Universidade de Brasília
Instituto de Ciências Biológicas
Programa de Pós-Graduação em Ecologia

Implicações e aprendizados do manejo integrado do fogo no Cerrado: estudo de caso no Parque Nacional da Chapada das Mesas (PNCM)

Lívia Carvalho Moura

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"Não é o mais forte que sobrevive,
nem o mais inteligente, mas o que
melhor se adapta às mudanças."
Charles Darwin

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Sumário

Resumo

A política de zero tolerância ao fogo no Cerrado contribuiu, dentre outros fatores, para o aumento da flamabilidade de ambientes pirofíticos e conflitos entre as autoridades governamentais e comunidades rurais por décadas e mesmo séculos. A partir de 2014 teve inicio o Programa Piloto de Manejo Integrado do Fogo (MIF), em que queimas controladas foram implementadas em algumas unidades de conservação (UCs) do Cerrado na tentativa de diminuir a extensão de áreas atingidas por incêndios, bem como proteger vegetações sensíveis ao fogo, como matas de galeria, de incêndios. Estas queimas controladas estão sendo conduzidas no início da estação seca, e em locais estratégicos sob condições meteorológicas amenas, geralmente no final da tarde quando a umidade relativa do ar é maior e a temperatura do ar se reduz gradualmente. Embora outras pesquisas tenham caracterizado o comportamento e efeito de diferentes frequências de fogo e épocas de queima do ano em comunidades arbóreas no Cerrado, faltam informações referentes a queimas realizadas no final de tarde e início da estação seca, as quais tendem a apresentar menores intensidades de fogo. Experimentos colaborativos e as estratégias utilizadas por outros países podem subsidiar ações de manejo. Esta tese foi desenvolvida em uma das três primeiras UCs do Cerrado a implementar o MIF, o Parque Nacional da Chapada das Mesas, e está dividida em três capítulos: (1) revisão bibliográfica sintetizando as políticas de manejo do fogo nas principais regiões savânicas do hemisfério sul; (2) caracterização do comportamento de 31 queimas experimentais no precoce (maio), ao meio-dia com umidade relativa >50% e no final da tarde com umidade relativa <50%, e queimas tardias (setembro) em áreas de cerrado sentido restrito com histórico de queima de dois e três anos; e (3) quantificação dos efeitos dessas queimas experimentais em comunidades lenhosas (~4.900 fustes de espécies arbórea). De maneira geral, intervalos de queima maiores (três anos) apresentaram intensidades de fogo maiores do que intervalos menores (dois anos). Queimas precoces no fim de tarde com umidade relativa >50% causam impactos similares na vegetação arbórea que queimas precoces ao meio-dia com umidade relativa <50%. Porém a intensidade de fogo, o consumo de combustível e calor liberado é menor quando a umidade relativa do ar é maior, a precipitação anual e a quantidade de combustível disponível para a queima é menor. Dessa forma, queimas precoces no fim de tarde podem ser úteis para atingir os objetivos do MIF em unidades de conservação no Cerrado a curto-prazo. Esta tese foi desenvolvida para aumentar as informações científicas disponíveis para pesquisadores, instituições ambientais e comunidades rurais que utilizam o fogo como instrumento para o manejo da terra.

Palavras-chave: época do fogo, incêndio, política do fogo, queima controlada, savana, vegetação arbórea.

Abstract

The zero tolerance fire policy in the Cerrado (Brazilian savanna) contributed, among other factors, to increase the flammability of fire-prone environments and conflicts between the authorities and rural communities for decades and even centuries. Since 2014, an Integrated Fire Management (IFM) Pilot Program started, where controled burns were implemented in some Cerrado protected areas (PAs) in the attempt to reduce the extension of areas burned by wildfires, as well as to protect fire sensitive vegetation, such as riparian forests, from wildfires. These controled burns are being conducted during early dry season, and in strategic places under mild weather conditions,generally in the evening when air relative humidity is higher and air temperature gradually reduces. Although other researches have characterized fire behaviour and the effects of different fire frequencies and seasonalities on Cerrado woody communities, there is a lack of information on the behavior and effects of evening burns in the beginning of the dry season, which tend to be of lower fire intensity. Collaborative experiments and the strategies used by other countries can subsidize management actions. This dissertation was developed in one of the first three Cerrado PA to implement IFM, the Chapada das Mesas National Park, and is divided in three chapters: (1) literature review synthesizing management policies in the largest savanna regions in the southern hemisphere; (2) characterization of fire behaviour in 31 experimental burns in early dry season, around mid-day with relative humidity <50% and evening with relative humidity >50%, and in the late dry season burns in open typical cerrado areas, with two and three years time since fire; and (3) quantification of the effects of these experimental burns in woody communities (~4,900 stems). Generally, longer fire intervals (three years) presented higher fire intensities than shorter intervals (two years). Early dry season burns in the evening with relative humidity >50% cause similar impacts to woody vegetation when compared to burns in the same season mid-day with relative humidity <50%. However, fire intensity, fuel consumption and heat released was lower when there was a combination of high relative humidity and low annual precipitation and fuel load. Thus, early dry season, evening burns can be useful to accomplish the IFM objectives in Cerrado PAs in short-term. This dissertation was developed to enhance the scientific information available for researchers, environmental institutions and rural communities who use fire as a land management tool.

Keywords: fire policy, fire season, prescribed burn, savanna, wildfire, woody vegetation.

Introdução geral

A presença do fogo na natureza é datada de aproximadamente 420 milhões de anos (Scott & Glasspool, 2006). O regime de fogo, assim como outros distúrbios ecológicos, atua juntamente com a disponibilidade de água e nutrientes no solo no processo de seleção e distribuição de espécies vegetais em diferentes ecossistemas (Bond & Keeley, 2005; Bowman et al., 2009). Regimes de fogo são especialmente caracterizados pela frequência e época das queimas. As características de cada queima são afetadas pela disponibilidade de combustível e de fatores abióticos, como condições meteorológicas e geomorfológicas (Whelan, 1995). Diferentes regimes de fogo ao longo de milhares de anos influenciaram o estabelecimento de ecossistemas dependentes, adaptados e sensíveis ao fogo (Bowman et al., 2009; Myers, 2006; Whelan, 1995).

Regimes de fogo naturais são iniciados principalmente por raios em períodos de transições entre estação seca-chuvosa e chuvosa-seca, em frequências e extensões variadas, geralmente influenciados pelo relevo, tipo e quantidade de vegetação e condições abióticas no momento das queimas (Allan & Southgate, 2002; Bond & Archibald, 2003; França et al., 2007; Ramos-Neto & Pivello, 2000; Russell-Smith et al., 2007; van Wilgen et al., 2004). Estes regimes contibuiram para o estabelecimento de ambientes pirofíticos e para a expansão de gramíneas C4 no final do Mioceno, uma vez que muitas espécies apresentam características adaptativas ao fogo (Bond, 2008; Keeley & Rundel, 2005). Esses ambientes pirofíticos são caracterizados pela produção de biomassa fina -graminóides, demais ervas e subarbustos com ramos até 0,6 cm de diâmetro (Luke & McArthur, 1978) – durante a estação chuvosa, seguida de um período seco, onde esta biomassa se transforma em material combustível, favorecendo a propagação do fogo (Bowman et al., 2009; Lehmann et al., 2014; Whelan, 1995).

A savana é considerada o bioma mais pirofítico do mundo, responsável por cerca de 86% de todas as queimas (Mouillot & Field, 2005), onde 30% de toda a produtividade primária do planeta é produzida sob uma taxa de precipitação geralmente acima de 700 mm por ano (Accatino et al., 2010; Grace et al., 2006; Sankaran et al., 2005). De forma geral, a biomassa fina é produzida mais rapidamente em formações savânicas e campestres, levando a intervalos menores de queimas do que em formações florestais (Bond & Keeley, 2005; Fidelis et al., 2010; Miranda et al., 2009). Fisionomias campestres e savânicas, em geral, apresentam substrato graminoso-herbáceo contínuo, o que favorece fogos de superfície com maiores velocidades de propagação do fogo, e são compostas por espécies mais adaptadas ao fogo quando comparadas a formações florestais (Higgins et al., 2000; Hoffmann et al., 2012).

Embora o fogo ocorra de forma natural em quase todos os continentes do mundo, a partir de 50.000-100.000 anos antes do presente (AP) o uso intensivo do fogo pelo Homem (Bar-Yosef, 2002; Bowman et al., 2009) provocou mudanças nos regimes de fogo naturais – em que as épocas de queimas foram diversificadas, passando a ocorrer inclusive em períodos em que não há fontes de ignição naturais e em frequências maiores do que as naturais, especialmente em ecossistemas florestais (Archibald et al., 2009; Kurz & Apps, 1999) - e consequentemente transformações nas paisagens (Bar-Yosef, 2002; Bowman et al., 2011), como a perda de áreas florestais (Whitlock et al., 2010), e a extinção de espécies, como por exemplo a extinção da megafauna na Austrália (Murphy et al., 2012). Em savanas australianas e no Cerrado, estudos indicam que práticas de queima de comunidades rurais e indígenas favorecem a criação de mosaicos de vegetação queimada e não-queimada, que podem impedir a propagação de incêndios no auge da estação seca, e beneficiar a reprodução, dispersão e forrageamento de algumas espécies (Bird et al., 2012; Bliege Bird et al., 2008; Jones, 1969; McGregor et al., 2010; Mistry & Bizerril, 2011; Mistry et al., 2005; Pivello, 2006; Russell-Smith et al., 2009; Welch et al., 2013). Porém, muitas vezes os objetivos de manejo dessas comunidades não estão relacionados com a conservação da natureza, e sim com a proteção de determinados ambientes como áreas de moradia, agricultura ou ambientes florestais importantes para a produção de madeira ou coleta de frutos e água (Borges et al., 2016; Eloy et al., n.d.; Leonel, 2000; Melo & Saito, 2011; Mistry, 1998). Dessa forma, o fogo contribuiu para o estabelecimento e desenvolvimento de populações humanas (Bowman et al., 2011; Pyne, 1993, 2016), bem como para a manutenção de algumas fitofisionomias como as campestres e savânicas, em que espécies arbóreas e gramíneas coexistem (Accatino et al., 2010; Bond, 2016; Bond et al., 2005; Higgins et al., 2000; Scholes & Archer, 1997).

Apesar da ocorrência natural do fogo nas savanas e de sua utilização como ferramenta de manejo por diversas populações nesses ambientes, políticas de exclusão do fogo foram implementadas pelos europeus durante o período colonial na África subsariana (Archibald, 2016; Laris & Wardell, 2006; Pyne, 1997a; Wardell et al., 2004), América do Sul (Durigan & Ratter, 2016; Mistry et al., 2016; Pivello, 2006; Sletto, 2008) e norte da Austrália (Edwards et al., 2015; Ritchie, 2009; Russell-Smith et al., 2007), devido ao interesse econômico voltado para ecossistemas florestais e a falta de compreensão dos europeus da dinâmica ecológica de ecossistemas savânicos. Estas políticas de exclusão do fogo contribuíram para (i) o aumento da flamabilidade de ambientes pirofíticos, onde amplas áreas (milhares de hectares) com histórico de queima semelhante e continuidade de combustível disponível para queima com grande potencial para propagação de incêndios se formaram, (Bond & Parr, 2010; Durigan & Ratter, 2016; Pivello, 2006; Russell-Smith et al., 2007); (ii) conflitos entre as autoridades governamentais e comunidades tradicionais que fazem uso do fogo, uma vez que essas populações foram proibidas de utilizar suas ferramentas de manejo como meio de subsistência (Batista et al., 2018; Eriksen, 2007; Kull, 2002; Laris & Wardell, 2006; Mistry & Bizerril, 2011; Morrison & Cooke, 2003; Moura & Viadana, 2011); e/ou (iii) ao adensamento da vegetação lenhosa em locais onde as condições climáticas e edáficas são favoráveis (Archibald, 2016; Bond et al., 2005; Honda & Durigan, 2016; O'Connor et al., 2014), o que pode reduzir a disponibilidade de combustível fino e o risco da propagação do fogo (Andela et al., 2017; Bradstock, 2010; Pinheiro & Durigan, 2009; Pinheiro, 2016).

