UNIVERSIDADE FEDERAL DE MINAS GERAIS
INSTITUTO DE CIÊNCIAS BIOLÓGICAS
DEPARTAMENTO DE BIOLOGIA GERAL
Programa de Pós-Graduação em Ecologia, Conservação e Manejo da Vida Silvestre

Fire in the Canastra National Park: background, challenges and solutions

Eugênia Kelly Luciano Batista

Belo Horizonte
2017

UNIVERSIDADE FEDERAL DE MINAS GERAIS

INSTITUTO DE CIÊNCIAS BIOLÓGICAS
DEPARTAMENTO DE BIOLOGIA GERAL
Programa de Pós-Graduação em Ecologia, Conservação e Manejo da Vida Silvestre

Fire in the Canastra National Park: background, challenges and solutions

Eugênia Kelly Luciano Batista

Orientador: Prof. Dr. José Eugênio Cortes Figueira
Co-orientador: Prof. Dr. Christian Niel Berlinck
Co-orientador: Prof. Jeremy Russell-Smith

Tese apresentada ao curso de Pós-Graduação em Ecologia, Conservação e Manejo da Vida Silvestre do Instituto de Ciências Biológicas da Universidade Federal de Minas Gerais, como requisito para obtenção do título de Doutorado em Ecologia.

Belo Horizonte

2017

À minha pequena grande Família, dedico.

AGRADECIMENTOS

À minha mãe (Graça) e ao meu esposo (Joselmar), o meu agradecimento maior. Agradeço pelo apoio incondicional em todas as etapas deste trabalho e em todas as decisões que tomei durante esta jornada. Agradeço por terem me ouvido falar durante horas sobre meu trabalho, sempre com atenção e interesse; pelas contribuições e sugestões que fizeram e principalmente por todas aquelas conversas que nos momentos difíceis me fizeram ter a certeza de que tudo ia dar certo. Agradeço por nunca terem duvidado que eu conseguisse chegar até aqui. Pela sincera lealdade, por todos os ensinamentos, esforços e por cada sábio conselho. Obrigada por terem enxugado minhas lágrimas por tantas vezes, por me fazerem sorrir e por nunca terem deixado eu me sentir sozinha. Com muita paciência, vocês me reergueram a cada tombo e me fizeram perceber que a vida era muito mais bonita do que parecia naqueles momentos de escuridão. Vocês me provaram que não há nada mais importante na vida de alguém do que a presença das pessoas certas. À vocês, o meu mais profundo e sincero amor e gratidão.

Ao meu orientador e amigo, Prof. Dr. José Eugênio (o Zé), pelo apoio e pelas valiosas contribuições científicas. Por ter respondido minhas dúvidas e apartado minhas ansiedades no *hangouts* nas horas mais inconvenientes do dia com toda a paciência do mundo. Brigas? Claro, mas como pai e filha. Nunca como inimigos! Você (sempre) foi um grande amigo e (sempre) acreditou em mim. Isso me motivou a ser cada vez melhor. Obrigada por tudo!

To my supervisor Dr. Jeremy Russell-Smith for the opportunity, guidance and support I received in Australia. Thank you for welcoming me into your lovely home, entrusting me with your car and your personal and professional reference without asking anything in return. Thank you for the rapid return on all my demands. Your confidence in my potential motivated me to be better! Thanks also to the Australian research team (Andrew Edwards, Cameron Yates, Jay Evans, Dominique Lynch and Kamaljit Sangha) for their friendship, patience (especially with my terrible English), pleasant interaction and shared knowledge. I always felt at home, even though I was in the other side of the world!

À minha “Equipe Torrada”, composta pelos estagiários mais eficientes, competentes, dedicados e divertidos do mundo: Ana Paula Otoni, Marcella Menezes Pinto, Caio Pereira Carvalho de Souza e Henrique Godoy Corsetti Purcino. Sem vocês esse trabalho não teria acontecido e eu teria surtado no meio da Canastra!!!!!!! E o Caio...

com todo o seu deboísmo, me ajudando a salvar fotos em jpeg 300dpi (eu não saberia nem por onde começar) para a publicação. Vocês são sensacionais!

Aos professores e mestres do Instituto de Geociências da UFMG (Lussandra Martins, Úrsula Ruchkys, Marcos Antônio Timbó, Philippe Maillard e Rodrigo Nóbrega) por terem compartilhado comigo sua expertise e me ensinado conhecimentos básicos e avançados de geoprocessamento e sensoriamento remoto.

Ao amigo e Prof. Dr. Rodrigo Nóbrega, pela paciência, pelos sábios conselhos e por todas aquelas sextas-feiras dedicadas ao meu mapa de vegetação. Foram horas, dias, meses.... Se os "peixes" estão aqui é porque você me ensinou como "pesca-los".

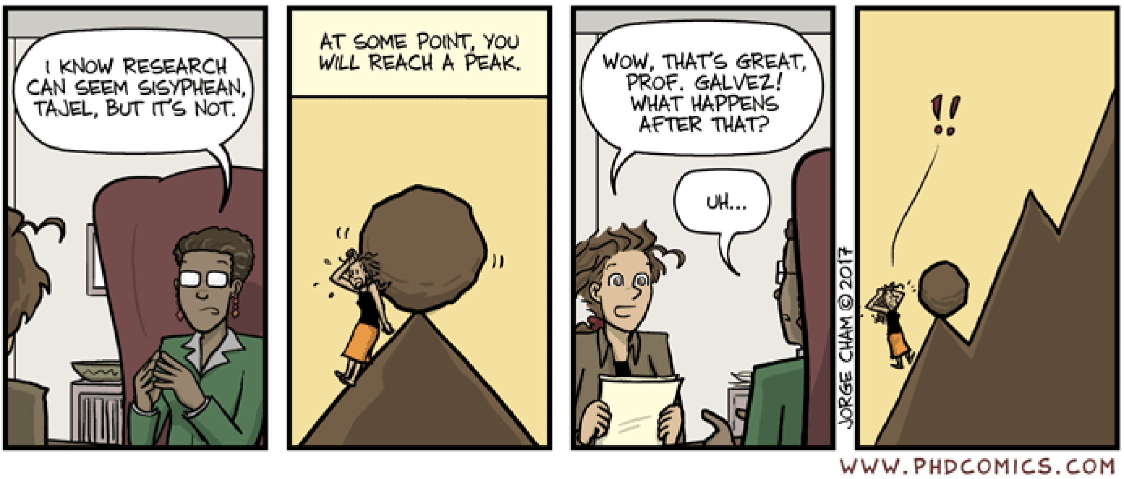
À Profa. Dra. Helena França pelo apoio e suporte nas interpretações dos mapeamentos e por ter dividido conosco seu grande conhecimento sobre o regime de fogo na Serra da Canastra.

Ao Parque Nacional da Serra da Canastra pela estrutura de alojamentos durante nossas coletas de campo. Ao brigadista Elder por ter nos guiado pelos labirintos da Babilônia.

À NASA por prover as imagens de satélite utilizadas nesta pesquisa.

À CAPES pela concessão da bolsa de doutorado, que em razão da ausência completa de apoio do Programa de Pós-Graduação em Ecologia, Conservação e Manejo da Vida Silvestre e do Governo brasileiro por meio de suas instituições de "fomento à Pesquisa", acabou sendo utilizada para financiar minhas coletas de campo (diárias do motorista da UFMG, alimentação para a equipe, equipamentos, etc.), a apresentação dos meus resultados no Savanna Meeting em 2017 na África do Sul e meu doutorado sanduíche na Austrália em 2016. Apesar de ter comprometido meu sustento e ter gerado enormes transtornos econômicos, considero que foram grandes investimentos na minha edificação pessoal e profissional.

A todos vocês e muitos outros que indiretamente contribuíram para esta tese, os meus sinceros agradecimentos.


I KNOW RESEARCH CAN SEEM SISYPHEAN, TAJEL, BUT IT'S NOT.
AT SOME POINT, YOU WILL REACH A PEAK.
WOW, THAT'S GREAT, PROF. GALVEZ! WHAT HAPPENS AFTER THAT?
UH...
JORGE CHAM © 2017
!!
WWW.PHDCOMICS.COM

SUMÁRIO

LISTA DE FIGURAS

LISTA DE TABELAS

RESUMO

Todos os anos nós assistimos aterrorizados a queima e destruição de centenas de hectares de vegetação nativa. São milhares de plantas e animais mortos direta ou indiretamente por incêndios de grandes extensões e alta severidade, majoritariamente provocados por ações humanas. Manejar o fogo em áreas protegidas do Cerrado, especialmente em zonas de conflito de interesses, não e uma tarefa trivial. Há décadas a legislação brasileira tem promovido políticas de combate e supressão total do fogo na tentativa de conservar a biodiversidade nas savanas sem considerar que o fogo tem moldado a evolução de plantas e ciclos biogeoquímicos por milhares de anos nesses ecossistemas. Neste trabalho, utilizamos o Parque Nacional da Serra da Canastra (PNSC) para ilustrar como as atuais políticas de supressão podem resultar em regimes de fogo inapropriados e divergentes dos objetivos de conservação desejados. No primeiro capítulo, publicado no Journal of Environmental Management, abordamos o longo histórico de ocupação e uso do fogo para agropecuária muito antes do ParNa Canastra ser criado, o desastroso processo de desocupação da área do parque e o estabelecimento da política de supressão de fogo após sua criação, que levaram ao regime de fogo de incêndios frequentes e fora de controle na zona regularizada do parque. Reconstituímos o regime de fogo histórico (2000-2015) no parque por meio de análises de geoprocessamento e sensoriamento remoto, utilizando imagens dos satélites Landsat. Demonstramos que i) a região alta do parque é um sistema de memória curta, que se recupera rapidamente dos incêndios, sujeitando-o a incêndios frequentes e de grande extensão; ii) os regimes de fogo são diferentes entre as regiões regularizadas e não regularizadas e iii) o uso tradicional de fogo promove tanta queima quanto os incêndios fora de controle, apontando para a urgência da resolução de conflitos entre administradores do parque e população local e da necessidade do uso do fogo de forma controlada para preservação da biodiversidade. No segundo capítulo, nós discutimos como um regime de fogo inadequado pode causar alterações nos ecossistemas e desencadear a perda de habitats e espécies. Na tentativa de minimizar esses impactos e atingir os objetivos de conservação da biodiversidade, são propostos limiares máximos e mínimos relacionados aos diferentes aspectos do regime de fogo, tais como: frequência, sazonalidade e extensão. Padrões muito distantes dos limites aceitáveis deverão alertar gestores sobre a possibilidade de que o regime de fogo vigente possa comprometer a biodiversidade a médio e longo prazos.

ABSTRACT

Every year we watched terrified at the burning and destruction of hundreds of acres of native vegetation. Thousands of plants and animals are killed directly or indirectly by fires with large extensions and high severity, mostly caused by human actions. Managing fire in protected areas of the Cerrado, especially in areas with conflict of interest, is not a trivial task. For decades, Brazilian legislation has promoted policies to combat and suppress total fire in an attempt to conserve biodiversity in savannas without considering that fire has shaped the evolution of plants and biogeochemical cycles for thousands of years on these ecosystems. In this study, we used the example of the Canastra National Park (CNP) to illustrate how current suppression policies can result in inappropriate fire regimes distant from the desired biodiversity conservation goals. In the first chapter, recently published in the Journal of Environmental Management, we have addressed the long history of occupation and use of fire for agricultural purposes long before ParNa Canastra was created, the violent and disastrous regularization process implemented by the Government and the establishment of fire suppression policies after its creation, that led to the fire regime of frequent and severe fires in the Canastra region. We reconstructed the historical fire regime (2000-2015) in the CNP through GIS and remote sensing analyzes using images from the Landsat satellites. We show that i) the Canastra region is a short memory system, which recovers rapidly after burning, being subject to frequent and extensive fires; (ii) fire regimes are different between regulated and non-regulated areas; and (iii) traditional fire use promotes as much burning as out-of-control fires, highlighting the urgent need to settle conflicts between park managers and the local population, in addition to use prescribed fires aiming to deliver the biological conservation outcomes. In the second chapter, we used the accumulated knowledge available in the literature to discuss how an inappropriate fire regime can promote changes in ecosystems and result in the loss of habitats and species. In an attempt to reduce these negative effects and achieve the biodiversity conservation goals, maximum and minimum thresholds related to the different aspects of fire regimes are proposed, such as: frequency, seasonality and extension. Fire regimes far beyond acceptable thresholds should act as a warning to managers about the possibility that current fire patterns may not be able to guarantee the biodiversity conservation in medium and long term.

CAPÍTULO I - AN EVALUATION OF CONTEMPORARY SAVANNA FIRE REGIMES IN THE CANASTRA NATIONAL PARK, BRAZIL: OUTCOMES OF FIRE SUPPRESSION POLICIES

Resumo

O fogo tem moldado a evolução de plantas e ciclos biogeoquímicos por milhares de anos em ecossistemas savânicos, mas as alterações nos regimes de queima provocadas pelo uso antrópico da paisagem têm ameaçado os esforços de conservação da biodiversidade. Em unidades de conservação do Cerrado, as políticas de supressão do fogo ainda são predominantes, refletindo uma herança cultural que considera o fogo como um elemento causador de impactos negativos nos ecossistemas. Neste estudo, nós comparamos os regimes de incêndios no Parque Nacional da Serra da Canastra (PNSC) associados a diferentes tipos de manejo de fogo por meio de imagens Landsat no período de 16 anos (2000-2015). Nos campo limpos do chapadão da Canastra, as políticas de supressão do fogo são sancionadas e aplicadas pelo governo, enquanto na região da Babilônia, o manejo do fogo é aplicado pelos proprietários rurais com o objetivo de renovar pastagens e plantações. O regime de fogo no chapadão da Canastra é caracterizado por poucos incêndios, severos e de grande extensão, recorrendo em intervalos de dois anos. Por outro lado, o regime de fogo no vale da Babilônia é caracterizado por muitos incêndios pequenos que se iniciam no começo da estação seca, enquanto nos chapadões da Babilônia, os incêndios de grandes extensões são mais frequentes. Nossos resultados ilustram os grandes desafios envolvidos no manejo de fogo em ecossistemas savânicos inseridos em áreas de conflito de interesses. Nós sugerimos que o planejamento do manejo de fogo no PNSC leve em consideração: i) os conflitos entre as políticas do governo e as necessidades das comunidades locais; e ii) a adoção de práticas estratégicas para o estabelecimento de regimes de fogo ecologicamente sustentáveis. Este estudo levanta discussões relevantes para a conservação da biodiversidade em savanas brasileiras e sul-americanas, em geral.

Palavras-chave: cerrado; incêndios antrópicos; regime de fogo; supressão do fogo; conservação da biodiversidade

Abstract

Fire has shaped plant evolution and biogeochemical cycles for millions of years in savanna ecosystems, but changes in natural fire regimes promoted by human land use threaten contemporary conservation efforts. In protected areas in the Brazilian savannas (Cerrado), the predominant management policy is fire suppression, reflecting a cultural heritage which considers that fire always has a negative impact on biodiversity. Here we compare resultant fire-regimes in Canastra National Park (CNP), southeast Brazil, associated with areas under and without fire suppression management, based on a 16-year Landsat imagery record. In open grasslands of the Canastra plateau (CP), firefighting is undertaken under government-sanctioned regulation, whereas in the Babilonia sector, non-sanctioned fire management is undertaken by small farmers to promote cattle grazing and cropping. Fire regimes in the Canastra sector are characterized by few, very large, late dry season wildfires recurring at intervals of two years. Fire regimes in lowlands of the Babilonia sector are characterised by many small-scale, starting at the beginning of the dry season (April-July). In Babilonia uplands fire regimes are characterized by higher frequencies of large fires. The study illustrates major challenges for managing fire-prone areas in conflict-of-interest regions. We suggest that management planning in CNP needs to effectively address: i) managing conflicts between CNP managers and local communities; and ii) fire management practices in order to achieve more ecologically sustainable fire regimes. The study has broader implications for conservation management in fire-prone savannas in South America generally.

Keywords: savanna; human-caused fires; fire regime; fire suppression; biodiversity conservation

1. Introduction

Wildfires have performed an important function in the development and expansion of savannas since the late Miocene (Beerling and Osborne, 2006; Edwards et al., 2010; Simon et al., 2009). The climate changes that have taken place from this period have created favorable conditions for the expansion of flammable C4 grasses, which are conducive for carrying fire. In a feedback process, more open landscapes have favored the higher productivity and lower decomposition rates of C4 grasses, promoting the gradual replacement of fire-sensitive woodlands by fire-prone savannas and grasslands (Cerling et al., 1997; Epstein et al., 1997; Keeley and Rundel, 2005; Pagani, 1999).

Most Cerrado plant lineages exhibit an array of syndromes associated with adaptations to fire (Gignoux et al., 1997), and started to diversify ~4 Mya, coinciding with the regional expansion of the savanna biome and dominance of C4 grasses. Despite the influence of fire shaping plant evolution and global biogeochemical cycles in savannas for millions of years, recent changes in fire regimes promoted by accelerating human land use are incurring significant deleterious impacts on some vegetation types, even in fire-prone ecosystems (Andersen et al., 2005; Murphy et al., 2010; Russell-Smith et al., 2012).

The role of human activities in changing fire regime patterns has been considered at a global scale (Bowman et al., 2011, 2009; Chuvieco and Justice, 2010). Modern humans have changed natural fire regimes by clearing forests, promoting grazing, introducing plants, suppressing fires or modifying the season and amount of ignitions. On the one hand the increase in frequency of wildfires can stimulate the fire-grass cycle, eliminating woody or fire-sensitive species and promoting the maintenance of open ecosystems (Bond and Keeley, 2005; Bond et al., 2005; Ratajczak et al., 2014; Staver et al., 2011). On the other, the suppression of fires for long periods can result in vegetation encroachment (Sankaran *et al.* 2005; Smit *et al.* 2010; Murphy and Bowman 2012; Scott *et al.* 2012), or result in high-intensity wildfires due to accumulation of large amounts of flammable biomass (D’Antonio and Vitousek, 1992; Vogl J, 1979). These destructive fires, in turn, result in high greenhouse gas emissions (Edwards et al., 2015; Heckbert et al., 2012) and biodiversity losses (Andersen et al., 2005; Clarke, 2002; Hoffmann, 1999; Oliveira et al., 2015). International experience indicates that prescribed fire management in fire-prone savanna biomes can substantially reduce the risk of frequent late season fires and resultant impacts on fire-vulnerable biodiversity elements (Brockett et al., 2001;

Burrows, 2008; Russell-Smith et al., 2013a; Van Wilgen et al., 2014), and reduce greenhouse emissions (Russell-Smith, 2016; Russell-Smith et al., 2015, 2013b).

The Cerrado covers an area of approximately two million km^2 of Central Brazil and parts of Bolivia and Paraguay (Cardoso Da Silva and Bates, 2002). The biodiversity is impressive and thus it has been identified as a global biodiversity hotspot with more than 10 000 plant species, 161 mammal species, 837 bird species, 120 reptile species and 150 amphibian species, all with high rates of endemism (44, 11.8, 1.4, 20, and 30%, respectively) (Cardoso Da Silva and Bates, 2002; Kier et al., 2005; Klink and Machado, 2005; Myers et al., 2000). The Cerrado is characterized by a mosaic of vegetation types ranging from grasslands, open scrublands to dense woodlands (Coutinho, 2002; Eiten, 1978, 1972), whose spatial distribution is regulated, amongst other factors, by soil type and topography, and patterning in the timing, intensity and frequency of fire (Durigan et al., 1994; Kauffman et al., 1994; Mistry, 1998; Moreira, 2000).

The contemporary Brazilian savanna has been threatened by the absence of a consistent fire policy, and most protected areas in Brazil have continued to apply total fire suppression policies (Durigan and Ratter, 2016; Schmidt et al., 2016). These policies reflect a cultural heritage which considers that fire regimes have a significant negative impact on biodiversity, and ignore the requirement that fire is essential for the dynamics and balance of savanna ecosystems (Figueira et al., 2016). The same assumption, added to the misunderstanding that the savanna biome is a product of forest degradation (Bond and Parr, 2010; Sankaran and Ratnam, 2013), has justified the total suppression of fire in other South American countries (Bilbao et al., 2010; Myers et al., 2006, 2004).

In this study we assess the effects of current Brazilian fire suppression policy on fire regimes in a major Cerrado protected area, the Canastra National Park (CNP). Specifically, we review the historical regional land use from the 18th century, measure fuel accumulation rates and describe contemporary (2000-2015) fire regime characteristics (frequency, seasonality, size distribution) for two regions of CNP with different management policies: the government-managed Canastra region, in which a total fire suppression policy is applied; and the small landholder-managed Babilonia region, where non-sanctioned fire management is undertaken to promote an array of livelihood activities (cattle grazing and cropping). The study illustrates the significant challenges associated with implementing ecologically sustainable fire management programs in Brazilian protected areas, and South America more generally.

2. Methods

2.1. Study area

The Canastra National Park (CNP) was created in April 1972 and currently comprises an area of 197 928 ha. It is located in the State of Minas Gerais, Southeast Brazil, between latitudes 20°05’20”S and 20°11’30”S and longitudes 46°55’10”W and 46°57’25”W (Fig.1). CNP extends over six municipalities: São João Batista do Glória, São Roque de Minas, Delfinópolis, Vargem Bonita, Capitólio e Sacramento.

Originally, CNP was restricted to the Canastra Plateau, which extends over an area of 71 503 ha. However, in 2005, when a new Management Plan was released, the CNP was enlarged to cover an additional area of 126 425 ha called the Babilonia region (Fig.1). Because of ongoing land use conflicts in the region, and to allow the extraction of mineral resources (diamond), recently it has been proposed to restrict the park extent to the Canastra region solely. In January 2017, the grazing of cattle has been permitted in unconsolidated Park lands, and which has revived conflicts with the Chico Mendes Institute for Biodiversity Conservation (ICMBio), the Government Agency responsible for the management of CNP.

