12

BARBOSA, L. G.; ALVES, M. A. S.; GRELLE, C. E. V. Actions against sustainability: Dismantling of the environmental policies in Brazil. **Land Use Policy**, v. 104, p. 105384, 5 2021. ISSN 0264-8377. 75

BARBOSA, M. L. F.; HADDAD, I.; NASCIMENTO, A. L. da S.; SILVA, G. M. da; VEIGA, R. M. da; HOFFMANN, T. B.; SOUZA, A. R. de; DALAGNOL, R.; STREHER, A. S.; PEREIRA, F. R. S.; ARAGãO, L. E. O. C. de; ANDERSON, L. O. Compound impact of land use and extreme climate on the 2020 fire record of the brazilian Pantanal. **Global Ecology and Biogeography**, v. 31, p. 1960–1975, 10 2022. ISSN 1466-8238. Available from: <<https://onlinelibrary.wiley.com/doi/full/10.1111/geb.13563https://onlinelibrary.wiley.com/doi/abs/10.1111/geb.13563https://onlinelibrary.wiley.com/doi/10.1111/geb.13563>>. 78

BARLOW, J.; HAUGAASEN, T.; PERES, C. A. Effects of ground fires on understorey bird assemblages in amazonian forests. **Biological Conservation**, v. 105, p. 157–169, 2002. Available from: <<www.elsevier.com/locate/biocon>>. 76

BEHLING, H.; PILLAR, V. D. P.; ORLóCI, L.; BAUERMANN, S. G. Late quaternary araucaria forest, grassland (campos), fire and climate dynamics, studied by high-resolution pollen, charcoal and multivariate analysis of the Cambará do Sul core in southern Brazil. **Palaeogeography, Palaeoclimatology, Palaeoecology**, v. 203, p. 277–297, 2 2004. ISSN 0031-0182. 77

BOND, W. J.; WOODWARD, F. I.; MIDGLEY, G. F. The global distribution of ecosystems in a world without fire. **New Phytologist**, v. 165, p. 525–538, 2 2005. ISSN 0028646X. 1

BOWMAN, D. M.; BALCH, J.; ARTAXO, P.; BOND, W. J.; COCHRANE, M. A.; D'ANTONIO, C. M.; DEFRIES, R.; JOHNSTON, F. H.; KEELEY, J. E.; KRAWCHUK, M. A.; KULL, C. A.; MACK, M.; MORITZ, M. A.; PYNE, S.; ROOS, C. I.; SCOTT, A. C.; SODHI, N. S.; SWETNAM, T. W. The human dimension of fire regimes on earth. **Journal of Biogeography**, v. 38, p. 2223–2236, 12 2011. ISSN 1365-2699. Available from: <<https://onlinelibrary.wiley.com/doi/full/10.1111/j.1365-2699.2011.02595.xhttps://onlinelibrary.wiley.com/doi/abs/10.1111/j.1365-2699.2011.02595.xhttps://onlinelibrary.wiley.com/doi/10.1111/j.1365-2699.2011.02595.x>>. 13, 74, 75

BRANDO, P. M.; BALCH, J. K.; NEPSTAD, D. C.; MORTON, D. C.; PUTZ, F. E.; COE, M. T.; SILVéRIO, D.; MACEDO, M. N.; DAVIDSON, E. A.; NóBREGA, C. C.; ALENCAR, A.; SOARES-FILHO, B. S. Abrupt increases in amazonian tree mortality due to drought-fire interactions. **Proceedings of the National Academy of Sciences of the United States of America**, v. 111, p. 6347–6352, 2014. ISSN 10916490. 3

BURTON, C.; BETTS, R. A.; JONES, C. D.; FELDPAUSCH, T. R.; CARDOSO, M.; ANDERSON, L. O. El Niño driven changes in global fire 2015/16. **Frontiers in Earth Science**, v. 8, 2020. ISSN 22966463. 1

BURTON, C.; KELLEY, D. I.; JONES, C. D.; BETTS, R. A.; CARDOSO, M.; ANDERSON, L. South American fires and their impacts on ecosystems increase with continued emissions. **Climate Resilience and Sustainability**, v. 1, 2022. ISSN 2692-4587. 1

CABRAL, A. I.; SILVA, S.; SILVA, P. C.; VANNESCHI, L.; VASCONCELOS, M. J. Burned area estimations derived from Landsat ETM+ and OLI data: comparing genetic programming with maximum likelihood and classification and regression trees. **ISPRS Journal of Photogrammetry and Remote Sensing**, v. 142, p. 94–105, 8 2018. ISSN 0924-2716. 15

CAMPANHARO, W. A.; LOPES, A. P.; ANDERSON, L. O.; SILVA, T. F. da; ARAGãO, L. E. Translating fire impacts in southwestern amazonia into economic costs. **Remote Sensing**, v. 11, 4 2019. ISSN 20724292. 7

CASSINO, R. F.; LEDRU, M. P.; SANTOS, R. de A.; FAVIER, C. Vegetation and fire variability in the central cerrados (Brazil) during the pleistocene-holocene transition was influenced by oscillations in the sasm boundary belt. **Quaternary Science Reviews**, v. 232, p. 106209, 3 2020. ISSN 0277-3791. 77

CHEN, D.; PEREIRA, J. M.; MASIERO, A.; PIROTTI, F. Mapping fire regimes in China using MODIS active fire and burned area data. **Applied Geography**, v. 85, p. 14–26, 8 2017. ISSN 0143-6228. 15

CHERGUI, B.; FAHD, S.; SANTOS, X.; PAUSAS, J. G. Socioeconomic factors drive fire-regime variability in the mediterranean basin. **Ecosystems**, v. 21, p. 619–628, 6 2018. ISSN 14350629. Available from: <<https://link.springer.com/article/10.1007/s10021-017-0172-6>>. 71

CHRISTIAN, T. J.; KLEISS, B.; YOKELSON, R. J.; HOLZINGER, R.; CRUTZEN, P. J.; HAO, W. M.; SAHARJO, B. H.; WARD, D. E. Comprehensive laboratory measurements of biomass-burning emissions: 1. emissions from indonesian, african, and other fuels. **Journal of Geophysical Research: Atmospheres**, v. 108, 12 2003. ISSN 01480227. 9