As proibições de uso do fogo em áreas de savana onde comunidades locais utilizavam o fogo como ferramenta de manejo (Bowman et al., 2011; Pyne, 1997b; Scott et al., 2016), como na África subsaariana (Eriksen, 2007; Kull, 2002; Laris & Wardell, 2006; Wardell et al., 2004) e no Cerrado (Batista et al., 2018; Mistry & Bizerril, 2011; Moura & Viadana, 2011; Pivello, 2006), causaram descontentamento em populações locais aumentando o número de queimas de litígio e consequentemente a ocorrência de incêndios no final da estação seca. No geral, com a política de supressão do fogo no Brasil, os incêndios no Cerrado são comuns em UCs no final da estação seca (agosto-setembro) e em muitos casos são atribuídos ao homem (Fiedler et al., 2006; França, 2010; Medeiros & Fiedler, 2004; Moura & Viadana, 2012; Pereira et al., 2004) e em outros a raios (França et al., 2007; Ramos-Neto & Pivello, 2000), e quando em condições favoráveis (combustível e meteorologia) tendem a atingir grandes extensões de áreas (muitas vezes > 50.000 ha) que incluem vegetações resistentes e sensíveis ao fogo (Barradas, 2017; Batista et al., 2018; Pereira Júnior et al., 2014). Essas queimas no final da estação seca podem causar maior taxa de mortalidade ou topkill de espécies arbóreas do que as queimas no início da estação seca (Sato & Miranda, 1996; Williams et al., 1999)

Devido a ocorrência de grandes incêndios que geraram prejuízos ambientais e econômicos, bem como conflitos entre populações locais e autoridades ambientais países como Austrália (Russell-Smith et al., 2009, 2017), Estados Unidos (Freeman et al., 2017; Ryan et al., 2013) e Canadá (Lee et al., 2002; Stocks & Martell, 2016) têm implementado ações de manejo de fogo nas últimas décadas. Essas ações incluem queimas controladas em épocas do ano estratégicas, sob condições meteorológicas e qualidade e quantidade de combustível adequados para atingir objetivos específicos de manejo, como a conservação da biodiversidade – incluindo ambientes sensíveis e adaptados ao fogo - e reduzir conflitos com as populações locais, custos com combates a incêndios e a extensão de áreas queimadas por incêndios.

Além de reduzir a ocorrência e extensão de incêndios, a Austrália, possui programas com objetivos claros de redução de emissões de gases de efeito (GEE). Esses programs ao incorporar conhecimento ecológico tradicional (CET) dos aborígenes nas técnicas de queima de áreas silvestres obtiveram maior efetividade de manejo, e possibilitaram a geração de renda através da venda de créditos de carbono pela redução das emissões de GEE, por meio da redução de incêndios no final da estação seca resultantes de queimas controladas realizadas especialmente no início da estação seca (Preece, 2007; Price et al., 2012; Russell-Smith, 2016; Russell-Smith et al., 2013). Por outro lado, regimes de queimas na mesma época (independentemente da época do ano) e mesma frequência podem interromper o ciclo de vida de algumas espécies de plantas e levar populações a extinção (Armstrong & Phillips, 2012; Whelan et al., 2002).

Experiências de manejo como estas estão incentivando queimas controladas no inícios da estação seca e manejo integrado de fogo (MIF) em outras regiões pirofíticas, como em algumas zonas rurais e reservas ambientais africanas (Beatty, 2011;

Goldammer & de Ronde, 2004; Hoffmann, 2013; Kepe, 2005; Moore et al., 2002; Ouedraogo & Balma, 2012; UNU, 2015; Wardell et al., 2004) e recentemente (2014) em algumas áreas protegidas brasileiras (Schmidt et al., 2016, 2018). Dentre outras tentativas de manejo, regimes naturais de queima foram estabelecidos em algumas áreas do Parque Nacional Kruger na savana sul africana (Bond & Archibald, 2003; van Wilgen et al., 2004), que levaram a um intervalo de retorno do fogo de 13 anos, duas vezes maior comparado aquele estabelecido por queimas controladas (6 anos). Esse regime de queima natural foi marcado pela ocorrência de incêndios no período entre setembro e novembro (van Wilgen et al., 2000) e levou ao adensamento da vegetação lenhosa do parque (Bond & Archibald, 2003).

Dentre esses diferentes sistemas de manejo que estão sendo implementados em fisionomias savânicas, são desenvolvidas muitas pesquisas científicas voltadas para a caracterização de queimas de manejo e acompanhamento de seus efeitos na vegetação (Bowman et al., 2007; Dickinson & Ryan, 2010; Driscoll et al., 2010; Edwards et al., 2003; Moore et al., 2002; Petty et al., 2015; Russell-Smith et al., 2003b; Schmidt et al., 2018; van Wilgen & Biggs, 2011a; Van Wilgen et al., 2014). No Cerrado, foram reportadas diversas pesquisas voltadas para o comportamento e efeitos do fogo na vegetação lenhosa em diversas fisionomias e regiões, avaliando queimas em diferentes épocas do ano – início, meio e fim da estação seca - e frequências de fogo, para do Cerrado (Coutinho, 1982; Fidelis et al., 2010; Hoffmann et al., 2009, 2011; Honda & Durigan, 2016; Kauffman et al., 1994; Medeiros & Miranda, 2005; Miranda et al., 2010; Pivello & Coutinho, 1996; Ribeiro et al., 2012; Rissi et al., 2017; Sato, 2003; Sato & Miranda, 1996; Schmidt et al., 2017; Silva et al., 2010; Souchie et al., 2017). As condições meteorológicas são comumente medidas e indicadas como uma das variáveis que mais influenciam o comportamento e efeito do fogo entre o início e final da estação seca em trabalhos relacionados a queimas em vegetações de cerrado sentido restrito e campo sujo (Gomes et al., 2018; Hoffmann et al., 2011; Rissi et al., 2017). Embora as condições meteorológicas – como temperatura, umidade relativa e velocidade do vento, muitas vezes associadas ao período do dia – sejam um dos critérios utilizados na tomada de decisão de queimas de manejo (Schmidt et al., 2018), o uso destas variáveis como parâmetro/critério de pesquisa junto a queimas experimentais dentro da mesma época ainda não foi reportado para fisionomias do Cerrado.

O papel da pesquisa aplicada no auxílio à gestão de UC e elaboração/implementação de políticas públicas na área ambiental é amplamente reconhecido. No entanto, a contribuição de resultados de pesquisa para a gestão podem muitas vezes ser limitados (Arruda et al., 2018; Dickinson & Ryan, 2010; Driscoll et al., 2010; Gomes et al., 2018; Russell-Smith et al., 2003a). Para aumentar a contribuição da ecologia aplicada à gestão é recomendada a realização de experimentos conjuntos. Assim, a criação e a manutenção de ambientes de aprendizagem, que agreguem diferentes atores sociais, como pesquisadores, gestores e comunidades locais, são essenciais para a integração entre pesquisa, gestão e conhecimento tradicional (Berkes et al., 2000; Christensen, 2005; Perry et al., 2018; Petty et al., 2015; van Wilgen & Biggs, 2011b). Neste contexto, além das contribuições científicas e da formação da autora, os experimentos realizados para a execução desta tese de doutorado contribuíram também para integrar conhecimentos técnicos, científicos e tradicionais voltados para o Parque Nacional da Chapada das Mesas, MA, uma das primeiras UC do Cerrado a implementar ações de manejo de fogo, quebrando o paradigma da Política de "fogo-zero" em vigência no Brasil por séculos.

Esta pesquisa teve por objetivo identificar sistemas antropogênicos de manejo do fogo em savanas do hemisfério sul e avaliar o comportamento do fogo e os efeitos de diferentes épocas, frequências e condições meteorológicas de queima em vegetações lenhosas do Cerrado. Especificamente, foi possível descrever, pela primeira vez no Cerrado, o comportamento do fogo durante queimas precoces (início da estação seca) de baixa intensidade, feitas em condições metereológicas amenas, que são utilizadas em queimas controladas de manejo de fogo. Estas queimas foram comparadas com queimas precoces em condições ambientais menos amenas (menores umidades e maiores temperaturas) e queimas tardias (final da estação seca), simulando incêncios. O acompanhamento de áreas experimentais antes, durante e depois destas queimas possibilitou verificar os efeitos destes diferentes regimes de fogo sobre a vegetação lenhosa e comparar com a dinâmica da vegetação em áreas protegidas do fogo por até cinco anos. Para isto esta tese foi dividida em três capítulos que visam contribuir com as respectivas temáticas: (1) contexto histórico em que as políticas de fogo foram estabelecidas e os padrões de manejo utilizados na implementação destas políticas nas maiores extensões de savana do hemisfério sul e suas respectivas consequências empíricas na interação socioeconomica,, e meio ambiente; (2) caracteristicas do comportamento do fogo em áreas de cerrado sentido restrito aberto, considerando queimas em diferentes épocas,frequências e condições meteorológicas ; e (3) efeitos ecológicos de queimas em diferentes épocas, frequências e condições meteorológicas na vegetação arbórea em áreas de cerrado sentido restrito aberto.Cada capítulo desta tese foi escrito em forma de artigo na língua inglesa.

No capítulo um desta tese foram avaliadas as mudanças nas abordagens de queima em três regiões savânicas - Cerrado, norte da Austrália e sul da África - do período pré-colonial até a atualidade. Ao contextualizar política e economicamente os diferentes sistemas de manejo do fogo e reportar algumas consequências socioeconomicas e ambientais destes sistemas a partir da literatura científica, foi possível identificar os aprendizados e desafios associados a políticas de fogo sustentáveis e viáveis para regiões pirofíticas.

O capítulo dois apresenta as características do comportamento do fogo em diferentes sistemas de manejo em uma das três primeiras unidades de conservação (UCs) a implementar o MIF no Cerrado: Parque Nacional da Chapada das Mesas (PNCM), localizado no sul do Maranhão. Estes sistemas foram avaliados em áreas com histórico de queima entre dois e três anos em fisionomias de cerrado sentido restrito e foram representados pelas seguintes queimas experimentais: (1) queimas no final da estação seca (setembro), simulando o regime de fogo predominante em UCs do Cerrado regulamentadas pela política de exclusão do fogo; (2) queimas no início da estação seca (maio) próximas do meio-dia, quando a temperatura do ar atinge o máximo e a umidade relativa é menor que 50%; e (3) queimas no início da estação seca (maio) no final da tarde, quando a temperatura do ar diminui e a umidade relativa é maior que 50%. As queimas experimentais realizadas no início da estação seca representam as queimas implementadas pelo programa do MIF e, por isso, foram utilizados os mesmos parâmetros estabelecidos para as tomadas de decisão antes de cada queima prescrita. Não há registro do acompanhamento científico de queimas de manejo no Cerrado, e queimas realizadas em condições metereológicas amenas tendem a ter menor intensidade e menor velocidade de propagação da frente de fogo. Pode-se levantar a hipótese que isto levaria a maiores tempo de permanência de altas temperaturas que poderiam causar maior mortalidade de plantas, hipóteses que foram testadas no capítulo três.

Finalmente, no capítulo três, foram avaliados os efeitos destes sistemas de manejo do fogo em comunidades de espécies arbóreas em relação a: porcentagem de sobrevivência e topkill; número de recrutamentos; mudança na densidade de indivíduos, área basal e altura. Parcelas permanentes de pesquisa foram estabelecidas em fisionomias de cerrado sentido restrito no PNCM para possibilitar o monitoramento das comunidades arbóreas no período de 2015 a 2017. Para testar as implicações dos diferentes sistemas de manejo, o experimento foi dividido em: (a) parcelas sob regime bienal de queimas no final da estação seca (setembro); (b) parcelas sob regime bienal e trienal de queimas no início da estação seca (maio) iniciadas próximo ao meio-dia com a umidade relativa do ar abaixo de 50%; (c) parcelas sob regime bienal e trienal de queimas no início da estação seca (maio) iniciadas no final da tarde com a umidade relativa do ar acima de 50%; e (d) parcelas protegidas do fogo por quatro e cinco anos fogo.

