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HUMAN-CLIMATE INTERACTIONS SHAPE FIRE REGIMES IN THE CERRADO OF SÃO PAULO STATE, BRAZIL

Dhemerson E. CONCIANI[1*]; Lucas SANTOS[2]; Thiago Sanna Freire SILVA[3]; Giselda DURIGAN[4]; Swanni T. ALVARADO[5 6]

[1] Universidade Estadual Paulista (UNESP), Instituto de Biociências, Departamento de Ecologia, Avenida 24-A 1515, 13506-900, Rio Claro, Brazil. * dhemerson.conciani@unesp.br

[2] Centro Universitário Faculdades Integradas de Ourinhos, Departamento de Agronomia, Ourinhos, São Paulo, Brazil.

[3] Biological and Environmental Sciences, Faculty of Natural Sciences, Stirling University, Stirling, FK9 4LA, UK.

[4] Instituto Florestal, Floresta Estadual de Assis, Cx Postal 104, CEP 19807-300, Assis, São Paulo, Brazil.

[5] Universidade Estadual do Maranhão (UEMA), Programa de Pós-graduação em Agricultura e Ambiente, Balsas (Maranhão), Brasil.

[6] Universidade Estadual do Maranhão (UEMA), Programa de Pós-graduação em Geografia, Natureza e Dinâmica do Espaço, São Luís, Maranhão, Brasil.

ABSTRACT

The Cerrado is the most diverse tropical savanna in the world. As a fire-prone ecosystem, natural fire in the Cerrado shapes plant communities and drives evolutionary processes. Human activities and landscape management can alter natural fire regimes and reshape Cerrado dynamics, making biodiversity conservation a challenge, particularly in densely populated areas. We reconstructed the historical fire regime of three protected areas (PA) and their buffer zones in São Paulo state to understand how current fire exclusion policies are affecting fire regimes and to measure how human-climate-fire relationships can change in areas under different land management. We used Landsat satellite imagery, from 1984 to 2017, with 30 meters of spatial resolution and 16 days of temporal resolution. In total, we mapped 49,471 hectares of burned area, and we detected variations in fire frequency and fire size among sites. PA dominated by open savanna in Itirapina concentrated 93% of all observed fires, while PA dominated by forest-like formations in Assis represented only 2% of the fires. Annual rainfall showed a very weak relationship (R2 = 0.04) with annual total burned area, while the rainfall split between dry and wet seasons showed a tendency to have a fuel moisture effect which determined the vegetation available to burn in the dry season ($R^2 = 0.09$). Fire regimes in PA were similar to those observed in buffer zones suggesting that fire-exclusion policies do not effectively prevent fires in PA that are surrounded by an anthropic matrix where fire is often used. When we included human factors in addition to rainfall, our models explained 44% of variation of burned areas. We conclude that fire regimes in São Paulo Cerrado have been modified by humans and that fire exclusion is not a suitable policy for protected areas in this fire-prone ecosystem.

Key- words: savannas**;** protected areas; anthropic landscapes; forestry; Landsat; fire management

Introduction

Prior to the rapid land cover conversion from natural areas to agriculture lands observed in Brazil during the last century, the Cerrado biome covered an area of ca. 2 million km², about 25% of the Brazilian territory (Durigan & Ratter, 2016). Globally classified as a humid savanna, the Cerrado comprises a complex mosaic of vegetation types, ranging from open grasslands to forest-like formations (Coutinho, 1990; Eiten, 2001; Ribeiro & Walter, 2017). Currently, The Cerrado biome contains 13 127 known plant species (Overbeck et al., 2015), of which ca. 4 400 are endemic (Klink & Moreira, 2002), and it has been considered the most diverse savanna in the world in terms of plant, avian, mammal and amphibian species (Murphy et al., 2016).

These characteristics have ranked the Cerrado among the global hotspots for biodiversity conservation (Myers et al., 2000). Increasing anthropogenic activities such as forestry, agriculture and livestock farming have had severe impacts on this ecosystem, representing the main threat to Cerrado conservation (Alencar et al., 2020). Still, only 7% of its total area is under legal protection in Brazil (Soares-Filho et al., 2014). In the state of São Paulo, Cerrado vegetation originally covered 14% (35 000 km²) of state's total area (Brito et al., 1997), but the currently remaining 8 353 fragments represent only 5.7% (2 000 km²) of this original cover, with 99.5% of these composed of small patches (< 4 km²) of Cerrado vegetation within a matrix of cropland, pasture and dense urban areas (Kronka et al., 2005). Of these remnants, only 12% (~250 km² - 0.7% of original cover) are currently under protection (Fiori & Fioravanti, 2001).

As with other savannas, fire is one of the most important drivers of vegetation dynamics, shaping the composition, structure and function of plant communities (Coutinho, 1990; Miranda et al., 2009). This dynamic emerged around 4 millions of years ago with the expansion of C_4 grasses and global savannas (Keeley &Rundel, 2005; Pagani et al.,1999). A natural fire regime is determined mainly by climate, where the interaction between accumulated rainfall and seasonality determine fuel build-up during the wet season and fuel moisture during the dry season (Alvarado et al., 2020; Bradstock, 2010; van der Werf et al., 2008). Before the arrival of humans, fires occurred late in the dry season or in the beginning of the rainy season, ignited mainly by lightning (Dias, 2006). However, when humans began to use fire for domestic activities about ~100 000 years ago (Goldammer, 1993; Vale, 2002), they modified fire regimes, reshaping the climate-fire relationship (Bird et al., 2012). Human activities can cause landscape fragmentation, increasing fire ignitions or suppressing fire occurrence, which can also alter fire seasonality and modify the fire dynamic (Archibald, 2016; Bowman et al., 2009).

Tracking fire regime changes is therefore essential to the effective conservation of this threatened and fragmented vegetation. Remote sensing tools allow us to monitor more than three decades of these changes and reconstruct contemporary ecological dynamics (Lentile et al., 2006). Particularly useful are the Landsat series image products, including the Landsat 5 TM sensor, launched in 1984, and the Landsat 7 ETM+ and Landsat 8 OLI sensors, which are still in operation. Landsat imagery, which offers a good compromise between spatial resolution (30 m) and moderate temporal resolution (16 days), can be used to reconstruct fire occurrence at local (Alvarado et al., 2018; Smith et al., 2007) and regional scales (INPE, 2018) and to identify burn scars over a period of more than 30 years.