CHUVIECO, E.; MARTíNEZ, S.; ROMáN, M. V.; HANTSON, S.; PETTINARI, M. L. Integration of ecological and socio-economic factors to assess global vulnerability to wildfire. **Global Ecology and Biogeography**, v. 23, p. 245–258, 2 2014. ISSN 1466822X. 8, 11

CHUVIECO, E.; MOUILLOT, F.; WERF, G. R. van der; MIGUEL, J. S.; TANASSE, M.; KOUTSIAS, N.; GARCíA, M.; YEBRA, M.; PADILLA, M.; GITAS, I.; HEIL, A.; HAWBAKER, T. J.; GIGLIO, L. Historical background and current developments for mapping burned area from satellite earth observation. **Remote Sensing of Environment**, v. 225, p. 45–64, 5 2019. ISSN 0034-4257. 15

COCHRANE, M. A.; RYAN, K. C. **Fire and fire ecology: concepts and principles**. [S.l.: s.n.], 2009. 9, 10, 11, 12

COLLINS, B. M.; EVERETT, R. G.; STEPHENS, S. L. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests. **Ecosphere**, v. 2, p. 1–14, 4 2011. ISSN 2150-8925. Available from: <<https://onlinelibrary.wiley.com/doi/full/10.1890/ES11-00026.1https://onlinelibrary.wiley.com/doi/abs/10.1890/ES11-00026.1https://esajournals.onlinelibrary.wiley.com/doi/10.1890/ES11-00026.1>>. 71

COLLINS, L.; BRADSTOCK, R. A.; CLARKE, H.; CLARKE, M. F.; NOLAN, R. H.; PENMAN, T. D. The 2019/2020 mega-fires exposed australian ecosystems to an unprecedented extent of high-severity fire. **Environmental Research Letters**, v. 16, p. 044029, 3 2021. ISSN 1748-9326. Available from: <<https://iopscience.iop.org/article/10.1088/1748-9326/abeb9ehttps://iopscience.iop.org/article/10.1088/1748-9326/abeb9e/meta>>. 8, 71

DAMIAN, J. M.; DURIGAN, M. R.; CHERUBIN, M. R.; MAIA, S. M. F.; OGLE, S. M.; CAMARGO, P. B. de; FERREIRA, J. N.; OLIVEIRA-JÚNIOR, R. C. de; CERRI, C. E. P. Deforestation and land use change mediate soil carbon changes in the eastern brazilian Amazon. **Regional Environmental Change**, v. 21, p. 1–12, 2021. ISSN 1436378X. 3

DELGADO, R. C.; SANTANA, R. O. de; GELSLEICHTER, Y. A.; PEREIRA, M. G. Degradation of south american biomes: what to expect for the future? **Environmental Impact Assessment Review**, v. 96, 9 2022. ISSN 01959255. 75

FIDELIS, A. Is fire always the “bad guy”? **Flora**, v. 268, p. 151611, 7 2020. ISSN 0367-2530. 77

FIDELIS, A.; ALVARADO, S. T.; BARRADAS, A. C. S.; PIVELLO, V. R. The year 2017: megafires and management in the Cerrado. **Fire**, v. 1, p. 49, 12 2018. ISSN 2571-6255. Available from: <<https://www.mdpi.com/2571-6255/1/3/49/htmhttps://www.mdpi.com/2571-6255/1/3/49>>. 77

FLORES, B. M.; FAGOAGA, R.; NELSON, B. W.; HOLMGREN, M. Repeated fires trap amazonian blackwater floodplains in an open vegetation state. **Journal of Applied Ecology**, v. 53, p. 1597–1603, 10 2016. ISSN 13652664. 76

FOLEY, J. A.; DEFRIES, R.; ASNER, G. P.; BARFORD, C.; BONAN, G.; CARPENTER, S. R.; CHAPIN, F. S.; COE, M. T.; DAILY, G. C.; GIBBS, H. K.; HELKOWSKI, J. H.; HOLLOWAY, T.; HOWARD, E. A.; KUCHARIK, C. J.; MONFREDA, C.; PATZ, J. A.; PRENTICE, I. C.; RAMANKUTTY, N.; SNYDER, P. K. Global consequences of land use. **Science**, v. 309, p. 570–574, 2005. 1

FORMAN, R. T. T. **Land mosaics: the ecology of landscapes and regions**. [S.l.: s.n.], 1995. ISBN 0521474620. 14

FORMAN, R. T. T.; GODRON, M. **Landscape ecology**. [s.n.], 1986. ISBN 978-0-471-87037-1. Available from: <<https://www.wiley.com/en-us/Landscape+Ecology-p-9780471870371>>. 13, 14

GARCIA, L. C.; SZABO, J. K.; ROQUE, F. de O.; PEREIRA, A. de M. M.; CUNHA, C. N. da; DAMASCENO-JúNIOR, G. A.; MORATO, R. G.; TOMAS, W. M.; LIBONATI, R.; RIBEIRO, D. B. 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**, v. 293, p. 112870, 9 2021. ISSN 0301-4797. 75

GEIST, H. J.; LAMBIN, E. F. Proximate causes and underlying driving forces of tropical deforestation. **BioScience**, v. 52, p. 143–150, 2002. ISSN 14678306. 14

GIGLIO, L.; BOSCHETTI, L.; ROY, D. P.; HUMBER, M. L.; JUSTICE, C. O. The collection 6 MODIS burned area mapping algorithm and product. **Remote Sensing of Environment**, v. 217, p. 72–85, 11 2018. ISSN 0034-4257. 20

GODFREY, B. J.; BROWDER, J. O. Disarticulated urbanization in the brazilian amazon. **Geographical Review**, v. 86, p. 441, 7 1996. ISSN 00167428. 14

GOLDING, N.; BETTS, R. Fire risk in amazonia due to climate change in the HadCM3 climate model: potential interactions with deforestation. **Global Biogeochemical Cycles**, v. 22, 12 2008. ISSN 08866236. 7, 8

HARDESTY, J.; MYERS, R.; FULKS, W. Fire, ecosystems and people: a preliminary assessment of fire as a global conservation issue. **George Wright Forum**, v. 22, p. 78–87, 2005. 2, 3, 11, 17