Esta tese apresenta os primeiros resultados do acompanhamento científico de atividades de manejo de fogo no Cerrado. O delineamento experimental realizado, assim como todas as medidas realizadas em campo, contaram com a participação de gestores ambientais, moradores e brigadistas do Parque Nacional da Chapada das Mesas bem como outros funcionários do Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio), responsável pela gestão de UCs federais no Brasil. Esta tese foi escrita em forma de artigos na língua inglesa, em que cada capítulo corresponde a um artigo.

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Capítulo 1 - The legacy of fire suppression management policies on traditional livelihoods and ecological sustainability in three fire-prone continental savanna settings: impacts, consequences, new directions

O legado das políticas de manejo de supressão do fogo nos modos de vida tradicionais e na sustentabilidade ecológica em três cenários de savana continentais pirofíticos: impactos, consequências, novos rumos

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Abstract

Land occupation and management systems have defined fire regimes and intercontinental landscapes for millennia. The savanna biome is responsible for 86% of all fire events, contributes to 10% of the total carbon emissions annually and is home to 10% of the human population. During the European colonization fire suppression policies were instituted in many tropical savannas, markedly disrupting traditional fire management practices and transforming ecosystems. In this paper we assess savanna burning approaches from pre-colonial to contemporary eras in three regions: northern Australia, southern Africa and Brazil. In these three regions, fire suppression policies have led to (i) conflicts between government authorities and local communities; (ii) frequent late dry season wildfires and/or (iii) woody encroachment. Such consequences have required changes to historical fire management policies, including recognition and incorporation of traditional ecological knowledge in contemporary savanna fire management contexts. The application of integrative and flexible fire management programmes, mostly implementing prescribed early dry season fires, has substantially reduced the frequency of more severe late dry season fires and greenhouse gas emissions, as well as generating income opportunities for rural and traditional communities. However, prescribed burns should not include only one fire season, but a varying seasonal and frequency range, providing management for different species to persist. Additionally, when associated to environmental compensations from multinational companies or income generation to rural populations, these abatement programmes may stimulate over-fire management activities. We present a brief history of fire management policies in these three important fire-prone savanna regions, and identify ongoing challenges for implementation of culturally and ecologically sustainable, integrated fire management policies.

Keywords: Cerrado; fire regime; fire suppression; integrated fire management; prescribed fire; traditional ecological knowledge.

1. Introduction

Natural fire regimes (initiated by lightning or volcanos) have been molding landscapes in almost all continents on Earth for approximately 420 million years (Bowman et al., 2009; Scott & Glasspool, 2006). These natural fires were characterized by occurring mostly during rainy seasons and/or in the transition periods from dry to rainy seasons; their extents and frequencies varied according to local and regional conditions (Allan & Southgate, 2002; Bond & Archibald, 2003; França et al., 2007; Ramos-Neto & Pivello, 2000; Russell-Smith et al., 2007; van Wilgen et al., 2004). Fire has also contributed to the evolution of *Homo erectus* species since the lower Pleistocene, from at least 1.6 million years ago (Bellomo, 1994). Over the past 50,000 years the recurrent use of fire for foraging and agriculture has shaped modern fire regimes characterized by human-influenced fire frequency and seasonality (Bowman et al., 2011). The savanna is the most fire-prone biome in the world, due to the characteristic continuous grassy fuel layer that becomes increasingly flammable with the progression of the lengthy (5-8 month) dry season (Giglio et al., 2013; van der Werf et al., 2017). Worldwide, savannas produce 30% of the earth's net primary production (Grace et al., 2006), accounts for 86% of all fire events (Mouillot & Field, 2005), resulting in ~10% contribution of total carbon emissions annually (van der Werf et al., 2010). Africa is responsible for 71% of all savanna CO_2 emissions, South America for 12% and Australia for 7.3% (UNU, 2015). The biome is home to approximately 10% of the human population (White et al., 2000), many of whom rely on the use of fire for their livelihoods (Laris, 2002; Mistry et al., 2005, 2016; Russell-Smith et al., 2009; UNU, 2015; Welch et al., 2018).

The European colonization of savanna regions since the 16th Century was marked by the official prohibition in the use of fire (Pyne, 1997b). Fire suppression policies neglected and disrupted local agriculture and landscape management practices and led to profound ecosystem changes. These changes ranged from woody encroachment (of native and exotic species) due to reduced fire frequency (O'Connor et al., 2014), where there may occur increases in carbon stocks but also a reduction of species richness of plants and animals (Abreu et al., 2017; Maravalhas & Vasconcelos, 2014; Parr et al., 2012), ground water recharge and grazing potential (Angassa & Baars, 2000; Gray & Bond, 2013; Stevens et al., 2017), to extensive accumulation of flammable ground fuels and increased incidence of wildfires, favouring opened savannas and grassland physiognomies (Bond & Parr, 2010; Pyne, 1997b; Ryan et al., 2013; UNU, 2015). Such wildfires are generally unwanted due to high fire-fighting costs, risks of destruction of infra-structure, risks to human lives and health and high environmental impacts in rural and natural areas. In recent decades there has being growing recognition of the value and effectiveness of traditional (Indigenous, Aboriginal) fire management approaches for addressing contemporary fire management issues in fire-prone savannas —at least in a few southern African countries, parts of north America (Ryan et al., 2013), more recently in Brazil (Schmidt et al., 2018), and especially in northern Australia (Preece, 2007; Russell-Smith et al., 2017; UNU, 2015). However, as a legacy of colonial history, traditional savanna fire management practices are still officially stigmatized and/or prohibited in most African (Laris & Wardell, 2006), Asian (van Vliet et al., 2013), and South American savannas (FAO, 2007; Russell-Smith et al., 2017; UNU, 2015; Welch et al., 2018). When this approach is applied it is mainly focused on the reduction of the extension of wildfires - incentivized economically by carbon markets with the reduction of greenhouse gas (GHG) emissions (Gibbs et al., 2007; Petty et al., 2015).

In this contribution we aim to assess the antecedents of savanna burning approaches in three fire-prone continental savanna settings – northern Australia, southern Africa and Brazil. Our focus is especially concerned with: (i) the history of fire management in these regions; (ii) the contributions and limitations of local populations' management techniques that have been used for some generations locally – also known as traditional ecological knowledge (TEK) - to contemporary savanna fire management policies; and (iii) the challenges associated with incorporating and maintaining traditional practices in legal frameworks to better achieve sustainable fire management in these and other fire-prone regions. We hypothesize that fire policies on southern hemisphere savannas have been driven by economic demands and development since the colonial era (~1600), and that fire suppression policies were a consequence of this approach that led into sociocultural and environmental problems. In addition, fire management policies have been introduced to reduce wildfires extents and the economic losses caused by them in the last decades.

Synthesizing the management and policies experiences related to fire that have been adopted in Australia and Africa for the past decades may inform and help advance the implementation of adequate fire management policies in the Brazilian savanna, where this initiative is still incipient (Schmidt et al. 2018).

2. Natural fire regimes in savannas

Fire has been a natural evolutionary force in savannas leading to physiological and morphological adaptations – such as thick barks, fruit protection and subterranean organs and high belowground biomass that allows for resprouting after fire passage (Bond & Midgley, 2003; Bond & van Wilgen, 1996; Coutinho, 1990; Dantas & Pausas, 2013; Lamont et al., 2011; Simon & Pennington, 2012) - depending on vegetation type, fire intervals and severity, as well as water and nutrients availability (Bond & Keeley, 2005; Bond & Scott, 2010; Bowman et al., 2009; Dantas et al., 2013; Keeley et al., 2011; Power et al., 2008; Scott & Glasspool, 2006; Simon et al., 2009; Whelan, 1995). Commonly, natural fires caused by lightning occur during the rainy season and in transition periods, between wet-dry or dry-wet seasons, which are not always followed by rain, and, therefore, capable of burning large landscapes when fuel load is continuous, homogeneous and flammable (Bond & Archibald, 2003; Fiedler et al., 2006; Ramos-Neto & Pivello, 2000; Russell-Smith et al., 2007). Previous studies have recorded natural fire return intervals for savannas between 1-13 years and burned areas from 1 to more than 100,000 ha (Balfour & Howison, 2002; Bliege Bird et al., 2008; França et al., 2007; Ramos-Neto & Pivello, 2000; Russell-Smith et al., 2007; van Wilgen et al., 2000).

3. An overview of traditional use of fire in savannas

Homo species have used fire as a tool and influenced fire regimes in Africa over the past 400,000 years (Bird & Cali, 1998),Australia’s tropical savannas over at least 60,000 years (Clarkson et al., 2017; Roberts et al., 1993), and in Brazil over at least 32,000 years (Guidon & Delibrias, 1986). The use of fire is recognized as an essential tool underpinning the development and expansion of hunting and foraging societies (Bird & Cali, 1998; Jones, 1969). In the Holocene, fire was essential to the development of cattle domestication 10,000-8,000 years BP (before present) in the eastern Sahara in Africa (Marshall & Hildebrand, 2002), plant domestication in Northern Brazil around 8,000 years BP (Clement et al., 2010), and 5,000 years BP in western Africa (Phillipson, 2005). Overall, ancient anthropogenic burning has markedly contributed to human evolution (Bowman et al., 2011) and, at the same time, contributed to the creation of high-diversity landscapes, where the formation of small-scale successional mosaics increased local-scale biodiversity and prevented losses of important habitat resources (Bliege Bird et al., 2008; Bowman et al., 2016; Trauernicht et al., 2015).

In present times, fire continues to be used worldwide by traditional societies for multiple purposes, including: land clearing for agriculture, hunting, stimulating plant flowering/fruiting for harvesting purposes, regenerating pasture species, reducing fuels, increasing visibility, extending habitat zones, avoiding and selecting species, pest elimination, communication, protection, ceremonies and rituals, (e.g. Garde et al., 2009; Mistry et al., 2016; Shaffer, 2010). The frequency and the selection of areas to burn has depended on management goals, environmental conditions and resource availability (Bowman et al., 2011; Welch et al., 2018).

“Slash and burn" (swidden) agriculture is one of the most ancient cultivation systems (Iversen, 1956), which gradually became commonly practiced among Brazilian Indigenous groups (Dean, 1996; Piperno, 2011) and some African societies (Harlan, 1976). To implement this itinerant cultivation system in Brazil, small (usually < 1ha) forested patches are cut down during the end of the dry season so the biomass is ready to burn as soon as the rainy season starts, releasing nutrients from ashes to cultivated plants (Pedroso Júnior et al., 2008). North Australian Aborigines by contrast practiced "fire stick farming", which consisted of setting either small savanna fires with a smoldering fire-stick to hunt small game and gather plant resources while they traversed their traditional estates (Jones, 1969), or highly organized fire-drives to hunt larger game (Altman, 2009).

Prior to European colonization, repeated burning in small (typically hectare-scale) contiguous patches was a common strategy used by African, Brazilian and Australian Indigenous people to create mosaics with different fire histories and burnt extents (Bliege Bird et al., 2008; Garde et al., 2009; Laris, 2002; Welch et al., 2013, 2018). In these respective regions, burning would commence as fuels dried (cured) in the early dry season (EDS), from April to May, and extend throughout the dry season until the onset of the rainy season (September to March). Strategically burnt patches—for example reinforcing natural features such as wetlands, streams, rocky areas—served also as effective firebreaks thereby reducing the risk of extensive late dry season (LDS) wildfires (generally August – October) (e.g. Garde et al., 2009; Laris, 2002; Welch et al., 2018).