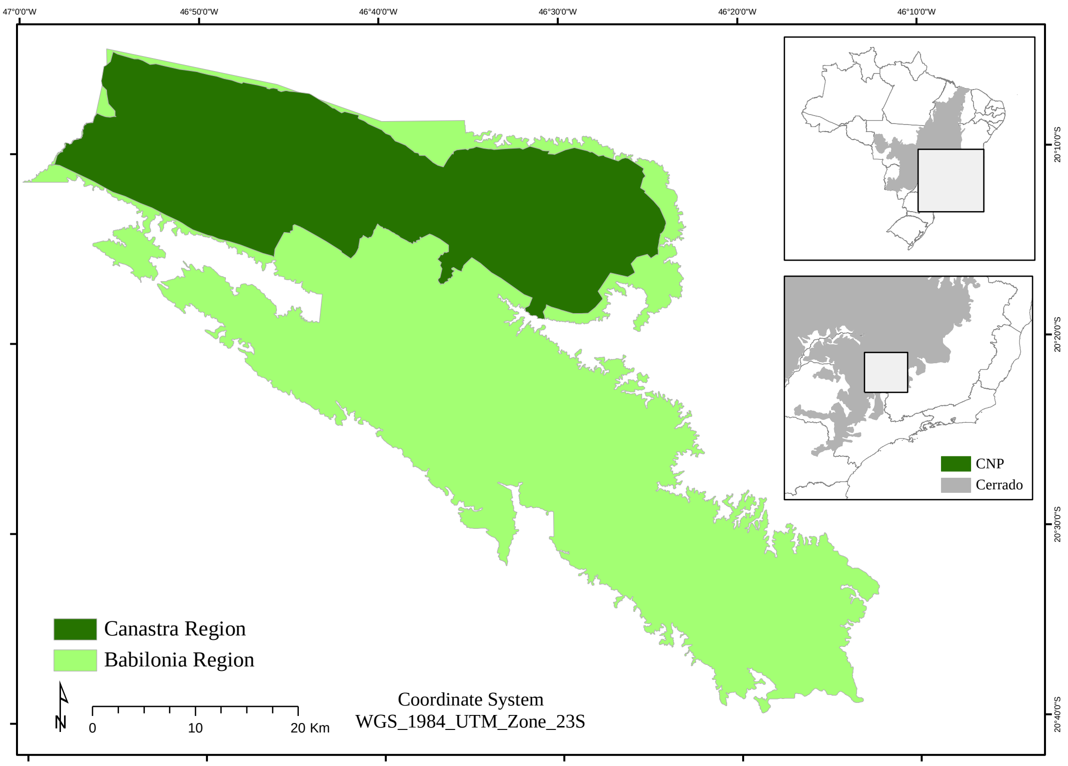


Figure 1. Geographical position of the Canastra National Park (CNP) inserted in the Cerrado biome and divided into two regions: Babilonia and Canastra.

The region is characterised by valleys surrounded by steep slopes and alternating plateaus with NW-SE orientation (elevation 690-1500 m). The geological structure comprises Middle Proterozoic quartzite that shapes the local landscape, either in raised zones represented by the plateau, or in slopes and valleys carved in softer materials based on shales, phyllites and clayey quartzite. Soils, where present, are shallow or moderately deep infertile latosols.

Vegetation cover is mostly open grasslands with scattered emergent shrubs and trees. Wooded savannas may occur on deeper latosols with clayey texture, and forests are usually associated with water courses, at interflows, and on soils of greater fertility. The valleys are dominated by crops and pastures planted with invasive grasses (*Melinis minutiflora* and *Brachiaria sp*.). The CNP supports a variety of grazing and browsing mammal species, notably cervids such as pampas deer (*Ozotoceros bezoarticus*) and gray brocket deer (*Mazama gouazoubira*).

2.1.1. Tracks and Roads

The road map was elaborated by using ten scenes of RapidEye constellation, which were acquired from the Brazilian Environment Ministry (MMA). All the scenes were joined but, due to temporal difference between them, the histograms of bands needed to be harmonized in order to make contrast and the brightness compatible. The layout edition was done manually in natural color with the software ArcGis 10.0 (Fig.2a).
The road infrastructure in CNP differs between regions. On the Canastra Plateau the roads are larger but smaller in number, and used as firebreaks by the park managers (Fig.2b). The Babilonia region is heavily dissected by small rural, poorly maintained roads connecting farms (Fig.2c).

2.1.2. Land cover and vegetation

A land cover map of CNP was created by using OLI/Landsat-8 scenes acquired in the late dry season of 2015, and joined to create an image covering the whole protected area. Late dry season imagery was used to avoid the high cloud density typical of rainy seasons. Mapping was performed using Geographic Object-Based Image Analysis (GEOBIA) with the software eCognition Developer 8.7, applying standard techniques (Flanders et al., 2003). Mapping surfaces included standard vegetation indices: Plant Senescence Reflectance Index (PSRI), Normalized Difference Vegetation

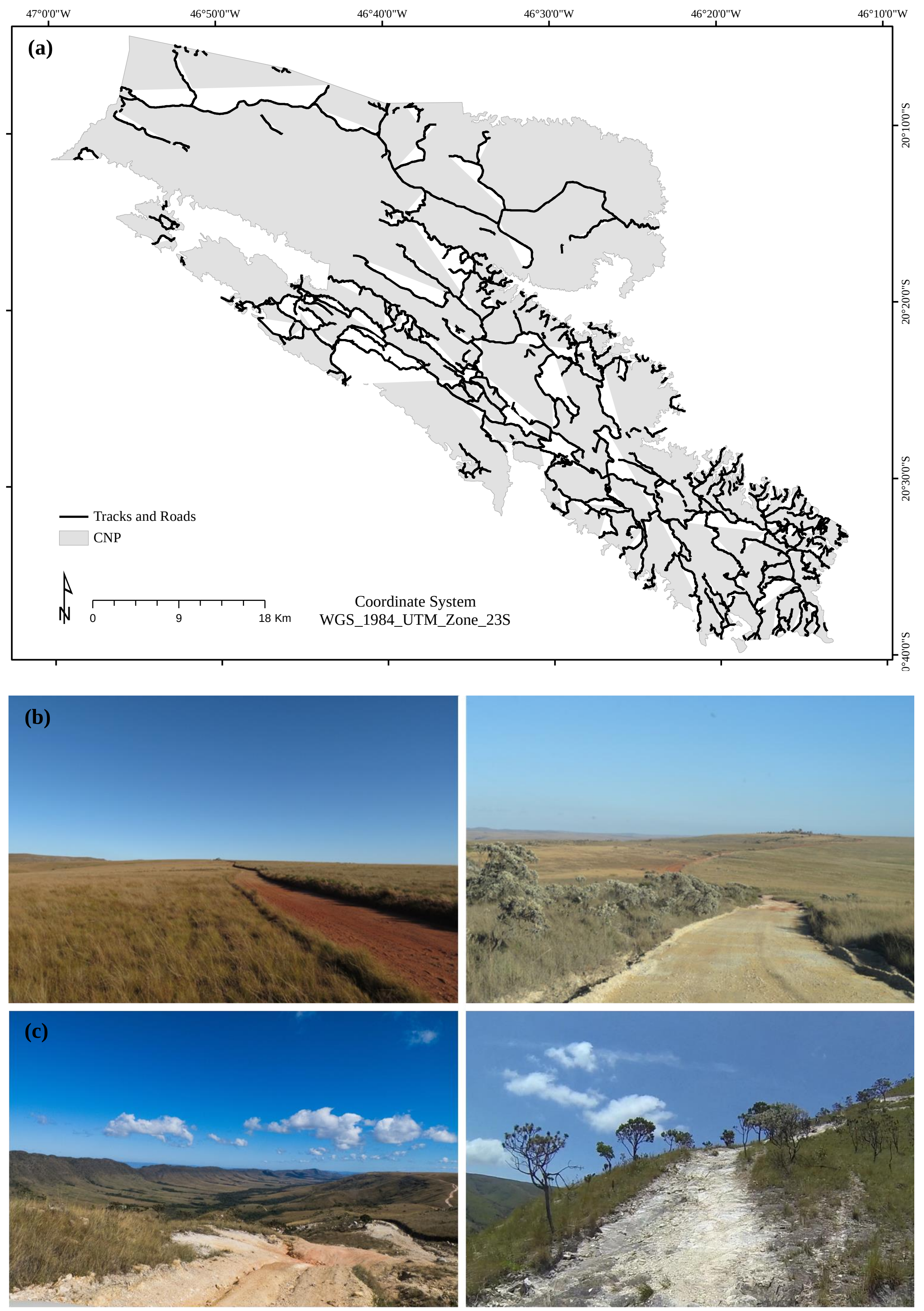


Figure 2. (a) Roads and tracks in the CNP; (b) Main roads, which act as firebreaks in the Canastra Plateau and (c) Poorly maintained roads in the Babilonia region.

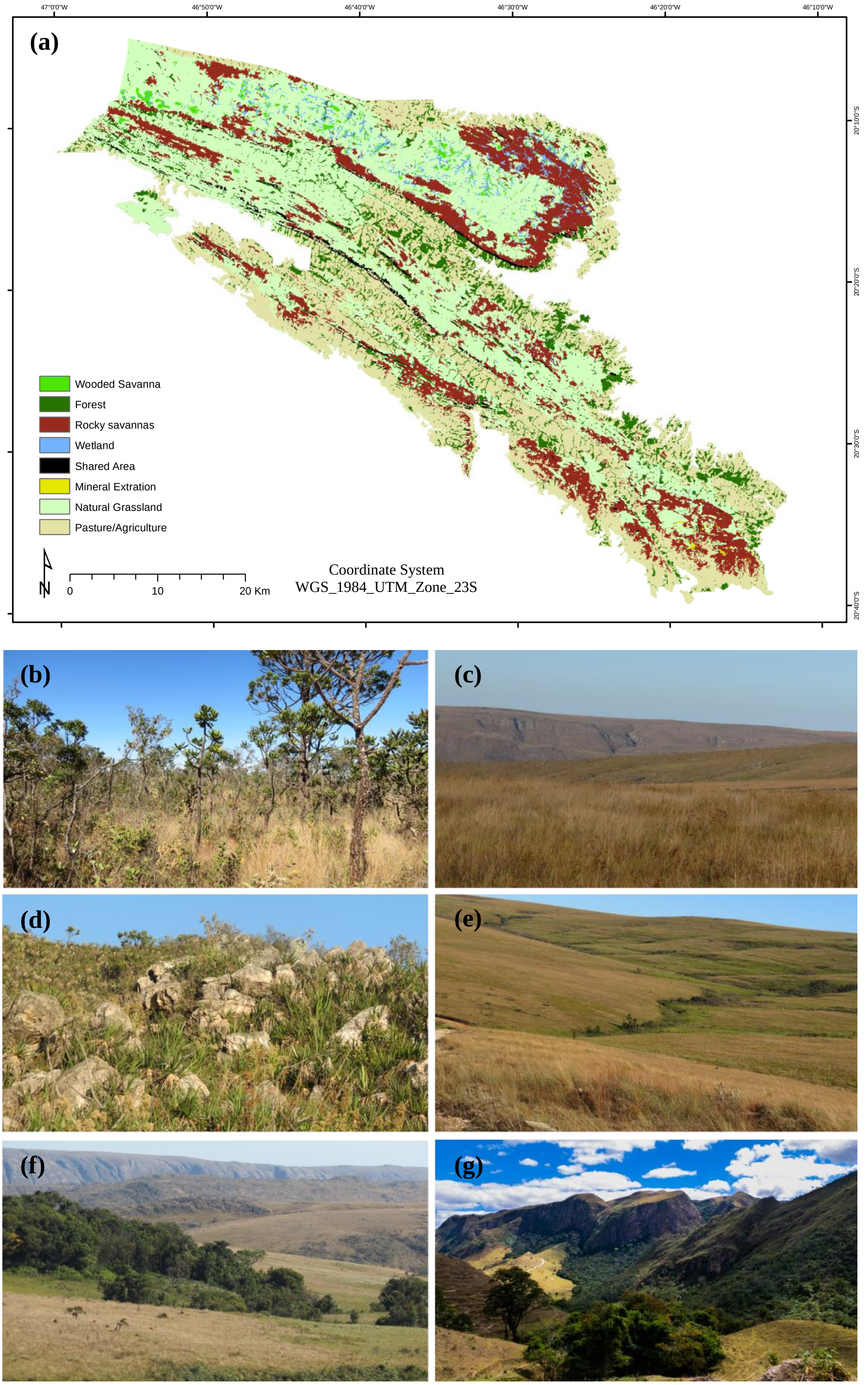


Figure 3. (a) Vegetation types in the CNP. (b) Wooded savanna; (c) Natural grasslands; (d) Rocky savannas; (e) Wetlands; (f) Forest patches in Canastra plateau; (g) Forest patches in Babilonia valley.

Index (NDVI), Visible Atmospherically Resistant Index (VARI), Visible Green Index (VIg), Triangular Vegetation Index (TVI), Modified Triangular Vegetation Index (MTVI), Char Soil Index (CSI), Normalized Burn Ratio (NBR), Mid-Infrared Burn Index (MIRBI), Soil Adjusted Vegetation Index (SAVI), Enhanced Vegetation Index (EVI), and elevation and slope surfaces derived from ASTER (Advanced Spaceborne Thermal Emission and Reflection Radiometer) imagery.

There are three main land types in CNP: (1) valleys, predominantly occupied by pastures and agricultural activities (PA, 28% of CNP); (2) natural woody vegetation, comprising wooded savannas (WS, 1% of CNP) on flat to gently undulating terrain, and riparian and mesophilic forests (FO, 9% of CNP) associated with watercourses; and (3) natural grasslands (GR, 41% of CNP) which predominate on flat surfaces of the Canastra Plateau, and rupestrian grasslands (RG, 17% of CNP) on more stony soils (Fig. 3).

2.1.3. Climate and seasonality

Climatic aspects have great influence on the characteristics of the fire regime. High precipitation rates favor the accumulation of biomass fuel that can dry quickly in the dry season. The climate in CNP is markedly seasonal with high rainfall in the wet season (~ 1500 mm), from October to April, and a dry season from May through September (~ 10% of annual precipitation). Temperatures throughout the year vary across the Park, ranging between 16º to 20ºC in the Canastra sector, and from 18º to 23ºC in the Babilonia region. According to the Köppen classification, the climate in the Canastra region is Cwa, with dry winters and hot summers (Alvares et al., 2013).

To evaluate the extent of the dry season, a daily time-series of rainfall and temperature over the sixteen-year period (2000-2015) was acquired from the Energy Company of Minas Gerais (CEMIG), and downloaded from the online database of the National Institute of Meteorology (INMET). Because of differences in elevation and the distance between Canastra and Babilonia regions, two weather stations were used as a data source: the first, located in São Roque de Minas, represented the weather conditions in the Canastra region, while the latter, located in Passos, represented conditions in the Babilonia region. A Pearson's correlation matrix was generated aiming to identify similarities between data from these weather stations and others in the same region. Gaps in the time-series were interpolated from above sources using linear regression.

For both weather stations the water balance was calculated for the sixteen-year period (2000-2015) through the Thornthwaite and Mather (1955) method, described by

Tubelis and Nascimento (1980). This method calculates water availability in the soil accessible to vegetation. Water balance (excess, deficiency) is computed from precipitation inputs and evapotranspiration outputs. The dry season in CNP was defined by the months in which the water deficit and recharge (outputs > inputs) occurred in both meteorological stations (April to October) (Fig.4). On this basis, the early dry season (EDS) was defined as the period from April to July, and the late dry season (LDS) as the period from August to October. The ratio between LDS and EDS was used to provide an estimate of the seasonal distribution of burned areas in each management zone.

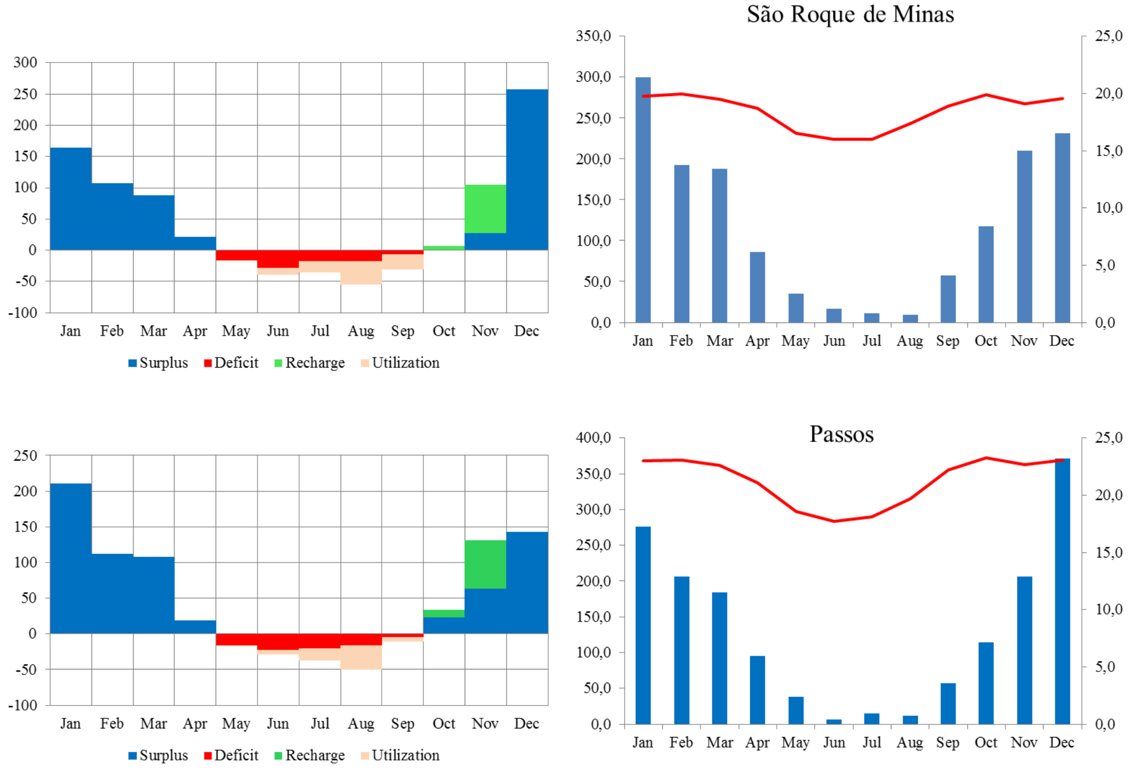


Figure 4. Monthly water balance in São Roque de Minas and Passos, 2000 to 2015. Due to low amount of water in the system, the dry season was defined as the period from April to October.

2.2. Fire management zones

For this study, we divided the whole area of CNP into three fire management zones (Fig.5): (1) Canastra plateau (CP - 71 503ha), a large natural and continuous grassland interspersed with small highly disconnected forest fragments, low density of roads, and government-regulated 'zero-fire' policy; (2) the Babilonia plateau (BP - 47 331), which has similar characteristics, but in addition to being dissected by high density of roads, non-sanctioned fire management is undertaken to promote an array of livelihood activities (cattle grazing, cropping); and (3) the Babilonia valley (BV - 79 093), where small farmers usually manage their properties using prescribed fire and introduced grasses in a farming landscape dissected by larger forest patches.

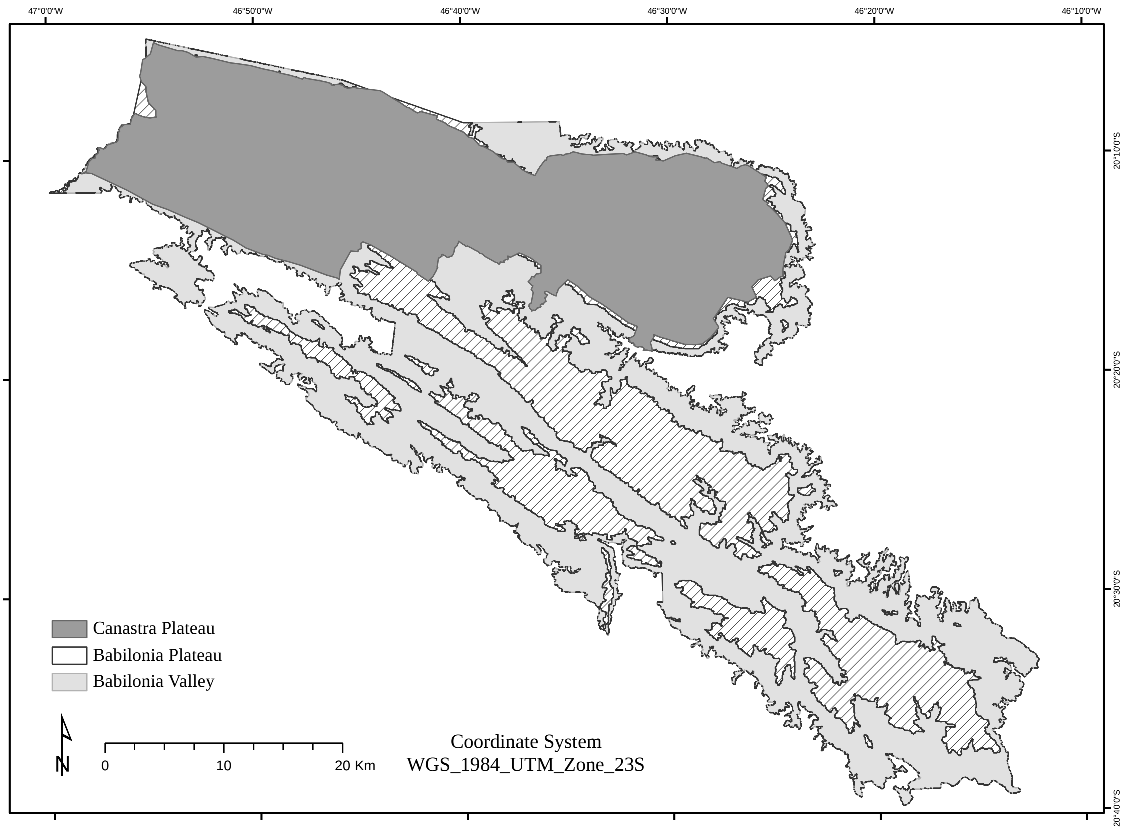


Figure 5. Fire management zones in the CNP.

2.3. Burned area mapping

Burned area mapping was performed using Landsat imagery, including Thematic Mapper (TM), Enhanced Thematic Mapper plus (ETM+), and Operational Land Imager (OLI). Assembled data comprised a time series of 455 images (path/row 220/074 and 219/074) from 2000 to 2015, downloaded from the U.S. Geographical Survey (http://earthexplorer.usgs.gov). All useful (cloud-free) images were analyzed for the purposes of reducing omission errors, since burn scars in tropical savannas usually disappear in a few weeks (Pereira, 2003).

The Landsat-based burned area classification was conducted by visual interpretation of the orbital images and manual scanning of the burned polygons, with ArcGis 10.0. This manual technique was selected assuming that automatic procedures often confuse burned areas with other elements of the landscape that have similar spectral patterns, such as water courses, cloud shadows or topographical features. To better discriminate fire scars in the landscape, images were manipulated as Red-Green-Blue color composites, in bands from the mid-infrared, near-infrared and red, respectively. The burned polygons were validated by using the Fire Occurrence Reports (ROIs) provided by CNP as a guide. The date of each event was estimated through the overlapping of the polygons with daily hotspots from the MODIS (Moderate Resolution Imaging Spectroradiometer) and GOES (Geostationary Operational Environmental) sensors, available from the Wildfires Database of the National Institute for Space Research (INPE). This method provided a close estimation of the fire occurrence date, since Landsat images have lower (16-day) temporal resolution.

We separated individual years into five categories according to the percentage of area burned: very small (<1% of the land surface in the fire management zone burned); small (1-10% burned); medium (10-30% burned); large (30-50% burned); and very large (>50% burned). Fire events were assigned into five size classes: <10 hectares; 10-100 hectares; 100-1000 hectares; 1000-10 000 hectares; 10 000-100 000 hectares.

2.4. Fire frequency

Fire frequency analysis was performed with the software ArcGis 10.0, where all polygons occurring in the same year were summed to create an annual burned area layer. Annual fire maps were used to generate fire frequency distributions, where each pixel assumed numerical values corresponding to the number of fires superimposed over the 16 years.