HARRISON, S.; MARLON, J.; BARTLEIN, P. Fire in the earth system. **Changing Climates, Earth Systems and Society**, p. 21–48, 2010. 9

HAWBAKER, T. J.; VANDERHOOF, M. K.; BEAL, Y. J.; TAKACS, J. D.; SCHMIDT, G. L.; FALGOUT, J. T.; WILLIAMS, B.; FAIRAUX, N. M.; CALDWELL, M. K.; PICOTTE, J. J.; HOWARD, S. M.; STITT, S.; DWYER, J. L. Mapping burned areas using dense time-series of Landsat data. **Remote Sensing of Environment**, v. 198, p. 504–522, 9 2017. ISSN 0034-4257. 15

HEBERT-DUFRESNE, L.; PELLEGRINI, A. F.; BHAT, U.; REDNER, S.; PACALA, S. W.; BERDAHL, A. M. Edge fires drive the shape and stability of tropical forests. **Ecology Letters**, v. 21, p. 794–803, 6 2018. ISSN 14610248. 76

HERRERA, A. H.; BALLERA, B. L.; TORO-MANRíQUEZ, M. D.; LENCINAS, M. V.; PASTUR, G. J. M.; RAM, H. H. Changes in vegetation of flooded savannas subject to cattle grazing and fire in plains of Colombia. **Land**, v. 10, p. 1–14, 1 2021. ISSN 2073445X. 75

HUMBER, M. L.; BOSCHETTI, L.; GIGLIO, L.; JUSTICE, C. O. Spatial and temporal intercomparison of four global burned area products. **International Journal of Digital Earth**, v. 12, p. 460–484, 4 2019. ISSN 17538955. Available from: <<https://www.tandfonline.com/doi/abs/10.1080/17538947.2018.1433727>>. 15

ICHOKU, C.; KAHN, R.; CHIN, M. Satellite contributions to the quantitative characterization of biomass burning for climate modeling. **Atmospheric Research**, v. 111, p. 1–28, 7 2012. ISSN 0169-8095. 15

JEZEQUEL, C.; OBERDORFF, T.; TEDESCO, P. A.; SCHMITT, L. Geomorphological diversity of rivers in the Amazon Basin. **Geomorphology**, v. 400, p. 108078, 3 2022. ISSN 0169-555X. 18

JONES, G. M.; KRAMER, H. A.; WHITMORE, S. A.; BERIGAN, W. J.; TEMPEL, D. J.; WOOD, C. M.; HOBART, B. K.; ERKER, T.; ATUO, F. A.; PIETRUNTI, N. F.; KELSEY, R.; GUTIéRREZ, R. J.; PEERY, M. Z. Habitat selection by spotted owls after a megafire reflects their adaptation to historical frequent-fire regimes. **Landscape Ecology**, v. 35, p. 1199–1213, 5 2020. ISSN 15729761. 7

JUAREZ-OROZCO, S. M.; SIEBE, C.; FERNáNDEZ, D. F. y. Causes and effects of forest fires in tropical rainforests: a bibliometric approach. **Tropical Conservation Science**, v. 10, 10 2017. ISSN 19400829. Available from: <<https://journals.sagepub.com/doi/10.1177/1940082917737207>>. 11

JURVELIUS, M. **Forest fires (prediction, prevention, preparedness and suppression)**. [S.l.: s.n.], 2004. 334-339 p. ISBN 978-0-12-145160-8. 2

JUSTICE, C. O.; GIGLIO, L.; KORONTZI, S.; OWENS, J.; MORISETTE, J. T.; ROY, D.; DESCLOITRES, J.; ALLEAUME, S.; PETITCOLIN, F.; KAUFMAN, Y. The modis fire products. **Remote Sensing of Environment**, v. 83, p. 244–262, 11 2002. ISSN 0034-4257. 15

JUSTINO, F.; MéLO, A. S. de; SETZER, A.; SISMANOGLU, R.; SEDIYAMA, G. C.; RIBEIRO, G. A.; MACHADO, J. P.; STERL, A. Greenhouse gas induced changes in the fire risk in Brazil in echam5/mpi-om coupled climate model. **Climatic Change**, v. 106, p. 285–302, 5 2011. ISSN 01650009. 7, 8

KEANE, R. **Disturbance regimes and the historical range of variation in terrestrial ecosystems**. [S.l.: s.n.], 2013. 568-581 p. ISBN 9780123847195. 2

KEELEY, J. E. Fire intensity, fire severity and burn severity: a brief review and suggested usage. **International Journal of Wildland Fire**, v. 18, p. 116–126, 2 2009. ISSN 1448-5516. Available from: <<https://www.publish.csiro.au/wf/WF07049>>. 11

KNAPP, G. **The legacy of european colonialism**. [S.l.: s.n.], 2007. 74

KUMAR, S.; GETIRANA, A.; LIBONATI, R.; HAIN, C.; MAHANAMA, S.; ANDELA, N. Changes in land use enhance the sensitivity of tropical ecosystems to fire-climate extremes. **Scientific Reports**, v. 12, 2022. ISSN 20452322. 3

LAMBIN, E. F.; STRAHLER, A. H. Indicators of land-cover change for change-vector analysis in multitemporal space at coarse spatial scales. **International Journal of Remote Sensing**, v. 15, p. 2099–2119, 7 1994. ISSN 13665901. Available from: <<https://www.tandfonline.com/doi/abs/10.1080/01431169408954230>>. 14

LARSEN, A. E.; REICH, B. J.; RUMINSKI, M.; RAPPOLD, A. G. Impacts of fire smoke plumes on regional air quality, 2006–2013. **Journal of Exposure Science and Environmental Epidemiology**, v. 28, p. 319–327, 12 2017. ISSN 1559-064X. Available from: <<https://www.nature.com/articles/s41370-017-0013-x>>. 7

LEAL, I. R.; TABARELLI, M.; SILVA, J. M. C. **Ecologia e conservação da Caatinga**. [S.l.]: Universidade Federal do Pernambuco (UFPE), 2003. 18