4. Traditional fire management systems and fire policies

4.1 Southern African savanna

During the 18th and 19th centuries gradual transference of land ownership from traditional local authority to the state domain enabled European colonial authorities to consolidate exploitation of African lands, labour and resources (Roe et al., 2009). Natural resource management policies during the colonial era were a central component of extending European political control into rural African landscapes (Neumann, 1998). Resources, such as wildlife and timber, were progressively placed under central regulatory authority with the rights of local people alienated over time (Roe et al., 2009). Disruption of Indigenous cultural traditions and practices was highly successful with, amongst others, "civilizing missions" that promoted racial segregation to divide and control local populations (Conklin, 1998).

Traditional fire management systems were discouraged or prohibited based on the view that Indigenous burning regimes threatened resources, property, the social hierarchies of rigid colonial societies, and were environmentally degrading (Laris, 2002; Mistry, 2000, 2002; Pearce, 2000; Pyne, 1993, 1997a, b). Fire suppression policies reduced numbers of controlled burns changing traditional anthropogenic fire regimes – which happened throughout the dry season, forming seasonal-mosaic burned patches (Laris et al., 2016) - across southern Africa to almost exclusive LDS fires (Shaffer, 2010). Extensive LDS wildfires are frequent (often annual-biennial) and extensive (100 – 300km^2) in remote sparsely populated landscapes (e.g. protected areas - PAs), greatly reduced in areas utilized by cattle, and further reduced in areas of cultivation and settlement (Archibald et al., 2009, 2010).

According to fire experiments and satellite information database, a history of frequent LDS fires in African savanna can result in a reduction of woody density and canopy cover, (Frost & Robertson, 1987; Furley et al., 2008; Menault & Cesar, 1982; Ryan & Williams, 2011), damage to fire-prone ecosystems and loss of biodiversity (Archibald, 2016; Laris & Wardell, 2006), significant greenhouse gas (GHG) emissions (Korontzi, 2005; Lehsten et al., 2009; Scholes et al., 1996; van der Werf et al., 2010) or conversely facilitate woody encroachment (Archibald, 2016; Bond & Parr, 2010; O’Connor et al., 2014). Fire prohibition led to conflicts between governmental officials and Indigenous or rural communities, given impacts of LDS wildfires on people’s livelihoods, and often resulted in protest arson fires (Eriksen, 2007; Kull, 2002).

Starting in the late 1950’s, newly independent nations typically adopted colonially-derived political structures and, in many instances, extended central authority over lands and resources further alienating local rights (Alden Wily, 2008; Mamdani, 1996). Colonial approaches to fire management were among those retained and have largely continued to the present via the emphasis on ‘green conditionality’ in the disbursement of aid and loans to developing nations (Bryant & Bailey, 1997; Mistry, 2002; Suchet, 2010). Policy for commercial agriculture, timber and PAs paralleled developments in ecological knowledge and has oscillated between no-burn to prescribed burns and lightning strike regimes (Kepe, 2005; van Wilgen et al., 2000). Fire regimes developed to conserve and manage southern African savannas are primarily based on natural, experimental or archaeological evidence (Govender et al., 2006; Hall, 1984; Roques et al., 2001; van Wilgen et al., 2003, 2004), consistently ignoring the existence of TEK and burning practices in contemporary rural communities (Shaffer, 2010). Although these rural fire practices aimed sustaining peoples’ livelihoods rather than conserving surrounded ecosystems, the sustainability of these environments was indirectly and commonly considered in management strategies, once without their natural resources human settlements would be unbearable (Chazdon et al., 2009; Chukwuone, 2008; Eriksen, 2007; Jones, 2004; Laris, 2002; Trollope, 2011). The traditional fire management practices that have persisted, in varying forms, throughout southern Africa have generally been unsanctioned and transformed into arson fire that only increased wildfire occurrence (Kuhlken, 1999; Kull, 2002). Fire management in PAs can also be based on biodiversity and especially landscape management purposes. For example, in Kruger National Park frequent prescribed fires in severe abiotic conditions (<30% relative humidity, > 30°C and in areas with no rain for more than 30 days) are recommended to kill trees, avoid wood encroachment and favor open vegetation types, which are preferred by large herbivores (Bond & Archibald, 2003).

The emergence of Community-Based Natural Resource Management (CBNRM) in the 1980s initiated a dramatic shift in natural resource use rights of local people on communal lands (Jones & Murphree, 2001; Jones, 2004). Driven by political reform (independence), legislated decentralized approaches devolved substantial rights over wildlife and wildlife-based revenue to local people in Zimbabwe and Namibia. Innovative joint venture tourism enterprises, based on traditional ecological knowledge, developed into Zimbabwe’s CAMPFIRE program (1989-2001) covering over 40,000 km^2 of communal land and generating $20 million in revenue, and Namibia’s Communal Area Conservancies currently cover more than 14% of the country, involve over 200,000 people and earn $2.5 million p.a. (NACSO, 2012). These have played a key role in the development of CBNRM throughout the region and across sub-Saharan Africa (Suich et al., 2009).

Community-based Fire Management (CBFiM) and Integrated Fire Management (IFM) concepts developed alongside the 'community forestry' branch of CBNRM in the late 1990s to reduce conflicts, firefighting costs and environmental losses (Goldammer & de Ronde, 2004; Pricope & Binford, 2012; Walters, 2015)– such as the burning of fire-sensitive vegetation, and the extinction of local populations of fire sensitive plant species or even fire adapted species within fire-resistant vegetation, which under frequent fires occurring in the same season can interrupt the populations' life cycles (Armstrong & Phillips, 2012; Myers, 2006; Schmidt et al., 2005; Whelan et al., 2002). Early initiatives tended to reproduce colonial approaches to fire management through training community work forces to implement externally driven prevention and suppression activities in communal areas, for example the Namibia-Finland Forestry Programme, and Working on Fire in South Africa.

Traditional fire knowledge was promoted as a central component of CBFiM and IFM programs in the late 2000s through growing recognition of contemporary Indigenous African, Australian and South American communities elsewhere using controlled burns to expand or maintain subsistence livelihood activities and resources, and reduce wildfires (Kepe, 2005; Kepe & Scoones, 1999; Kull, 2002; Laris, 2002; Mistry et al., 2005; Sheuyange et al., 2005; Yibarbuk et al., 2001). Programs in Botswana (UNDP, 2010), Mozambique (CARE/WWF, 2013), Namibia (FAO, 2011; UNU, 2015), Swaziland, Tanzania (Sills et al., 2014), Zambia (TNC, 2018) and Zimbabwe have significantly improved community livelihoods, reduced wildfires and, to varying degrees, been adopted into national and international fire management policies.

At present, such programs rely upon international development funding and remain relatively limited in duration and extent. In recent years, significant attempts have been made to incentivize community-based EDS fire management approaches to reduce GHG emissions reduction in comparison with frequent wildfires. Such fire management schemes need also to maintain savanna physiognomies, avoiding woody encroachment when abiotic conditions are favorable (Beatty, 2011; Hoffmann, 2013;

UNU, 2015). However, the application of these management systems are not being implemented extensively and, therefore, only initial results have been reported for wildfire extent reduction and savanna conservation (FAO, 2011; Goldammer & de Ronde, 2004; Goldammer et al., 2004; Kojwang, 2000; Moore et al., 2002). A Verified Carbon Standard methodology was approved recently for quantifying GHG emissions reductions from the implementation of EDS burning in miombo woodlands in Tanzania, Mozambique and Malawi (Sills et al., 2014; "VCS Methodology VM0029 Methodology for Avoided Forest Degradation through Fire Management," 2015).

4.2 Northern Australian savanna

European colonization of northern Australia commenced in the mid-19th Century and was associated with the establishment of large beef cattle pastoral leases and incipient mining activities. The colonizers effectively gave no recognition to Aboriginal interests in land under a mythical legal concept termed '*terra nullius'*, which ignored the reality of Aboriginal occupation now known to span at least sixty millennia (Clarkson et al. 2017). Catastrophic population declines are well documented by the start of the 20th Century associated mostly with disease, translocation and, in some regions, massacres (Levitus, 2009; Ritchie, 2009; Roberts, 2005).

In consequence, traditional fire regimes across the northern savannas, marked by seasonal-mosaics forming patches of areas burned throughout the dry season with different fire frequencies (1-10 years) (Bird et al., 2005, 2012; Bliege Bird et al., 2008; Bowman et al., 2004; Chase & Sutton, 1981; McGregor et al., 2010; Russell-Smith et al., 1997; Yibarbuk et al., 2001) were substantially interrupted and replaced with a new regime dominated by extensive (often >1000 km^2) and frequent (>0.3 fires p.a.) LDS wildfires (Edwards et al., 2015; Morrison & Cooke, 2003; Russell-Smith et al., 2007). These contemporary fire regimes have had, and in many locations continue to have, very significant deleterious impacts on a variety of ecosystem attributes, including: fire-vulnerable plant taxa and assemblages (Bowman & Panton, 1993; Freeman et al., 2017a; Russell-Smith et al., 2012), granivorous birds (Franklin et al., 2005), small mammals (Lawes et al., 2015; Woinarski et al., 2011), and greenhouse gas emissions (Edwards et al., 2015, 2018). In some locations, traditional management practices continue to the present day, most notably in the Arnhem Land region of the Northern Territory (Garde et al., 2009; Yibarbuk et al., 2001).

In 1976 and 1993 the Australian Government instituted legislation giving further opportunities for Aboriginal people to reclaim land—if claimants could provide evidence of their traditional association with that land. The 1976 legislation, which applies only to the 1.42 million km^2 Northern Territory, has allowed Aboriginal people to reclaim over 50% of that jurisdiction as freehold title; the 1993 legislation has enabled Aboriginal people nationally to claim lands under a joint 'Native Title' arrangement which does not extinguish pre-existing land use titles. Collectively, these recent developments have enabled Aboriginal people to currently reclaim 56% of the 1.2 million km^2 north Australian savannas region receiving mean annual rainfall of at least 600 mm (Archer et al., 2018). Despite being relatively "land rich", regional Aboriginal people remain severely economically and socially disadvantaged (Russell-Smith et al., 2017).

Land reclamation by Aboriginal claimants has also been associated with a strong desire to reinstitute cultural practices, including responsibilities for fire management of traditional estates. This intent has not been thwarted by colonial legislative instruments (e.g. bushfire management Acts of the three north Australian jurisdictions—Queensland, Northern Territory, Western Australia) given tacit understanding that the risks of LDS wildfire need to be managed by various means, including allowance for the lighting, under permitted conditions, of preventative fires (Preece, 2007). General acceptance of the pragmatic wisdom of Aboriginal landscape-scale fire management approaches are enshrined in regional biodiversity conservation applications—for example, underpinning fire management in the World Heritage property, Kakadu National Park (Director of National Parks, 2016).

Since the early 2000s, direct support for the implementation of Aboriginal landscape-scale savanna fire management initiatives has been afforded through development of Australia's formal savanna burning GHG emissions abatement programme - as part of meeting Australia's Kyoto reporting targets. Foremost amongst those developments has been the West Arnhem Land Fire Abatement (WALFA) project, formally established in 2006 to deliver effective fire management and associated industrial-scale GHG emissions abatement over a 28,000 km^2 region, under an offset arrangement with a multi-national corporate (Russell-Smith et al., 2013). Over the past decade WALFA has substantially exceeded the contracted 100,000 $t.CO_2$-e annual abatement by applying prescribed fires in the EDS, which transformed the former LDS-dominated fire regime to one where annual fire extent is reduced and the great majority of burning is delivered through prescribed EDS management. This project is also providing broader ecosystem health, maintaining fire adapted vegetation, and community benefits by generating conservation stewardship options where pastoral lands have been marginalized and LDS had frequently been occurring (Price et al., 2012; Russell-Smith et al., 2013). In contrast, more productive, fertile and humid areas have been subjected to very limited prescribed fires in Australia and are undergoing woody encroachment(Stevens et al., 2017; Williams et al., 2002).