2.5. Fire rotation period and fuel accumulation

To determine the fire rotation period (FRP), annual burned maps of the three management zones were converted into a binary raster with 30m of spatial resolution, assigning the value 1 for burned areas and 0 for unburnt. By paired analyzes of annual maps, we determined for each pixel the time interval between two consecutive fires. We verified the occurrence of annual, biannual, triennial and so on, up to the maximum interval of 15 years and finally, we calculated the area burned at each fire rotation. We used the software IDRISI Taiga to perform this analysis.

To evaluate the rate of fuel accumulation in CNP natural grasslands, biomass sampling was undertaken in August 2015. The sampling sites were stratified by mapping of time-since-last-fire and divided into three time classes: < 1 year; 1 - 3 years; > 3 years. For each time class, five sampling points, each with ten 0.5 X 0.5 m quadrats, were established giving a total of 150 samples / time class. All fine biomass in sample quadrats was collected and subsequently separated into live and dead components, then oven-dried and weighed.

We also measured fuel connectivity considering just the grass layer. At each of above 5 biomass sampling points, ground cover was estimated at five random linear 5 m transects, giving a total of 75 transects. The coverage ratio was calculated based on the sum of ground portions covered by vegetation in each transect, divided by its total length. A critical threshold of 60% cover may be responsible for creating the connectivity needed to provide the spatial flammable substrate over which fire can spread indefinitely (Abades et al., 2014; Finney, 2001; Loehle, 2004).

2.6. Statistical analyzes

To test for statistical differences in the proportion of burned area per year and EDS-LDS sub-seasons between management zones, non-parametric tests (Friedman's test) were performed since normality or homogeneity of variance was violated. If a significant value was obtained, we used a post-hoc test to evaluate which datasets were different. To do this, we used the R packages, PMCMR and rcompanion. A Friedman's test based on weighted averages was also used to compare times burned and fire rotation periods between the three fire management zones.

To assess whether fires in a given year can affect the patterns of burning in the following year, we used Pearson correlation coefficient for each fire management zone and then we compared the three regression lines using an analysis of covariance (ANCOVA).

A permutation test followed by post hoc tests (R packages rcompanion and coin), were used to compare the ratios (*BA:NF*) between burnt areas and number of fires in the three zones of CNP.

We conducted a Chi-square test to compare fire sizes among management zones. Even though the fires may be initiated in the Babilonia region and later invade the Canastra plateau, we assumed that all observations are independent because the fire spreading is not an obvious process and depends on the particular characteristics of each site.

A General Linear Model (GLM) was run to identify significant effects of time after fire on the ground cover. GLMs were fitted using the function glm from the R package stats. Concurrently, we applied a Kruskal-Wallis test to examine the influence of time on the total biomass accumulation (fine fuel) and dry-wet ratio. We have chosen

Kruskal-Wallis tests instead of One-way ANOVA, because the assumptions of normality and equal variance on the scores across groups were not met.

3. Results

3.1. Historical background and fire policies in the CNP

The first social conflicts in Canastra region emerged with the gold-mining in the 18th century and have persisted even after the exhaustion of auriferous resources, when lands that were once dominated by Indigenous people and fugitive slaves were occupied by large landowners and small farmers (Barbosa, 2007; Fernandes, 2012; Ferreira, 2013). The agro-pastoral system that was implemented in the region during the 19th century was characterized by large farms for livestock and family-based agriculture developed on small land holdings. In this traditional system, the cattle remained in the lowlands, close to the slopes of the mountain during the rainy season (October to April). Soon after the first rainfalls, extensive areas of natural pastures on the plateau were burned by local farmers, allowing the access of cattle to vegetation with high nutritional value during the dry season (May to September) (Barbosa and Júnior, 2006; Goulart, 2013; Saint-Hilaire, 1975). In general, the management of these natural pastures was made on plots with biennial rotation and use of firebreaks around springs, forests and pasture borders.

Although the Park was created in 1972, the efforts to promote landholding regularization began in 1974, when many farms were taken over by the government on the grounds that these properties would be of social interest. This claim enabled the government to remove families from their properties with no obligation to pay reimbursement, except by Public Debt Security. These government securities could last from 20 to 30 years, until such time as they could be repaid (Ferreira, 2015). Conflicts have grown steadily in subsequent decades, including from 2005, when a new Management Plan promoted the integration of more areas within the limits of the Park. This Plan of Management restricted access to natural resources and imposed policies reinforcing the criminalization of fire use practices in pasture renewal. Without alternatives, farmers have kept using fire illegally to manage their properties or as a way of protest, resulting in undesirable fire regimes characterized by large and severe wildfires (Moura, 2013).

From an institutional perspective, local residents still represent the main threat to the conservation goals of the protected area (IBAMA, 2005). In response, managers have applied fire suppression policies including implementation of five meter-wide firebreaks constructed along roads and tracks.

Historically, Brazilian legislation aimed to avoid or restrict the use of fire, especially in protected areas (Schmidt et al., 2016). Recently, the New Brazilian Forestry Code (Law 12 651/2012) has reiterated the requirement for seeking prior permission for using fire in protected areas in accordance with respective Management Plans, but only under prior approval of the managing agency responsible for the protected area. However, despite all this preventive legislation, large, intense late dry season wildfires are common in Brazilian protected areas (Gonçalves et al., 2011; Júnior et al., 2013; Mesquita et al., 2011; Silva et al., 2011), emphasizing the gap between fire management policies and realities. For instance, fires burnt large areas on the CP in 2002, including the habitat of mergansers *Mergus octosetaceus*, a Critically Endangered species at the global level according to IUCN criteria (BirdLife International, 2015), and in Brazil (BRASIL, 2014; Hughes et al., 2006). Only recently, in 2014, strategies were proposed with the aim of joining research with management as part of the development of an Integrated Fire Management Program in the CNP (Souza et al., 2016).

3.2. Burned Area

Over the 16-year assessment period (2000 to 2015), a total of 925 543 ha was burned, which is equivalent to almost four times the size of CNP. An annual mean of 22 069 ha (30.8% of the CP area) of the Canastra plateau was burnt, while the Babilonia plateau burnt at an annual average of 20 826 ha (44% of the BP area), and the Babilonia valley 14 952 ha (18.9% of the BV area) (Tab.1). The proportion of areas burned annually differ significantly between BP and the other two fire management zones (Friedman's test = 18.375, $p = 0.0001023$; post hoc tests were significant between BP-BV, p-value < 0.001 and BP-CP, p-value $= 0.032$).

Fires in a given year affect negatively burned areas in following years for the three management zones (BP, $r = -0.559$; BV, $r = -0.763$; CP, $r = -0.644$) (Fig.6). The analysis of covariance revealed that there were no significant differences in intercepts ($t = -1.315$, p-value $= 0.200$) and angular coefficients ($t = 1.448$, p-value $= 0.159$) of the regression lines relating burned area with burned area in the previous year (covariate) for CP and BP. However, the intercepts ($t = 3.942$, p-value < 0.001) and the angular coefficients ($t = -4.500$, p-value < 0.001) of the regression lines differ when comparing CP+BP and BV.

Table 1. Mean annual burned area in each fire management zone: Babilonia Plateau (BP), Babilonia Valley (BV) and Canastra Plateau (CP).

	Burned Area (hectares)		
Year	BP	BV	CP
2000	18848	12234	21733
2001	20159	11891	14860
2002	19891	19599	<u>51146</u>
2003	<u>26331</u>	14348	3671
2004	20285	16169	20786
2005	21485	11934	11922
2006	22649	16095	29964
2007	<u>24367</u>	17627	35868
2008	19871	13305	1027
2009	7057	6124	<u>454</u>
2010	<u>28299</u>	27221	<u>44498</u>
2011	19791	10927	<u>702</u>
2012	<u>25945</u>	22334	<u>58294</u>
2013	9645	6037	<u>492</u>
2014	<u>31200</u>	26896	<u>44921</u>
2015	17398	6483	12758
Mean Burned Area	20826	14952	22069

Note: highlighted values correspond to years where very large (>50%) or very small (<1%) areas were burned

After 2008/2009, there has been an increase in the frequency of very large fires (>50% of the region burned) in CP and BP. Whereas a maximum of two very large burns occurred in the period 2000 - 2009, they have occurred three times thereafter. This latter trend was linked to the occurrence of two previous years of low burning (2008 and 2009), in particular in the CP (Tab.1).

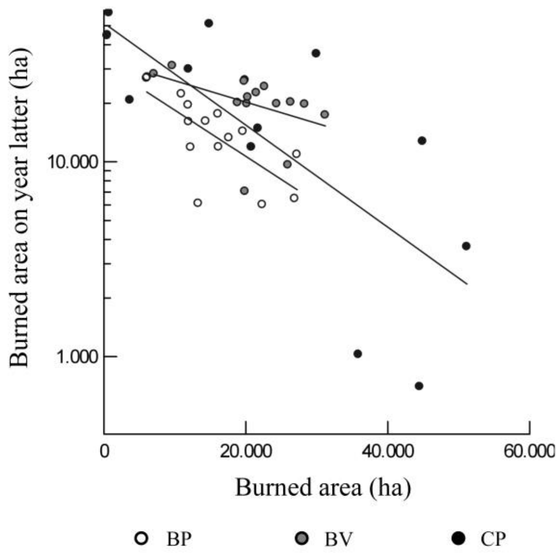


Figure 6. Total burned area as a function of burned area in preceding year for each fire management zone.

From the total area affected by fire in CNP, 85% was burnt in the LDS months (August to October), and 12% in EDS months (April to July). Although this pattern is repeated in each management zone (Fig.7), the fire activity in Babilonia region (BP and BV) was clearly distinguishable from the Canastra zone due to its higher proportion of areas burned in the EDS (Friedman's test = 24.5, 2 df, p-value <0,0001; post hoc tests were significant between BP-CP, p-value < 0.0001 and BV-CP, p-value <0,0001).

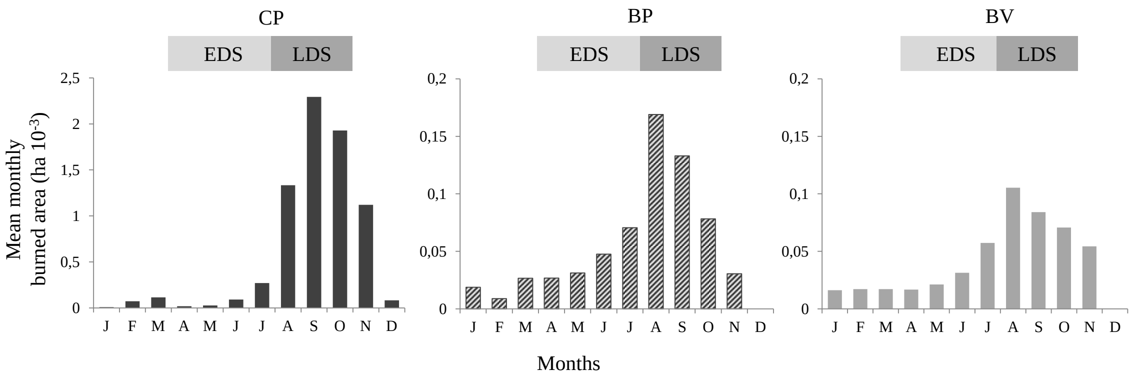


Figure 7. Mean monthly burned area in each fire management zone, 2000 to 2015, where CP = Canastra plateau, BP = Babilonia plateau, BV = Babilonia valley.

Figure 8 illustrates the monthly distribution of fires in CNP from 2013 to 2015. The map shows spatially how fires ignited in the BV and BP zones at the beginning of the dry season (EDS) extend into the CP at the end of the dry season (LDS).

Over the study period, 275 individual fire events were recorded in CP, 2975 in BP and 3374 in BV, totaling 6624 events in CNP. The ratios between burned area and number of fires (*BA:NF*) differ among the three zones of CNP (Chi-squared permutation test =

19.843, p-value < 0.0001; and p-value < 0.05 for all post hoc pairwise comparisons) (Fig.9).

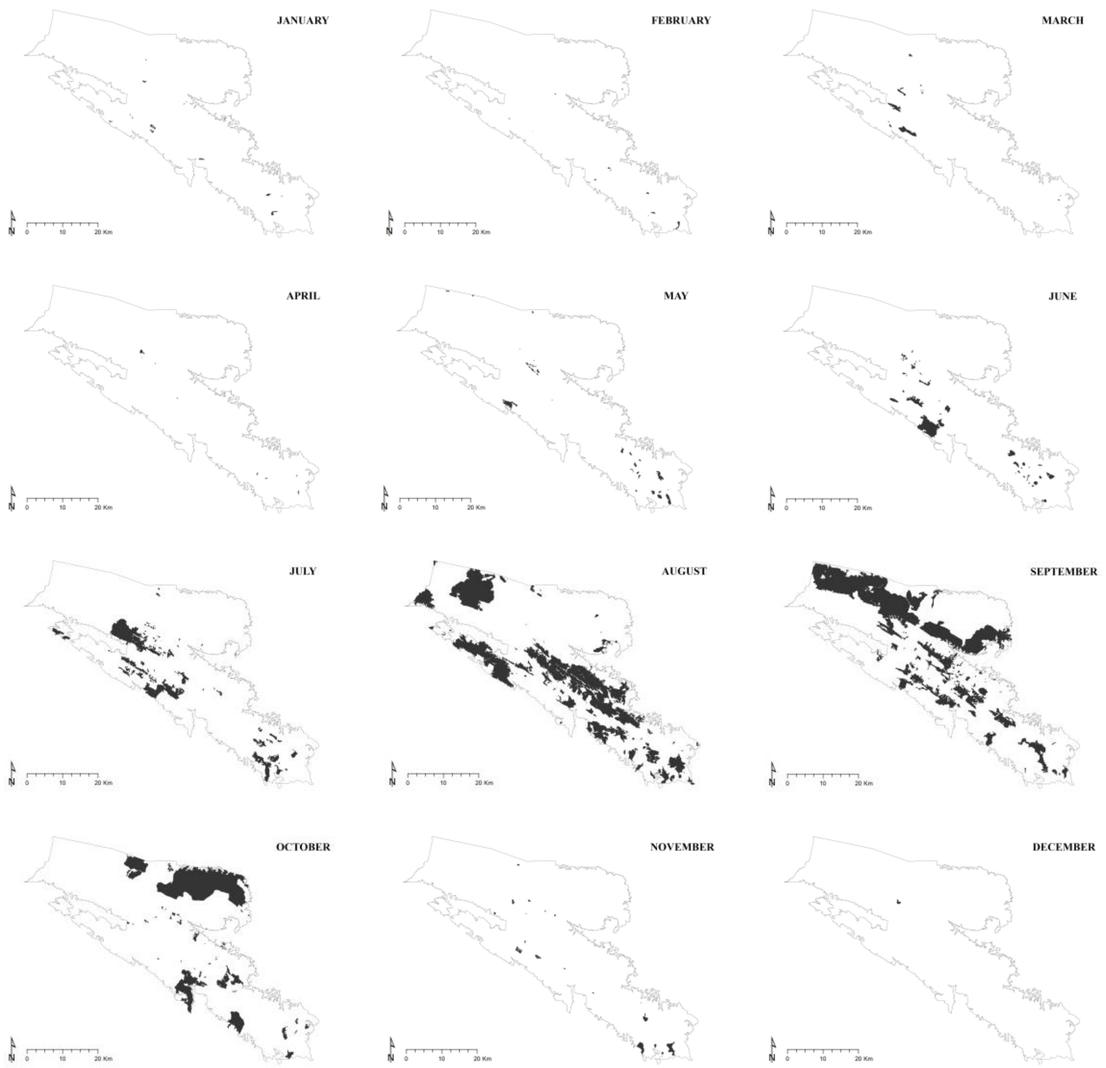


Figure 8. Monthly distribution of fires in CNP from 2013 to 2015.

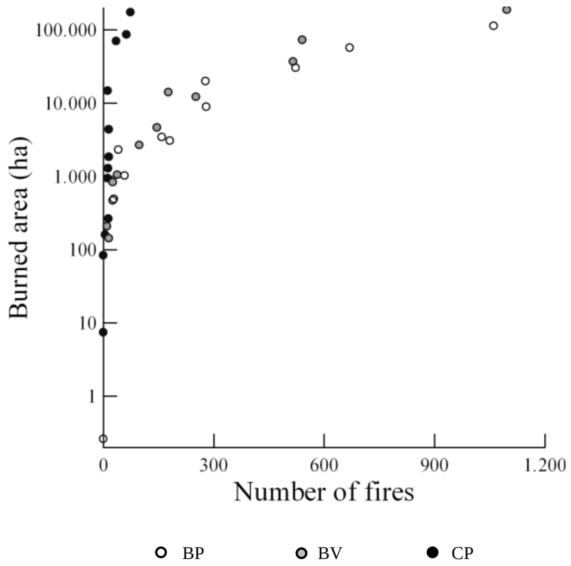


Figure 9. Annual burned area as a function of number of fires in each fire management zone, 2000 - 2015, where CP = Canastra plateau, BP = Babilonia plateau, BV = Babilonia valley. Note the highest number of fires in BP and BV in relation to the CP.

The distribution of fire sizes was significantly distinct between the three fire management zones (X2 = 29865.5, 2 df, CP-BP; X2 = 38064.1, 2 df, CP-BV; X2 = 39.73, 3 df, BP-BV; p-value < 0.001 in each case) and sub-seasons (X2 = 1432.6, 4 df, LDS-EDS; p-value <0.001). In general, 97-98% of burned areas were <1000 hectares in BP and BV, whereas big fires, > 1000 hectares, occurred in CP at relatively high frequency (17%). Over the entire period analyzed, > 80% of fire events were < 100 hectares during the early dry season (EDS). As the humidity decreased from the start of the late dry season (LDS), the proportion of large fires (>100 hectares) increased, especially in CP where fires > 1000 hectares comprised 25% of total fire events (Fig.10).

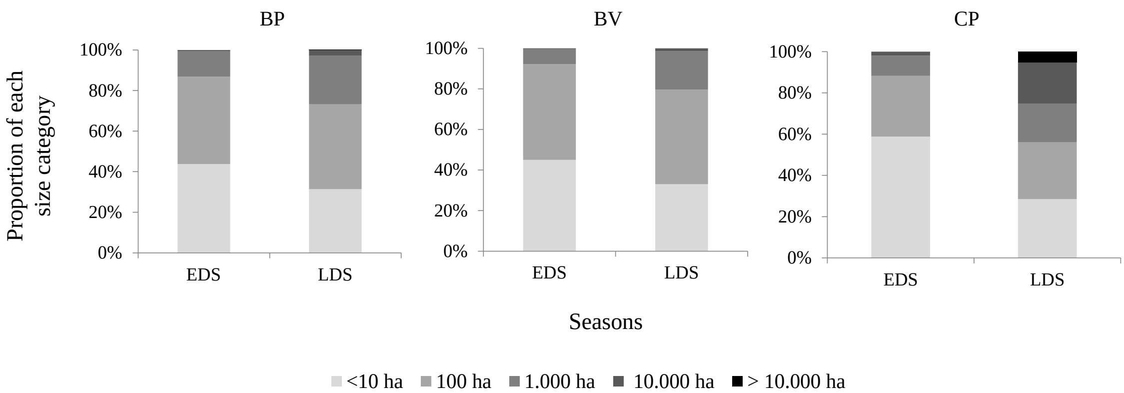


Figure 10. Proportion of fires in each fire size category for EDS and LDS periods in the three management zones, where CP = Canastra plateau, BP = Babilonia plateau, BV = Babilonia valley.

3.3. Fire frequency

In general, fire frequencies were high, particularly in the BP, where approximately 72% of the area experienced more than 6 fires over the assessment period. This percentage is relatively high compared which those achieved in the other two management zones (CP=23%; BV=40%). Although BP has burned more often than other two zones, differences between the three areas were not statistically significant (Friedman's test = 4.5, 2 df, p-value = 0,1054) (Fig. 11).

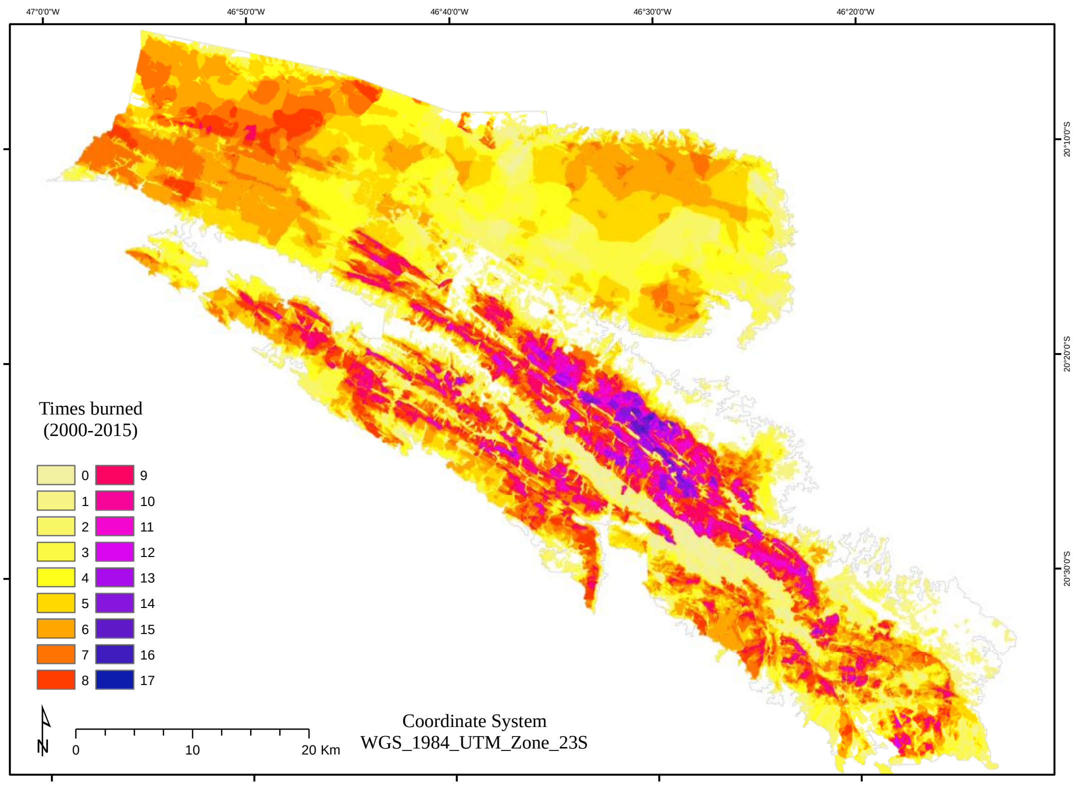


Figure 11. Times burned from 2000-2015 in the CNP.

3.4. Fire Rotation Period (FRP) and Fuel Accumulation

There are significant differences in FRP between management zones (Friedman chi-squared = 16.3, 2 df, p-value = 0.0002887; post hoc tests were significant between BP-BV, p-value = 0.001 and BP-CP, p-value = 0.017). The most common fire return period for CNP was 2 years, although 1-year return periods were relatively frequent in BP (Fig.12).