LEHMANN, C. E.; ARCHIBALD, S. A.; HOFFMANN, W. A.; BOND, W. J. Deciphering the distribution of the savanna biome. **New Phytologist**, v. 191, p. 197–209, 7 2011. ISSN 1469-8137. Available from: <<https://onlinelibrary.wiley.com/doi/full/10.1111/j.1469-8137.2011.03689.xhttps://onlinelibrary.wiley.com/doi/abs/10.1111/j.1469-8137.2011.03689.xhttps://nph.onlinelibrary.wiley.com/doi/10.1111/j.1469-8137.2011.03689.x>>. 19

LEWIS, S. L.; BRANDO, P. M.; PHILLIPS, O. L.; HEIJDEN, G. M. V. D.; NEPSTAD, D. The 2010 Amazon drought. **Science**, v. 331, p. 554, 2 2011. ISSN 00368075. 74

LI, F.; LAWRENCE, D. M.; BOND-LAMBERTY, B. Impact of fire on global land surface air temperature and energy budget for the 20th century due to changes within ecosystems. **Environmental Research Letters**, v. 12, 2017. ISSN 1748-9326. 1, 7

LI, S.; SPARROW, S. N.; OTTO, F. E.; RIFAI, S. W.; OLIVERAS, I.; KRIKKEN, F.; ANDERSON, L. O.; MALHI, Y.; WALLOM, D. Anthropogenic climate change contribution to wildfire-prone weather conditions in the Cerrado and arc of deforestation. **Environmental Research Letters**, v. 16, p. 094051, 9 2021. ISSN 1748-9326. Available from: <<https://iopscience.iop.org/article/10.1088/1748-9326/ac1e3ahttps://iopscience.iop.org/article/10.1088/1748-9326/ac1e3a/meta>>. 74

LIBONATI, R.; PEREIRA, J. M.; CAMARA, C. C. D.; PERES, L. F.; OOM, D.; RODRIGUES, J. A.; SANTOS, F. L.; TRIGO, R. M.; GOUVEIA, C. M.; MACHADO-SILVA, F.; ENRICH-PRAST, A.; SILVA, J. M. Twenty-first century droughts have not increasingly exacerbated fire season severity in the brazilian Amazon. **Scientific Reports**, v. 11, 2021. ISSN 20452322. 3

LONG, T.; ZHANG, Z.; HE, G.; JIAO, W.; TANG, C.; WU, B.; ZHANG, X.; WANG, G.; YIN, R. 30 m resolution global annual burned area mapping based on landsat images and Google Earth Engine. **Remote Sensing**, v. 11, p. 1–30, 2 2019. ISSN 2072-4292. Available from: <<https://www.mdpi.com/2072-4292/11/5/489/htmhttps://www.mdpi.com/2072-4292/11/5/489>>. 14

MAEZUMI, S. Y.; ROBINSON, M.; SOUZA, J. de; URREGO, D. H.; SCHAAN, D.; ALVES, D.; IRIARTE, J. New insights from pre-columbian land use and fire management in amazonian dark earth forests. **Frontiers in Ecology and Evolution**, v. 6, 8 2018. ISSN 2296701X. 76

MANN, H. B. Nonparametric tests against trend. **Econometrica**, v. 13, p. 245, 7 1945. ISSN 00129682. 23, 26

MARENGO, J. A.; SOUZA, C. M.; THONICKE, K.; BURTON, C.; HALLADAY, K.; BETTS, R. A.; ALVES, L. M.; SOARES, W. R. Changes in climate and land use over the amazon region: current and future variability and trends. **Frontiers in Earth Science**, v. 6, p. 425317, 12 2018. ISSN 22966463. 12

MARENGO, J. A.; TOMASELLA, J.; ALVES, L. M.; SOARES, W. R.; RODRIGUEZ, D. A. The drought of 2010 in the context of historical droughts in the Amazon region. **Geophysical Research Letters**, v. 38, 6 2011. ISSN 00948276. 74

MARLIER, M. E.; BONILLA, E. X.; MICKLEY, L. J. How do brazilian fires affect air pollutionand public health? **GeoHealth**, v. 4, 8 2020. ISSN 24711403. 7

MARTIN, D. A. Linking fire and the United Nations Sustainable Development Goals. **Science of the Total Environment**, v. 662, p. 547–558, 4 2019. ISSN 18791026. 7

MARTINS, P. I.; BELéM, L. B. C.; SZABO, J. K.; LIBONATI, R.; GARCIA, L. C. Prioritising areas for wildfire prevention and post-fire restoration in the brazilian Pantanal. **Ecological Engineering**, v. 176, p. 106517, 3 2022. ISSN 0925-8574. 75

MATAVELI, G. A. V.; SILVA, M. E. S.; FRAN'A, D. de A.; BRUNSELL, N. A.; OLIVEIRA, G. de; CARDOZO, F. da S.; BERTANI, G.; PEREIRA, G. Characterization and trends of fine particulate matter (pm2.5) fire emissions in the brazilian Cerrado during 2002-2017. **Remote Sensing**, v. 11, 10 2019. ISSN 20724292. 7

MATAVELI, G. A. V.; SILVA, M. E. S.; PEREIRA, G.; CARDOZO, F. D. S.; KAWAKUBO, F. S.; BERTANI, G.; COSTA, J. C.; RAMOS, R. D. C.; SILVA, V. V. D. Satellite observations for describing fire patterns and climate-related fire drivers in the brazilian savannas. **Natural Hazards and Earth System Sciences**, v. 18, p. 125–144, 1 2018. ISSN 16849981. 15

MAURIN, O.; DAVIES, T. J.; BURROWS, J. E.; DARU, B. H.; YESSOUFOU, K.; MUASYA, A. M.; BANK, M. van der; BOND, W. J. Savanna fire and the origins of the 'underground forests' of Africa. **New Phytologist**, v. 204, p. 201–214, 2014. ISSN 14698137. 2