In 2013 the Australian Government formally accredited a savanna burning GHG emissions accounting methodology (CoA, 2013), as part of its then Carbon Farming Initiative (CFI) programme. Further refinements to that emissions abatement method have continued (CoA, 2015) under the revised Emissions Reduction Fund (ERF) programme, and new complementary savanna burning carbon sequestration components are in preparation (e.g. Cook et al., 2016). Under the ERF, the Australian Government purchases carbon credits (where a credit = 1 $t.CO_2$-e) from contracted project suppliers currently at ~USD 10 per credit. Additionally there is a nationally strong voluntary market where business entities buy savanna-burning credits at significantly higher prices to meet their corporate social responsibility requirements. Although some environmental benefits can come from this trade, since greater investments are needed for implementing fire management activities, it is questionable whether conservation interests and goals will generally guide the transactions.

As at early 2018 there were 78 formally ERF-registered savanna burning projects of which 32 are on Aboriginal lands, encompassing 25% of the entire 1.2 million km^2 savanna region above the 600 mm mean annual rainfall isohyet. Importantly, while the application of savanna burning projects may adequately complement the cultural requirements of Aboriginal landowners (Garde et al., 2009; Russell-Smith et al., 2009), contemporary regulatory and administrative requirements add levels of complexity which have the potential to disenfranchise full landowner participation (Perry et al., 2018).

This Australian experience is encouraging the development of similar approaches in other savanna regions, especially southern Africa and most recently in Brazil (Russell-Smith et al., 2017; Schmidt et al., 2018; UNU, 2015).

4.3 Brazilian savanna (cerrado)

European colonization started in the Atlantic rainforest along the Brazilian coast in the 1500's, focusing on Brazilwood (*Paubrasilia echinata)* extraction, and sugarcane and coffee plantations. Fire was used to implement all three economic activities, incorporating and adapted from traditional Indigenous techniques. Approximately 100 years after Europeans arrived, the use of fire became less tolerated, due to its damaging effects on sensitive rainforest vegetation (Dean, 1996). In the 1700's, livestock and agricultural activities expanded inland through the Brazilian savanna (Silva et al., 2012). Europeans progressively occupied Indigenous territories and consequently large extents of native savanna landscape were transformed to pastures, monoculture crops and settlements. Indigenous populations have been significantly reduced to less than 10% of their pre-Columbian number, due to diseases and conflicts (Pivello, 2006).

As well as native vegetation being replaced by exotic grasses and extensive monocultures, Indigenous traditional burning systems and activities were significantly reduced (Nevle & Bird, 2008). However, some of their techniques were adapted and incorporated in agricultural and livestock production systems (Pivello, 2011). Major differences between traditional Indigenous and farmers' burning practices included the size of the areas managed, fire frequency, vegetation type, and purposes. Moreover, the indiscriminate use of fire in monocultures and pastures, as well as the accumulation of unmanaged fuel loads in remote areas, led to changes in the fire regime maintained by Indigenous people, characterized by burnings throughout the dry season, usually small patches and in a frequency of 1-6 years, depending on the management goal (Borges et al., 2016; Melo & Saito, 2012, 2011; Mistry et al., 2005; Pivello, 2006; Welch et al., 2013, 2018); the Brazilian savanna became mostly burnt by extensive LDS wildfires rather than small patchy burns (Pereira Júnior et al., 2014; Pivello, 2011). The recurrence of large wildfires resulted in a zero fire tolerance policy in the whole country, regardless of the vegetation type or seasonal period (Durigan & Ratter, 2016). Consequences of this policy for the savannas included forest encroachment in less frequently burnt locations and where abiotic conditions were favorable (Abreu et al., 2017). This situation is mostly restricted to small, well-structured and PA where the zero-fire policy is actually effective due to isolation from anthropogenic fire sources or, more commonly rapid fire-fighting responses. In most Cerrado PA, however, the zero fire tolerance policy leads to large areas burned by wildfires in the LDS in savannas dominated by fine biomass due to the combination of homogeneous and continuous layers of fuel load and recurrent sources of ignitions (natural and arson) (Batista et al., 2018; Pereira Júnior et al., 2013; Pivello, 2006; Ramos-Neto & Pivello, 2000); landscape homogenization (Durigan & Ratter, 2016); changes to fauna distribution and species richness (Briani et al., 2004; Frizzo et al., 2011; Maravalhas & Vasconcelos, 2014); and increases in GHG emissions due to frequent and extensive LDS wildfires (Pinto & Bustamante, 2010).

Throughout its history, Brazil has pursued a development policy where agriculture and livestock have been identified as the main economic activities (Silva et al., 2012), resulting in deforestation and immeasurable biodiversity losses, including within the savanna region (Klink & Machado, 2005). Nevertheless, in 1961, the establishment of the first three PAs in central Brazil was an important step towards recognizing the environmental values of savanna ecosystems. However, these PAs forbade human settlement and the direct use of natural resources even when these overlapped with Indigenous peoples' and small farmers' territories (MMA, 2006), where commonly patchy-burnings mosaics took place (Pivello, 2006; Welch et al., 2013) and could have benefit species richness by creating different disturbance histories, frequencies and burned fragments that helped stopping wildfires. Consequently, PAs have become a source of conflict (Diegues, 2000), frequently involving the use of fire by local communities, for livelihood purposes or protest (Batista et al., 2018; Mistry & Bizerril, 2011). The natural tendency of unmanaged fire-prone savanna ecosystems to develop highly flammable fuel load (continuous, homogeneous, and dry), together with illegal fire ignitions, has rendered Brazilian savanna PAs increasingly vulnerable to LDS wildfires (Durigan & Ratter, 2016;

Pivello, 2006). This is especially deleterious to biodiversity because PAs are commonly the only fragments of native vegetation in large landscapes. When such fragments are hit by large wildfires, large proportion of the native areas are homogenized and burned simultaneously, causing high animal mortality especially due to lack of resources after the fires in a fragmented landscape (Silveira et al., 1999).

After decades of frustrated attempts to reconsider zero-fire policies, and scientific evidence of the ecological role of fire in the Brazilian savannas (Maravalhas & Vasconcelos, 2014; Miranda, 2010; Pivello, 2011; Simon et al., 2009), the national legislation was changed in 2012 to explicitly allow for fire management for conservation purposes, such as landscape and biodiversity maintenance that guides all Brazilian PAs, in fire-prone environments (Schmidt et al., 2018). A binational Brazil-German project collaboration implemented the first Integrated Fire Management program in the Brazilian savanna, 2012 - 2016 (the “Cerrado-Jalapão project”). This was the first politically sanctioned trial for using prescribed burns to reduce wildfires, reduce GHG emissions, and support biodiversity conservation outcomes in Brazilian savanna PAs. Other goals of the project were to: (1) enhance improved dialogue between environmental managers, scientists and local communities, and; (2) reduce fire-fighting costs (Schmidt et al., 2018).

The Cerrado-Jalapão IFM pilot programme trialled the implementation of low intensity, EDS (April to July) fires, commencing in 2014 at three PAs. Subsequently, the trial was extended and in 2017 included eight PAs and 11 Indigenous reserves. The burning program was annually planned, implemented, monitored and evaluated jointly by policymakers and stakeholders, including representatives from international, national and state environmental institutions, researchers, and local communities. The Cerrado-Jalapão IFM project resulted in up to 57% less LDS fires and up to 75% increase in EDS fires over three years of implementation (Mistry et al., 2018; Schmidt et al., 2018), stimulating broader acceptance of fire management and specifically prescribed fires in both public and private lands. The implications of this fire management programme on woody encroachment have not been reported yet, however these areas continue to burn in a frequency of 2-4 years in the EDS that helps to keep opened physiognomies.

The Cerrado-Jalapão IFM programme has contributed significantly to retraction of the zero-fire paradigm in Brazilian fire-prone ecosystems that will likely spread to other Brazilian and South American regions. The programme has also helped to push further legislative changes regarding the hiring of wildland fire-fighters and fire-managers—promoting the training and hiring of community-based fire brigades, and encouraging local communities' participation in landscape and fire management decisions and activities in communal, public and private areas.

Preliminary data indicate that fire management following this model may contribute to reduced GHG fire emissions, which can help bringing investments to fire management activities, but on the other hand can overcome financial interests in detriment of conservation goals. Further GHG monitoring will help quantify resultant benefits and pave the way for future financing of enhanced fire management actions through carbon abatement programs.

5. Savanna fire management policies: contributions from traditional systems and implications for institutional change

Indigenous fire practices, socio-economic conditions and land management policies have rarely been constructively aligned throughout the long colonial history of savanna occupation. This mismatch can be attributed to the prioritization of economic development policies detrimental to broader socio-ecological values. Fire suppression policies, initially focused on conserving forest resources, were introduced to uphold centralized economic interests, resulting in broad-scale impacts on local livelihoods and land use practices, conflicts between state authorities and local communities, and variable environmental changes. Changing these colonial fire management paradigms, particularly policies emphasizing fire suppression, is a long-term social and political process, which must overcome institutional resistance and lack of investment in community engagement, governance and technological infrastructure (Russell-Smith et al., 2017; UNU, 2015).

In recent decades there has been growing support from at least some savanna state authorities to formally recognize traditional fire knowledge systems in contemporary management contexts. In substantial part this recognition has been driven by the magnitude of recurrent fire management issues associated particularly with the impacts of extensive LDS wildfires on both rural livelihoods and ecological values, such as losses of fire sensitive species and ecosystems. As well, there has been growing understanding of the dynamic role which fire regimes play in maintaining healthy savanna ecosystems, which should not only include EDS or LDS seasonality (Freeman et al., 2017b; Laris et al., 2016), but a varying seasonal and frequency range, providing management for different species to persist. As illustrated by our inter-continental comparative assessment, traditional savanna fire knowledge systems and practices contribute a number of valuable common practical lessons pertinent to contemporary management contexts. For example, application of fine-scale strategic EDS fire management can: (1) significantly reduce the extent and impact of LDS savanna fires; (2) reduce GHG emissions that is a governmental management goal to meet international settlements and accordance; (3) reduce the impacts on biodiversity assets and values, such as the maintenance of fire sensitive and adapted species; hence (4) generate income and opportunities for regional traditional communities through GHG emissions abatement and community-based fire management (CBFiM) schemes (Russell-Smith, 2016; Schmidt et al., 2018; UNU, 2015); and (5) allow for the undertaking of prescribed LDS management applications under safely contained conditions when necessary (e.g. using relatively intense fires to avoid unwanted woody thickening and encroachment) and with specific biodiversity conservation goals. ;

Effective fire management policies in fire-prone ecosystems are dependent on the knowledge related to historical fire regimes, such as the natural fire regimes under which many species evolved, adequate legal frameworks, institutional capacities, and broad community support and engagement to protect human lives and property, manage destructive wildfires, and conserve natural ecosystems (Moritz et al., 2014; Pereira et al., 2012; Stephens et al., 2009). While national fire management policy frameworks typically reflect top-down control, recognition of traditional management knowledge and practice in formal policy frameworks has obvious benefits for enhancing fire management effectiveness (Dombeck et al., 2004; Foster et al., 2003; Mistry et al., 2016; Myers, 2006; Silva et al., 2010). A bottom-up framework should consider an integrated approach, where governmental authorities together with local communities and scientists discuss and build management plans and strategies to meet both goals, when possible (Christensen, 2005; Driscoll et al., 2010; Wells et al., 1992). However, for many developing countries in the post-colonial era, transitioning from state-centric fire suppression policies to local-level inclusive approaches poses significant challenges, including lack of local-level governance and well defined policy processes, and institutional and community capacity to deal with socially contentious fire management issues (Brown, 2003; Mistry et al., 2016; Russell-Smith et al., 2017; Tacconi et al., 2006; UNU, 2015).