In the present study, we reconstructed 33 years of fire occurrence within and around three Cerrado protected areas in São Paulo state, Brazil. These areas share similar environmental conditions in terms of climate and vegetation type and management policies, and they are all surrounded by a complex landscape characterized by severe land cover changes. Moreover, the same institution has managed these areas and applied a fire exclusion policy since the 1980s. Given that, we hypothesized that because this fire exclusion policy was equally applied to all areas, no differences in fire regime should be observed between them. We thus aimed to answer the following questions: i) Are there differences in fire regime among protected areas under the same environmental and management conditions? ii) Are these differences, if existent, driven primarily by climate conditions or by human activities?

Methods

Study sites

This study covered three protected areas in São Paulo state, Brazil, comprising natural ecosystems and silvicultural stands enclosed within areas of public land under varying protection status, which are surrounded by private rural and/or urban lands. To quantify the influence of the surrounding anthropogenic activities on fire regimes, we delineated a buffer of 7 km around each study site, covering in total 14 624 ha of protected areas and 78 153 ha of buffer zones (Table 1). Buffer zones comprised land uses such as sugarcane plantations, food crops (soybean, corn, rice), pasture, forestry, industrial areas, and occasionally unprotected remnants of native vegetation (cerradão and ecotonal zones with seasonal tropical forest) (Figure 1). In São Paulo state, the use of fire as a management tool was permitted until the year 2000 and was widely used to renew pasture lands and facilitate sugarcane harvesting. Since then, legal constraints on burning have been imposed (state law 11 241/ 2002), which has gradually reduced the use of fire.

Table 1. Description of study sites; NA means information is not available.

Site	Protected area type	Conservation status	Area (ha)	% of open savanna
Assis	Ecological Station – AES	Full preservation since 1984	1 755	1.2
	State Forest– ASF	Sustainable use since 1959	2 672	0
	Buffer	Buffer zone	15 420	*NA*
Santa Bárbara	Ecological Station – SBES	Full preservation since 1984	2 796	49.3
	State Forest – SBSF	Sustainable use since 1964	1 593	0
	Buffer	Buffer zone	20 394	*NA*
Itirapina	Ecological Station – IES	Full preservation since 1984	2677	91.6
	Experimental Station– IExS	Sustainable use since 1957	3 131	0
	Buffer	Buffer zone	42 339	*NA*

The native vegetation remaining in the study sites represents a gradient of different Cerrado vegetation types, from open grasslands to forest-like vegetation (“cerradão”). The Assis Ecological Station (AES) is dominated by "cerradão", as it is located at a transitional area to seasonal tropical forest (E. D. S. Pinheiro & Durigan, 2009); Santa Bárbara Ecological Station (SBES) comprises open savannas and "cerradão" (Abreu et al., 2017) and Itirapina Ecological Station (IES) is mainly covered by open savannas (Zanchetta et al., 2006). Sustainable use areas are represented by the Assis State Forest (ASF), Santa Bárbara State Forest (SBSF) and Itirapina Experimental Station (IExS), all characterized by plantations of exotic species of *Pinus* and *Eucalyptus* interspersed with small patches of "cerradão" (Gurgel-Garrido et al., 1997). Management of these areas is under the purview of the São Paulo Forestry Institute (Instituto Florestal).

Species composition of cerrado vegetation is very similar among the three study sites. The dominant woody species in Assis are *Copaifera langsdorffii* (Desf.) Kuntze, *Protium heptaphyllum* (Aubl.) Marchand, *Ocotea corymbosa* (Meisn.) Mez, *Vochysia tucanorum* Mart., and *Stryphnodendron rotundifolium* Mart. (Pinheiro & Durigan, 2012). In Itirapina, among the dominant species are *V. tucanorum*, *Myrcia guianensis* (Aubl.) DC., *Myrsine umbellata* Mart., *Qualea grandiflora* Mart. and *Ocotea pulchella* Mart. (Giannotti, 1988). In Santa Bárbara, the most abundant trees are *O. corymbosa*, *C. langsdorffii*, *Miconia ligustroides* (DC.) Naudin, *M. umbellata*, and *S. rotundifolium* (Neto, 1991). Although the proportions of the dominant species vary among sites, all species mentioned above occur in the three sites, possibly due to the similarities in climate and soil properties (deep, sandy, with low fertility and low soil water holding capacity) among them.