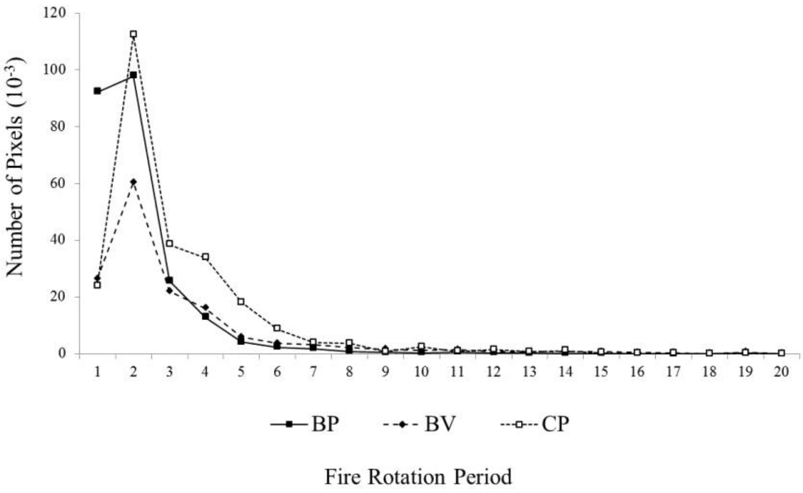


Figure 12. Fire rotation period from 1995-2015.

These results are consistent with the fuel accumulation analysis performed in CP, which indicated a rapid recovery of grasses, typical of short memory ecosystems. In the CP, time after fire appeared to be an important variable in estimates not only of the total biomass accumulation, but also of the ratio between dry and wet biomass (Kruskal-Wallis tests = 7280 and 11180 respectively; 2 df ; all p-value < 0.05). Just one year after fire, grass biomass (dead plus live) attained values in the order of 3.5 t/ha, with a dead/live matter ratio substantially > 1.0 (Fig.13). Between 1 and 3 years after fire, grass biomass varied from 4.3 to 7.1 t/ha, after which biomass accumulation decelerated, tending to level off at ~ 10 t/ha.

These fuel accumulation patterns also reflect an increase in fuel connectivity. GLM analysis detected differences in the ground cover (dependent variable) according to the time since fire (F= 26.094; p-value < 0.001). Just one year after fire, the grass layer already exceeded 60% cover, allowing easy percolation of fire across the landscape (Fig.14).

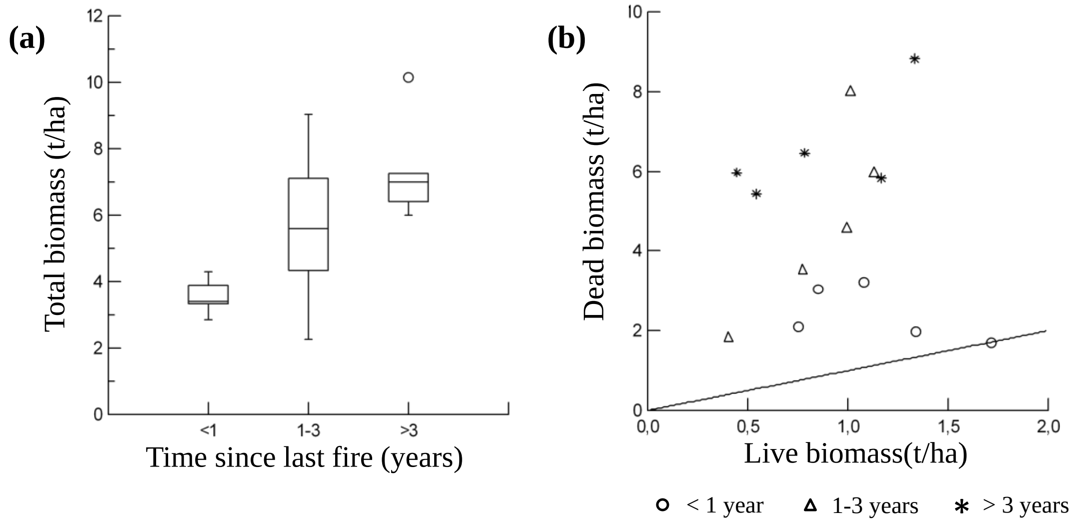


Figure 13. (a) Grass fuel accumulation rates with time since last fire. (b) Dead and live biomass accumulation, where the line represents the point at which the dead biomass is equal to living biomass.

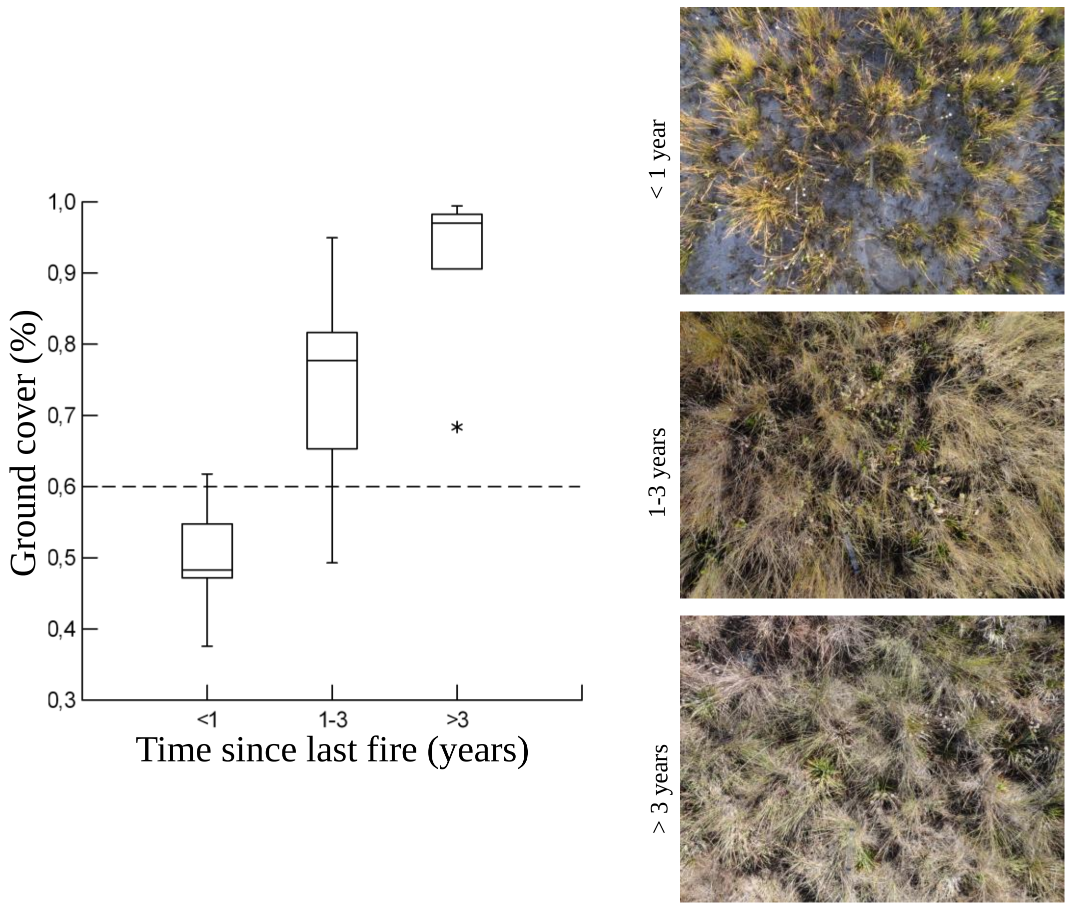


Figure 14. Connectivity of grasses represented by ground cover percentage as a function of time since last fire. The horizontal line represents the theoretical percolation threshold of fire spread (60% of the ground covered by fuel material).

4. Discussion

Fire regimes in the CNP mainly result from fire management practices, but can also be influenced by the particular features from each site. In the CP, government agencies and managers have performed fire suppression policies establishing firebreaks and fighting any fire that begins, regardless of the season when it happens. The landscape structure is highly homogeneous and fires percolate readily. Small forest patches afford ineffective barriers for fire spread. By contrast, in the BV, grassy fuels are widely discontinuous due to the higher density of roads and tracks, and elongated forest patches associated with rivers and slopes which can act as effective barriers to fire spread across the valley. Mostly, however, fires in this zone generally tend to stop at the limits of neighboring properties. Landowners typically establish firebreaks around their properties and practice alternating burning so that areas burned in a given year will reduce the risk of fires occurring in the following year.

Considering this whole framework, we have shown that fire regimes differ markedly between the three fire management zones of the CNP. The annual mean of extent of burning is lower in BV, mainly because of the non-sanctioned fire management undertaken by small farmers, and also the effects of larger forest patches. In contrast, large areas are burned on average annually in the CP, essentially due to landscape structure and fuel homogeneity created by fire suppression practices. In the BP, landowners also use fires to manage pastures, but they neither practice alternating burning (they put fires every year) nor carry out firebreaks around their properties. In a highly homogeneous landscape, fires easily get out of control and can be very extensive, in the same way as the CP. These observations imply that a feasible means for reducing the size of fires in the CP and BP zones could involve the creation of landscape/fuel discontinuities through prescribed and controlled burning.

In general, if large areas are burned in a given year, small areas remain to be burned in the following year. This trend is weaker in BV, simply because fires are better planned and controlled. By contrast, in the highlands, management practices create the necessary conditions for large fires, with few areas remaining for burning in the following year. This is even more evident after 2009, when unexpected rains occurred in September, reducing the length of the dry season and the risk of wildfires. After two years (2008-2009) without burning, a large amount of biomass built-up, forming a continuous fuel bed where fire could readily percolate. Consequently, in 2010 more than 50% of the total area was burned. Since then, especially on the CP, a ‘boom and bust’ cycle of very large fires followed by small fires has been established.

Natural fires in the CNP are mostly concentrated in October, when vegetation is still dry enough to burn and the incidence of lightning is substantially increased. On the other hand, anthropogenic fires occur throughout the year, especially over the course of the dry season (April to October). The great majority of fires in CNP can be attributed to human ignitions (Fig.7), especially as a result of land use and social conflicts. As mentioned earlier, landowners in the BV zone usually burn different portions of their properties every two years, or every year in the case of BP, for the purpose of rejuvenating pastures and clearing land. Farmers start burning early in the year and let fires escape onto the CP in the last months of the dry season. These actions are often intentional and linked to social conflicts generated in opposition to CNP policies, including fire suppression.

Finally, we have also demonstrated that CNP is a short-memory ecosystem with very rapid fuel accumulation rates. Just one year after fire dry biomass is higher than living biomass; before three years ground cover is > 60%, a threshold from which fires can easily percolate across the landscape (especially in LDS windy conditions) (Abades et al., 2014; Finney, 2001; Loehle, 2004). Since landowners use fires to manage their properties every year, the system becomes dependent on fuel accumulation rates and availability based on previous burning (bottom-up control). For that reason, management recommendations should encourage the monitoring and control of fuel loads rather than the total suppression of fires.

As described in Australian studies, late dry season fires are potentially destructive because they are typically severe, less patchy (smaller and fewer unburned patches) and often grow to a large size (Oliveira et al., 2015; Yates et al., 2008). Unfortunately, there are relatively few studies that have attempted to provide clear links between fire regimes in Brazilian savannas and effects on wildlife and biodiversity values. Nevertheless, available data suggests that fire regimes characterised by large, extensive and severe late dry season fires are likely to make major adverse effects on biodiversity values (Tab.2).

4.1. Implications for management

From the official perspective of Brazilian policy-makers and most conservation reserves, fire is a destructive agent that promotes loss of species irrespective of temporal and spatial patterns. However, in other international contexts the planned use of fire is recommended for reducing fuels and to avoid inappropriate wildfire regimes. In Australia, for example, Indigenous (Aboriginal) knowledge and practices have been incorporated into conservation and standard land management practices in savanna landscapes in order to reduce the impacts of LDS fires and reduce greenhouse gas (GHG) emissions (Russell-Smith, 2016; Russell-Smith et al., 2015, 2013b). In South Africa, fires, including those caused by lightening, are used to manage fuel loads, improve pastures for wildlife in grasslands, maintain populations of obligate seeder plants within tolerable thresholds, and help manage woody encroachment (Brockett et al., 2001; Van Wilgen, 2009; Van Wilgen et al., 2014).

As noted in the introduction to this paper, fire management issues facing fire-prone CNP are symptomatic of ongoing conflicts between official fire exclusion policies in conservation reserves, and the livelihood aspirations and imperatives of embedded and surrounding local communities, both in Brazil, and throughout South America generally. Some of the primary barriers to effective fire management in CNP and Brazilian protected areas concern: lack of information about, and failure to understand the ecological role of, fire in ecosystems; failure to link the causes of fire problems with appropriate solutions; legal constraints; and limited resources. There is deficient available knowledge about fire regimes, which patterns are appropriate for a given ecosystem, and how to distinguish detrimental and beneficial fires. The focus on fire suppression and criminalizing all fire use fails to recognize the legitimate roles both of Indigenous and local peoples, and biodiversity conservation managers, in the undertaking of ecologically sustainable fire management practices in fire-dependent Cerrado ecosystems.

We contend that an adaptive, inclusive approach between conservation agency authorities and local communities needs to be adopted for defining, and then collaboratively implementing, appropriate fire regimes which deliver benefits both for sustainable long-term biodiversity conservation and livelihood outcomes. In this regard, the application of market-based savanna burning greenhouse gas emissions reduction programs as undertaken in fire-prone northern Australian savannas, provides a powerful incentive for facilitating that change (Russell-Smith et al., 2015).

Furthermore, reducing the proportion of frequent, extensive, and relatively severe LDS fires requires increasing both the spatial heterogeneity and non-randomness of fire ignitions through prescribed EDS burning, such that patches of unburnt vegetation are compartmentalized between recently burned areas and an enhanced system of new, strategically placed firebreaks. EDS fires typically are of lower intensity and more patchy, providing refuges and sources of food for fauna. Conversely, critical habitats which require longer fire return intervals for vegetation persistence (e.g. forests), should be the focus both of prescribed fire management efforts as well as back-burning to contain undesirable wildfires.

5. Conclusion

Achieving appropriate biodiversity conservation targets in CNP is a complex challenge keeping in mind five main issues: (1) relatively frequent fire is essential to maintain the open and grassy physiognomies of plateau areas especially, as well as dependent species; (2) LDS fires typically are more severe and extensive than EDS fires and are difficult to manage; (3) very large fires are likely to impact on populations of slow-moving species, and those with small home ranges and arboreal habits; (4) the conservation requirements of fire-vulnerable shrubs and trees in Cerrado and forest habitats; and (5) fire regimes in CNP are driven mainly by human activities and exacerbating conflicts between park authorities and local farmers.

To achieve these conservation demands requires the development not only an strategic plan for prescribed burning, but also a dialogue and understanding between CNP authorities and the local farming community. Both parties face significant fire management issues. With goodwill, it should be eminently feasible to agree to and implement fire regimes which meet both the livelihood requirements of farmers, and evidence-based conservation management requirements over extensive areas of CNP where limited grazing is undertaken.

Table 2. Fire regime vulnerability components and potential threats to biodiversity conservation.

	Fire regime vulnerability
Large Fires	*Increased predation pressure due to lack of shelter or scarcity of food and resources after fires may increase mortality rates. *Can result in loss of structural complexity at the landscape and decrease in habitat specialists populations, for example: ants (Morais and Benson, 1988); small mammals - e.g. *Gracilinanus agilis* (Briani et al., 2004; Henriques et al., 2006; Mendonça et al., 2015a); lizards (Faria et al., 2004); birds - e.g. *Mergus octosetaceus* (Hughes et al., 2006; Lamas, 2006); butterflies (Marini-Filho and Martins, 2010). *Large homogenously burnt areas reduces the rates of post-fire recolonization (Mendonça et al., 2015a). *Direct mortality of slow-moving animals and those unable to fly or with small home ranges: giant anteaters - *Myrmecophaga tridactyla* (Redford, 1994; Silveira et al., 1999); snakes (Koproski et al., 2006); galling and leaf-miners (Marini-Filho, 2000); caterpillars (Diniz et al., 2011).
Severe fires	*Severe fires cause the death of stems (top-kill), reducing the mean plant size and size-specific seed production (Hoffmann, 1998). *Fire produces an immediate reduction in sexual reproductive success by causing abortion of flowers and fruits, interrupting the reproductive cycle and favouring vegetative sprouting. Most Cerrado plants produce flowers and fruits at the end of the dry season and beginning of the wet season (August - October) (Falleiro, 2011; Lenza and Klink, 2006; Pilon et al., 2015), when severe late dry season fires will destroy flower buds, flowers, developing fruit, and mature seeds, greatly reducing seed availability (Hoffmann, 1998). *Several animals in the Cerrado are nesting and generating puppies at the end of the dry season (Falleiro, 2011). *Severe fires may adversely affect fire-sensitive habitats (e.g. forests: Hoffmann et al., 2009) and the wildlife that depends on them (e.g. habitat specialists or animals seeking refuge during large fires: (Alho, 2005; Coelho et al., 2008; Hughes et al., 2006; Marini-Filho and Martins, 2010)). *Although fossorial animals may be able to avoid incineration, the thermal stress, smoke toxicity and increased predation pressure due to lack of shelter or scarcity of food and resources after severe fires may increase mortality rates (Massochini Frizzo et al., 2011). Spiders, mites and Ophelia (Arachnida), are commonly described as species that suffer significant population reduction after severe fires (White et al., 2010). *Severe fires can substantially reduce the density of seed banks, particularly those of more sensitive taxa or those that are not deeply buried in the soil (Camargos et al., 2013).

Table 2. Fire regime vulnerability components and potential threats to biodiversity conservation.

	Fire regime vulnerability
Frequent fires	*In the long term, frequent fires will generate shifts in community composition by favoring plants capable of resprouting over obligate seeders (Hoffmann and Moreira, 2002). *Frequent fires can lead monocarpic species to population declines through the exhaustion of seed banks and the mortality of reproductive individuals (Figueira et al., 2016). *In Eriocaulaceae species, fire promotes an increase in the number of reproductive individuals and, for polycarpic species, the increased reproductive effort may negatively impact growth, survival and the production of inflorescences in years after burning. (Neves et al., 2011). *Too frequent fires can not only exclude shrubs and trees and favor grasses and forbs, but could also lead populations to decline by the exhaustion of seed banks, seedlings mortality, induced reproduction of young/small individuals and early death in semelparous species (Alves and Silva, 2011; Figueira et al., 2016). *The abundance of trees and large shrubs tends to be reduced by fire, whereas subshrubs are favored. Fire reduces the woody plant cover as a whole regardless of physiognomy, but denser formations experience the greatest declines (Moreira, 2000).

6. References

Abades, S.R., Gaxiola, A., Marquet, P. a., 2014. Fire, percolation thresholds and the savanna forest transition: A neutral model approach. J. Ecol. 102, 1386–1393. doi: 10.1111/1365-2745.12321

Alho, C.J.R., 2005. Intergradation of habitats of non-volant small mammals in the patchy cerrado landscape, in: Arquivos Do Museu Nacional. Rio de Janeiro, pp. 41–48.

Alvares, C.A., Stape, J.L., Sentelhas, P.C., De Moraes Gonçalves, J.L., Sparovek, G., 2013. Koppen's climate classification map for Brazil. Meteorol. Zeitschrift 22, 711–728. doi: 10.1127/0941-2948/2013/0507

Alves, R.J.V., Silva, N.G. Da, 2011. O Fogo é Sempre um Vilão nos Campos Rupestres ? Biodiversidade Bras. 2, 120–127.

Andersen, A.N., Cook, G.D., Corbett, L.K., Douglas, M.M., Eager, R.W., Russell-Smith, J., Setterfield, S. a., Williams, R.J., Woinarski, J.C.Z., 2005. Fire frequency and biodiversity conservation in Australian tropical savannas: Implications from the Kapalga fire experiment. Austral Ecol. 30, 155–167.

Barbosa, C., 2007. Territórios de vida e trabalho dos pequenos produtores de queijo da Serra Da Canastra: um estudo sobre a relação entre produção camponesa e espaços naturais protegidos nas nascentes do rio São Francisco, Minas Gerais. Master's thesis, Universidade Federal de Uberlândia.

Barbosa, C., Júnior, J.C., 2006. Tem gente no Parque: um estudo sobre a agricultura e áreas protegidas nas nascentes do rio São Francisco - Serra da Canastra, MG, in: II Encontro de Grupos de Pesquisa. Uberlândia, MG, p. 23.

Beerling, D.J., Osborne, C.P., 2006. The origin of the savanna biome. Glob. Chang. Biol. 12, 2023–2031. doi: 10.1111/j.1365-2486.2006.01239.x

Bilbao, B. a., Leal, A. V., Méndez, C.L., 2010. Indigenous use of fire and forest loss in Canaima National Park, Venezuela. Assessment of and tools for alternative strategies of fire management in Pemón Indigenous Lands. Hum. Ecol. 38, 663–673. doi: 10.1007/s10745-010-9344-0

BirdLife International, 2015. Threatened Birds of the world. Lynx Editions, Barcelona and Cambridge, U.K.

Bond, W.J., Keeley, J.E., 2005. Fire as a global "herbivore": The ecology and evolution of flammable ecosystems. Trends Ecol. Evol. 20, 387–394. doi:10.1016/j.tree.2005.04.025

Bond, W.J., Parr, C.L., 2010. Beyond the forest edge: Ecology, diversity and conservation of the grassy biomes. Biol. Conserv. 143, 2395–2404. doi:10.1016/j.biocon.2009.12.012

Bond, W.J., Woodward, F.. I., Midgley, G.F., 2005. The Global distribtuion of ecosystems in a world without fire. New Phytol. 165, 525–538. doi : 10.1111/j.1469-8137.2004.01252.x

Bowman, D.M.J.S., 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., 2011. The human dimension of fire regimes on Earth. J. Biogeogr. 38, 2223–2236. doi:10.1111/j.1365-2699.2011.02595.x

Bowman, D.M.J.S., Balch, J.K., Artaxo, P., Bond, W.J., Carlson, J.M., Cochrane, M. a, Antonio, C.M.D., Defries, R.S., Doyle, J.C., Harrison, S.P., Johnston, F.H., Keeley, J.E., Krawchuk, M. a, Kull, C. a, Marston, J.B., Moritz, M. a, Prentice, I.C., Roos, C.I., Scott, A.C., Swetnam, T.W., Werf, G.R. Van Der, Pyne, S.J., 2009. Fire in the Earth System. Science (80-.). 324, 481–484. doi: 10.1126/science.1163886

BRASIL, 2014. Lista nacional oficial de especies da fauna ameacadas de extincao. Brasilia, DF, Brazil.

Briani, D.C., Palma, A.R.T., Vieira, E.M., Henriques, R.P.B., 2004. Post-fire succession of small mammals in the Cerrado of Brasil. Biodivers. Conserv. 13, 1023–1037. doi:10.1023/B:BIOC.0000014467.27138.0b

Brockett, B.H., Biggs, H.C., Van Wilgen, B.W., 2001. A patch mosaic burning system for conservation areas in southern African savannas. Int. J. Wildl. Fire 10, 169–183. doi: 10.1071/WF01024

Burrows, N.D., 2008. Linking fire ecology and fire management in south-west Australian forest landscapes. For. Ecol. Manage. 255, 2394–2406. doi:10.1016/j.foreco.2008.01.009

Camargos, V.L., Martins, S.V., Ribeiro, G.A., Carmo, F.M. da S., da Silva, A.F., 2013. Influência do fogo no banco de sementes do solo em Floresta Estacional Semidecidual. Cienc. Florest. 23, 19–28.