The practical benefits of co-opting and adapting traditional savanna fire management approaches has been demonstrated by local or regional-scale examples in each of the three fire-prone continental settings. In the sparsely settled and fire-prone savannas of northern Australia, regional fire management policies have tacitly accepted (while not fully acknowledging) the pragmatic utility of traditional Aboriginal seasonal fire management practices (Preece, 2007). Positive experience with that policy approach has, over the past decade, encouraged development of a formal (state-sanctioned) abatement program to meet international agreements over the reduction of GHG emission which, at the time of writing, is now being implemented across 300,000 km^2 of Australia's northern, higher rainfall (>600 mm p.a.) region, mostly on Aboriginal-owned lands (Russell-Smith et al., 2018). However, when associated to environmental compensations from multinational companies or income generation to rural populations, these abatement programmes may stimulate over-fire management activities (exceeding EDS fires and bringing damages to fire adapted ecosystems).

Although most savanna regions in South America and sub-Saharan Africa are still regulated by suppressive fire management measures, recent integrative approaches (e.g. CBFiM, IFM) are increasingly delivering optimistic results and expanding official fire acceptance of rural activities and Indigenous practices—including support for sustaining local livelihoods, and incorporation into PA management practices (Goldammer & de Ronde, 2004; Schmidt et al., 2018). Ironically, savanna PAs present challenging settings for implementing strategic fire management programmes since they currently comprise the most extensively burnt tenures in sub-Saharan African and South American continental contexts. As noted previously such conditions can be attributed to regionally high and continuous fuel loads associated with generally limited landscape fragmentation, fire use restrictions, and abundant sources of ignition from surrounding densely settled agricultural holdings (Archibald, 2016; Batista et al., 2018; Pereira Júnior et al., 2014; Schmidt et al., 2018). Inclusive policies which support the sustainable fire management practices of rural communities (including those whose livelihoods are reliant on PAs), and enhanced understanding of the dynamic role of fire regimes in maintaining healthy savanna ecosystems, are mandatory for eliminating conflicts, reducing wildfire issues, and conserving biodiversity.

As documented here, recent decades have witnessed the emergence (albeit tentative) of a profound shift to official acceptance and recognition of more community-based fire management approaches in three fire-prone continental savanna settings. Key points arising from that discussion can be summarized as a schematic (Fig. 1), illustrating fundamental differences in practical societal, livelihood, and environmental outcomes emanating from the implementation of repressive or inclusive fire management policy approaches, respectively. Importantly, the latter inclusive approach inherently requires ongoing refinement through adaptive monitoring and management processes (Gómez-Baggethun et al., 2013; Huffman, 2013; Roos et al., 2016), and, as witnessed recently on all three continents, it has helped energize development of CBFM and IFM approaches (Goldammer & de Ronde, 2004; Schmidt et al., 2018), and novel incentivized GHG emissions abatement and carbon sequestration schemes (Russell-Smith et al., 2013; UNU, 2015). After a long colonial history of repressive fire management policies, the importance of locality- and context-specific Indigenous and traditional fire management approaches is at last being recognized to help address global savanna fire management problems.

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Capítulo 2 - How does the Cerrado burn? Informing fire management in the Brazilian savanna

Como o Cerrado queima? Informando o manejo do fogo na savana brasileira

Abstract

Recurrent large wildfires have become a global problem threatening human lives and assets, as well as environmental sustainability; such wildfires are especially common in unmanaged fire-prone environments. Weather conditions and cured biomass in the late dry season (LDS, September) increase the chances of high-intensity fires that can be more severe and difficult to control. The Cerrado region, in central Brazil, is a tropical humid savanna where LDS frequent and large wildfires have prevailed in the past decades, especially in protected areas (PAs). Since 2014, prescribed early dry season (EDS, April-July) fires have been implemented within a pilot Integrated Fire Management (IFM) programme, where the main goals are to change LDS fire regime and fragment continuous fuel load in Cerrado PAs and Indigenous reserves. Under this perspective, we aimed to characterize fire intensity, fuel consumption and heat released, i.e. fire behaviour, comparing fires in: the beginning and end of the dry season (EDS and LDS), biennial and triennial frequencies, and relative humidity above and under 50%. Additionally, we identified the parameters most influencing fire behaviour under these conditions. We conducted 20 experimental fires in 2015 and 11 in 2017 in 12 experimental areas, where we recorded: weather conditions (air temperature, relative humidity and wind speed), rate of fire front spread and fuel consumption to calculate fire intensity. Our results show that EDS fires with air relative humidity below 50%, have similar fire intensities to LDS fires, but they are both different from EDS fires with relative humidity above 50%. This pattern shows that fire intensity is mostly related to weather variations, such as air temperature, air relative humidity and wind speed, which may vary along time of the day and season of the year, as shown by our experimental records. From our experiment, low-intensity, EDS fires with relative humidity above 50% in a lower rainfall year (2017) registered lower fuel consumption and heat released compared to the same conditions in a higher rainfall year (2015). Considering weather forecast to carry out higher-intensity fires are indispensable for fire management, including measurements in the field just before burning takes place. Acknowledging these biotic and abiotic local and regional conditions can lower human and equipment resources required for fire management, as well as better achieve conservation goals.

Keywords: fire behavior, fire seasonality, fuel load, prescribed fires, weather conditions

1. Introduction

Fire-prone ecosystems are characterized by the annual occurrence of ideal conditions for fire to propagate across the landscape, provided by continuous layer of fine fuel, lengthy dry season and systematic ignition source (Bowman et al., 2009; Whelan, 1995). In the Anthropocene era, human ignition sources have increased fire occurrence and burned extents (Bowman et al., 2011; Pyne, 1997; Roos et al., 2014). Which tends to enhance vegetation traits of flammability, e.g. favouring graminoids over woody plants and keeping opened physiognomies (Accatino et al., 2010; Bond et al., 2005; Miranda et al., 2002), where the main source of fuel lies in the standing structure of dead biomass (Kauffman et al., 1994; Pausas & Moreira, 2012). Recurrentlarge wildfires have become a global problem threatening human lives and assets, as well as environmental sustainability, such wildfires are especially common in unmanaged fire-prone environments (Edwards et al., 2015; Hantson et al., 2017; Moritz et al., 2014; Scott et al., 2016).

Savanna is one of the most fire-prone biome on earth (Giglio et al., 2013; van der Werf et al., 2017), responsible for the production of around 30% of the world's net primary production (Grace et al., 2006) and approximately 86% of all fire incidents (Mouillot & Field, 2005). Although fire occurs naturally within savanna physiognomies, large savanna areas are often hit by human initiated wildfires mainly because of the combination of unmanaged, highly flammable, fuel load (Archibald, 2016; Russell-Smith et al., 2017; UNU, 2015; Yates et al., 2008) and extreme fire weather conditions (Cheney & Sullivan, 2008; Gill et al., 1996; Moritz, 2003). In tropical savannas, anthropogenic wildfires commonly occur in the late dry season (LDS, from August to October in southern hemisphere and from December to February in the northern hemisphere) with high frequency (annual-triennial), burning large areas (>100,000 ha, Archibald, 2016; Cahoon Jr et al., 1992; Pereira Júnior et al., 2014; Pivello, 2006; Russell-Smith et al., 2007; Shaffer, 2010; Yates et al., 2008); whereas natural wildfires – originated by lightning – usually occur in the transition between wet and dry seasons, vary in the return interval (1-13 years) and burned extension (1 to >100,000 ha) (Balfour & Howison, 2002; França et al., 2007; Ramos-Neto & Pivello, 2000; van Wilgen et al., 2000). Weather conditions and cured biomass in the LDS increase the chances of high-intensity fires that can be more severe and difficult to control (Whelan, 1995; Williams et al., 1998). In tropical humid savannas (rainfall >1,100 mm), grasses grow fast providing enough cured fuel to burn annually by the end of the dry season (Accatino et al., 2010; Batmanian & Haridasan, 1985; Williams et al., 2002; Yates et al., 2008).

The Cerrado region, in central Brazil, is a tropical humid savanna where LDS frequent and large wildfires have prevailed in the past decades, especially in protected areas and reserves (PAs) (Barradas, 2017; Batista et al., 2018; França, 2010; Pereira Júnior et al., 2014; Pivello, 2006; Schmidt et al., 2018). Commonly, these wildfires are difficult to extinguish and generate large governmental expenses in firefighting operations (Schmidt et al., 2018). Since 2014, prescribed early dry season (EDS, April-July) fires have been implemented within a pilot Integrated Fire Management (IFM) programme. The main goals of this pilot IFM programme are to change LDS fire regime and fragment continuous fuel load in Cerrado PAs and Indigenous reserves where it has been implemented, avoiding large LDS wildfires (Schmidt et al., 2018). Fire behaviour is commonly used for evaluating how the vegetation burns under different fire regimes (mainly fire season and frequency) influenced by the abiotic and biotic conditions, such as type of fuel, fuel load, weather and slope (Whelan, 1995). Fuel consumption, rate of spread, flame height, intensity, heat released, air temperature and residence time of high temperatures during fire passage are one of the variables measured to determine fire behaviour and can help predicting the type of fire, impacts on the environment and burned extents (Cheney et al., 1993; Miranda et al., 1993; Shea et al., 1996; Trollope & Trollope, 2002; Ward et al., 1996), however it does predict directly the effects of fire in the biota (Rothermel & Deeming, 1980). Researches related to fire behaviour have previously contributed to decision making process (Christensen, 2005; Dickinson & Ryan, 2010; Govender et al., 2006; Lee et al., 2002), where fire seasonality and frequency and fuel type are generally the parameters used for comparing experimental fires (Kauffman et al., 1994; Miranda et al., 2010; Rissi et al., 2017; Trollope & Trollope, 2002; Williams et al., 1998). Although weather conditions are known to influence fire behaviour in Cerrado vegetation (Conceição & Pivello, 2011; Hoffmann et al., 2012; Miranda et al., 1993; Rissi et al., 2017) and it is being used as a parameter in management decisions within the implementation of the IFM – specially air relative humidity, associated to time of day (Schmidt et al., 2016, 2018), weather parameters have not yet been used as an experimental criteria to evaluate changes in fire behaviour within the same season and frequency.

Previous studies have shown that fire behaviour in the Cerrado is not responsive to fire seasonality alone but to the combination of vegetation type, fuel load, fuel moisture content and proportion of dead fuel, all of which related to the vegetation physiognomy, time since fire (TSF), annual precipitation and weather conditions (Hoffmann et al., 2012; Kauffman et al., 1994; Miranda et al., 2002, 2010; Rissi et al., 2017; Schmidt et al., 2017). Larger volume and drier proportion of fuel load are more likely to generate higher fire intensities, with higher rates of spread and fuel consumption in open Cerrado physiognomies (Kauffman et al., 1994; Miranda et al., 2010; Sato, 2003). When fuel moisture is high the heat required for fire to propagate is higher reducing the rate of spread compared to situations where fuel is drier (Cochrane, 2009; Rothermel, 1983). The heat released from a fire is higher when the rate of spread is slower given the same fireline intensity, since more heat is concentrated on the site (Rothermel & Deeming, 1980).

Fire management is yet to be intensively implemented in tropical savanna ecosystems, once changing fire exclusion paradigms is a long-term process. Local and scientific ecological knowledge have been guiding this transition in many management systems (Driscoll et al., 2010; Goldammer et al., 2004; Mistry et al., 2018; Myers, 2006; Russell-Smith et al., 2009b; Schmidt et al., 2018), and because institutional management goals maybe different from the rural communities' management needs, the adaptive use of fire and monitoring and evaluating the implementation of these management systems is recommended (Christensen, 2005; Driscoll et al., 2010; Eriksen, 2007; Kaufmann et al., 2003). Understanding fire behaviour under different fire regimes can help inform management systems and also prevent economic and human resources losses (Spring et al., 2007).

Under this perspective, we aimed to characterize fire intensity, fuel consumption and heat released, i.e. fire behaviour, comparing fires in: the beginning and end of the dry season (EDS and LDS), biennial and triennial frequencies, and relative humidity above and under 50%. Additionally, we identified the parameters most influencing fire behaviour under these conditions. We hypothesized that fires in the EDS, shorter return interval (biennial) and relative humidity above 50% results in lower fire intensities and fuel consumption. The low rate of fire spread under these conditions will keep heat released similar between EDS fires but lower compared to LDS fires. We expect that, weather conditions and fuel load are the main drivers explaining the differences among these experimental fires. .