MCLAUCHLAN, K. K.; HIGUERA, P. E.; MIESEL, J.; ROGERS, B. M.; SCHWEITZER, J.; SHUMAN, J. K.; TEPLEY, A. J.; VARNER, J. M.; VEBLEN, T. T.; ADALSTEINSSON, S. A.; BALCH, J. K.; BAKER, P.; BATLLORI, E.; BIGIO, E.; BRANDO, P.; CATTAU, M.; CHIPMAN, M. L.; COEN, J.; CRANDALL, R.; DANIELS, L.; ENRIGHT, N.; GROSS, W. S.; HARVEY, B. J.; HATTEN, J. A.; HERMANN, S.; HEWITT, R. E.; KOBZIAR, L. N.; LANDESMANN, J. B.; LORANTY, M. M.; MAEZUMI, S. Y.; MEARNS, L.; MORITZ, M.; MYERS, J. A.; PAUSAS, J. G.; PELLEGRINI, A. F.; PLATT, W. J.; ROOZEBOOM, J.; SAFFORD, H.; SANTOS, F.; SCHELLER, R. M.; SHERRIFF, R. L.; SMITH, K. G.; SMITH, M. D.; WATTS, A. C. Fire as a fundamental ecological process: research advances and frontiers. **Journal of Ecology**, v. 108, p. 2047–2069, 9 2020. ISSN 13652745. 1

MEDEIROS, M. B. de; FIEDLER, N. C. Incêndios florestais no Parque Nacional da Serra da Canastra: desafios para a conservação da biodiversidade. **Ciência Florestal**, v. 14, p. 157–168, 6 2004. ISSN 1980-5098. Available from: <<https://periodicos.ufsm.br/cienciaflorestal/article/view/1815>>. 76

MELCHIORRE, A.; BOSCHETTI, L. Global analysis of burned area persistence time with MODIS data. **Remote Sensing**, v. 10, p. 750, 5 2018. ISSN 2072-4292. Available from: <<https://www.mdpi.com/2072-4292/10/5/750/htmhttps://www.mdpi.com/2072-4292/10/5/750>>. 15

MERTENS, B.; LAMBIN, E. Modelling land cover dynamics: integration of fine-scale land cover data with landscape attributes. **International Journal of Applied Earth Observation and Geoinformation**, v. 1, p. 48–52, 1 1999. ISSN 1569-8432. 14

METZGER, J. P. O que é ecologia de paisagens? **Biota Neotropica**, v. 1, p. 1–9, 2001. ISSN 1676-0611. Available from: <<https://www.scielo.br/j/bn/a/Jbchd6rjY35PGkY5BHPz63S/?format=html&lang=pt>>. 14

MOUILLOT, F.; SCHULTZ, M. G.; YUE, C.; CADULE, P.; TANSEY, K.; CIAIS, P.; CHUVIECO, E. Ten years of global burned area products from spaceborne remote sensing — a review: analysis of user needs and recommendations for future developments. **International Journal of Applied Earth Observation and Geoinformation**, v. 26, p. 64–79, 2 2014. ISSN 1569-8432. 15

NAVEH, Z.; LIEBERMAN, A. S. **Landscape ecology**. [s.n.], 1994. ISBN 978-0-387-94059-5. Available from: <<http://link.springer.com/10.1007/978-1-4757-2331-1>>. 14

NEPSTAD, D.; MCGRATH, D.; STICKLER, C.; ALENCAR, A.; AZEVEDO, A.; SWETTE, B.; BEZERRA, T.; DIGIANO, M.; SHIMADA, J.; MOTTA, R. S. D.; ARMIJO, E.; CASTELLO, L.; BRANDO, P.; HANSEN, M. C.; MCGRATH-HORN, M.; CARVALHO, O.; HESS, L. Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. **Science**, v. 344, p. 1118–1123, 6 2014. ISSN 10959203. Available from: <<https://www.science.org/doi/10.1126/science.1248525>>. 14

OVERBECK, G. E.; MüLLER, S. C.; FIDELIS, A.; PFADENHAUER, J.; PILLAR, V. D.; BLANCO, C. C.; BOLDRINI, I. I.; BOTH, R.; FORNECK, E. D. Brazil's neglected biome: the south brazilian campos. **Perspectives in Plant Ecology, Evolution and Systematics**, v. 9, p. 101–116, 12 2007. ISSN 1433-8319. 77

OVERBECK, G. E.; MüLLER, S. C.; PILLAR, V. D.; PFADENHAUER, J. Fine-scale post-fire dynamics in southern brazilian subtropical grassland. **Journal of Vegetation Science**, v. 16, p. 655–664, 12 2005. ISSN 1654-1103. Available from: <<https://onlinelibrary.wiley.com/doi/full/10.1111/j.1654-1103.2005.tb02408.xhttps://onlinelibrary.wiley.com/doi/abs/10.1111/j.1654-1103.2005.tb02408.xhttps://onlinelibrary.wiley.com/doi/10.1111/j.1654-1103.2005.tb02408.x>>. 77

OVERBECK, G. E.; PFADENHAUER, J. Adaptive strategies in burned subtropical grassland in southern Brazil. **Flora - Morphology, Distribution, Functional Ecology of Plants**, v. 202, p. 27–49, 2 2007. ISSN 0367-2530. 77

PAUSAS, J. G.; KEELEY, J. E. A burning story: the role of fire in the history of life. **BioScience**, v. 59, p. 593–601, 7 2009. ISSN 00063568. 1

_____. Wildfires as an ecosystem service. **Frontiers in Ecology and the Environment**, v. 17, p. 289–295, 6 2019. ISSN 15409309. 3

PEREIRA, A. C.; OLIVEIRA, S. L.; PEREIRA, J. M.; TURKMAN, M. A. A. Modelling fire frequency in a cerrado savanna protected area. **PLOS ONE**, v. 9, p. e102380, 7 2014. ISSN 1932-6203. Available from:

<<https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0102380>>. 76

PEREIRA, G.; SIQUEIRA, R.; ROSáRIO, N. E.; LONGO, K. L.; FREITAS, S. R.; CARDOZO, F. S.; KAISER, J. W.; WOOSTER, M. J. Assessment of fire emission inventories during the South American Biomass Burning Analysis (SAMBBA) experiment. **Atmospheric Chemistry and Physics**, v. 16, p. 6961–6975, 6 2016. ISSN 16807324. 15