Cardoso Da Silva, J.M., Bates, J.M., 2002. Biogeographic Patterns and Conservation in the South American Cerrado: A Tropical Savanna Hotspot. Bioscience 52, 225–233.

Cerling, T.E., Harris, J.M., Macfadden, B.J., Leakey, M.G., Quade, J., Eisenmann, V., Ehleringer, J.R., 1997. Global vegetation change through the Miocene/Pliocene boundary. Nature 389, 153–158.

Chuvieco, E., Justice, C., 2010. Relations between human factors and global fire activity, in: Chuvieco, E., Li, J., Yang, X. (Eds.), Advances in Earth Observation of Global Change. Springer Netherlands, Dordrecht, pp. 187–199. doi: 10.1007/978-90-481-9085-0_14

Clarke, P.J., 2002. Habitat islands in fire-prone vegetation: do landscape features influence community composition? J. of Biogeogr. 29, 1–8.

Coelho, C.M., De Melo, L.F.B., Sábato, M.A.L., Magni, E.M.V., Hirsch, A., Young, R.J., 2008. Habitat Use by Wild Maned Wolves (Chrysocyon brachyurus) in a Transition Zone Environment. J. Mammal. 89, 97–104. doi: 10.1644/06-MAMM-A-383.1

Coutinho, L.M., 2002. O bioma do cerrado, in: Klein, A.L. (Ed.), Eugen Warming E O Cerrado Brasileiro: Um Século Depois. Editora UNESP, São Paulo, SP, pp. 77–91.

D'Antonio, C., Vitousek, P.M., 1992. Biological invasions by exotic grasses, the grass fire cycle, and global change. Annu. Rev. Ecol. Syst. 23, 63–87.

Diniz, I.R., Higgins, B., Morais, H.C., 2011. How do frequent fires in the Cerrado alter the lepidopteran community? Biodivers. Conserv. 20, 1415–1426. doi:10.1007/s10531-011-0034-z

Durigan, G., Leitão-Filho, H. de F., Rodrigues, R.R., 1994. Phytosociology and structure of a frequently burnt cerrado vegetation in SE-Brazil. Flora 189, 53–160.

Durigan, G., Ratter, J.A., 2016. The need for a consistent fire policy for Cerrado conservation. J. Appl. Ecol. 53, 11–15. doi: 10.1111/1365-2664.12559

Edwards, A., Russell-Smith, J., Meyer, M., 2015. Contemporary fire regime risks to key ecological assets and processes in north Australian savannas. Int. J. Wildl. Fire 24, 857–870. doi: 10.1071/WF14197

Edwards, E.J., Osborne, C.P., Strömberg, C.A.E., Smith, S.A., 2010. The origins of C4 grasslands: integrating evolutionary and ecosystem science. Science 328, 587–590. doi: 10.1126/science.1177216

Eiten, G., 1978. Delimitation of the Cerrado concept. Vegetatio 36, 169–178.

Eiten, G., 1972. The Cerrado vegetation of Brazil. Bot. Rev. 38, 201–338.

Epstein, H.E., Lauenroth, W.K.K., Burke, I.C., Coffin, D.P., 1997. Productivity patterns of C3 and C4 functional types in the U.S. Great Plains. Ecology 78, 722–731.

Falleiro, R.D.M., 2011. Resgate do manejo tradicional do Cerrado com fogo para proteção das Terras Indígenas do oeste do Mato Grosso: um estudo de caso. Biodiversidade Bras. 2, 86–96.

Faria, A.S., Lima, A.P., Magnusson, W.E., 2004. The effects of fire on behaviour and relative abundance of three lizard species in an Amazonian savanna. J. Trop. Ecol. 20, 591–594. doi:10.1017/S0266467404001798

Fernandes, V.S.R., 2012. Entre a regulação e a emancipação social : desafios à continuidade do lugar frente ao Parque Nacional da Serra da Canastra - MG. Universidade Federal de Minas Gerais (UFMG), Belo Horizonte, MG.

Ferreira, G.H.C., 2015. O Parque Nacional da Serra da Canastra – MG : algumas propostas , conflitos e incertezas territoriais. Rev. Cerrados 13, 111–139.

Ferreira, G.H.C., 2013. A regularização fundiária no Parque Nacional da Serra da Canastra e a expropriação camponesa : da baioneta à ponta da caneta. São Paulo, SP.

Figueira, J.E.C., Ribeiro, K.T., Ribeiro, M.C., Jacobi, C.M., França, H., Neves, A.C.O., Conceição, A.A., Mourão, F.A., Souza, J.M., Knegt, C.A. de M., 2016. Fire in rupestrian grasslands: plant response and management, in: Fernandes, G.W. (Ed.), Ecology and Conservation of Mountain-Top Grasslands in Brazil. Springer International Publishing Switzerland, Belo Horizonte, pp. 1–34.

Finney, M. a., 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. For. Sci. 47, 219–228.

Flanders, D., Hall-Beyer, M., Pereverzoff, J., 2003. Preliminary evaluation of eCognition object-based software for cut block delineation and feature extraction. Can. J. Remote Sens. 4, 441–452.

Gignoux, J., Clobert, J., Menaut, J.C., 1997. Alternative fire resistance strategies in savanna trees. Oecologia 110, 576–583.

Gonçalves, C.N., Mesquita, F.W., Lima, N.R.G., Coslope, L.A., Lintomen, B.S., 2011. Recorrência dos Incêndios e Fitossociologia da Vegetação em Áreas com Diferentes Regimes de Queima no Parque Nacional da Chapada Diamantina. Biodiversidade Bras. 1, 161–179.

Goulart, E.M.A., 2013. Viagens do naturalista Saint-Hilaire por toda Província de Minas Gerais. Graphar, Ouro Preto, MG.

Heckbert, S., Russell-Smith, J., Reeson, A., Davies, J., James, G., Meyer, C., 2012.

Spatially explicit benefit-cost analysis of fire management for greenhouse gas abatement. Austral Ecol. 37, 724–732. doi:10.1111/j.1442-9993.2012.02408.x

Henriques, R.P.B., Briani, D.C., Palma, a R.T., Vieira, E.M., 2006. A simple graphical model of small mammal succession after fire in the Brazilian cerrado. Mammalia 70, 226–230. doi: 10.1515/MAMM.2006.044

Hoffmann, W.A., 1999. Fire and population dynamics of woody plants in a neotropical savanna : matrix model projections. Ecology 80, 1354–1369.

Hoffmann, W.A., 1996. The effects of fire and cover on seedling establishment in a neotropical savanna. J. Ecol. 84, 383–393.

Hoffmann, W.A., Adasme, R., Haridasan, M., De Carvalho, M.T., Geiger, E.L., Pereira, M. a B., Gotsch, S.G., Franco, A.C., 2009. Tree topkill, not mortality, governs the dynamics of savanna-forest boundaries under frequent fire in central Brazil. Ecology 90, 1326–1337.

Hoffmann, W.A., Moreira, A.G., 2002. The role of fire in population dynamics of woody plants, in: The Cerrados of Brazil: Ecology and Natural History of a Neotropical Savanna. Columbia University Press, p. 424.

Hoffmann, W.A., Solbrig, O.T., 2003. The role of topkill in the differential response of savanna woody species to fire. For. Ecol. Manage. 180, 273–286. doi:10.1016/S0378-1127(02)00566-2

Hoffmann, W. a., 1998. Post-burn reproduction of woody plants in a neotropical savanna: The relative importance of sexual and vegetative reproduction. J. Appl. Ecol. 35, 422–433.

Hughes, B., Dugger, B., Cunha, H.J., Lamas, I., Goerck, J., Lins, L., Silveira, L.F., Andrade, R., Bruno, S.F., Rigueira, S., Barros, Y.D.M., 2006. Action Plan for the Conservation of the Brazilian Merganser (Mergus octosetaceus). IBAMA, Brasilia, DF.

IBAMA, 2005. Plano de Manejo do Parque Nacional da Serra da Canastra. Brasília, DF.

IBGE, 2012. Manual Técnico da Vegetação Brasileira, Manuais técnicos em Geociências. Rio de Janeiro, RJ.

IBGE, 2004. Mapa de Vegetação do Brasil. Rio de Janeiro, RJ.

Júnior, A.D.C.P., Bretos, A.L.K., Antonioli, V., 2013. Mudanças no regime do fogo no bioma Cerrado: o caso do Parque Estadual do Jalapão, in: XVI Simpósio Brasileiro de Sensoriamento Remoto. Foz do Iguaçu, PR, pp. 6401–6408.

Kauffman, J.B., Cummings, D.L., Ward, D.E., 1994. Relationships of fire, biomass and nutrient dynamics along a vegetation gradient in the Brazilian Cerrado. J. Ecol. 82, 519–531.

Keeley, J.E., Rundel, P.W., 2005. Fire and the Miocene expansion of C4 grasslands. Ecol. Lett. 8, 683–690. doi: 10.1111/j.1461-0248.2005.00767.x

Kier, G., Mutke, J., Dinerstein, E., Ricketts, T.H., Küper, W., Kreft, H., Barthlott, W., 2005. Global patterns of plant diversity and floristic knowledge. J. Biogeogr. 32, 1107–1116. doi:10.1111/j.1365-2699.2005.01272.x

Klink, C.A., Machado, R.B., 2005. A conservação do Cerrado brasileiro. Megadiversidade 1, 147–155.

Koproski, L., Mangini, P.R., Pachaly, J.R., Batista, A.C., Soares, R.V., 2006. Impactos do fogo sobre serpentes (Squamata) no Parque Nacional de Ilha Grande (PR/MS), Brasil. Arq. Ciências Veterinárias e Zool. da Unipar 9, 129–133.

Lamas, I.R., 2006. Census of Brazilian Merganser Mergus octosetaceus in the region of Serra da Canastra National Park, Brazil, with discussion of its threats and conservation. Bird Conserv. Int. 16, 145. doi: 10.1017/S0959270906000220

Lenza, E., Klink, C.A., 2006. Comportamento fenológico de espécies lenhosas em um cerrado sentido restrito de Brasília, DF. Rev. Bras. Botânica 29, 627–638. doi:10.1590/S0100-84042006000400013

Loehle, C., 2004. Applying landscape principles to fire hazard reduction. For. Ecol. Manage. 198, 261–267. doi:10.1016/j.foreco.2004.04.010

Marini-Filho, O.J., 2000. Distance-Limited Recolonization of Burned Cerrado by Leaf-Miners and Gallers in Central Brazil. Environ. Entomol. 29, 901–906. doi:10.1603/0046-225X-29.5.901

Marini-Filho, O.J., Martins, R.P., 2010. Nymphalid butterfly dispersal among forest fragments at Serra da Canastra National Park, Brazil. J. Insect Conserv. 14, 401–411. doi:10.1007/s10841-010-9271-9

Massochini Frizzo, T.L., Bonizário, C., Borges, M.P., Vasconcelos, H.L., 2011. Revisão dos efeitos do fogo sobre a fauna de formações savânicas do Brasil. Oecologia Aust. 15, 365–379. doi:10.4257/oeco.2011.1502.13

Mendonça, A.F., Armond, T., Camargo, A.C.L., Camargo, N.F., Ribeiro, J.F., Zangrandi, P.L., Vieira, E.M., 2015. Effects of an extensive fire on arboreal small mammal populations in a neotropical savanna woodland. J. Mammal. 96, 368–379. doi:10.1093/jmammal/gyv038

Mesquita, F.W., Rodrigo, N., Lima, G., Gonçalves, C.N., Berlinck, C.N., Lintomen, B.S., 2011. Histórico dos Incêndios na Vegetação do Parque Nacional da Chapada Diamantina , entre 1973 e abril de 2010 , com base em Imagens Landsat. Biodiversidade Bras. ICMBio 228–246.

Mistry, J., 1998. Fire in the savannas of Brazil: an ecological review. Prog. Phys. Geogr. 22, 425–448. doi: 10.1177/030913339802200401

Morais, H.C., Benson, W.W., 1988. Recolonização de vegetação de cerrado após queimada por formigas arborícolas. Rev. Bras. Biol. 48, 459–466.

Moreira, A.G., 2000. Effects of fire protection on savanna structure in Central Brazil. J. Biogeogr. 27, 1021–1029.

Moura, L.C., 2013. Fogo no Parque Nacional da Serra da Canastra/MG: Abordagem dos aspectos fisiográficos e humanos na concepção de uma proposta de manejo integrado de fogo. Rio Claro, SP.

Murphy, B.P., Bowman, D.M.J.S., 2012. What controls the distribution of tropical forest and savanna? Ecol. Lett. 15, 748–758. doi: 10.1111/j.1461-0248.2012.01771.x

Murphy, B.P., Russell-Smith, J., Prior, L.D., 2010. Frequent fires reduce tree growth in northern Australian savannas: Implications for tree demography and carbon sequestration. Glob. Chang. Biol. 16, 331–343. doi: 10.1111/j.1365-2486.2009.01933.x

Myers, N., Mittermeier, R. a, Mittermeier, C.G., da Fronseca, G. a B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.

Myers, R., O'Brien, J., Mehlman, D., Bergh, C., 2004. Evaluación del Manejo del Fuego en los Ecosistemas de Tierras Altas de la República Dominicana. Arlington, VA.

Myers, R., O'Brien, J., Morrison, S., 2006. Descripción general del manejo del fuego en las sabanas de Pino Caribe (Pinus caribaea) de la Mosquitia, Honduras. Arlington, VA.

Neves, A.C.D.O., Bedê, L.C., Martins, R.P., 2011. Revisão sobre os Efeitos do Fogo em Eriocaulaceae como Subsídio para a sua Conservação. Biodiversidade Bras. 1, 50–66.

Oliveira, S.L.J., Campagnolo, M.L., Price, O.F., Edwards, A.C., Russell-Smith, J., Pereira, J.M.C., 2015. Ecological implications of fine-scale fire patchiness and severity in tropical savannas of Northern Australia. Fire Ecol. 11, 10–31. doi: 10.4996/fireecology.1101010

Pagani, M., 1999. Late Miocene atmospheric CO2 concentrations and the expansion of C4 grasses. Science 285, 876–879.

Pereira, J.M.C., 2003. Remote sensing of burned areas in tropical savannas. Int. J. Wildl. Fire 12, 259. doi: 10.1071/WF03028

Pilon, N.A.L., Udulutsch, R.G., Durigan, G., 2015. Padrões fenológicos de 111 espécies de Cerrado em condições de cultivo. Hoehnea 42, 425–443. doi:10.1590/2236-8906-07/2015

Ratajczak, Z., Nippert, J.B., Briggs, J.M., Blair, J.M., 2014. Fire dynamics distinguish grasslands, shrublands and woodlands as alternative attractors in the central great plains of North America. J. Ecol. 102, 1374–1385. doi: 10.1111/1365-2745.12311

Redford, K.H., 1994. The edentates of the Cerrado. Edentata 1, 4–10.

Russell-Smith, J., 2016. Fire management business in Australia's tropical savannas: Lighting the way for a new ecosystem services model for the north? Ecol. Manag. Restor. 17, 1–4. doi: 10.1111/emr.12201

Russell-Smith, J., Cook, G.D., Cooke, P.M., Edwards, A.C., Lendrum, M., Meyer, C.P., Whitehead, P.J., 2013a. Managing fire regimes in north Australian savannas: Applying Aboriginal approaches to contemporary global problems. Front. Ecol. Environ. 11, 55–63. doi:10.1890/120251

Russell-Smith, J., Edwards, A.C., Price, O.F., 2012. Simplifying the savanna: the trajectory of fire-sensitive vegetation mosaics in northern Australia. J. Biogeogr. 39, 1303–1317. doi:10.1111/j.1365-2699.2012.02679.x

Russell-Smith, J., Monagle, C., Jacobsohn, M., Beatty, R.L., Bilbao, B., Millán, A., Vessuri, H., Sánchez-Rose, I., 2013b. Can savanna burning projects deliver measurable greenhouse emissions reductions and sustainable livelihood opportunities in fire-prone settings? Clim. Change 1–15. doi: 10.1007/s10584-013-0910-5

Russell-Smith, J., Yates, C.P., Edwards, A.C., Whitehead, P.J., Murphy, B.P., Lawes, M.J., 2015. Deriving multiple benefits from carbon market-based savanna fire management: An Australian example. PLoS One 10, 1–21. doi:

10.1371/journal.pone.0143426

Saint-Hilaire, A. de, 1975. Viagem as Nascentes do Rio São Francisco. Itatiaia, Belo Horizonte.

Sankaran, M., Hanan, N.P., Scholes, R.J., Ratnam, J., Augustine, D.J., Cade, B.S., Gignoux, J., Higgins, S.I., Le Roux, X., Ludwig, F., Ardo, J., Banyikwa, F., Bronn, A., Bucini, G., Caylor, K.K., Coughenour, M.B., Diouf, A., Ekaya, W., Feral, C.J., February, E.C., Frost, P.G.H., Hiernaux, P., Hrabar, H., Metzger, K.L., Prins, H.H.T., Ringrose, S., Sea, W., Tews, J., Worden, J., Zambatis, N., 2005. Determinants of woody cover in African savannas. Nature 438, 846–849. doi:10.1038/nature04070

Sankaran, M., Ratnam, J., 2013. African and Asian Savannas, in: Levin, S. (Ed.), Encyclopedia of Biodiversity. Elsevier Inc., pp. 58–74.

Schmidt, I.S., Fonseca, C.B., Ferreira, M.C., Sato, M.N., 2016. Implementação do Programa Piloto de Manejo Integrado do Fogo em três Unidades de Conservação do Cerrado. Rev. Biodiversidade Bras. 6, 55–70.

Scott, K., Setterfield, S. a., Douglas, M.M., Parr, C.L., Schatz, J., Andersen, A.N., 2012. Does long-term fire exclusion in an Australian tropical savanna result in a biome shift? A test using the reintroduction of fire. Austral Ecol. 37, 693–711. doi:10.1111/j.1442-9993.2012.02379.x

Silva, D.M., Loiola, P.P., Rossati, N.B., 2011. Os efeitos dos regimes de fogo sobre a vegetação de cerrado no parque nacional das emas, GO : Considerações para a conservação da diversidade. Biodiversidade Bras. 1, 26–39.

Silveira, L., Rodrigues, F.H.G., Jácomo, a T. a, Filho, J. a D., 1999. Impact of wildfires on the megafaune of Emas National Park, central Brazil. Oryx 33, 108–114.

Simon, M.F., Grether, R., de Queiroz, L.P., Skema, C., Pennington, R.T., Hughes, C.E., 2009. Recent assembly of the Cerrado, a neotropical plant diversity hotspot, by in situ evolution of adaptations to fire. Proc. Natl. Acad. Sci. U. S. A. 106, 20359–20364. doi: 10.1073/pnas.0903410106

Smit, I.P.J., Asner, G.P., Govender, N., Kennedy-Bowdoin, T., Knapp, D.E., Jacobson, J., 2010. Effects of fire on woody vegetation structure in African savanna. Ecol. Appl. 20, 1865–1875.

Souza, R.O., Fieker, C.Z., Gonçalves, M., Bruno, S.F., 2016. Estratégias de integração entre pesquisa e manejo do fogo no Parque Nacional da Serra da Canastra como parte do desenvolvimento de um Programa de Manejo Integrado do Fogo. Rev. Biodiversidade Bras. 6, 205–219.

Staver, C.C., Archibald, S., Levin, S.A., 2011. The global extent and determinants of savanna and forest as alternative biome states. Science 334, 230–232. doi: 10.1126/science.1210465

Thornthwaite, C.W., Mather, J.R., 1955. The water balance. Publications in Climatology, Centerton, New Jersey.

Tubelis, A., Nascimento, F.J.L. do, 1980. Meteorologia Descritiva: fundamentos e aplicações brasileiras. Nobel, São Paulo, SP.

Van Wilgen, B.W., 2009. The evolution of fire management practices in savanna protected areas in South Africa. S. Afr. J. Sci. 105, 343–349.

Van Wilgen, B.W., Govender, N., Smit, I.P.J., MacFadyen, S., 2014. The ongoing development of a pragmatic and adaptive fire management policy in a large African savanna protected area. J. Environ. Manage. 132, 358–368. doi: 10.1016/j.jenvman.2013.11.003

Veloso, H.P., Filho, A.L.R.R., Lima, J.C.A., 1991. Classificação da vegetação brasileira adaptada a um sistema universal. Rio de Janeiro, RJ.

Vogl J, R., 1979. Some Basic Principles of Grassland Fire Management. Environ. Manage. 3, 51–57.

White, B.L.A., Ribeiro, G.T., Souza, R.M. e, 2010. Efeito do fogo na população de artrópodes e seu uso como bioindicadores para conservação ambiental, in: Santos, A.C. dos, Filho, J.D., Soares, M.J.N., Randow, P.C.B.D. (Eds.), Pensar a (In)sustentabilidade. Redes, Porto Alegre, p. 265.

Yates, C.P., Edwards, A.C., Russell-Smith, J., 2008. Big fires and their ecological impacts in Australian savannas: Size and frequency matters. Int. J. Wildl. Fire 17, 768–781. doi: 10.1071/WF07150

CAPÍTULO II - FIRE MANAGEMENT IN BRAZILIAN SAVANNAS: CHALLENGES AND SOLUTIONS - PROPOSING THRESHOLDS OF POTENTIAL CONCERN FOR CANASTRA NATIONAL PARK, SOUTHEAST BRAZIL

Resumo

Quando inseridos no contexto de manejo ecológico do fogo, os gestores de áreas protegidas buscam garantir a resiliência e auto-sustentabilidade dos ecossistemas nativos, utilizando todo o conhecimento acumulado sobre a flora e fauna, suas populações e comunidades e suas respostas aos padrões do fogo. Os sistemas naturais são altamente complexos, geralmente compostos por muitos elementos que interagem de forma não linear ou caótica. Sendo assim, eles podem responder de forma imprevisível à superação de seus limiares, definidos como o ponto a partir do qual pequenas alterações nos elementos de controle catalisam mudanças abruptas nas feições do ecossistema. A determinação de algumas variáveis ecológicas estratégicas e suas respostas aos padrões espaciais e temporais do fogo pode nortear a proposição de limiares adequados às diferentes unidades da paisagem. Uma avaliação contínua desses limiares de regime de fogo permite a readequação das práticas de manejo por meio de um processo adaptativo que busca a convergência com os objetivos de conservação da biodiversidade. O uso de limiares específicos de regime de fogo para cada fisionomia com o objetivo de monitorar os resultados das ações de manejo tem ganhado reconhecimento entre ecologistas e gestores de áreas protegidas. O roteiro proposto neste trabalho é um primeiro passo nessa direção. Nós apresentamos limiares específicos de regime de fogo que poderiam ser utilizados para nortear as ações de manejo no Parque Nacional da Serra da Canastra, fizemos uma análise retrospectiva dos regimes em relação a esses limiares e propusemos variáveis ecológicas estratégicas que poderiam ser empregadas no monitoramento dos resultados dessas ações de manejo.