Identifying the parameters that influence fire behaviour along a gradient of biotic and abiotic conditions broadens management possibilities and provides valuable scientific information to measure and predict environmental consequences of fire regimes. To our knowledge this was the first time fire behaviour was characterized under high humidity (>50% air humidity) conditions within the Brazilian savanna, even though managed fires are commonly carried out in such conditions.

2. Material and methods

2.1 Study area

The Chapada das Mesas National Park is located in northern Cerrado (CMNP, 7°19’0”S and 47°20’6”W), in the northeast of Brazil (Fig. 1). The CMNP was created in 2005 with 160,000 ha aiming to conserve the natural landscape and the local biodiversity. However, the Brazilian fire exclusion attempt in PAs did not prevent wildfires from burning large extents of the CMNP burned extents by wildfires during the dry season throughout the park since its creation (ICMBio, 2017, Fig. 2). The combination of fire-prone vegetation, high air temperature (average 35°C) and average annual rainfall of 1,500-1,800 mm, concentrated in seven months (October-April, INMET 2017), makes the region highly flammable and, consequently, frequently burned by wildfires. The park is predominantly covered by savanna physiognomies, mostly open savanna on well-drained, sandy soils. Most of the fire-sensitive vegetation consist in riparian and swampy forests that grow along watercourses.

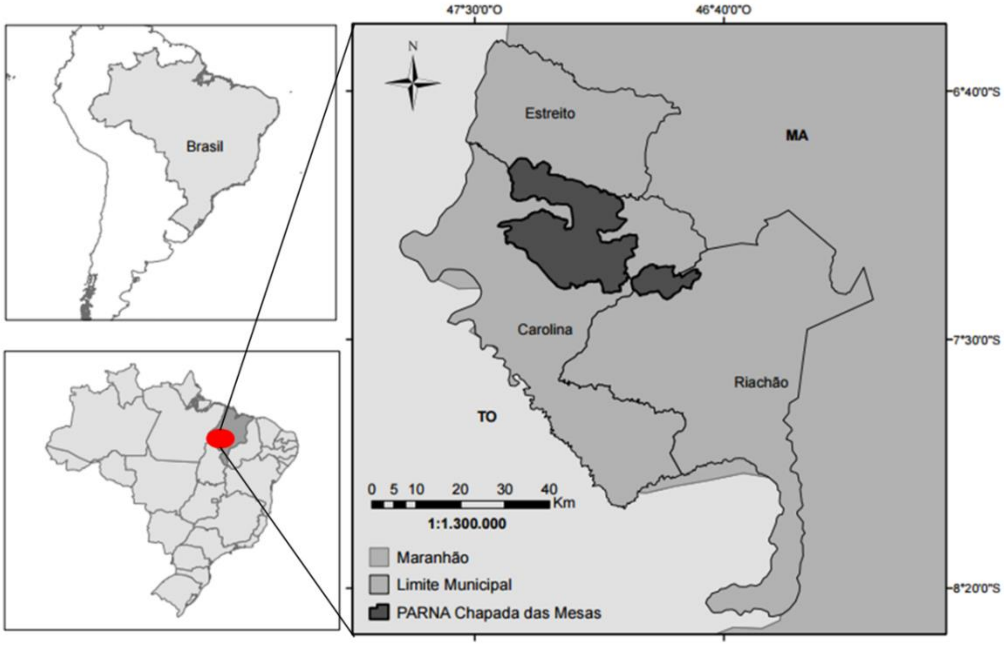

Fig. 1. Chapada das Mesas National Park location in regional, national and continental scales. Font: Victor Ferreira, 2015.

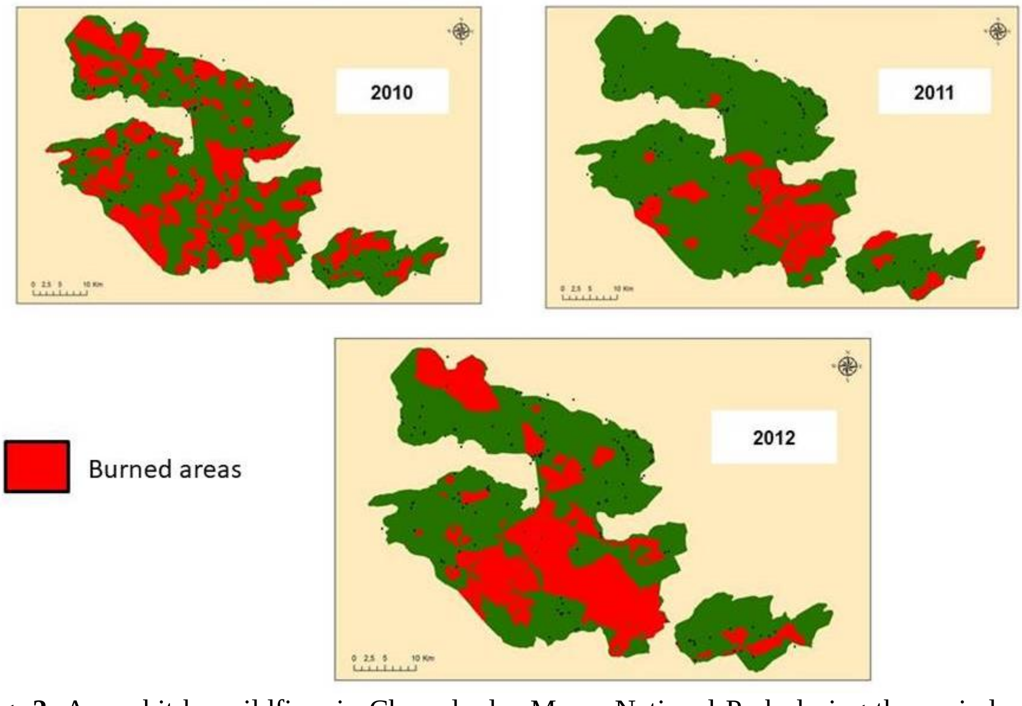

Fig. 2. Areas hit by wildfires in Chapada das Mesas National Park during the period of 2010-2012. Font: ICMBio, 2017.

The CMNP was one of the first three PAs to implement the IFM pilot program in 2014, especially because of its excessive wildfire occurrences and the managers' interest in participating. As the implementation of fire management is still very new to Brazilian PA managers, environmental authorities are still seeking parameters and indicators that can help to determine when to start prescribed fires on the field (Schmidt et al., 2018). Thus, our experimental design was previously discussed with managers and focused on questions that may support management decisions, such as how fire behaves under different weather conditions in two periods of the day, two seasons (early vs. late dry season) and fire intervals (two and three years).

2.2 Fire treatments

We based our experimental fire tretaments in the management prescribed fires performed by PA managers. During fire management, EDS prescribed fires are carried out yearly, in different areas from April until June by the PA's staff. Prescribed management fires are precursory in Brazil (Schmidt et al., 2018) and, therefore, several types of fire are being tested and still little is known about their behaviour during these burns. Managers tend to favour safe fires, which are those carried out in lower temperature and higher air humidity conditions, commonly started during late afternoon or early night.

To describe fire behaviour in different abiotic conditions, and fire history (two and three years since last fire). Experimental treatments were based on the prescribed burning practices, and encompassed experimental fires in the early-dry season both during the day (air humidity <50%) and during late-afternnon or night (air humidity >50%) as well as in late dry season to simulate wildfires. Hence, we implemented the following experimental fire treatments: (i) low-intensity EDS fires (Low-EF): evening fires (17:30-19:00), when air relative humidity was >50% in May; (ii) high-intensity, EDS fires (High-EF): mid-day fires (11:00-16:30), when air relative humidity was <50% also in May; (iii) LDS fires (LF): mid-day fires (12:00-14:30) carried out in September (Table 1).

All research areas were chosen by the CMNP's staff in open savanna physiognomies to represent main environmental conditions and in fragments previously isolated by firebreaks (Fig. 3). We selected a total of 12 experimental areas: four were unburned for two years and received Low-EF and High-EF experimental fires in May 2015 and May 2017 (biennial); four areas were unburned for three years and received Low-EF and High-EF experimental fires in 2015 (triennial) and the additional four areas received only the LF experimental treatments that for safety reasons were only applied in areas unburned for two years (biennial, Fig 4), one of these areas LF was hit by a wildfire in 2017. These studied fire intervals (2 and 3 years) represent the fire interval of most areas managed by prescribed fires in the Park. Each experimental fire was undertaken in areas of at least one hectare (Fig. 5) Fire lines in LF and High-EF treatments were pulled downwind to forward-burn, while evening Low-EF lines were pulled upwind to back-burn, since these are the way prescribed fires are implemented by the PA managers.

Table 1. Number and characteristics of the 31 experimental fires carried out in open savanna areas, located in the CMNP, in 2015 and 2017. LF = high-intensity, LDS mid-day fires (September); High-EF = high-intensity, EDS mid-day fires (May); and Low EF = low-intensity, EDS evening fires (May). Each experimental fire was at least one hectare and experimental areas separated by at least 100m.

Fire type/ treatments	Time since fire (years)	Number of fires 2015	Number of fires 2017
Low-EF	two	4	4
Low-EF	three	4	0
High-EF	two	4	4
High-EF	three	4	0
LF	two	4	3*
Total	-	20	11

*One area was burned by a wildfire before we carried the experimental fire in 2017

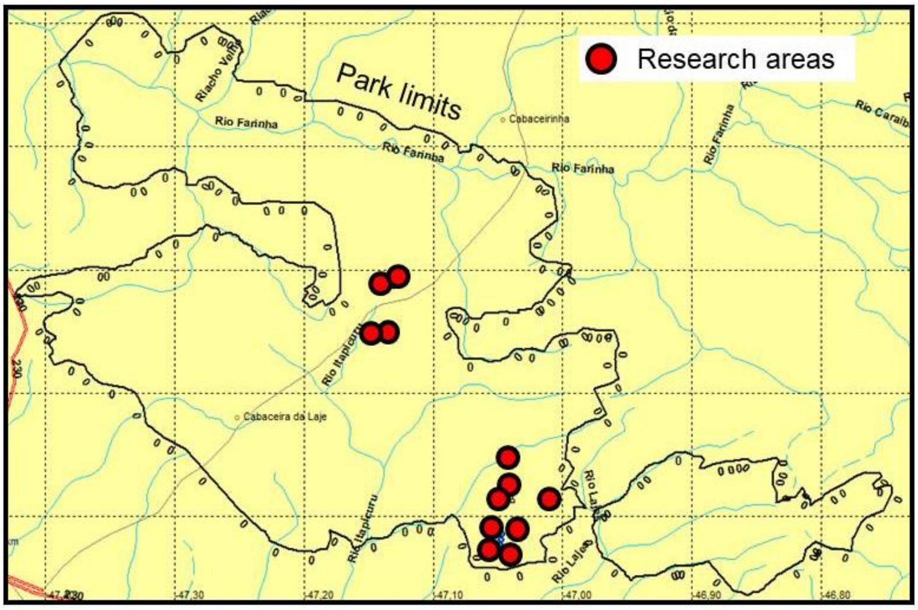

Fig. 3. Limits of the Chapada das Mesas National Park in Maranhão, Brazil with the 12 study areas where we carried out 31 experimental biennial late-dry season fires, as well as biennial and triennial early-dry season fires, under high (>50%) and low (<50%) air humidity in 2015 and 2017.