PESSOA, A. C. M.; ANDERSON, L. O.; CARVALHO, N. S.; CAMPANHARO, W. A.; JUNIOR, C. H. S.; ROSAN, T. M.; REIS, J. B.; PEREIRA, F. R.; ASSIS, M.; JACON, A. D.; OMETTO, J. P.; SHIMABUKURO, Y. E.; SILVA, C. V.; PONTES-LOPES, A.; MORELLO, T. F.; ARAGãO, L. E. Intercomparison of burned area products and its implication for carbon emission estimations in the Amazon. **Remote Sensing**, v. 12, p. 1–24, 11 2020. ISSN 2072-4292. Available from: <<https://www.mdpi.com/2072-4292/12/23/3864/htmhttps://www.mdpi.com/2072-4292/12/23/3864>>. 15

PIVELLO, V. R.; VIEIRA, I.; CHRISTIANINI, A. V.; RIBEIRO, D. B.; MENEZES, L. da S.; BERLINCK, C. N.; MELO, F. P.; MARENGO, J. A.; TORNQUIST, C. G.; TOMAS, W. M.; OVERBECK, G. E. Understanding Brazil's catastrophic fires: causes, consequences and policy needed to prevent future tragedies. **Perspectives in Ecology and Conservation**, v. 19, p. 233–255, 2021. ISSN 2530-0644. 2, 3, 12

POPP, A.; CALVIN, K.; FUJIMORI, S.; HAVLIK, P.; HUMPENöDER, F.; STEHFEST, E.; BODIRSKY, B. L.; DIETRICH, J. P.; DOELMANN, J. C.; GUSTI, M.; HASEGAWA, T.; KYLE, P.; OBERSTEINER, M.; TABEAU, A.; TAKAHASHI, K.; VALIN, H.; WALDHOFF, S.; WEINDL, I.; WISE, M.; KRIEGLER, E.; LOTZE-CAMPEN, H.; FRICKO, O.; RIAHI, K.; VUUREN, D. P. Land-use futures in the shared socio-economic pathways. **Global Environmental Change**, v. 42, p. 331–345, 1 2017. ISSN 0959-3780. 75

PORTNER, H. O.; ROBERTS, D. C.; TIGNOR, M.; POLOCZANSKA, E. S.; MINTENBECK, K.; ALEGRíA, A.; CRAIG, M.; LANGSDORF, S.; LOSCHKE, S.; MOLLER, V.; OKEM, A.; RAMA, B. **Climate Change 2022: Impacts, Adaptation and Vulnerability**. Cambrigde University Press, 2022. Available from: <<https://www.ipcc.ch/report/ar6/wg2/chapter/summary-for-policymakers/>>. 11

RAMOS-NETO, M. B.; PIVELLO, V. R. Lightning fires in a brazilian savanna national park: rethinking management strategies. **Environmental Management**, v. 26, p. 675–684, 12 2000. ISSN 0364152X. Available from: <<https://link.springer.com/article/10.1007/s002670010124>>. 76

RIBEIRO, M. C.; MARTENSEN, A. C.; METZGER, J. P.; TABARELLI, M.; SCARANO, F.; FORTIN, M.-J. The brazilian Atlantic Forest: a shrinking biodiversity hotspot. **Biodiversity Hotspots**, p. 405–434, 2011. Available from: <<https://link.springer.com/chapter/10.1007/978-3-642-20992-5_21>>. 18

RICHARDS, P. D.; MYERS, R. J.; SWINTON, S. M.; WALKER, R. T. Exchange rates, soybean supply response, and deforestation in South America. **Global Environmental Change**, v. 22, p. 454–462, 5 2012. ISSN 09593780. 74

RISSER, P. G. **Landscape ecology: state of the art**. [s.n.], 1987. 3-14 p. Available from: <<https://link.springer.com/chapter/10.1007/978-1-4612-4742-5_1>>. 14

ROESCH, L. F. W.; VIEIRA, F. C. B.; PEREIRA, V. A.; SCHüNEMANN, A. L.; TEIXEIRA, I. F.; SENNA, A. J. T.; STEFENON, V. M. The brazilian Pampa: a fragile biome. **Diversity**, v. 1, p. 182–198, 12 2009. ISSN 1424-2818. Available from: <<https://www.mdpi.com/1424-2818/1/2/182/htmhttps://www.mdpi.com/1424-2818/1/2/182>>. 19

ROSAN, T. M.; ARAGãO, L. E.; OLIVERAS, I.; PHILLIPS, O. L.; MALHI, Y.; GLOOR, E.; WAGNER, F. H. Extensive 21st-century woody encroachment in South America's savanna. **Geophysical Research Letters**, v. 46, p. 6594–6603, 6 2019. ISSN 1944-8007. Available from: <<https://onlinelibrary.wiley.com/doi/full/10.1029/2019GL082327https://onlinelibrary.wiley.com/doi/abs/10.1029/2019GL082327https://agupubs.onlinelibrary.wiley.com/doi/10.1029/2019GL082327>>. 77

ROY, D. P.; LEWIS, P. E.; JUSTICE, C. O. Burned area mapping using multi-temporal moderate spatial resolution data—a bi-directional reflectance model-based expectation approach. **Remote Sensing of Environment**, v. 83, p. 263–286, 11 2002. ISSN 0034-4257. 15

SALAZAR, A.; BALDI, G.; HIROTA, M.; SYKTUS, J.; MCALPINE, C. Land use and land cover change impacts on the regional climate of non-amazonian South America: a review. **Global and Planetary Change**, v. 128, p. 103–119, 5 2015. ISSN 09218181. 74

SANTANA, R. O. de; DELGADO, R. C.; SCHIAVETTI, A. The past, present and future of vegetation in the central Atlantic Forest corridor, Brazil. **Remote Sensing Applications: Society and Environment**, v. 20, 11 2020. ISSN 23529385. 76

SCHOLES, R. J.; ARCHER, S. R. Tree-grass interactions in savannas. **Annual Review of Ecology, Evolution, and Systematics**, v. 28, p. 517–561, 1997. 1

SCHULZ, C.; WHITNEY, B. S.; ROSSETTO, O. C.; NEVES, D. M.; CRABB, L.; OLIVEIRA, E. C. de; LIMA, P. L. T.; AFZAL, M.; LAING, A. F.; FERNANDES, L. C. de S.; SILVA, C. A. da; STEINKE, V. A.; STEINKE, E. T.; SAITO, C. H. Physical, ecological and human dimensions of environmental change in Brazil's Pantanal wetland: synthesis and research agenda. **Science of The Total Environment**, v. 687, p. 1011–1027, 10 2019. ISSN 0048-9697. 19