Palavras-chave: Limiares de preocupação; Manejo adaptativo; Monitoramento; Queimas prescritas.

Abstract

When adept at the ecological management of fire, managers of savanna protected areas strive to ensure the resilience and self-sustainability of native ecosystems based on accumulating knowledge of flora and fauna species, populations and communities and their response to fire regimes. Natural systems are often highly complex and generally composed of many elements that interact in a non-linear or chaotic way. Thus, they may respond unpredictably to exceeding their thresholds, defined as the point from which small changes in the control elements catalyze abrupt changes in the ecosystem features. The determination of some strategic ecological variables and their responses to the spatial and temporal patterns of fire can guide the proposition of appropriate thresholds for different landscape units. A continuous assessment of these fire regime thresholds allows readjustment of management practices through an adaptive process that seeks convergence with biodiversity conservation objectives. The use of thresholds of potential concern to monitor the outcomes of management actions has gained recognition between ecologists and land managers. The guide proposed here is a first step in that direction. We present specific fire-related thresholds that could be used as a guide for management actions in the Canastra National Park, we made a retrospective assessment of fire regimes in relation to these thresholds and suggested ecological features that could be employed to monitor the outcomes of these actions.

Keywords: Thresholds of Potential Concern, Adaptive management, Monitoring, Prescribed fires

1. Introduction

Although fire is a natural component of savanna ecosystems shaping plant evolution and global biogeochemical cycles for millions of years (Simon et al., 2009), modern humans have changed the background levels of natural fire activity (Bowman et al., 2011; Chuvieco and Justice, 2010). Humans influence fire patterns in different ways, not only by changing fuel structure and/or connectivity but also suppressing and igniting few or many fires in different weather conditions (Howe, 1994). Despite the continuous improvement of fire use practices, humans often are not able to control the fires they initiate resulting in destructive and uncontrolled wildfires. Faced with the challenge of aligning biodiversity conservation goals with livelihood requirements of landowners, scientists, policymakers and managers around the world have discussed potential solutions to fire management in protected areas especially concerned with the sustainability of these ecosystems.

In South America, a cultural heritage which advocates that fire produces only deleterious impacts on biodiversity has led government agencies and managers to apply fire suppression policies in savanna protected areas (Bilbao et al., 2010; Durigan and Ratter, 2016). On the other hand, the intentional use of fire to maintain desired conditions in fire-prone ecosystems has progressively gained importance between scientists and managers in other international settings. In Australia and South Africa, for example, managers have used prescribed burning in protected areas for a number of purposes, including to mitigate the severity of wildfires by controlling seasonality and reducing the build-up of flammable fuel loads, and to maintain biodiversity values (Price et al., 2012; Russell-Smith, 2016; Russell-Smith et al., 2013; Van Wilgen et al., 2014).

Other researchers and practitioners have operated under a restoration ecology paradigm trying to replicate the patterns of fires ignited by lightning or pre-industrial humans period (Ramos-Neto and Pivello, 2000). The conceptual framework behind this approach holds that simulating “natural fire regimes” is the best alternative to achieve particular biodiversity or fuel reduction outcomes. However, it is important to consider that many protected areas are effectively islands surrounded by pastures and agriculture (Clarke, 2008; Freeman et al., 2017), with associated high risks of human caused ignition. Therefore, unplanned fires must be incorporated into decision-making processes, especially in areas surrounded by human activities (Van Wilgen, 2009). Furthermore, parameters of a “natural fire regime” are both currently unknown and will also change over time (Van Wilgen et al., 1998). Observing the restricted impacts of natural fires on fauna and flora, managers in Brazilian protected areas (i.e. Jalapão National Park, Campos Amazônicos National Park and Chapada dos Guimarães National Park) have applied low-intensity prescribed burning in order to simulate “natural fires” and reduce fuel buildup in an attempt to reduce the intensity and size of subsequent wildfires.

A patch mosaic burning (PMB) is a fire management strategy that aims to introduce heterogeneity across space and time into the landscape by using a mosaic of patches representative of a range of fire histories in order to maximize biodiversity requirements (Brockett et al., 2001). This approach is supported by the assumption that a greater range of fire regimes (pyrodiversity) begets diversity in fire-prone plant communities, which, in turn, can increase faunal diversity (Brockett et al., 2001; Parr and Brockett, 1999; Parr and Andersen, 2006). Given that species respond differently to different fire regimes, it follows that fire-prone systems include as many successional states as possible, large enough to support successional-specific species.

However, a major challenge confronting ecologists and land managers in applying PMB is how much pyrodiversity is required for maintaining the diversity of animals, as well as plants? Or how much pyrodiversity can be considered (un)natural? In an attempt to define the levels at which fire should be managed for the benefit of all species of the community, much effort has been made to describe the “Vital Attributes” of particular fire-sensitive plant species (Gosper et al., 2013; Noble and Slatyer, 1980), assuming that the requirements of these key species will encapsulate the needs of other species in the community. One problem with this assumption is that the maintenance of a given key plant species at a site does not ensure the conservation of the entire community (Clarke, 2008). For most animal species it is desirable to preserve the functional and structural attributes of the vegetation that provide key resources for survival and reproduction (Croft et al., 2016). For that reason, in addition to enabling the survival of sensitive species, other ecological features of major importance to animals also must be considered in fire management plans.

To effectively manage fire in protected areas it is essential to monitor ecological outcomes (Van Wilgen, 2009). Seen in these terms, a growing approach that has gained recognition with ecologists and land managers is the integration of thresholds of potential concern (TPC) as part of adaptive fire management. Assuming that natural systems are characterized by nonlinear dynamics, ecological thresholds could be defined as the point at which there is an abrupt change in the ecosystem features produced by small changes in the drivers (Foley et al., 2015; Groffman et al., 2006). Using these ecosystem features as a guide, practitioners can develop an integrated understanding of how drivers, e.g. fire regimes, are likely to affect key features of the ecosystem. Sometimes it may be necessary to implement management actions in order to prevent the system reaching a tipping point. In such cases long-term monitoring is important because although some pressures, such as fire regimes, can be manipulated by management actions, it may take years until the effects of that action can be seen.

The employment of TPC in fire management planning aims to maintain or enhance the resilience of native ecosystems based on accumulating knowledge of the responses of flora and fauna species, populations and communities to fire regimes (Edwards and Russell-Smith, 2009; Foley et al., 2015; Groffman et al., 2006; Van Wilgen et al., 2014, 1998). Population declines or eventual local extinction can be anticipated where thresholds are exceeded.

Incorporating ecological thresholds into management is not a trivial task, especially given limited data often available for establishing appropriate thresholds in many ecosystems. Despite these limitations, the process of identifying and refining thresholds is an important component of the adaptive management cycle and will enhance environmental decision-making and management over time. To assist land managers, we demonstrate how ecological thresholds can be incorporated into decision-making processes concerning fire management in the Canastra National Park (CNP), Southeast Brazil.

2. An overview of the fire management context

When fire suppression policies are applied in fire-prone savanna ecosystems the result is a large accumulation of woody and grassy biomass. In the particular case of the CNP, it is important to emphasize that it becomes a short-memory system, with rapid fuel accumulation and curing of grass and litter at the beginning of the dry season associated with low canopy cover. The uplands are characterized by an extensive matrix of grasses interspersed by small disconnected forest patches. Together, these features make the system highly vulnerable to the occurrence of large fires, which in turn creates uniformity in fire history and landscape structure, increasing the chances of subsequent large fire events. Generally fires are initiated by farmers in the vicinity of, or in properties embedded within, the CNP. These fires invade the uplands and quickly become out of control. Pastoral management undertaken by landowners is performed on plots with biennial rotation, allowing the fire to occur annually and making the system entirely dependent on the fuel accumulation rates (bottom-up control). On the other hand, social conflicts intrinsic to the historical background and current fire management policies increase the risk of unplanned fires late in the dry season. These fires are potentially destructive because they are typically severe, less patchy (smaller and fewer unburned patches) and usually grow to a larger size. A detailed description of the fire context in the CNP can be found in Batista et al. (2018) and is summarized in Figure 15.

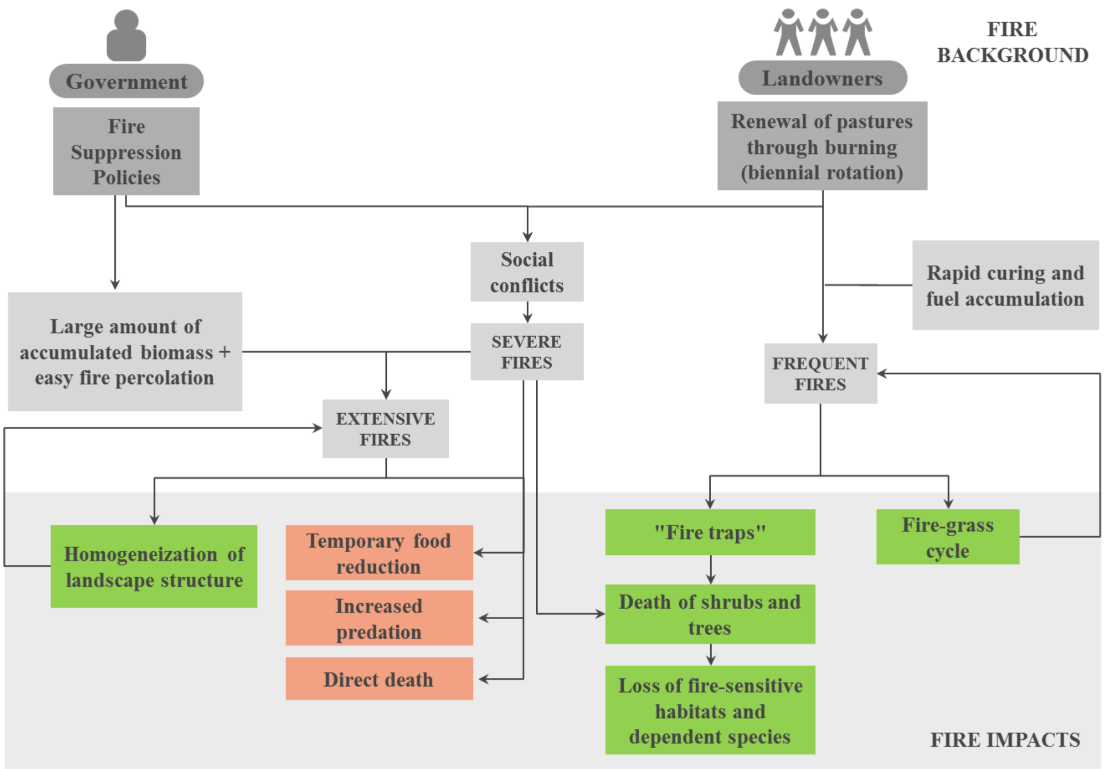


Figure 15. Simplified conceptual diagram showing the background and impacts of fire regimes in the CNP. The government has influenced fire regimes in the CNP
implementing fire suppression policies, while landowners have ignited fires to manage their properties. Fire suppression polices go against people’s livelihoods, generating social conflicts, in addition to promoting large biomass accumulation. Despite the prohibitions, landowners usually burn pastures late in the dry season, resulting in severe unplanned fires that can easily become out of control. A large amount of accumulated biomass in a homogeneous landscape results in extensive fires that may cause negative impacts on populations of animals in addition to promoting the openness and homogenization of landscape structure. On the other hand, homogeneous open landscapes favor fire percolation promoting extensive fires. Severe fires also kill shrubs and trees. The use of fire by landowners in a biennial rotation, associated with the rapid fuel accumulation, results in high frequency of burning, reinforcing the fire-grass cycle and keeping woody elements in small and vulnerable classes-size. The death of trees and shrubs, particularly on the edges of forest patches, enables fire incursions, loss of biodiversity, and damaging the quality of these habitats as well as its ability to provide resources and services for dependent fauna.

3. Broad Ecological Objectives

The main reason for the creation of CNP in 1972 was to preserve the source of the São Francisco River. However, the Park supports a wide diversity of wildlife, with critically endangered species adapted to grasslands, wooded savannas and forest environments. According to the CNP's Management Plan (IBAMA, 2005), the protected area should aim to: (1) ensure that recharging zones and river springs will be protected at long term; (2) maintain the diversity of physiognomies at the landscape (forests, wooded savannas, rocky savannas and grasslands); (3) promote connectivity and gene flow between populations of native species; and (4) protect populations of endangered and endemic species.

To achieve these biodiversity conservation outcomes in fire-prone CNP requires the prior consideration of some general principles:

I. Fire management must be both preventive and adaptive ("learning by doing");
II. Fire management should consider the livelihood aspirations and imperatives of embedded and surrounding local communities;
III. Unplanned fires should be considered likely to occur and need to be managed for;
IV. Plant and animal needs, and structural and functional habitat attributes, are important for maintaining ecosystem values;
V. It is essential to continually gather information on the biology and ecological requirements of species (especially fire-sensitive species, immobile or dependent animals and those with small home ranges), to inform the development and implementation of appropriate adaptive fire management;
VI. Romero and Nakajima (1999) reported 17 areas of endemism in the CNP based on the distribution of 45 endemic species, including species not yet described and others that are not found in any other protected area. Population parameters and ecological requirements of these species should be continuously monitored.

4. Fire Management Units (FMU)

The delineation of a set of Fire Management Units (FMU) presented here was based on the vegetation types mapping (Fig. 16) and their respective fire sensitivity. The vegetation map was created by using OLI/Landsat-8 scenes acquired in the late dry season of 2015. Mapping was performed using Geographic Object-Based Image Analysis

(GEOBIA) with the software eCognition Developer 8.7. Fire sensitivity was based on published information sources and accumulating knowledge of flora and fauna species, populations and communities and their response to fire regimes, as well as particular edaphic characteristics.

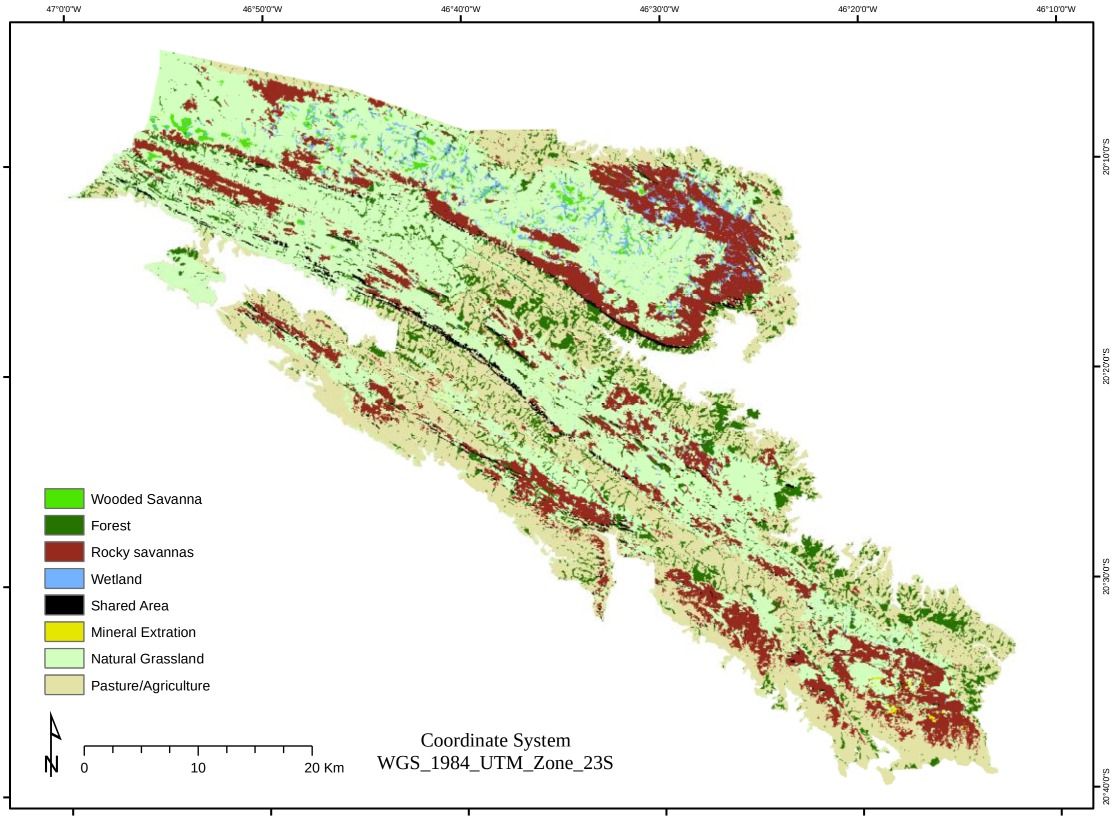


Figure 16. Vegetation map of the CNP.

The above delineation reveals three distinct FMUs: (1) Forests, (2) Grasslands/wooded savannas, and (3) Rocky savannas (Fig. 17). For present purposes pastures, crops and mining areas are not considered here because they are located on private property not yet regulated by the government. We propose specific objectives for each FMU (Tab.3) to guide planning and implementation of management actions, but it is important to highlight the importance of monitoring the ecological outcomes and to adapt actions continuously, as necessary.

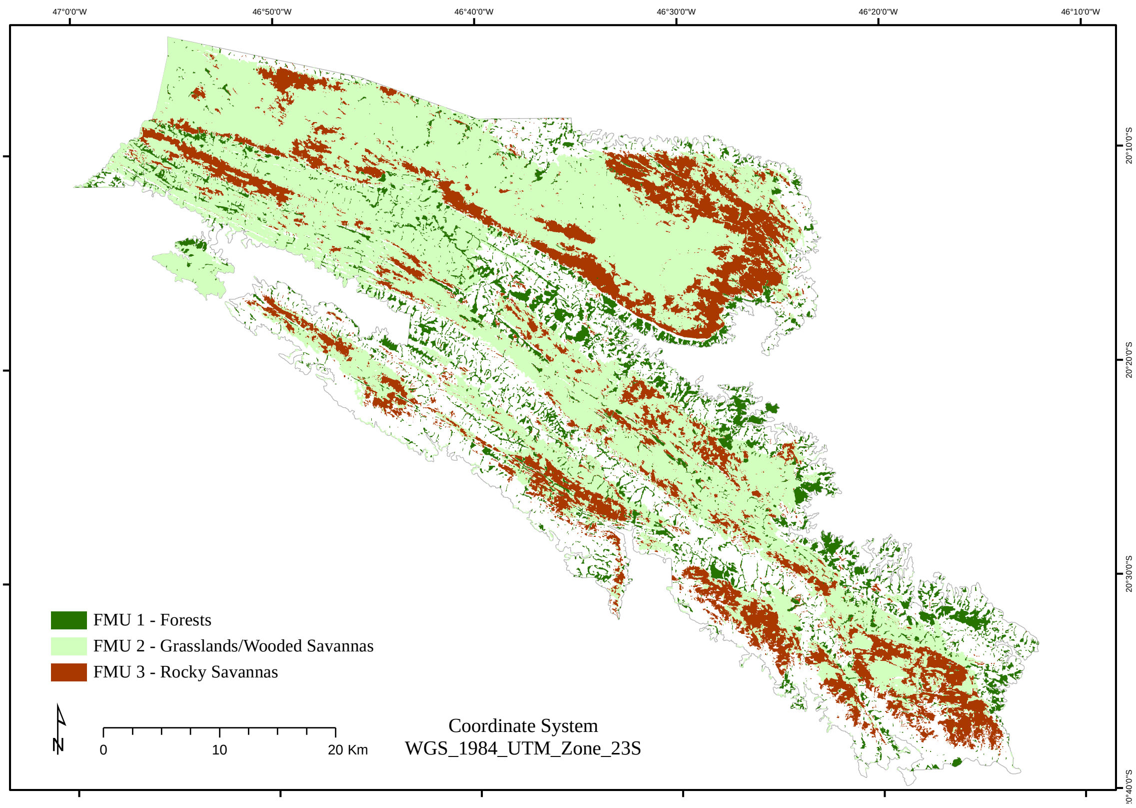


Figure 17. Fire management units based on the vegetation map and published information sources.

5. Specific concerns and management proposals

5.1. FMU 1 - Forests

<u>Specific concerns</u>: Patches of fire-sensitive vegetation often occur within fire-prone tropical savannas, and may act as important refugia for animals and plants (Radford et al., 2013). The thin bark of forest species (Hoffmann and Solbrig, 2003) makes them particularly vulnerable to fires, while repeated topkill of saplings, sprouts and young trees prevents recruitment into adult size classes. After burning and death of trees, forest edges suffer biomass collapse and microclimate changes increasing openness and making them susceptible to subsequent fire incursions (Cochrane, 2003). Didham et al. (1999) suggested that open-canopied edges may enforce more adverse conditions for organisms within forest fragments, so that fragments with open edges will have a smaller unaffected core area than those with closed edges. This edge effect can be even more pronounced in smaller fragments, where it has been observed a lower number of rare and shade-tolerant tree species, lower canopy height, higher temperature, higher evaporative drying rate, lower leaf litter moisture content, and lower litter depth (Didham et al., 1999; Hill and Curran, 2003). In other words, small fragments, such as those found in the uplands of the CNP, are more vulnerable to extinction from current fire regimes and less able to support

biodiversity of plants and animals. In the course of this study, trees killed by fire were observed on the edges of forest patches in the Canastra sector.

Several species present in the CNP depend partially or entirely on the resources granted by forests, and so the maintenance of these habitats, even in small fragments, is of major concern (IBAMA, 2005). Marini-Filho and Martins (2010) provided evidence that the CNP is an environment capable of sustaining forest dependent butterflies in metapopulations, suggesting that the protection of large forest fragments located in lowlands and outside the Park may be necessary to promote colonization of smaller forest fragments inside the plateau. The giant anteater (*Tamandua tetradactyla*), for instance, usually selects forest edges but is more frequently observed in forested habitats which provides them with more opportunities to flee from predators by climbing up trees, and where they can also feed on ants and termites. The Brazilian merganser (*Mergus octosetaceus*), one of the most threatened birds in the Americas and categorized as critically endangered due to its small and declining populations, uses these habitats, particularly the riparian forests, for feeding and breeding (BirdLife International, 2015; Hughes et al., 2006; Lamas, 2006).

In general, arboreal species are more negatively affected by fire in forest habitats than fossorial species, which can survive wildfires by staying in underground holes while fire passes (Lindenmayer et al., 2013; Vieira and Marinho-Filho, 1998; Whelan, 1995). Mendonça et al. (2015) reported collapse of populations of the gracile mouse (*Gracilinanus agilis*), an arboreal marsupial also present in the CNP, after a severe fire that burned most of gallery forests in another reserve, suggesting direct mortality or emigration. Although fossorial animals may be able to avoid incineration, the thermal stress, smoke toxicity and increased predation pressure due to lack of shelter or scarcity of food and resources after fire may increase mortality rates (Leahy et al., 2015; Whelan, 1995).