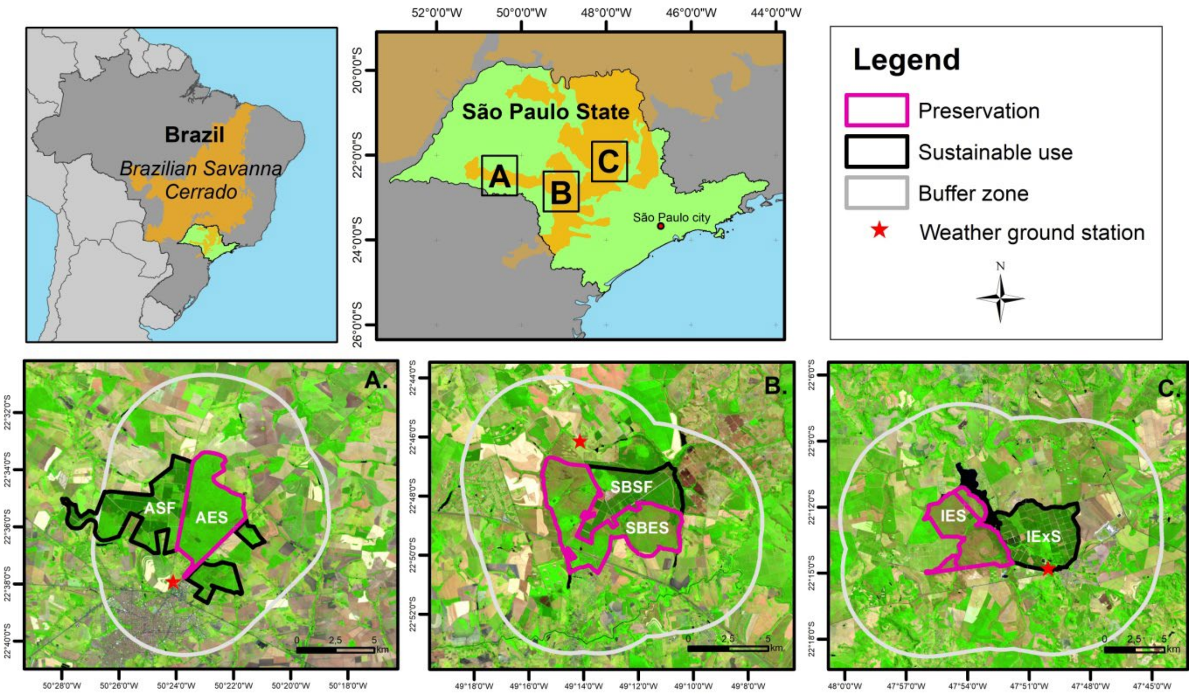

Figure 1. Location of the study areas: **A**) Assis Ecological Station (AES), Assis State Forest (ASF) and Assis buffer zone; **B**) Santa Bárbara Ecological Station (SBES), Santa Bárbara State Forest (SBSF) and Santa Bárbara buffer zone; **C**) Itirapina Ecological Station (IES), Itirapina Experimental Station (IExS) and Itirapina buffer zone. Background from Landsat 8 (OLI) satellite images, false color composition (from 2017-09; bands 6, 5, 4)

Annual rainfall at the studied sites ranged from 1 100 to 1 450 mm between 1985 and 2017 (Agência Nacional das Águas, ANA). Seasonality is defined by the two characteristic seasons of tropical savanna climates: a warm and rainy season (monthly mean temperature 22–23 °C; monthly rainfall > 100 mm) between October and March (~80% of annual rainfall); and a cold and dry season (monthly mean temperature 17–19 °C; monthly rainfall < 100 mm) between April and September (~20% of annual rainfall). Demographic density in the buffer zones is estimated at 206 inhabitants/km² in Assis,14 inhabitants/km² in Santa Bárbara and 27 inhabitants/km² in Itirapina (IBGE, 2014).

Management history

The State Forests—Assis (ASF) and Santa Bárbara (SBSF)—are protected under both the Brazilian National Conservation System (Sistema Nacional de Unidades de Conservação, SNUC, National Law 9985/ 2000) and environmental state laws. These areas, as well as the Experimental Station of Itirapina (IExS), were created in the mid-20th century to develop scientific research focused on sustainable use of exotic woody species of *Pinus* and *Eucalyptus*. A fire suppression policy was adopted then to prevent economic losses caused by accidental and arson fires.

Conversely, Ecological Stations— Assis (AES), Santa Bárbara (SBES) and Itirapina (IES)—comprise one of the most restrictive categories of the SNUC, having a strict full protection status. These areas were created during the 1980s, when the natural cover remaining within pre-existing sustainable use areas were re-categorized to improve nature conservation. None of these areas are open to public visitation and only allow scientific research and guided environmental education. After these areas were established as protected areas, the same fire suppression policy already adopted for sustainable use areas was maintained. Since then, woody encroachment due to fire suppression has been observed in all these areas, as reported by Pinheiro & Durigan (2009) and Abreu et al. (2017) for Assis and Santa Bárbara, respectively.

Remote sensing data: mapping burn scars

We defined the fire season for the study sites using the Active Fire product (MOD14) from the *Moderate Resolution Imaging Spectroradiometer* (MODIS) sensor onboard the TERRA satellite, covering the period 2004–2014 for São Paulo state. To delimit burn scars, we then acquired 805 multispectral surface reflectance images, from Landsat 5, 7 and 8, for the period 1985–2017 (Supplementary material S1), for path and row (WRS-2) 220/75 (Itirapina), 221/76 (Santa Bárbara) and 222/76 (Assis), from *Earth Resources Observation and Science* (EROS) *Center Science Processing Architecture* (ESPA; https://espa.cr.usgs.gov/). All additional spatial data were reprojected to Universal Transverse Mercator (zones 22S or 23S) using the WGS-84 reference datum to match the acquired imagery. Due to the location and size of the studied areas, burn scars mapping was not affected by the known scan line corrector mirror (SLC-off) failure of ETM+.

Rainfall data

For each study site, we obtained data on monthly rainfall and observed days of rain, for the period 1984–2017, from the closer weather ground stations

(Figure 1) operated by the Brazilian National Water Agency (*Agência Nacional de Águas*, ANA, http://www.snirh.gov.br/hidroweb).

Tree cover and land use data

Land use and cover was obtained from Projeto MapBiomas - Collection 5 (https://mapbiomas.org/). We extracted these data for the years 1985, 1995, 2005 and 2019 to produce annual land cover maps (Supplementary material Fig. S2), to compare land cover changes over time for the three sites, and to show all the differences and similarities between sites. We obtained tree cover estimation for all study sites, using the Global Forest Cover Change Tree Cover Multi-Year Global 30m V003 product (GFCC30TC V003), for the years 2000, 2005, 2010 and 2015 to evaluate changes in tree cover between sites and between areas with different conservation statuses over time. We characterized these differences using boxplots (Supplementary material Fig. S3). These data were provided by NASA Earth Data repository (https://search.earthdata.nasa.gov/).

Data processing and analysis

Fire season

We determined the fire season by combining monthly counts of active fires for the period 2004–2014 (MOD14 - Terra) and mean monthly rainfall (from weather ground stations) for the same period. We applied a threshold of >= 70 mm to the monthly rainfall to determine the rainy (October–March) and dry seasons (April–September), and as expected, most active fires occurred during the dry season.

Fire regime reconstruction

We detected and delineated every observable burn scar in all acquired Landsat images by combining the short-wave and near infrared bands, following the method proposed by Alvarado et al. (2017). For TM and ETM+, we created false color compositions using bands 5 (short-wave infrared, 0,55 - 1,75 μm), 4 (near infrared, 0,76 - 0,90 μm) and 3 (visible red, 0,63 - 0,69 μm). For the OLI sensor, we created false color compositions using bands 6 (short-wave infrared, 1,57 - 1,65 μm), 5 (near infrared, 0,85 - 0,88 *μ*m) and 4 (visible red, 0,64 - 0,67 *μ*m). We standardized brightness and contrast for all scenes of the same sensor to minimize reflectance variation and reduce observation error.