SCOTT, A. C.; GLASSPOOL, I. J. The diversification of paleozoic fire systems and fluctuations in atmospheric oxygen concentration. **Proceedings of the National Academy of Sciences of the United States of America**, v. 103, p. 10861–10865, 7 2006. ISSN 00278424. 1

SEN, P. K. Estimates of the regression coefficient based on Kendall's tau. **Journal of the American Statistical Association**, v. 63, p. 1379–1389, 1968. ISSN 1537274X. 23, 26

SHIMABUKURO, Y. E.; DUTRA, A. C.; ARAI, E.; DUARTE, V.; CASSOL, H. L. G.; PEREIRA, G.; CARDOZO, F. da S. Mapping burned areas of mato grosso state brazilian Amazon using multisensor datasets. **Remote Sensing**, v. 12, p. 1–23, 11 2020. ISSN 2072-4292. Available from: <<https://www.mdpi.com/2072-4292/12/22/3827/htmhttps://www.mdpi.com/2072-4292/12/22/3827>>. 14, 15

SILVA-JUNIOR, C. A. da; TEODORO, P. E.; DELGADO, R. C.; TEODORO, L. P. R.; LIMA, M.; PANTALEãO, A. de A.; BAIO, F. H. R.; AZEVEDO, G. B. de; AZEVEDO, G. T. de O. S.; CAPRISTO-SILVA, G. F.; ARVOR, D.; FACCO, C. U. Persistent fire foci in all biomes undermine the Paris agreement in Brazil. **Scientific Reports**, v. 10, p. 1–14, 2020. ISSN 2045-2322. 1

SILVA, P. S.; NOGUEIRA, J.; RODRIGUES, J. A.; SANTOS, F. L.; PEREIRA, J. M.; DACAMARA, C. C.; DALDEGAN, G. A.; PEREIRA, A. A.; PERES, L. F.; SCHMIDT, I. B.; LIBONATI, R. Putting fire on the map of brazilian savanna ecoregions. **Journal of Environmental Management**, v. 296, 2021. ISSN 10958630. 3

SILVEIRA, L.; RODRIGUES, F. H. G.; JACORNO, A. T. A. D.; DINIZ, J. A. F. Impact of wildfires on the megafauna of Emas National Park, central Brazil. **Oryx**, v. 33, p. 108–114, 4 1999. ISSN 1365-3008. Available from: <<https://www.cambridge.org/core/journals/oryx/article/impact-of-wildfires-on-the-megafauna-of-emas-national-park-central-brazil/EE0856D30ECDBFC9F947D97EB88A5177>>. 77

SIMON, M. F.; GRETHER, R.; QUEIROZ, L. P. D.; SKEMAE, C.; PENNINGTON, R. T.; HUGHES, C. E. Recent assembly of the Cerrado, a neotropical plant diversity hotspot, by in situ evolution of adaptations to fire. **Proceedings of the National Academy of Sciences of the United States of America**, v. 106, p. 20359–20364, 2009. ISSN 00278424. 2, 77

SINGLA, S.; ELDAWY, A. Distributed zonal statistics of big raster and vector data. **Proceedings of the 26th ACM International Symposium on Advances in Geographic Information Systems**, p. 536–539, 11 2018. Available from: <<https://dl.acm.org/doi/10.1145/3274895.3274985>>. 24

SINGLETON, M. P.; THODE, A. E.; MEADOR, A. J. S.; INIGUEZ, J. M. Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. **Forest Ecology and Management**, v. 433, p. 709–719, 2 2019. ISSN 0378-1127. 71

SOUZA, C. M.; SHIMBO, J. Z.; ROSA, M. R.; PARENTE, L. L.; ALENCAR, A. A.; RUDORFF, B. F.; HASENACK, H.; MATSUMOTO, M.; FERREIRA, L. G.; SOUZA-FILHO, P. W.; OLIVEIRA, S. W. de; ROCHA, W. F.; FONSECA, A. V.; MARQUES, C. B.; DINIZ, C. G.; COSTA, D.; MONTEIRO, D.; ROSA, E. R.; VéLEZ-MARTIN, E.; WEBER, E. J.; LENTI, F. E.; PATERNOST, F. F.; PAREYN, F. G.; SIQUEIRA, J. V.; VIERA, J. L.; FERREIRA-NETO, L. C.; SARAIVA, M. M.; SALES, M. H.; SALGADO, M. P.; VASCONCELOS, R.; GALANO, S.; MESQUITA, V. V.; AZEVEDO, T. Reconstructing three decades of land use and land cover changes in brazilian biomes with landsat archive and earth engine. **Remote Sensing**, v. 12, p. 2735, 8 2020. ISSN 2072-4292. Available from: <<https://www.mdpi.com/2072-4292/12/17/2735/htmhttps://www.mdpi.com/2072-4292/12/17/2735>>. 20, 21

STAVER, A. C.; BRANDO, P. M.; BARLOW, J.; MORTON, D. C.; PAINE, C. E.; MALHI, Y.; MURAKAMI, A. A.; PASQUEL, J. del A. Thinner bark increases sensitivity of wetter amazonian tropical forests to fire. **Ecology Letters**, v. 23, p. 99–106, 1 2020. ISSN 14610248. 76

SULLIVAN, A.; BAKER, E.; KURVITS, T. **A rapid response assessment spreading like wildfire the rising threat of extraordinary landscape fires**. [s.n.], 2022. Available from: <<http://www.un.org/Depts/>>. 8, 11, 13

TEODORO, P. E.; SILVA-JUNIOR, C. A.; DELGADO, R. C.; LIMA, M.; TEODORO, L. P. R.; BAIO, F. H. R.; AZEVEDO, G. B. de; AZEVEDO, G. T. de O. S.; PANTALEãO, A. de A.; CAPRISTO-SILVA, G. F.; FACCO, C. U. Twenty-year impact of fire foci and its relationship with climate variables in brazilian regions. **Environmental Monitoring and Assessment**, v. 194, p. 1–17, 2 2022. ISSN 15732959. Available from: <<https://link.springer.com/article/10.1007/s10661-021-09702-x>>. 75