Particularly, forest avian carnivores and nectarivores are more sensitive and vulnerable to short-term changes caused by fires. It reflects the relatively high trophic status and large minimum-area requirements of avian carnivores, while avian nectarivores are generally small specialists with restricted ranges (Hill et al., 2011). By contrast, avian granivores and insectivores may be able to persist after fires because they can exploit resources in extensive grasslands or recently burned habitats. Based on these considerations, the relative abundance of those more sensitive guilds in the CNP can be used as an indicator of ecosystem health, since species with restricted dispersal ability will not survive in a fragmented environment that cannot sustain a metapopulation.

In conclusion, given the importance of forest habitats as refugia and a source of resources for dependent wildlife, management actions should focus on protecting and improving the quality of these habitats, minimizing the risk of severe fires and controlling the edge-effects.

Management proposals: This FMU includes riparian and mesophilic forests usually associated with watercourses or higher soil moisture. In the uplands, forests are limited to small patches with high insulation levels, while larger patches are located in the slopes and valleys. These remaining forests are important habitats for vulnerable and endemic species, and provide refugia for fire-sensitive species to escape high-intensity fires. It is advisable to enlarge the fragments as much as possible (considering the edaphic limitations) and reduce the edge-effects in order to protect the core areas, increase the biodiversity, the quality and availability of resources for sensitive or dependent wildlife. We propose that no deliberate burning should be performed inside these units, but natural fires ignited by lightning could simply be tolerated. Any late dry season fire should be extinct. In order to protect the forests of fire incursions, some fuel control should be performed next to the patch borders and the maintenance of long-unburnt areas around these forest patches should be encouraged through reducing the frequency of fires. We propose to do this by imposing a system of strategic firebreaks.

5.2. FMU 2 - Grasslands and Wooded Savannas

Specific concerns: The matrix in which forest patches are embedded is characterized by large areas of continuous grasslands, unrestricted flow of winds, intense insolation, and low humidity. Together, these features produce ideal conditions for fire percolation, making it easy for fire to burn over extensive areas. In these physiognomies, fire has a key role not only removing the biomass accumulation that inhibits the survival and growth of additional herbaceous plants, but also converting them to ash and nutrients (Vogl J, 1979). In some species, germination may be stimulated by fire or reproduction may be enhanced through flowering response. Studies have demonstrated that post-burn plants usually grow larger and more vigorously in addition to producing more flowers and seeds (Fidelis et al., 2010). Fire exclusion can lead to loss of grassland species, mostly forbs, since fire can remove above-ground biomass and, consequently, open new microsites for plant establishment and survival (Fidelis et al., 2012).

Despite the apparent balance between fire and savanna ecosystems, fire regimes currently imposed by human practices also may have negative impacts on biodiversity. Firstly, the homogenization of fire histories through large areas can lead to the loss of species for long periods of time, since the landscape becomes unable to ensure resources for species in all successional stages (Briani et al., 2004; Kelly et al., 2012; Lindenmayer et al., 2016; Taylor et al., 2005). Additionally, the recolonization of burned areas by species with low dispersal capacity will be more difficult when large areas are affected, because it depends on the migration of individuals from nearby areas across unsuitable habitat (Mendonça et al., 2015b). Berry et al. (2015) recommended maintaining large areas of intact unburnt vegetation and minimizing the extent of large homogenously burnt areas in order to facilitate rapid rates of post-fire recolonization. The retention of older vegetation in fire-prone ecosystems should also be encouraged because it may act as a refuge, and the structural complexity found in these habitats may take decades to develop (Kelly et al., 2012; Russell-Smith et al., 2017).

Another concern regarding large fires is their potential to cause mass mortality. Basically, if fires are so large as to make it impossible for a particular animal to escape to unburnt areas, they will incur significant fitness costs and population decline (Lawes et al., 2015). Giant anteaters (*Myrmecophaga tridactyla*), for example, are slow-moving animals present in the CNP that can be easily burned during hot grassland fires because of their highly-flammable pelage (Redford, 1994). However, large fires can be especially deleterious to small, isolated populations of small mammals causing direct mortality, increased exposure to predators (Leahy et al., 2015), and destruction of neighbouring potential source areas. Even those small-mammal species that live in burrows or rock gaps can declines in abundance some time after fire, emphasizing the relevance of indirect fire effects (especially large-scale and low-patchiness fires) on the sustainability of these species (Lawes et al. 2015) . Otherwise, amphibians and reptiles are able to take shelter in underground burrows or find moist refugia as protection from fires (Pilliod et al., 2003). However, some fires may move too quickly, resulting in some fauna being vulnerable to direct mortality from fire (Koproski et al., 2006). In addition, severe fires (August-November) may intensify the mortality of animals due to higher intensity and speed of fire fronts, especially if considered that the birth of nestlings also happens between the end of the dry period and the beginning of the rains (Falleiro, 2011). These young individuals have displacement limitations and cannot escape or adequately protect themselves from the flames.

In addition to spatial patterns, the frequency with which fires have occurred lately (especially the severe ones) has generated major concerns among researchers and managers. Hoffmann (1998) showed that fire produces an immediate reduction in sexual reproductive success by causing abortion of flowers and fruits, interrupting the reproductive cycle and forcing the vegetative sprout. Disturbingly, the reproductive period of many species from Brazilian savannas happens at the same time as anthropogenic severe fires season (Falleiro, 2011). In the long term, frequent fires will generate shifts in the community composition by favoring plants capable of resprouting over obligate seeders. Obligate seeders are more susceptible to population decline and local extinction because mature individuals are killed by fire and regeneration depends exclusively on the seed bank (Burrows et al., 2008; Noble and Slatyer, 1980; Whelan, 1995). If a second fire occurs before developing obligate seeders reaches reproductive maturity, the permanence of the species on the site can be threatened. Given the significant dependence of these species on the seed bank to survive, particular attention should also appropriately be given to severe fires, in which high temperatures could damage seeds stored below ground. Several studies have reported substantial decreases in the density of seed banks after a fire, particularly those more sensitive or that were not deeply buried in the soil (de Camargos et al., 2013; Martinez-Orea et al., 2010; Tesfaye et al., 2004).

Neves et al. (2011) has warned against the decline observed in populations of several Eriocaulaceae species that occur in Cerrado, which has been attributed to an apparent increase in the frequency of fires. Fire promotes an increase in the number of reproductive individuals and for polycarpic species, the increased reproductive effort may negatively impact growth, survival and the production of inflorescences in years after burning. Moreover, frequent fires can lead monocarpic species to population declines through the exhaustion of seed banks and the mortality of reproductive individuals.

With these assumptions in mind, it is suitable to emphasize that fire regimes in the CNP, currently distinguished by large, frequent and severe fires, are possibly generating homogeneity in habitat structure and fire histories, favoring grasslands and eliminating woody species or keeping them in lower, non-reproductive and vulnerable height classes.

Nonetheless, despite the important role of fire in maintaining natural grassland species, it is apparent that, in the light of current fire regimes, the CNP is probably losing diversity of woody plants and wildlife.

Management proposals: This zone ranges from dense grasslands without shrubs or trees to physiognomies dominated by trees and shrubs often 3-8 m tall and having more than 30% crown cover (Oliveira-Filho and Ratter, 2002). Between these two extremes there are intermediate structures, varying in the density of woody elements. This high diversity of physiognomies can be attributed, amongst other factors, to fire frequency (Coutinho, 1990). As this unit is affected by frequent fires (mean fire return periods of between 1-2 years), usually ignited late in the dry season (high-intensity fires), woody components tend to be eliminated and give rise to dominance by grasses and forbs. Thus, for this unit, pyrodiversity is required in order to provide the woody saplings with more opportunities to be recruited into larger, reproductive and fire resistant size classes. A range of fire histories may also encapsulate the requirements of fauna species adapted to different sucessional stages and long-unburnt habitats could provide refugia and diversity of resources for dependent wildlife. We suggest reducing the fire frequency close to forest patches, by establishing long-unburnt areas (2-9 years) as a way to minimize the edge-effects on these habitats. Deliberate burning should be conducted in the grasslands before the late dry season months with the aim of increasing spatial heterogeneity, such that patches of unburnt vegetation are compartmentalized between recently burned areas and a system of strategically placed firebreaks. Early dry season fires typically are of lower intensity and more patchy, providing refugia and sources of food for fauna (Oliveira et al., 2015). In addition, patchy early dry season fires would allow seedling recruitment, nutritious leaf flush for herbivores and the protection of trees and shrubs, reducing mortality and topkill. The risk of unplanned fires entering the uplands of the CNP from the vicinity could be reduced by creating effective firebreaks in critical strategic situations. Good planning and design of strategic barriers can maximize effectiveness while reducing the area to be treated (Finney, 2001).

5.3. FMU 3 - Rocky savannas

Specific concerns: Rocky savannas also compose the mosaic of this landscape, being primarily determined by bottom-up edaphic control imposed by shallow, dry, hard, coarse-textured dystrophic soils derived from the nutrient-poor quartzite. This particular ecosystem is well adapted to natural fire regimes. Several species apparently require fire in order to flower and plants with special adaptations to withstand fires, including subterranean organs (e.g. lignotubers) constitute a large proportion of woody species (Figueira et al., 2016; Kolbek and Alves, 2008). Kolbek and Alves (2008) concluded that rocky savannas are largely consistent with the intermediate disturbance hypothesis: with fewer fire events, the buildup of organic debris tend to increase fire intensity to lethal levels for several plant species, whereas some populations decline due to limited recruitment and mortality caused by competition with surrounding grasses. On these ecosystems, the fire may kill part of vegetation, which does not usually have a deep soil to hide its gems and xylopodia. On the other hand, too frequent fire events can not only exclude shrubs and trees and favor grasses and forbs, but could also lead populations to decline by the exhaustion of seed banks, seedlings mortality, induced reproduction of young/small individuals and early death in semelparous species (Alves and Silva, 2011; Figueira et al., 2016).

<u>Management proposals</u>: This unit is characterized by herbaceous or shrubby vegetation with occasional presence of undeveloped trees. Rocky savannas form a mosaic with grasslands, but the FMU is easily recognized by its rocky outcrops with shallow soils. This unit has been burned frequently (mean fire return periods of 1-2 years), but deliberate burning should be limited to minimum intervals of 3-4 years, based on accumulation rates of biomass and probability of burning (Figueira et al., 2016; Oliveras et al., 2013). Although fire is possible before 3 years, this interval is apparently necessary for the reestablishment of plant communities in rocky savannas (Brito, 2011). Fire should not be prescribed in fixed rotation. Instead, free running burns performed early in the dry season or rainy season should be adopted, because such fires are less severe and destructive, maximizing the chances of survival.

6. Thresholds of Potential Concern (TPC)

Currently, thresholds have been expressed only in terms of fire patterns based on the assumption that pyrodiversity promotes biodiversity, and where monitoring is not usually accompanied by an assessment of the ecosystem state. However, the continuous evaluation of key ecological features can provide managers with evidence of management efficacy and it is crucial to define meaningful targets and thresholds of fire patterns. The guide proposed here is a first step in that direction. The fire-related thresholds that will be used as a guide to management actions, as well as the ecological features that could be used to monitor the outcomes of these actions are presented (Tab.4) and outlined in Sections 6.1-6.3. The proposed thresholds are a tradeoff between ecological requirements, and current challenges associated with achieving these thresholds in practice.

For the purpose of assessing the status of these thresholds, annual fire mapping covering the period (2000-2015) was performed for the entire Park. Methodological details are given in Batista et al. (2018). Fire metrics for each fire management unit were calculated using standard Geographic Information System (GIS) tools available in the software, ArcGis 10.0.

6.1. Thresholds for Forests

Critical thresholds: forests are critical habitats which require longer fire return intervals for their persistence. So, the use of fire is not recommended in these units that should be the focus of both fire suppression and prescribed burning to contain undesirable fire incursions. A threshold of no more than 5% of the area, including borders, should be burnt by severe fires in any one year.

Fires in this FMU burnt an average of 10% each year between 2000 and 2015. The 5% threshold was exceeded in 11 years in the sixteen-year period (Fig.18), especially in 2010, 2012 and 2014, when about 19% of this FMU was burned.

Ecological features: (1) attributes of the fragments such as area, canopy height and edge-effect (core area). The increase in area is advisable and the edges should be less openness; (2) presence or richness of late sucessional species (rare and shade-tolerant). The representability of these species may indicate greater quality of the fragment; (3) in ideal situations the fragments should present low temperatures and high moisture content on deep layers of litter; (4) good representability of exclusive and dependent species (metapopulations), or those in high trophic status, large minimum-area requirements and specialist with restricted ranges.

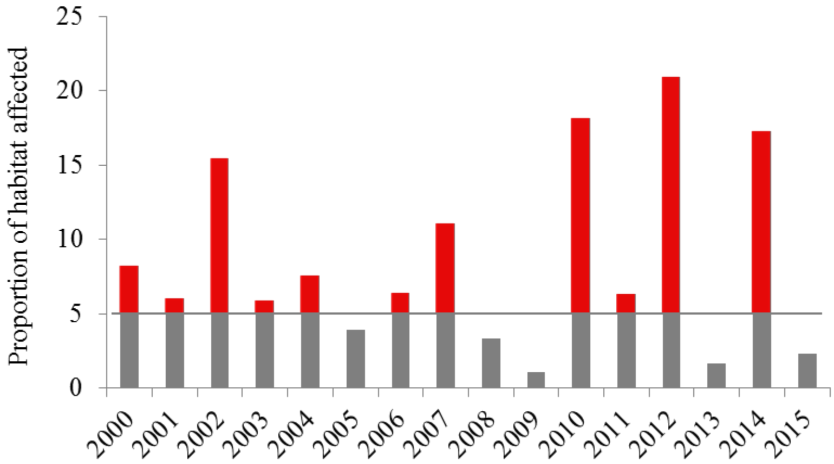


Figure 18. Sixteen-year retrospective assessment of thresholds metrics (< 5%) for severe fires (late dry season) affecting forest patches (FMU1) in the CNP. Note: grey columns indicate that thresholds are within acceptable limits and red columns that it is not.

6.2. Thresholds for Grasslands/Wooded savannas

<u>Critical thresholds</u>: the primary concern in this unit is the too frequent occurrence of large, high-intensity fires. Large fires may amplify the negative effects (indirect and direct) on vulnerable animals and promote homogeneity in the landscape. Low burning implies a lot of fuel accumulation and large fires in subsequent years. A threshold of 50% would be expected to burn annually in this unit. At least 20% of the area burning in a given year constitutes a lower threshold. Severe fires can kill even the most resistant plants and animals most likely to escape the flames. Because of these major implications on biodiversity, unplanned severe fires should be avoided, but up to 10% of the area expected to burn in a given year would be tolerated. Frequent and high-intensity fires have the potential to eliminate shrubs and trees, keeping them in vulnerable and non-reproductive size classes. We recommend fire-free intervals varying between 2-9 years, which is required for the persistence of vulnerable species, including subshrubs, shrubs and trees of Cerrado (Hoffmann, 1999). With respect to fire size, few studies in Brazil have delivered relevant ecological information to support any proposition. But from Australia, Radford (2012) demonstrated that required spatial scales of burning for small mammals are <100ha. Based on this information, we suggest that less than 10% of all fire management unit should be burnt by fires >100ha in extent. Management could encourage natural fires and implement prescribed fire management early in the dry season

in order to create a mosaic of different fuel ages and increase the spatial heterogeneity taking advantage of strategically placed firebreaks (natural or produced).

The thresholds for area to burn were reached in 9 years in a sixteen-year period (2000-2015) (Fig.19a). Particularly after 2008 and 2009, when less than 20% of this FMU was burned, more than 50% burnt in the following year. The actual area burnt by severe fires was higher than would have been expected. The 10% threshold has been exceeded in every year except 2008 (Fig.19b). With respect to fire size, great proportion of this FMU has been burned by fires > 100ha annually, except 2009 and 2013 (Fig.19c).

Ecological features: (1) carcasses of animals killed by fire. Although the absence of carcasses does not mean a reduction in mortality, their presence could indicate that fires are larger and more severe than it should be; (2) populations of obligate seeders, monocarpic species, small mammals (feeding guilds), amphibians and reptiles; (3) maintenance of large patches of older and unburned areas (particularly near to forest patches), keeping in mind that open grasslands are important for several species. So, the massive thickening of woody elements is not recommended;

6.3. Thresholds for Rocky savannas

<u>Critical thresholds</u>: in this unit, there is a concern that fires may be too frequent and too severe such that even fire-resistant species could not survive. On the other hand, long periods without fire will promote fine fuel accumulation, which may result in high-intensity fires. Management should consider reduce the frequency of fires into intermediate intervals (3-4 years). Recommended thresholds are that no more than 30% of these units should burn in any given year. In addition, it is advisable to reduce fire intensity, and the threshold in this case is that no more than 10% of the predicted area should burn in critical months (August-October).

Fires in this FMU burnt more than the 30% threshold in 9 years in a sixteen-year period (2000-2015) (Fig.20a). The incidence of severe fires was higher than the 10% threshold in all years (Fig.20b).

<u>Ecological features</u>: (1) biomass accumulation (grasses); (2) high richness and abundance of native species; (3) mortality of native species should not exceed the recruitment rates; (4) the seed bank must be able to maintain viable populations; (5) induced reproduction has to be carefully evaluated to ensure the persistence of species (particularly the semelparous) on the habitat.

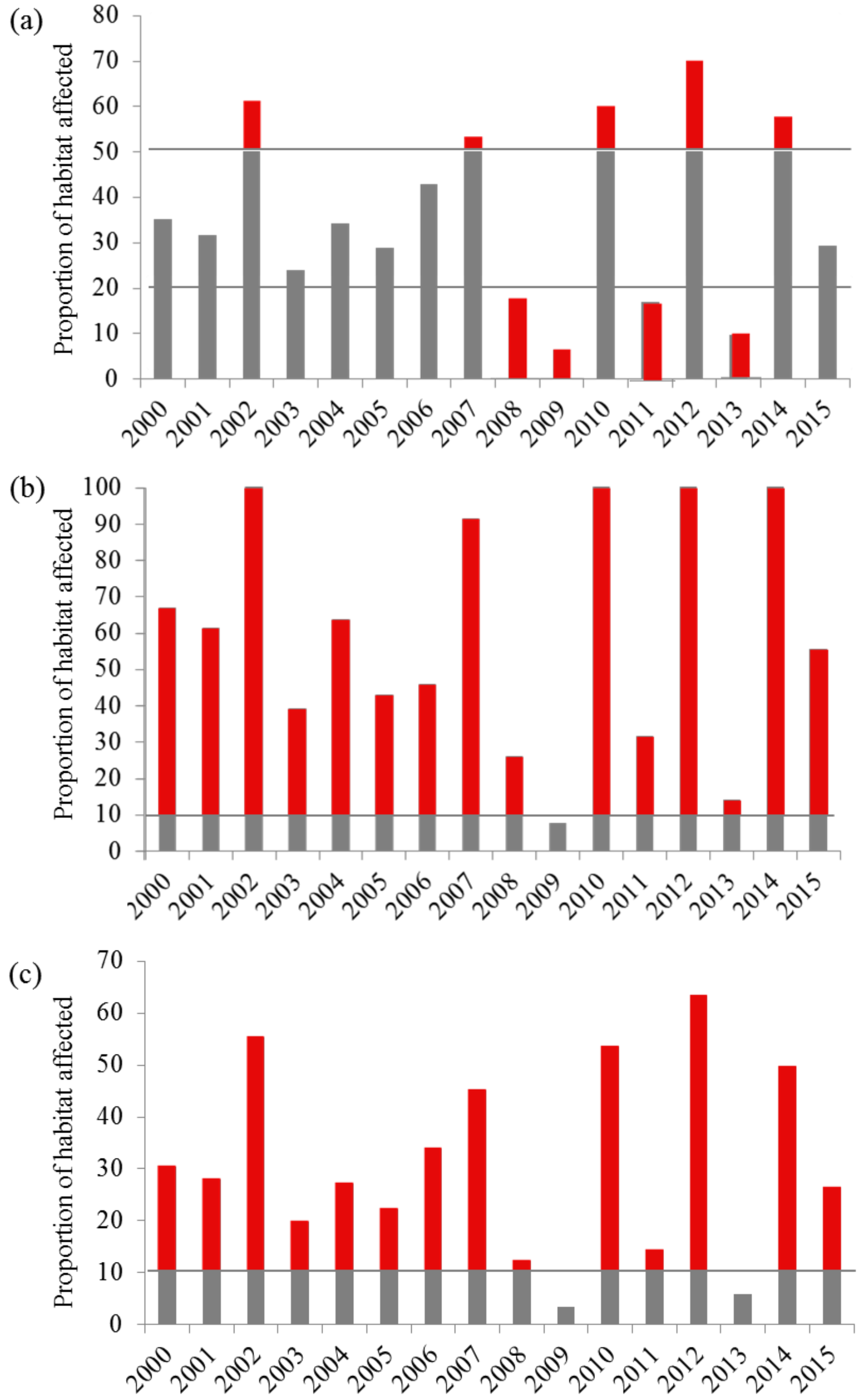


Figure 19. Sixteen-year retrospective assessment of thresholds metrics affecting grasslands and wooded savannas. (a) total area expected to burn; (b) area burnt by severe fires; (c) area affected by fires >100ha. Note: grey columns indicate that thresholds are within acceptable limits and red columns that it is not.

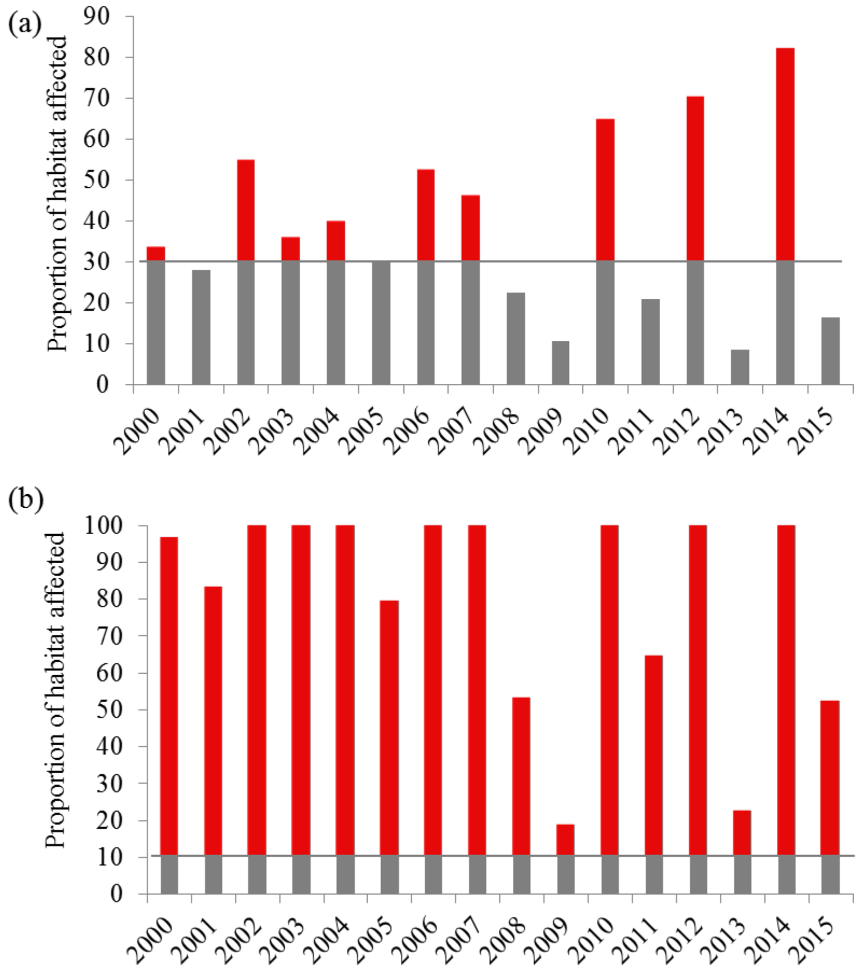


Figure 20. Sixteen-year retrospective assessment of thresholds metrics affecting rocky savannas. (a) total area expected to burn; (b) area burnt by severe fires. Note: grey columns indicate that thresholds are within acceptable limits and red columns that it is not.