We then performed visual detection and manual delineation of burn scars in every Landsat scene across each year using a standardized mapping scale of 1:10000. The resulting dataset contained information, in vector format, about the location, area and date of observation for every burn scar. Validation was performed by comparing our mapping results to the official fire records for each reserve. We then converted vector data into time series of annual burned (pixel value = 1) and non-burned (pixel value = 0) binary raster maps with 30-m pixel size, from which we calculated the following metrics (Table 2):

Table 2. Remote sensing derived metrics used to assess fire regimes.

Metric	Unit	Description
Fire count	*freq.*	A map of the total number of fire events observed over the 32 years studied, obtained from the per-pixel sum of all overlapped annual burn maps.
Density of ignitions	*Ignitions/ha*	Number of burn scars observed per year divided by the total area of each studied site.
Burned area	*ha*	Sum of individual vector burn-scar sizes (in hectares) per year or per month.
Accumulated burned area	*ha*	Sum of all annual burned area maps.
Relative burned area	*%*	Proportion between total burned area and total area of each site (in %).
Relative accumulated burned area	*%*	Sum of relative burned areas over time
Latest fire	*year*	Map for the year when the latest fire was observed, produced by reclassifying burned pixel values into year of observation, and then calculating the maximum per-pixel value among all overlapped annual raster maps.
Estimated time to burn the entire area	*years*	Time required to burn 100% of the area, calculated by a linear regression of the average native vegetation burned per year (%)

Burned area and human-climate relationships

Previous research has suggested that annual burned area responds significantly to 1.5 to 2 years of accumulated rainfall before the end of the dry season in savannas ecosystems (Archibald et al., 2010; Balfour & Howison, 2002). Similarly, burned area in South American savannas is mainly determined by the cumulative amount of rain during the last 6 months of the dry season (Alvarado et al., 2020). We therefore used Generalized Linear Models (GLM), assuming a binomial distribution error with the logit link function (Table 3), to test how one and two years of rainfall accumulation (including one and two wet seasons, respectively) prior to the end of the dry season would affect total burned area. We also tested the effects of rainfall accumulation during the dry season (6 months prior to the end of the fire season). We considered each fire year as starting at the beginning of the wet season (October) and ending at the end of the following dry season (September of next year). We expected to find positive correlations between wet season rainfall and burned area (more rain increases fuel build-up during the rainy season and, consequently, increases burned area) and negative correlations between dry season rainfall and burned area (more rain during the dry season increases fuel moisture retention, reducing burned area).

We included the human component in our models as the interaction between two factors. Firstly, management practices were included as "conservation status", with three protection levels: “preservation”, “sustainable use” and “buffer zone”. Secondly, we considered the differences caused by human activities, such as land use management and agropastoral practices between locals, by explicitly including a variable named "site", represented by the three municipalities: Assis, Santa Bárbara, and Itirapina. We found that land use was similar among the study sites, with a few differences over time (Supplementary material Fig S2), as were tree cover patterns (Supplementary material Fig. S3).

We then interpreted the Akaike Information Criterion (AIC), the R^2 and the ecological meaning of model coefficients to select the best-fitting model. We calculated R^2 using the “MuMIn” package of the R package (R Core Team, 2020), which has a specific function to calculate R^2 for generalized linear and mixed models (Nakagawa & Schielzeth, 2013). We applied the function “r.squaredGLMM()” to each resulting glm object, which outputs marginal (fixed effects alone) and conditional (mixed effects) R^2 values. However, since all variables were included in the model as fixed variables, there was no difference between marginal and conditional values, and we only retained the marginal R^2. All spatial analyses were performed using the packages “raster” (Hijmans & van Etten, 2012) and “rgdal” (Bivand et al., 2015). Descriptive statistics and multiple linear regressions were calculated in R 3.5.1 (R Core Team, 2018).

Table 3. Generalized linear model results. Relative burned area (BA, %) is summarized by fire year (FY: October–September). Accumulated fire-year rainfall is the sum of the wet season rainfall (October–March) and the dry season rainfall (April–September). Model in bold was considered the best candidate model according to the selection criteria. The symbol “:” represent the interaction between 2 factors.

Regression model	R²	AIC
BA = 1 fire year rainfall	0.04	9.75
BA = 24 months before the end of dry season rainfall + 1 dry season rainfall	0.07	11.75
BA = 1 wet season rainfall + 1 dry season rainfall	0.09	11.75
BA = Site	0.31	11.77
BA = Conservation status	0.18	11.78
BA = 1 wet season rainfall:site + 1 dry season rainfall:site	**0.44**	**19.77**
BA = 1 wet season rainfall:conservation status+ 1 dry season rainfall:conservation status	0.29	19.77
BA = 1 wet season rainfall:site+ 1 dry season rainfall:site+ conservation status	0.51	23.79
BA = 1 wet season rainfall + 1 dry season rainfall + site:conservation status	0.46	27.79
BA = 1 wet season rainfall:site + 1 dry season rainfall:site + site:conservation status	0.53	31.80

Table 4. Parameter estimates for the best candidate model (BA =1 wet season rainfall:site + 1 dry season rainfall:site)

Parameter	Coefficient
Intercept	-5
Wet season rainfall:Assis	0.00016
Wet season rainfall:Itirapina	0.00086
Wet season rainfall:Santa Bárbara	0.00086
Dry season rainfall:Assis	- 0.0069
Dry season rainfall:Itirapina	0.00079
Dry season rainfall:Santa Bárbara	- 0.0018

Results

Relationships between rainfall, site, conservation status and burned area

Contrary to our expectations, we did not find a strong correlation between rainfall and burned area. All models including rainfall variables only weakly explained annual burned area (coefficient of determination= 0.04, 0.09 and 0.07 respectively, Table 3). However, as expected, we did observe positive coefficients for wet season rainfall, and negative coefficients for dry season rainfall, when predicting burned area (Table 4). When wet and dry season rainfalls were included as separate variables, they explained only 9% of variation in total burned areas. When human influence expressed by the factor “site” was added to rainfall in both seasons, these variables together explained 44% of variation in the total burned area. This interaction was stronger than that obtained with the human component defined by “conservation status”, which explained only 29% of burned area with same AIC value of 19.77. The inclusion of interactions between the two human components (“site” x “conservation status”) or between both components and wet and dry season rainfall resulted in higher AIC values and only marginal gains of explanatory power in relation to our best model.

