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Research article

An evaluation of contemporary savanna fire regimes in the Canastra National Park, Brazil: Outcomes of fire suppression policies

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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 characterized by many small-scale, starting at the beginning of the dry season (EDS). 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.

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

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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, 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. 1a). 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. 1a). Because of ongoing land use conflicts in the region, and to allow the extraction of mineral resources, 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.

The climate 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 16° to 20 °C in the Canastra region, and from 18° to 23 °C in the Babilonia region. According to the Köppen classification, the climate in the Canastra National Park is Cwa, with dry winters and hot summers (Alvares et al., 2013).

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. The Babilonia region is heavily dissected by small rural, poorly maintained roads connecting farms.

CNP is located in the Cerrado phytogeographical domain, characterized as an area of grassy-woody savanna (IBGE, 2012, 2004; Veloso et al., 1991). 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.

2.2. Land use and vegetation map

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 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.

For this study, we divided the whole area of CNP into three fire management zones (Fig. 1b): (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

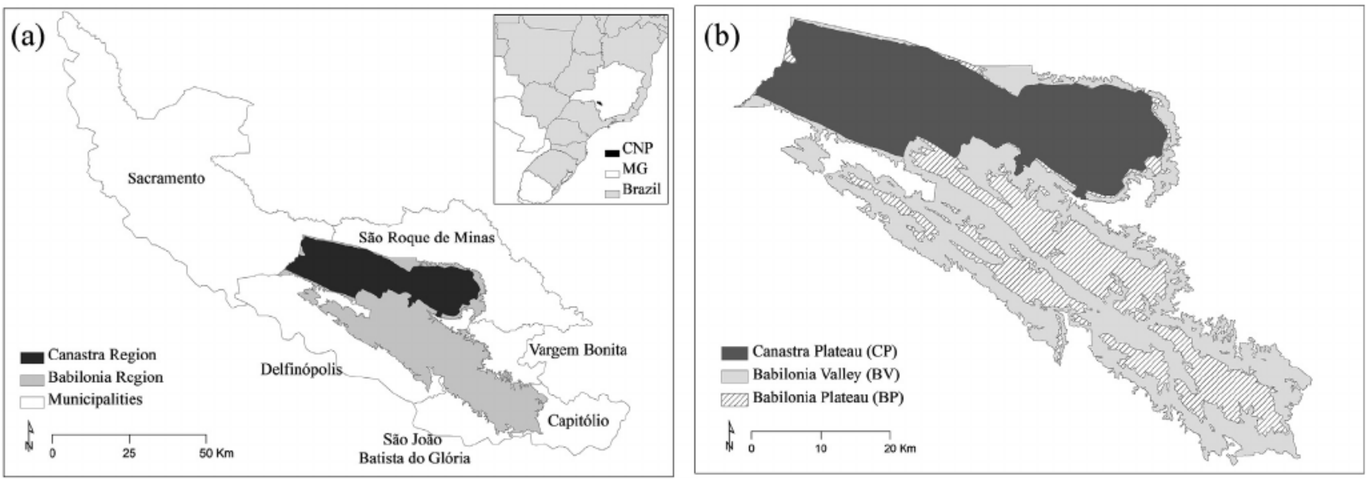

Fig. 1. (a) Location of the study area (CNP) within Minas Gerais State and municipalities. The protected area is divided into two regions: Canastra and Babilonia; (b) Fire management zones.

(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.

2.3. Seasonality

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 (outputs > inputs) occurred in both meteorological stations (April to October). On this basis, the early dry sub-season (EDS) was defined as the period from April to July, and the late dry sub-season (LDS) as the period from August to October. Natural burnings usually occur in October, a month that combines dry vegetation with the beginning of the rainy season and higher incidence of lightning.

2.4. 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) sensor, available from the Wildfires Database of the National Institute for Space Research (INPE) (https://prodwww-queimadas.dgi.inpe.br/bdqueimadas/). 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 ha; 10–100 ha; 100–1000 ha; 1000–10 000 ha; 10 000–100 000 ha.