7. Future challenges

The key challenge in the CNP, as well as most of Brazilian protected areas, is getting park authorities and local communities to work together to devise an appropriate effective fire management plan and then trial its implementation. In face of all variables to be considered, we realize that the guideline proposed here could not be applied throughout the whole Park at first. A realistic process requires following a road map by which all guidelines could be implemented and tested in order to define what resources, including time, funds, ecological information and community engagement would be necessary for the total implementation.

In addition to all ecological impacts discussed above, too frequent, severe and large fires certainly result in significant economic damages. Large resource amounts are expended on fire fighting in Brazil, generally. An average of R$15 million from the annual federal budget is destined to prevention and control of fires, which continues to be a challenge to Environmental Agencies and a serious threat to biological conservation. Just in Minas Gerais, a state that encompasses many protected areas of Brazilian Cerrado, more than R$20 million are aimed at fire prevention and control, mostly with airplanes and people in land. Neither costs nor fire damages have reduced significantly throughout the time, which indicates that our strategies, in addition to be very expensive, are probably ineffective because we are acting without thinking. Hence, the implementation of an adaptive fire management in Brazilian savannas becomes ecologically and economically needed.

Table 3. Specific ecological objectives and proposed fire management actions.

FMU	Specific ecological objectives	Fire management actions
FMU 1 Forests	(1) Prevent further loss of large trees. (2) Encourage recruitment of tree samplings into life stages where fires are not able to kill them. (3) Increase the biodiversity inside de fragments and the availability of resources. (4) Allow the establishment of shade-tolerant species.	(A) Suppression of severe fires inside forest patches. (B) Fuel reduction burns around forest patches. (C) Tolerance of natural (lightning) fires.
FMU 2 Grasslands/Wooded savanas	(1) Promote diversity of physiognomic types of the Cerrado. (2) Reduce the risk of large, high-intensity wildfires and consequently the risk of mortality, mainly for birds and mammals. (3) Avoid burning in recharge areas and river springs. (4) Create a mosaic of nutritious patches for large mammal herbivores	(A) Reduce the risk of unplanned (late dry season) fires entering the CNP from neighboring properties maintaining an adequate firebreak along critical borders. (B) Strategic placement of early season, low intensity fires to create firebreaks. (C) Deliberate prescribed burning mimicking lightning fires to promote a mosaic of different fuel ages. (D) Fuel reduction around recharge areas and river springs. (E) Tolerance of natural (lightning) fires.
FMU 3 Rocky savannas	(1) Maintaining species diversity. (2) Avoid severe, too frequent or too infrequent wildfires.	(A) Reduce the risk of unplanned (late dry season) fires entering the CNP from neighboring properties maintaining an adequate firebreak along critical borders. (B) Deliberate prescribed burning in patches and intervals of 3-4 years. (C) Tolerance of natural (lightning) fires.

Table 4. Fire Management Units (FMUs) with suggestions of ecological features that could be used to monitor the outcomes of management actions and the critical fire-related thresholds. * If the mortality is higher than recruitment, the population will decline up to the point where it could, indeed, be extinguished. ** The induced reproduction is important and necessary for some species, but it might be a threat to semelparous plants.

FMU	Suggestions of parameters to be assessed	Critical thresholds
FMU 1 Forests	1. ↑Core area/↓border width and openness 2. ↑Size and ↓isolation of fragments 3. ↑Presence/richness of rare and shade-tolerant tree species 4. ↑Canopy height 5. ↓Temperature inside the fragments, ↑litter moisture content and ↑depth 6. ↑Population size of: forest dependent species, high trophic status, large minimum-area requirements, small specialist with restricted ranges	(A) No more than 5% of forest fragments, including the borders, should be affected by severe fires.
FMU 2 Grasslands/Wooded savannas	1. ↓mortality of animals 2. ↑diversity of fire histories 3. Response of obligate seeders 4. Response of endemic, threatened and monocarpic species 5. Response of species with limited dispersal capacity 6. Populations of high trophic status, large minimum-area requirements and small specialist with restricted ranges	(A) At least 20%, but <50% of grasslands/wooded savannas should be burned annually. (B) <10% of burned area should be affected by severe fires. (C) <10% of the FMU should be burnt by individual fires >100ha in extent. (D) Fire-free intervals should vary between 2-9 years.
FMU 3 Rock savannas	1. ↓Grasses (biomass accumulation) 2. ↑Rickness and abundance of native species 3. *Recruitment/mortality 4. ↑Seed bank 5. **Induced reprodution 6. ↑Semelparous population 7. Response of endemic species	(A) < 30% of rocky savannas should burn in any given year. (B) < 10% of the burned area should be affected by unplanned severe fires. (C) Fire-free intervals should vary between 3-4 years.

8. References

Alves, R.J.V., Silva, N.G. Da, 2011. O Fogo é Sempre um Vilão nos Campos Rupestres ? Biodiversidade Bras. 2, 120–127.

Batista, E.K.L., Russell-Smith, J., França, H., Figueira, J.E.C., 2018. An evaluation of contemporary savanna fire regimes in the Canastra National Park, Brazil: outcomes of fire suppression policies. J. Environ. Manage. 205, 40-49. doi.org/10.1016/j.jenvman.2017.09.053

Berry, L.E., Lindenmayer, D.B., Driscoll, D. a., 2015. Large unburnt areas, not small unburnt patches, are needed to conserve avian diversity in fire-prone landscapes. J. Appl. Ecol. 52, 486–495. doi:10.1111/1365-2664.12387

Bilbao, B. a., Leal, A. V., Méndez, C.L., 2010. Indigenous use of fire and forest loss in Canaima National Park, Venezuela. Assessment of and tools for alternative strategies of fire management in Pemón Indigenous Lands. Hum. Ecol. 38, 663–673.

BirdLife International, 2015. Threatened Birds of the world. Lynx Editions, Barcelona and Cambridge, U.K.

Bowman, D.M.J.S., 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., 2011. The human dimension of fire regimes on Earth. J. Biogeogr. 38, 2223–2236.

Briani, D.C., Palma, A.R.T., Vieira, E.M., Henriques, R.P.B., 2004. Post-fire succession of small mammals in the Cerrado of Brasil. Biodivers. Conserv. 13, 1023–1037. doi:10.1023/B:BIOC.0000014467.27138.0b

Brito, J.C., 2011. Efeitos do fogo sobre a vegetação em duas áreas de campo rupestre na Chapada Diamantina, Bahia, Brasil. Universidade Estadual de Feira de Santana.

Brockett, B.H., Biggs, H.C., Van Wilgen, B.W., 2001. A patch mosaic burning system for conservation areas in southern African savannas. Int. J. Wildl. Fire 10, 169–183.

Burrows, N.D., 2008. Linking fire ecology and fire management in south-west Australian forest landscapes. For. Ecol. Manage. 255, 2394–2406.

Camargos, V.L., Martins, S.V., Ribeiro, G.A., Carmo, F.M. da S., da Silva, A.F., 2013. Influência do fogo no banco de sementes do solo em Floresta Estacional Semidecidual. Cienc. Florest. 23, 19–28.

Chuvieco, E., Justice, C., 2010. Relations between human factors and global fire activity, in: Chuvieco, E., Li, J., Yang, X. (Eds.), Advances in Earth Observation of Global Change. Springer Netherlands, Dordrecht, pp. 187–199.

Clarke, M.F., 2008. Catering for the needs of fauna in fire management: Science or just wishful thinking? Wildl. Res. 35, 385–394. doi:10.1071/WR07137

Cochrane, M. a, 2003. Fire science for rainforests. Nature 421, 913–919. doi:10.1038/nature01437

Coutinho, L.M., 1990. Fire in the ecology of the brazilian cerrado, in: Goldammer, J.G. (Ed.), Fire in the Tropical Biota. Springer-Verlag, Berlin, pp. 81–105.

Croft, P., Hunter, J.T., Reid, N., 2016. Forgotten fauna: Habitat attributes of long-unburnt open forests and woodlands dictate a rethink of fire management theory and practice. For. Ecol. Manage. 366, 166–174. doi:10.1016/j.foreco.2016.02.015

Didham, R.K., Lawton, J.H., Biotropica, S., Mar, N., Didham, R.K., Lawton, J.H., 1999. Edge Structure Determines the Magnitude of Changes in Microclimate and Vegetation Structure in Tropical Forest Fragments Determines the Magnitude of Changes in Edge Structure Microclimate and Vegetation Structure in Tropical Forest Fragments '. Biotropica 31, 17–30.

Durigan, G., Ratter, J.A., 2016. The need for a consistent fire policy for Cerrado conservation. J. Appl. Ecol. 53, 11–15.

Edwards, A.C., Russell-Smith, J., 2009. Ecological thresholds and the status of fire sensitive vegetation in western Arnhem Land, northern Australia: implications for management. Int. J. Wildl. Fire 18, 127–146.

Falleiro, R.D.M., 2011. Resgate do manejo tradicional do Cerrado com fogo para proteção das Terras Indígenas do oeste do Mato Grosso: um estudo de caso. Biodiversidade Bras. 2, 86–96.

Fernandes, V.S.R., 2012. Entre a regulação e a emancipação social : desafios à continuidade do lugar frente ao Parque Nacional da Serra da Canastra - MG. Universidade Federal de Minas Gerais (UFMG), Belo Horizonte, MG.

Fidelis, A., Blanco, C.C., Müller, S.C., Pillar, V.D., Pfadenhauer, J., 2012. Short-term changes caused by fire and mowing in Brazilian Campos grasslands with different long-term fire histories. J. Veg. Sci. 23, 552–562. doi:10.1111/j.1654-1103.2011.01364.x

Fidelis, A., Delgado-Cartay, M.D., Blanco, C.C., Müller, S.C., Pillar, V.D., Pfadenhauer, J., 2010. Fire intensity and severity in Brazilian Campos grasslands. Interciencia 35, 739–745.

Figueira, J.E.C., Ribeiro, K.T., Ribeiro, M.C., Jacobi, C.M., França, H., Neves, A.C.O., Conceição, A.A., Mourão, F.A., Souza, J.M., Knegt, C.A. de M., 2016. Fire in rupestrian grasslands: plant response and management, in: Fernandes, G.W. (Ed.), Ecology and Conservation of Mountain-Top Grasslands in Brazil. Springer International Publishing Switzerland, Belo Horizonte, pp. 1–34.

Finney, M. a., 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. For. Sci. 47, 219–228.

Foley, M.M., Martone, R.G., Fox, M.D., Kappel, C. V., Mease, L. a., Erickson, A.L., Halpern, B.S., Selkoe, K. a., Taylor, P., Scarborough, C., 2015. Using Ecological Thresholds to Inform Resource Management: Current Options and Future Possibilities. Front. Mar. Sci. 2, 1–12. doi:10.3389/fmars.2015.00095

Freeman, J., Kobziar, L., Cropper, W., 2017. A critical evaluation of the historical fire regime concept in conservation DRAFT. Conserv. Biol. 31, 1–25. doi:10.1111/cobi.12942

Gosper, C.R., Prober, S.M., Yates, C.J., 2013. Estimating fire interval bounds using vital attributes: Implications of uncertainty and among-population variability. Ecol. Appl. 23, 924–935. doi:10.1890/12-0621.1

Groffman, P.M., Baron, J.S., Blett, T., Gold, A.J., Goodman, I., Gunderson, L.H., Levinson, B.M., Palmer, M. a., Paerl, H.W., Peterson, G.D., Poff, N.L., Rejeski, D.W., Reynolds, J.F., Turner, M.G., Weathers, K.C., Wiens, J., 2006. Ecological thresholds: The key to successful environmental management or an important concept with no practical application? Ecosystems 9, 1–13. doi:10.1007/s10021-003-0142-z

Hill, J.K., Gray, M.A., Khen, C.V., Benedick, S., Tawatao, N., Hamer, K.C., 2011. Ecological impacts of tropical forest fragmentation: how consistent are patterns in species richness and nestedness? Philos. Trans. R. Soc. . Ser. B, Biol. Sci. 366, 3265–3276. doi:10.1098/rstb.2011.0050

Hill, J.L., Curran, P.J., 2003. Area, shape and isolation of tropical forest fragments: effects on tree species diversity and implications for conservation. J. Biogeogr. 30, 1391–1403. doi:10.1046/j.1365-2699.2003.00930.x

Hoffmann, W.A., 1999. Fire and population dynamics of woody plants in a neotropical savanna : matrix model projections. Ecology 80, 1354–1369.

Hoffmann, W.A., Solbrig, O.T., 2003. The role of topkill in the differential response of savanna woody species to fire. For. Ecol. Manage. 180, 273–286.

Hoffmann, W. a., 1998. Post-burn reproduction of woody plants in a neotropical savanna: The relative importance of sexual and vegetative reproduction. J. Appl. Ecol. 35, 422–433.

Howe, H.F., 1994. Managing Species Diversity in Tallgrass Prairie: Assumptions and Implications. Conserv. Biol. doi:10.1046/j.1523-1739.1994.08030691.x

Hughes, B., Dugger, B., Cunha, H.J., Lamas, I., Goerck, J., Lins, L., Silveira, L.F., Andrade, R., Bruno, S.F., Rigueira, S., Barros, Y.D.M., 2006. Action Plan for the Conservation of the Brazilian Merganser (Mergus octosetaceus). IBAMA, Brasilia, DF.

IBAMA, 2005. Plano de Manejo do Parque Nacional da Serra da Canastra. Brasília, DF.

Kelly, L.T., Nimmo, D.G., Spence-Bailey, L.M., Taylor, R.S., Watson, S.J., Clarke, M.F., Bennett, A.F., 2012. Managing fire mosaics for small mammal conservation: A landscape perspective. J. Appl. Ecol. 49, 412–421. doi:10.1111/j.1365-2664.2012.02124.x

Kolbek, J., Alves, R.J.V., 2008. Impacts of cattle, fire and wind in rocky savannas, southeastern Brazil. Acta Univ. Carolinae Environ. 22, 111–130.

Koproski, L., Mangini, P.R., Pachaly, J.R., Batista, A.C., Soares, R.V., 2006. Impactos do fogo sobre serpentes (Squamata) no Parque Nacional de Ilha Grande (PR/MS), Brasil. Arq. Ciências Veterinárias e Zool. da Unipar 9, 129–133.

Lamas, I.R., 2006. Census of Brazilian Merganser Mergus octosetaceus in the region of Serra da Canastra National Park, Brazil, with discussion of its threats and conservation. Bird Conserv. Int. 16, 145.

Lawes, M.J., Murphy, B.P., Fisher, A., Woinarski, J.C.Z., Edwards, A.C., Russell-Smith, J., 2015. Small mammals decline with increasing fire extent in northern Australia: Evidence from long-Term monitoring in Kakadu National Park. Int. J. Wildl. Fire 24, 712–722. doi:10.1071/WF14163

Leahy, L., Legge, S.M., Tuft, K., Mcgregor, H.W., Barmuta, L.A., Jones, M.E., Johnson, C.N., 2015. Amplified predation after fire suppresses rodent populations in Australia's tropical savannas. Wildl. Res. 42, 705–716.

Lindenmayer, D.B., Blanchard, W., MacGregor, C., Barton, P., Banks, S.C., Crane, M., Michael, D., Okada, S., Berry, L., Florance, D., Gill, M., 2016. Temporal trends in mammal responses to fire reveals the complex effects of fire regime attributes. Ecol. Appl. 26, 557–573. doi:10.1890/15-0575

Lindenmayer, D.B., Blanchard, W., McBurney, L., Blair, D., Banks, S.C., Driscoll, D., Smith, a. L., Gill, a. M., 2013. Fire severity and landscape context effects on arboreal marsupials. Biol. Conserv. 167, 137–148. doi:10.1016/j.biocon.2013.07.028

Marini-Filho, O.J., Martins, R.P., 2010. Nymphalid butterfly dispersal among forest fragments at Serra da Canastra National Park, Brazil. J. Insect Conserv. 14, 401–411. doi:10.1007/s10841-010-9271-9

Martinez-Orea, Y., Castillo-Argüero, S., Guadarrama-Chávez, M.P., Sánchez, I., 2010. Post-fire seed bank in a xerophytic shrubland. Boletín la Soc. Botánica México 86, 11–21.

Mendonça, A.F., Armond, T., Camargo, A.C.L., Camargo, N.F., Ribeiro, J.F., Zangrandi, P.L., Vieira, E.M., 2015. Effects of an extensive fire on arboreal small mammal populations in a neotropical savanna woodland. J. Mammal. 96, 368–379. doi:10.1093/jmammal/gyv038

Neves, A.C.D.O., Bedê, L.C., Martins, R.P., 2011. Revisão sobre os Efeitos do Fogo em Eriocaulaceae como Subsídio para a sua Conservação. Biodiversidade Bras. 1, 50–66.

Noble, I.R., Slatyer, R.O., 1980. The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances. Vegetatio 43, 5–21. doi:10.1007/BF00121013

Oliveira-Filho, A.T., Ratter, J.A., 2002. Vegetation physiognomies and woody flora of the Cerrado biome, in: Oliveira, P.S., Marquis, R.J. (Eds.), The Cerrados of Brazil: Ecology and Natural History of a Neotropical Savanna. University Presses of California, New York, pp. 91–120. doi:10.1663/0013-0001(2003)057[0656:DFABRE]2.0.CO;2

Oliveira, S.L.J., Campagnolo, M.L., Price, O.F., Edwards, A.C., Russell-Smith, J., Pereira, J.M.C., 2015. Ecological implications of fine-scale fire patchiness and severity in tropical savannas of Northern Australia. Fire Ecol. 11, 10–31.

Oliveras, I., Meirelles, S.T., Hirakuri, V.L., Freitas, C.R., Miranda, H.S., Pivello, V.R., 2013. Effects of fire regimes on herbaceous biomass and nutrient dynamics in the Brazilian savanna. Int. J. Wildl. Fire 22, 368–380. doi:10.1071/WF10136

Parr, C., Brockett, B., 1999. Patch-mosaic burning: a new pardigm for savanna fire management in protected areas? Koedoe.

Parr, C.L., Andersen, A.N., 2006. Patch mosaic burning for biodiversity conservation: A critique of the pyrodiversity paradigm. Conserv. Biol. 20, 1610–1619. doi:10.1111/j.1523-1739.2006.00492.x

Pilliod, D.S., Bury, R.B., Hyde, E.J., Pearl, C.A., Corn, P.S., 2003. Fire and amphibians in North America. For. Ecol. Manage. 178, 163–181. doi:10.1016/S0378-1127(03)00060-4

Price, O.F., Russell-smith, J., Watt, F., 2012. The influence of prescribed fire on the extent of wildfire in savanna landscapes of western Arnhem Land , Australia 297–305.

Radford, I.J., 2012. Threatened mammals become more predatory after small-scale prescribed fires in a high-rainfall rocky savanna. Austral Ecol. 37, 926–935. doi:10.1111/j.1442-9993.2011.02352.x

Radford, I.J., Andersen, A.N., Graham, G., Trauernicht, C., 2013. The fire refuge value of patches of a fire-sensitive tree in fire-prone savannas: Callitris intratropica in northern Australia. Biotropica 45, 594–601. doi:10.1111/btp.12050

Ramos-Neto, M.B., Pivello, V.R., 2000. Lightning fires in a Brazilian Savanna National Park: Rethinking management strategies. Environ. Manage. 26, 675–684.

Redford, K.H., 1994. The edentates of the Cerrado. Edentata 1, 4–10.

Romero, R., Nakajima, J.N., 1999. Espécies endêmicas do Parque Nacional da Serra da Canastra, Minas Gerais. Rev. Bras. Botânica 22, 259–265. doi:10.1590/S0100-84041999000500006

Russell-Smith, J., 2016. Fire management business in Australia’s tropical savannas: Lighting the way for a new ecosystem services model for the north? Ecol. Manag. Restor. 17, 1–4.

Russell-Smith, J., Cook, G.D., Cooke, P.M., Edwards, A.C., Lendrum, M., Meyer, C.P., Whitehead, P.J., 2013. Managing fire regimes in north Australian savannas: Applying Aboriginal approaches to contemporary global problems. Front. Ecol. Environ. 11, 55–63.

Russell-Smith, J., Evans, J., Edwards, A., Simms, A., 2017. Assessing ecological performance thresholds in fire-prone Kakadu National Park, northern Australia. Ecosphere 8, 1–23.

Simon, M.F., Grether, R., de Queiroz, L.P., Skema, C., Pennington, R.T., Hughes, C.E., 2009. Recent assembly of the Cerrado, a neotropical plant diversity hotspot, by in situ evolution of adaptations to fire. Proc. Natl. Acad. Sci. U. S. A. 106, 20359–20364.

Taylor, P., Pires, A.S., Fernandez, F.A.S., Freitas, D. De, Feliciano, B.R., Pires, A.S., Fernandez, F.A.S., Freitas, D.D.E., 2005. Studies on Neotropical Fauna and Environment Influence of edge and fire-induced changes on spatial distribution of small mammals in Brazilian Atlantic forest fragments. Stud. Neotrop. Fauna Environ. 40, 7–14. doi:10.1080/01650520412331333747

Tesfaye, G., Teketay, D., Assefa, Y., Fetene, M., 2004. The impact of fire on the soil seed bank and regeneration of Harenna Forest , Southeastern Ethiopia. Mt. Res. Dev. 24, 354–361.

Van Wilgen, B.W., 2009. The evolution of fire management practices in savanna protected areas in South Africa. S. Afr. J. Sci. 105, 343–349.

Van Wilgen, B.W., Biggs, H.C., Potgieter, a. L.F., 1998. Fire management and research in the Kruger National Park, with suggestions on the detection of thresholds of potential concern. Koedoe. doi:10.4102/koedoe.v41i1.248

Van Wilgen, B.W., Govender, N., Smit, I.P.J., MacFadyen, S., 2014. The ongoing development of a pragmatic and adaptive fire management policy in a large African savanna protected area. J. Environ. Manage. 132, 358–368.

Vieira, E.M., Marinho-Filho, J., 1998. Pre- and Post-fire habitat utilization by rodents of Cerrado from Central Brazil. Biotropica 30, 491–496. doi:10.1111/j.1744-7429.1998.tb00086.x

Vogl J, R., 1979. Some Basic Principles of Grassland Fire Management. Environ. Manage. 3, 51–57.

Whelan, J.. R., 1995. The Ecology of Fire. Cambridge University Press, Cambridge.