Examining the coefficients of the selected best model, relative burned areas are positively affected by wet season rainfall in all sites. Coefficients were similar for Itirapina and Santa Bárbara; in Assis, however, the value was ca. 1/5 of those obtained for the other sites. Conversely, dry season rainfall negatively affected burned area, with a coefficient value for Assis four times higher than that for Santa Bárbara. Interestingly, a slightly positive coefficient was observed for Itirapina, indicating that dry season rainfall is either neutral or slightly increases burned area in this site.

Spatial and temporal dynamics of fire

We detected 49 471 hectares of burned areas (BA), distributed among 1 607 ignitions between 1985 and 2017 for all three study sites. From this total, 93% of the ignitions and 91% of the total burned area were observed for Itirapina, followed by Santa Bárbara (6% of ignitions and 8% of total burned area) and Assis (2% of ignitions and 1% of total burned area). Small burn scars (less than 0.5% of relative area) were the most frequent burn events in all sites, while large burned areas were rarely observed (Figure 2A).

The Assis protected area (AES) was burned only once by a small burn (< 0.5% of total area), remaining otherwise without fire over the entire period analyzed (Figure 2B). We observed the same pattern of peaks in burned area for Santa Barbara (SBES) and Itirapina (IES) protected areas, characterized by 7- to 9-year periods without fire followed by a peak in burned area (Figure 2B). Considering the total extent of fire scars in the protected areas over 33 years, the amount of native vegetation burned per year is only 0.02% in Assis, 2.03% in Santa Bárbara and 3.37% in Itirapina. If we disregard overlapping fire scars, the situation is even worse (Table 5), with most areas in the three study sites not being burned over the last 33 years (99.3% in Assis, 57.2% in Santa Bárbara and 44.7% in Itirapina).

Table 5. Historical distribution of native vegetation areas burned at least once in 33 years (1985–2017) within the studied protected areas. *Time estimated to burn the entire preservation area was estimated using the historic burn rate (% native burned per year).

	Assis	Santa Bárbara	Itirapina
Native vegetation (ha)	1755	2796	2677
Native vegetation burned at least once in 32 years (ha)	12	1195	1480
Native vegetation never burned	1743	1601	1197
% mean native burned per year	0.02	1.33	1.72
% native vegetation never burned	99.3	57.2	44.7
***Time estimated to burn the entire reserve (years)**	5000	~ 75	~ 58

In general, the areas under sustainable use in all sites (ASF, SBSF and IExS) had a low increase in accumulated burned area over time, punctuated by abrupt increases when single wild fires burned large areas of pine plantations (Figure 2C).

We observed the lowest fire occurrence in Assis and Santa Bárbara buffer zones, where only a few areas burned during the study period (Figure 2D). In the Itirapina buffer zone, however, we observed high and continuous fire usage (Figure 2D).

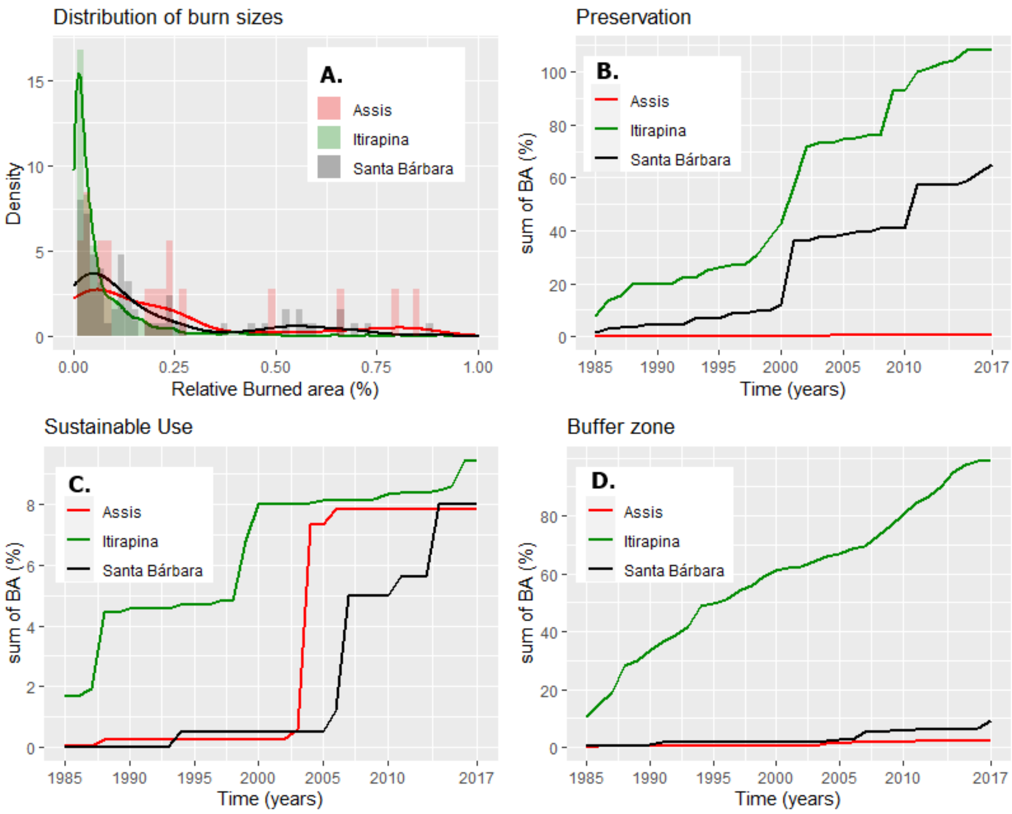

Figure 2.A. Burn-scar size distribution (density) per site, considering relative burned area of each burn (%). Relative accumulated burned area (%) between 1985 and 2017 for: **B.** protected areas; **C.** sustainable use areas; **D.** buffer zones.