2.5. 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.

2.6. 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 30 m 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 0.5×0.5 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).

2.7. 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 (Mistry, 1998). 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 5 m-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 (2000–2015), 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) (Table 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. 2). 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.

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 (Table 1).

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. 3), 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).

Fig. 4 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. 5).

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 ha in BP and BV, whereas big fires, > 1000 ha, occurred in CP at relatively high frequency (17%). Over the entire period analyzed, > 80% of fire events were ≤100 ha 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 ha) increased, especially in CP where fires > 1000 ha comprised 25% of total fire events.

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

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

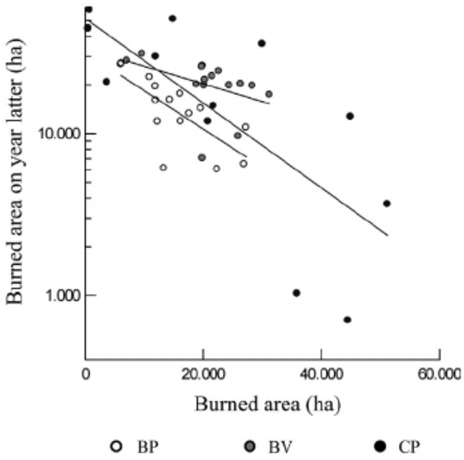

Fig. 2. Total burned area as a function of burned area in preceding year for each fire management zone.

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. S1).

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. 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. 6a and b). 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

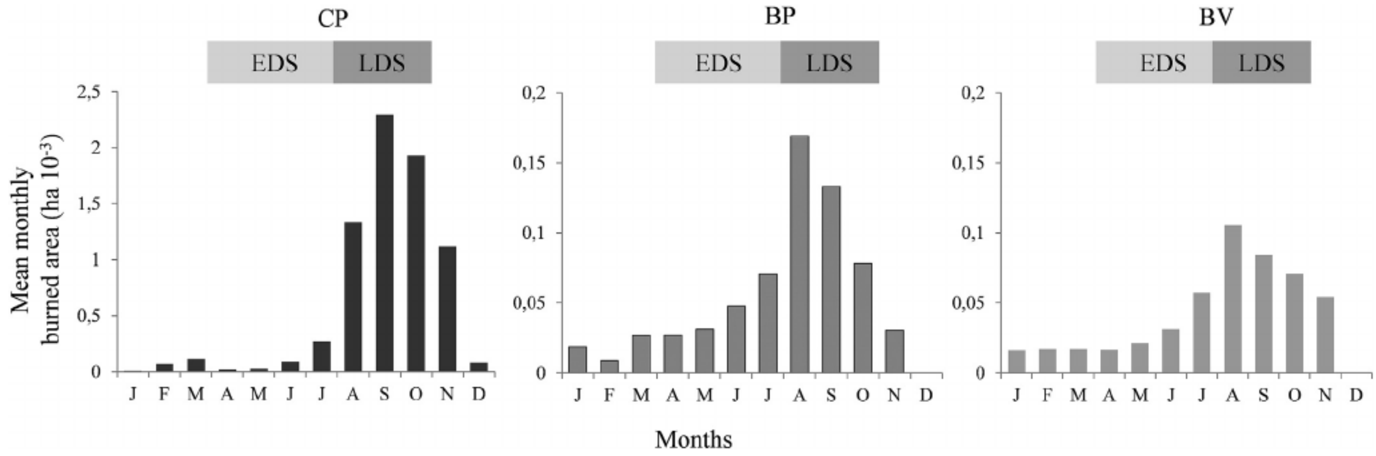

Fig. 3. Mean monthly burned area in each fire management zone, 2000 to 2015, where CP = Canastra plateau, BP = Babilonia plateau, BV = Babilonia valley.

exceeded 60% cover, allowing easy percolation of fire across the landscape (Fig. 6c).