TOMAS, W. M.; ROQUE, F. de O.; MORATO, R. G.; MEDICI, P. E.; CHIARAVALLOTI, R. M.; TORTATO, F. R.; PENHA, J. M.; IZZO, T. J.; GARCIA, L. C.; LOURIVAL, R. F.; GIRARD, P.; ALBUQUERQUE, N. R.; ALMEIDA-GOMES, M.; ANDRADE, M. H. S.; ARAUJO, F. A.; ARAUJO, A. C.; ARRUDA, E. C.; ASSUNçãO, V. A.; BATTIROLA, L. D.; BENITES, M.; BOLZAN, F. P.; BOOCK, J. C.; BORTOLOTTO, I. M.; BRASIL, M. da S.; CAMILO, A. R.; CAMPOS, Z.; CARNIELLO, M. A.; CATELLA, A. C.; CHEIDA, C. C.; CRAWSHAW, P. G.; CRISPIM, S. M.; JUNIOR, G. A.; DESBIEZ, A. L.; DIAS, F. A.; EATON, D. P.; FAGGIONI, G. P.; FARINACCIO, M. A.; FERNANDES, J. F.; FERREIRA, V. L.; FISCHER, E. A.; FRAGOSO, C. E.; FREITAS, G. O.; GALVANI, F.; GARCIA, A. S.; GARCIA, C. M.; GRACIOLLI, G.; GUARIENTO, R. D.; GUEDES, N. M.; GUERRA, A.; HERRERA, H. M.; HOOGESTEIJN, R.; IKEDA, S. C.; JULIANO, R. S.; KANTEK, D. L.; KEUROGHLIAN, A.; LACERDA, A. C.; LACERDA, A. L.; LANDEIRO, V. L.; LAPS, R. R.; LAYME, V.; LEIMGRUBER, P.; ROCHA, F. L.; MAMEDE, S.; MARQUES, D. K.; MARQUES, M. I.; MATEUS, L. A.; MORAES, R. N.; MOREIRA, T. A.; MOURãO, G. M.; NICOLA, R. D.; NOGUEIRA, D. G.; NUNES, A. P.; CUNHA, C. N.; OLIVEIRA, M. D.;

OLIVEIRA, M. R.; PAGGI, G. M.; PELLEGRIN, A. O.; PEREIRA, G. M.; PERES, I. A.; PINHO, J. B.; PINTO, J. O.; POTT, A.; PROVETE, D. B.; REIS, V. D. dos; REIS, L. K. dos; RENAUD, P. C.; RIBEIRO, D. B.; ROSSETTO, O. C.; SABINO, J.; RUMIZ, D.; SALIS, S. M.; SANTANA, D. J.; SANTOS, S. A.; SARTORI Ângela L.; SATO, M.; SCHUCHMANN, K. L.; SCREMIN-DIAS, E.; SEIXAS, G. H.; SEVERO-NETO, F.; SIGRIST, M. R.; SILVA, A.; SILVA, C. J.; SIQUEIRA, A. L.; SORIANO, B. M.; SOUSA, L. M.; SOUZA, F. L.; STRUSSMANN, C.; SUGAI, L. S.; TOCANTINS, N.; URBANETZ, C.; VALENTE-NETO, F.; VIANA, D. P.; YANOSKY, A.; JUNK, W. J. Sustainability agenda for the Pantanal wetland: perspectives on a collaborative interface for science, policy, and decision-making. **Tropical Conservation Science**, v. 12, 9 2019. ISSN 19400829. Available from: <<https://journals.sagepub.com/doi/10.1177/1940082919872634>>. 77

UHL, C.; KAUFFMAN, J. B.; BOONE, J. Deforestation, fire susceptibility, and potential tree responses to fire in the eastern Amazon. **Ecology**, v. 71, p. 437–449, 1990. 76

VASCONCELOS, H. L.; MARAVALHAS, J. B.; CORNELISSEN, T. Effects of fire disturbance on ant abundance and diversity: a global meta-analysis. **Biodiversity and Conservation**, v. 26, p. 177–188, 1 2017. ISSN 15729710. Available from: <<https://link.springer.com/article/10.1007/s10531-016-1234-3>>. 77

VEENENDAAL, E. M.; TORELLO-RAVENTOS, M.; MIRANDA, H. S.; SATO, N. M.; OLIVERAS, I.; LANGEVELDE, F. van; ASNER, G. P.; LLOYD, J. On the relationship between fire regime and vegetation structure in the tropics. **New Phytologist**, v. 218, p. 153–166, 4 2018. ISSN 14698137. 1

WEES, D. van; WERF, G. R. van der; RANDERSON, J. T.; ANDELA, N.; CHEN, Y.; MORTON, D. C. The role of fire in global forest loss dynamics. **Global Change Biology**, v. 27, p. 2377–2391, 2021. ISSN 1365-2486. 3

WERF, G. R. V. D.; RANDERSON, J. T.; COLLATZ, . J.; GIGLIO, L.; KASIBHATLA, P. S.; ARELLANO, A. F.; OLSEN, S. C.; KASISCHKE, E. S. Continental-scale partitioning of fire emissions during the 1997 to 2001 El Niño/La Niña period. **Science**, v. 303, p. 73–76, 2004. 1, 2

WERF, G. R. V. D.; RANDERSON, J. T.; GIGLIO, L.; COLLATZ, G. J.; MU, M.; KASIBHATLA, P. S.; MORTON, D. C.; DEFRIES, R. S.; JIN, Y.; LEEUWEN, T. T. V. Global fire emissions and the contribution of deforestation, savanna, forest, agricultural, and peat fires (1997–2009). **Atmospheric Chemistry and Physics**, v. 10, p. 11707–11735, 2010. ISSN 16807324. 1

WERF, G. R. V. der; MORTON, D. C.; DEFRIES, R. S.; GIGLIO, L.; RANDERSON, J. T.; COLLATZ, G. J.; KASIBHATLA, P. S. Estimates of fire emissions from an active deforestation region in the southern Amazon based on satellite data and biogeochemical modelling. **Biogeosciences**, v. 6, p. 235–249, 2009. ISSN 17264189. 14

WILLIAMS, F. A. Urban and wildland fire phenomenology. **Progress in Energy and Combustion Science**, v. 8, p. 317–354, 1982. 9