At the three study sites, burn scars had predominantly regular geometries (Figures 3A/B/C). The Assis preservation area had the lowest fire frequency, and the only observed burn scar was related to a shared burned scar within the contiguous sustainable use area and buffer zone (Figure 3A). Itirapina and particularly Santa Barbara protected areas had a higher fire frequency within their limits when compared to their buffer zones (Figure 3B/C), but fire scars spanning both protected area and buffer zones were also detected. The buffer zone of Itirapina had the highest fire count among all studied areas (Figure 3C). Concerning time since last fire, we observed that sites with a higher fire frequency tended to be burned more recently. In Santa Bárbara, the majority of burned areas were recent, while Itirapina had a pattern of large areas burned only once, followed by no recurrence of fire during the last 20 years.

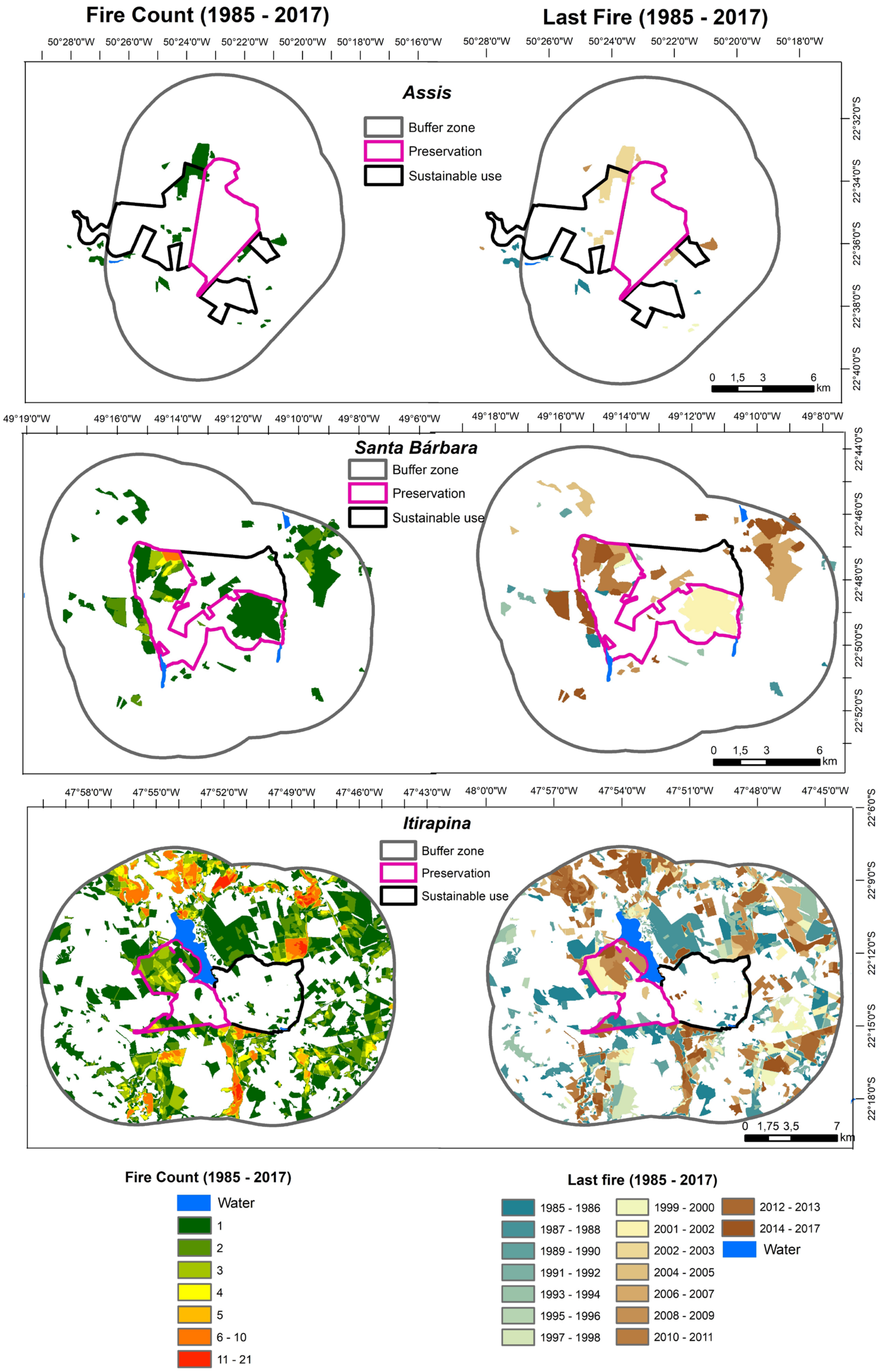

Figure 3. Fire count and year of the last fire from 1985 to 2017 for the regions of: **A.** Assis; **B.** Santa Bárbara; **C.** Itirapina.

Fire seasonality

In general, the peak of ignitions and burned area in all sites, independent of conservation status, was concentrated in mid and late dry season (July–September). A second, abnormal peak of ignitions and size of burned area was observed for the Itirapina protected area (Figure 4A/4B) and the Assis sustainable use area (Figure 4C/4D), both during the transition between the rainy and dry seasons in February–March. The Itirapina buffer zone had a higher fire frequency than the other buffer zones, burning in all months of the year, with increased fire activity during the dry season.