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. 3), 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 characterized by large, extensive and severe late dry season fires are likely to make major adverse effects on biodiversity values (Table S1).

4.1. Implications for management

From the official perspective of Brazilian policy-makers and most conservation reserves, fire is a destructive agent that

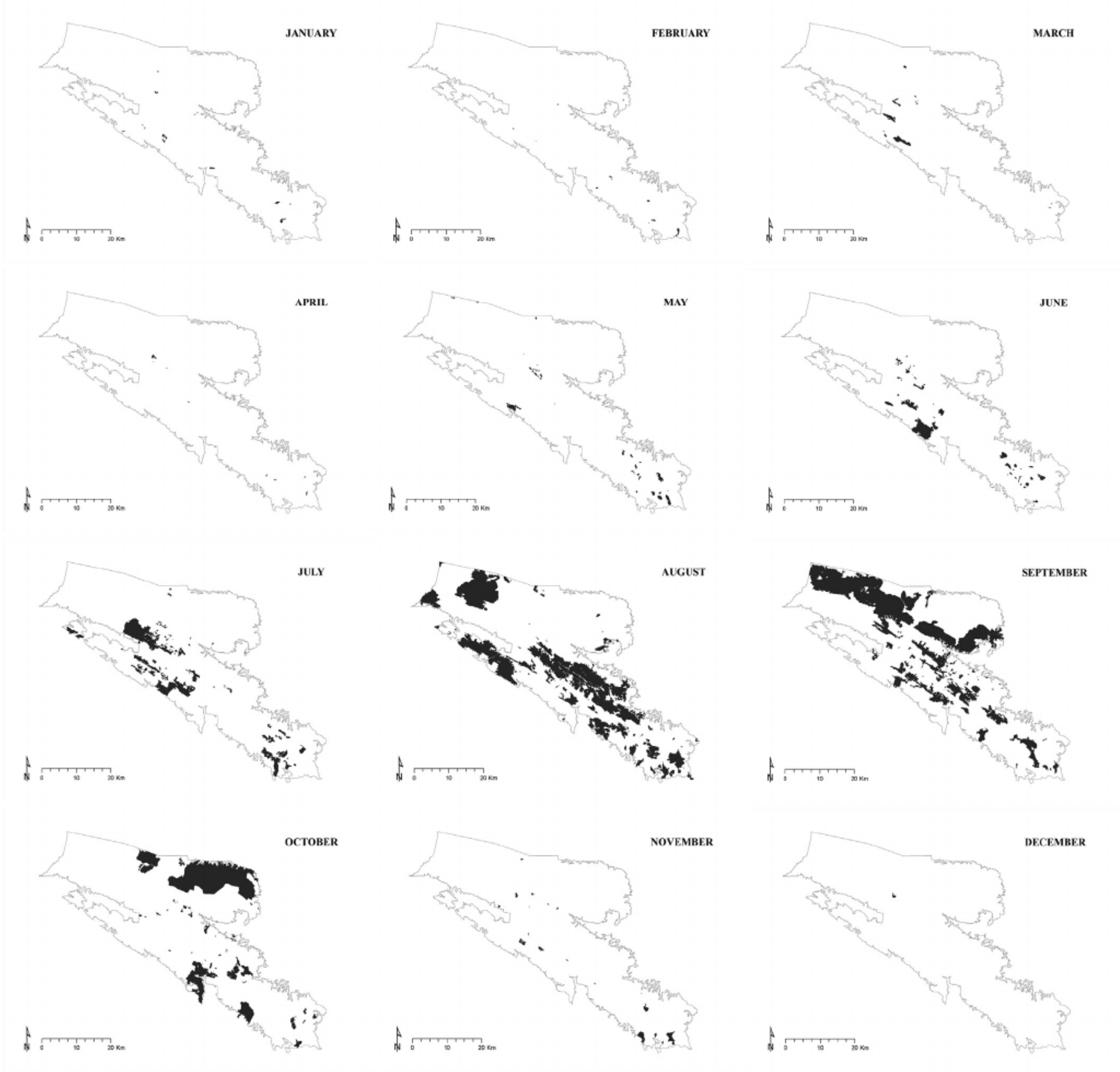

Fig. 4. Monthly distribution of fires in CNP from 2013 to 2015.