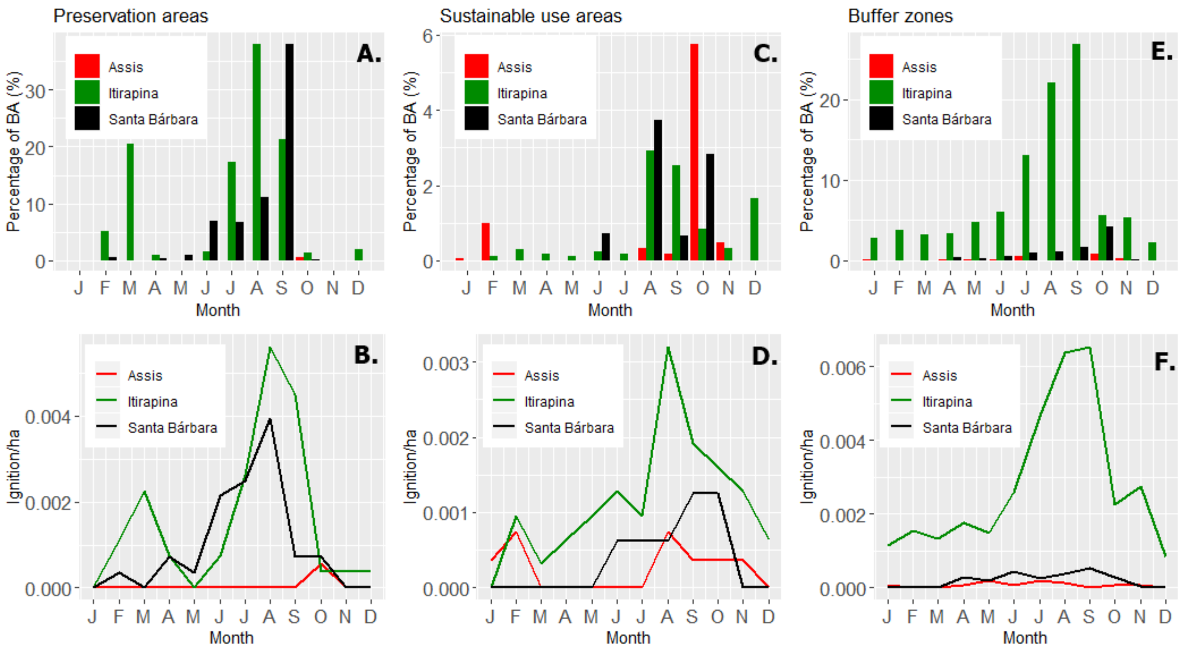

Figure 4. Monthly sum of relative burned area (%) from 1985 to 2017 and density of ignitions (ignition/ha) for: **A/B.** Preservation areas; **C/D.** Sustainable use areas; **E/F.** Buffer zones.

Discussion

Fire Seasonality

The peak of fire occurrence, in terms of number of ignitions and amount of burned areas, occurred by the end of the dry season and the beginning of the rainy season (October). This pattern has also been observed for other areas in the Cerrado, likely resulting from the balance between fuel load build-up during the rainy season and a decrease in fuel moisture during the dry season, which together determine the probability of ignition (Alvarado et al., 2017; Ramos-Neto & Pivello, 2000). The peak of ignitions and burned area during the late dry season (August–September) in human-impacted savannas is often related to the use of fire to support cattle grazing and cropping (Mataveli et al., 2018; Pivello, 2011). Within protected areas, the main cause of late dry season fires are lightning ignitions (when natural fire occurs) and accidental or arson fires (Batista et al., 2018). This pattern of human-driven fires in savannas during the late dry season has also been reported in several protected and non-protected areas in Brazil and Africa (Alvarado et al., 2017, 2018; Archibald, 2016; Fiedler et al., 2006; Medeiros &Fiedler, 2004; Pereira Júnior et al., 2014). Late fires are more intense than early fires and are often associated with a decrease in the density and diversity of the woody vegetation layer (Govender et al., 2006), potentially causing mid-term changes in plant communities.

Secondary ignition peaks were observed at the end of the rainy season (February–March). These lower peaks were described by Ramos-Neto & Pivello (2000) as natural fires caused by lightning. We observed fire scars at the end of the rainy season, characterized by regular geometries neighboring buffer zones and roads, indicating they were likely human set-fires and not caused by lightning.

Climate influences on fire regime

We found a positive effect of rainfall during the rainy season and a negative effect of rainfall during the dry season on annual burned area in most of our study sites except in Itirapina, where a slightly positive effect of dry-season rainfall on annual burned area was observed. In our study, the accumulated rainfall explained, in small part (9%), the extent of burned areas. This predictability is low compared to the 38% explanation obtained in Cerrado areas of Minas Gerais (Alvarado et al., 2017). We have not found a relevant relationship between accumulated rainfall and the amount of burned areas, as rainfall effects showed low R^2, suggesting that the human component can buffer the climate-fire relationship (Bird et al., 2012).

The amount of rainfall during the wet season determines productivity and fuel accumulation, while the amount of rainfall during the dry season determines the probability of ignition (Bradstock, 2010; Cochrane & Ryan, 2009). Moreover, among mesic savannas under similar climate conditions, differences in vegetation structure (e.g. percentage of grasses, shrubs and tree cover) alter fuel build-up, composition, moisture conditions and fuel availability in the dry season. These differences modify the fire-climate interaction and creating alternative stable states of vegetation maintained by fire-vegetation feedbacks (D'Onofrio, et al., 2018; Lasslop et al., 2018; Lehmann et al., 2014).

The positive effect of dry-season rainfall on burned areas observed in Itirapina contradicts what has been reported in the literature for wet savannas (Alvarado et al., 2020). A higher frequency of lightning with higher dry season rainfall could explain that result, but no empirical data about lightning fires was available. Given this almost neutral effect, we assume that the positive effect resulting from the model may be a spurious correlation. Also, human impact may be 'buffering' the climate-fire regime, as shown for Australian savannas (Bird et al., 2012).

Human influence

Burn scar geometries were regular in all sites, independent of conservation status, indicating a highly managed landscape (Cochrane & Ryan, 2009). São Paulo's state law (10 547/ 2001) requires the establishment and yearly maintenance of firebreaks of at least 3 meters width around infrastructure (e.g. roads, railways, power transmission lines) and 6 meters width around croplands, pastures and protected areas. These firebreak arrangements explain the appearance of regular burn scar geometries, as human activities can modify fire-climate relationships by increasing landscape fragmentation through roads, railways and infrastructure (Andela & van der Werf, 2014).

Since natural fires are rare in São Paulo state due to the massive presence of lightning rods, most observed fires were likely caused by humans. Roads can be an agent of such human-caused fires. While fragmenting the landscape, roads also enable people to travel through the savannas. Thus roads act both as firebreaks and potential ignition points at the same time. Accidental and arson fires ignited near preservation areas can spread into them and reshape fire regimes (Archibald, 2016; Daldegan et al., 2014). These modified fire regimes are considered among the threats to Cerrado conservation (Durigan et al., 2007).