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

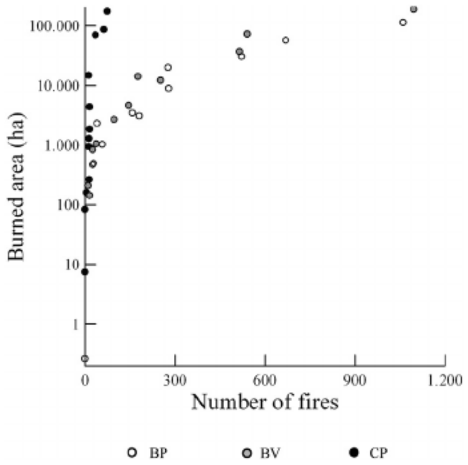

Fig. 5. Annual burned area as a function of number of fires in each fire management zone, 2000–2015, where CP = Canastra plateau, BP = Babilonia plateau, BV = Babilonia valley. Note the highest number of fires in BP and BV in relation to the CP.

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 a 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.

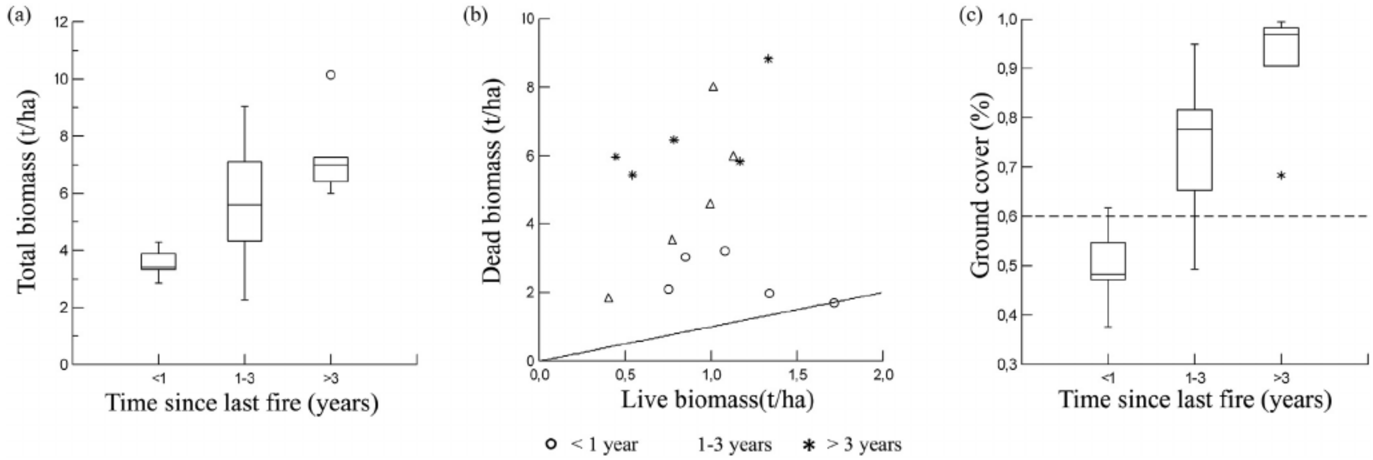

Fig. 6. (a) Grass fuel accumulation rates with time since last fire. (b) Dead and live biomass accumulation, where the line represents the point at which the dead biomass is equal to living biomass. (c) Connectivity of grasses represented by ground cover percentage as a function of time since last fire. The horizontal line represents the theoretical percolation threshold of fire spread (60% of the ground covered by fuel material).

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jenvman.2017.09.053.

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