Despite the study sites sharing similar management regimes, we found remarkable differences in fire regimes among sites, in terms of both the number of ignitions and annual burned area. In the buffer zones, fire regimes were highly human-managed, in terms of decreasing or increasing fire frequency. The differences between buffer zone fire regimes are related to differences in local agropastoral practices. Whereas fire is often used to renew large pastures in Itirapina and Santa Bárbara, the ranchers in Assis practice "pasture rotation". This practice involves dividing pastures into small plots and moving cattle among them over the year, thus preventing pasture exhaustion and stimulating regrowth without fire.

Moreover, the conversion of pastures into cropland, which automatically reduces fire frequency, has been observed in all study sites since 2005 (Supplementary material Fig S2). In Assis and Santa Bárbara, pastures transition to other cops (e.g. soybean, corn) or to sugarcane on flat terrain, where mechanized harvest and fire exclusion are practiced. In Itirapina, however, where pastures transition to sugarcane in high-slope areas, fire has been used until recently to facilitate sugarcane harvesting.

Concerning the effect of “conservation status” on fire regimes among our study sites, our results suggest that it corresponds, although in a discreet way, to the type of fire management that has been applied. While prescribed fire management has been implemented in agricultural areas in the buffer zones for decades, a successful fire exclusion policy has been simultaneously implemented inside the sustainable use forestry areas. This policy has not been as “successful” in the protected areas occupied by cerrado vegetation. When we considered “conservation status” effects alone in a model (coefficient of determination= 0.18), we observed a consistent increase of annual burned area in the cerrado preservation areas (B = 0.47 % yr^{-1}) and a decrease of annual burned area in sustainable use areas (B = -1.54 % yr^{-1}) when compared with the buffer zones.

Despite preservation and sustainable use areas being completely different in ecological terms, these areas are managed by a single institution and share similar fire exclusion policies and enforcement. We found a very low proportion of fire activity on native vegetation in the studied preservation areas, far below the 5-year fire return interval reported for Cerrado historical fire regime (Dias, 2006). The adopted fire-exclusion policies significantly reduced fire frequency and number of ignitions, increasing fire return intervals in all preservation areas studied. However, the reduced number of ignitions and increased fire return intervals may be associated to a high fuel load accumulation, leading to larger burned areas (Alvarado et al., 2018; Keeley et al., 1999; Miranda et al., 2009).

Large wildfires were observed in IES and SBES every 7–9 years of fire absence, highlighting that anthropic pressures can lock Cerrado into a fire paradox, where fire exclusion prevents fires in the short term but causes repeated wildfires in the midterm (Arévalo & Naranjo-Cigala, 2018). Fire occurrence was lower in AES, partly explained by lower fire pressures from the buffer zone. However, despite all buffer zones having similar land use patterns (Supplementary material Fig S2) and tree cover over time (Supplementary material Fig S3), as that of the sustainable use areas, we observed a large difference among the study sites in the dominant vegetation within the protected area. While AES was dominated by cerradão, a forest-like vegetation, IES and SBES were mostly occupied by grassland/savanna vegetation during the study period (Supplementary material Fig S2 and S3). The closed and evergreen canopy of the high tree biomass cerradão vegetation increases fuel moisture and decreases flammability (Barbosa & Fearnside, 2005; Miranda et al., 2009; Newberry et al., 2020). Thus, even if buffer zone fire pressures were stronger, native vegetation in AES would rarely burn as much as that in IES or SBES.

We found a similar fire regime in all sustainable use areas, which can be explained by the similarities in size, geometry and species composition (e. g. *Pinus* spp., *Eucalyptus* spp.) of the forestry stands. The tree composition and closed canopy of these forestry stands create low luminosity conditions and prevent the establishment of native Cerrado grasses and shrubs in the understory (Viani et al., 2010). However, the slow decomposition rates of pine needles and *Eucalyptus* leaves in the litter increases fuel load and thus the long-term flammability of the system, unlike the Cerrado's fast fuel build-up (Beutling et al., 2012; Ribeiro, 2013). Long-term fire absences and fuel load accumulation increases the probability of wildfires in arid pine ecosystems (Glitzenstein et al., 2003). In regions with humid climates, however, a trade-off between fuel loads and high moisture retention in the litter could limit ignitions and fire spread.

In terms of management, fire needs to be seen not as an enemy of Cerrado conservation, but as an ally. Once preservation areas become extremely fragmented and are under pressure from a complex anthropic matrix, prescribed fires, rather than fire exclusion, become crucial for the conservation of a fire-prone ecosystem like Cerrado, whose biodiversity is severely threatened by fire suppression (Abreu et al., 2017). In ecological terms, it is difficult to establish a suitable management without testing and assessing the effects of different fire regimes. Thus, management plans cannot be a “one-size-fits-all” recipe for prescribed fires—they need to include the prerogative of managers to test and apply management according to observed effects.

Conclusion

Fire regimes are highly variable in terms of frequency and extent of burned area among sites under similar climates, showing that other factors, largely human-related, have a stronger local effect than climate. The probabilities of accidental and arson fires are related to human-set fires in mid-late dry season. Our study detected a high proportion of the native vegetation not being subjected to any burning over the last 33 years, a direct result of the extensive fire exclusion policy adopted in Brazil 50 years ago. This has modified the fire regime of protected areas and decreased fire frequency in comparison to historical fire regimes of the Cerrado.

The exclusion of fire in protected areas of São Paulo state has severely affected their fire regimes by 1) precluding fires for much longer periods than historically observed for the biome, and 2) concentrating arson and accidental fires towards the end of the dry season, especially in sites where open savanna vegetation exists and fuel build-up is high. To mitigate the negative effects of these modified fire regimes on Cerrado ecosystems and their biodiversity, conservation policies need to change. Although fire prevention may be appropriate for sustainable use forestry areas, fire exclusion is not a suitable policy for protected areas of fire-prone ecosystems such as the Cerrado. New management strategies must therefore include legislation that supports protected area managers in planning and applying prescribed fires in Cerrado preservation areas.

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