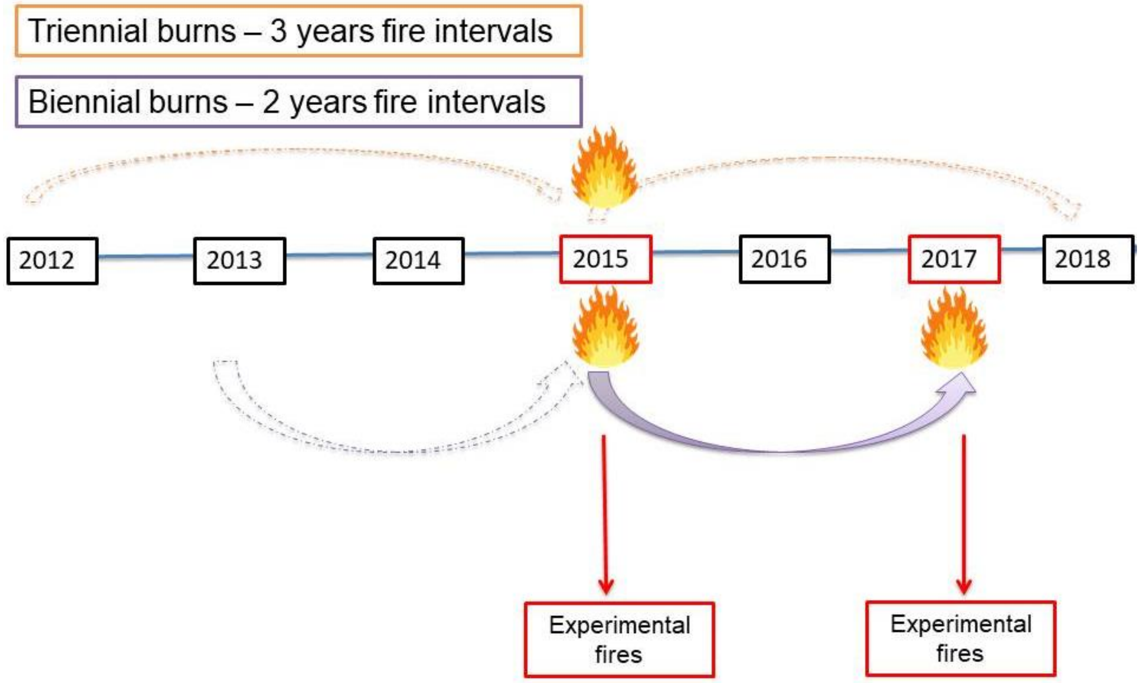

Fig. 4. Timeline showing fire history among the research areas and experimental fires we conducted in CMNP.

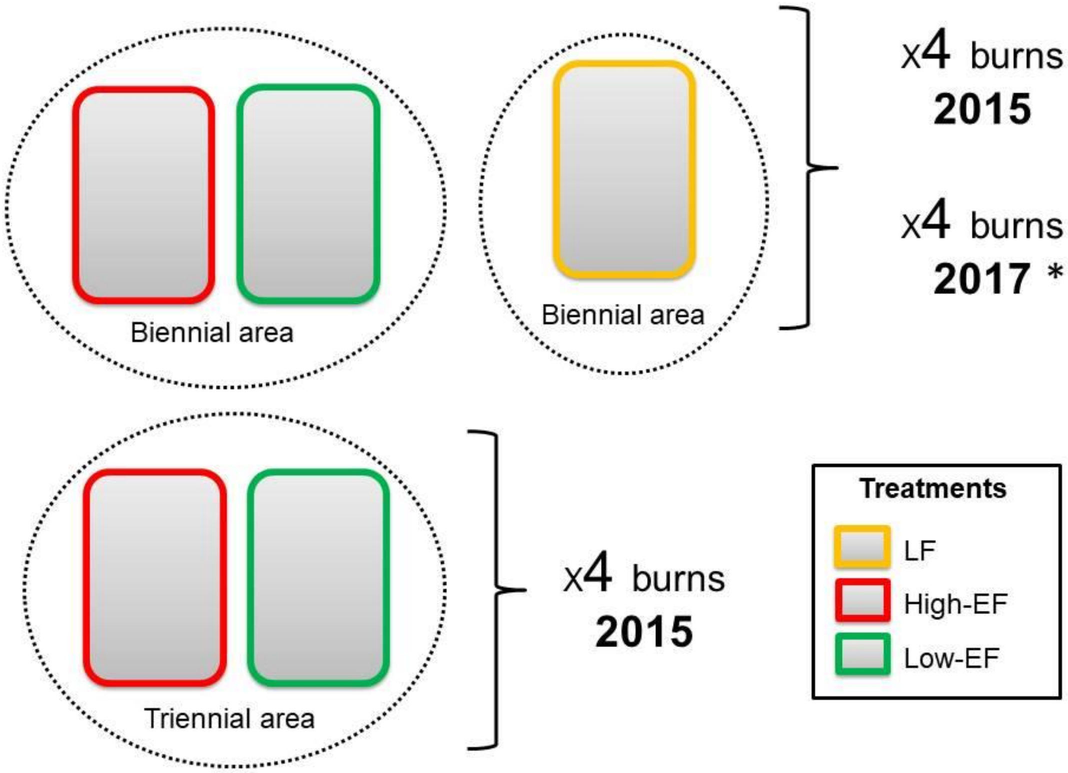

Fig. 5. Experimental design with fire treatments, time since fire, number of fires and burned years in the CMNP. Where: LF = LDS mid-day fires (September); High-EF = high-intensity EDS mid-day fires (May); and Low EF = low-intensity, EDS evening fires (May). *One of the LF experimental areas was hit by a wildfire in 2017 therefore experimental burn did not happen in this area.

2.3 Fire measurements and environmental conditions

For each experimental fire we recorded: weather conditions (air temperature, relative humidity and wind speed), rate of fire front spread and fuel consumption to calculate fire intensity. We calculated fire intensity and heat released according to the following equations (Byram, 1959): $I = h \times w \times r$ and $H = I / r$, where I = fire intensity ($kW\ m^{-1}$), h = heat yield constant for fine fuels ($kJ\ kg^{-1}$), w = weight of fuel consumed per unit area ($kg\ m^{-2}$), r = rate of fire spread ($m\ s^{-1}$) and H = heat released per unit area ($kW\ m^{-2}$). We considered heat yield from fine fuels as 15,500 $kJ\ kg^{-1}$ (Griffin & Friedel, 1984; Miranda et al., 2010; Pivello et al., 2010; Schmidt et al., 2017). We calculated fine fuel consumption by collecting all fine fuels (<6 mm in diameter) from the ground, considering the mean quantity of fuel within 12 plots (0.5 x 0.5 m) randomly allocated in each area, before and after the fires. The samples of fine fuel were dried at temperature between 60-70°C for three days before being weighted. The rate of spread was estimated visually, using stems, rocks or wood sticks as references. Air temperature and air humidity were measured using a portable weather station (Lutron EM-9000) to ensure that the weather conditions attended the prerequisite established for each treatment. For all experimental fires, the fine fuel consumption was calculated from the proportion: $(wi - wf)/wi$, where *wi* is the pre-fire, fine biomass (fuel load) and *wf* is post-fire, fine biomass. To calculate annual precipitation we used the online dataset from the Brazilian National Institute of Meteorology (INMET, from the meteorological station in the city of Carolina – state of Maranhão, which was ~100 km from the experimental areas), where we accounted for all rainfall from August in the previous year until July in the burned year. We considered the same precipitation in LF (burned in September) and EDS (burned in May), even though in May it had not completed a full year of rainfall in our annual estimate (August-July)in both study years (2015 and 2017) the precipitation in June and July was below 32 mm.

2.4 Statistical analysis

To test if fire intensity, fuel consumption and heat release (response variables) were different within biennial burning treatments and years (2015 and 2017) we applied linear mixed-effects models (Zurr et al., 2007), considering treatments and years as fixed effect variables and research areas as random effect variable. In this analysis LF, High-EF and Low-EF treatments from biennial fire frequency areas were included in the model, as well as both burned years (2015 and 2017), and our sample unit was 23 fires. We also tested if fire intensity, fuel consumption and heat released (response variables) were different between biennial and triennial treatments using linear mixed-effects models, where TSF (biennial and triennial fire frequencies) and High-EF and Low-EF treatments were considered the fixed effect variables, and area the random effect variable. From the results of these models we ran analysis of variance (ANOVA) and a posteriori test (Tukey). We did not include LF in the comparison between biennial and triennial TSF, because there was no equivalent treatment to compare it to within triennial areas. In this second analysis our sample unit was 16 fires and we only accounted for fires undertaken in 2015, when we burned both TSF areas. To fit data to normal distribution we transformed fire intensity in logarithm and fuel consumption in the inverse logistic, given by the formula: *log[p/(1 - p)]* , where p is the proportion value between 0 and 1 (Crawley, 2002). We considered significantly different when p value was below 0.05. The results are present with mean and standard error (±) values.

We built a correlation matrix to choose the parameters that most influenced significant differences in fire intensity, fuel consumption and heat released between the biennial treatments (LF, High-EF and Low-EF) and burned years (2015 and 2017), and then between biennial and triennial treatments (High-EF and Low-EF) in 2015 to avoid multicollinearity. Thus, the following parameters were tested for correlation: air humidity, air temperature, wind speed, fuel load, annual precipitation, fire treatment and TSF. Because fuel load, annual precipitation and TSF showed correlation between each other (>0.6, where values closer to 1 or -1 show more correlation between variables and values closer to 0 show less correlation), as well as air temperature, relative humidity, wind speed and treatment (>0.7 and < -0.68), we chose to use fuel load and relative humidity as the representative parameters. To evaluate if fuel load and relative humidity influenced differences in fire intensity, fuel consumption and heat released between only biennial fire frequency areas and then between biennial and triennial areas we ran linear mixed-effects models, considering the parameters our fixed effect variables and area the random effect variable. We used the R program version 3.5.0 (R-Core-Team, 2017) to run all analyses and produce all our graphics.

3. Results

3.2 Fire intensity

In 2015, there were significant differences among fires carried out in different fire intervals (two and three years, ANOVA, F=7, p=0.02) and treatments (F=8, p=0.01). The most intense fires were registered in triennial High-EF (4,400 ± 942 kWm^{-1}; Table 2), when the highest fire rate of spread (0.57 ± 0.11 $m\ s^{-1}$) and fuel load (0.582 ± 0.02 $kg\ m^{-2}$) were accounted in 2015 (Table 3). The intensity of these burnings was similar to the biennial High-EF (1,570 ± 456 $kW\ m^{-1}$, teste a posteriori p=0.2) and triennial Low-EF (1,339 ± 294 $kW\ m^{-1}$, p=0.2) and significantly more intense than the biennial Low-EF (640 ± 223 $kW\ m^{-1}$, p=0.01), when the lowest fire rate of spread (0.11 ± 0.04 $m\ s^{-1}$) and air temperature (28.5 ± 0.12 ºC), and highest relative humidity (64.2 ± 0.14 %) were registered in 2015 . The differences in fire intensity were best explained by relative humidity (t value=-3.87, p=0.0001) than by fuel load (t value=1.42 p=0.15; Fig. 6). Both treatments in triennial areas had higher fire intensities than the corresponding treatments in biennial areas. In triennial and biennial fires, Low-EF tended to be less intense than High-EF.

Table 2. Fire intensity, fuel consumption and heat released for EDS experimental fireswithin biennial and triennial fire frequency areas. Where: TSF = time since fire; High-EF = high-intensity, EDS mid-day fires (May); and Low-EF = low-intensity, EDS evening fires (May). Values represent the mean and ± standard error and lower case letters show significant difference ($p \leq 0.05$), according to a posteriori contrasts between fire treatment and TSF.

Treatment	TSF	Year	Numeber of fires	Fire intensity ($kW\ m^{-1}$)	Fine fuel consumption (%)	Heat released (kWm^{-2})
High EF	two	2015	4	1,570 ± 456 [ab]	81.5 ± 3.75 [a]	5,619±780 [a]
Low EF	two	2015	4	640 ± 223 [a]	87 ± 2.3 [a]	6,200±305 [a]
High EF	three	2015	4	4,400 ± 942 [b]	84.5 ± 2.8 [a]	7,634±348 [a]
Low EF	three	2015	4	1,339 ± 294 [ab]	85.5 ± 2.45 [a]	7,440±475 [a]

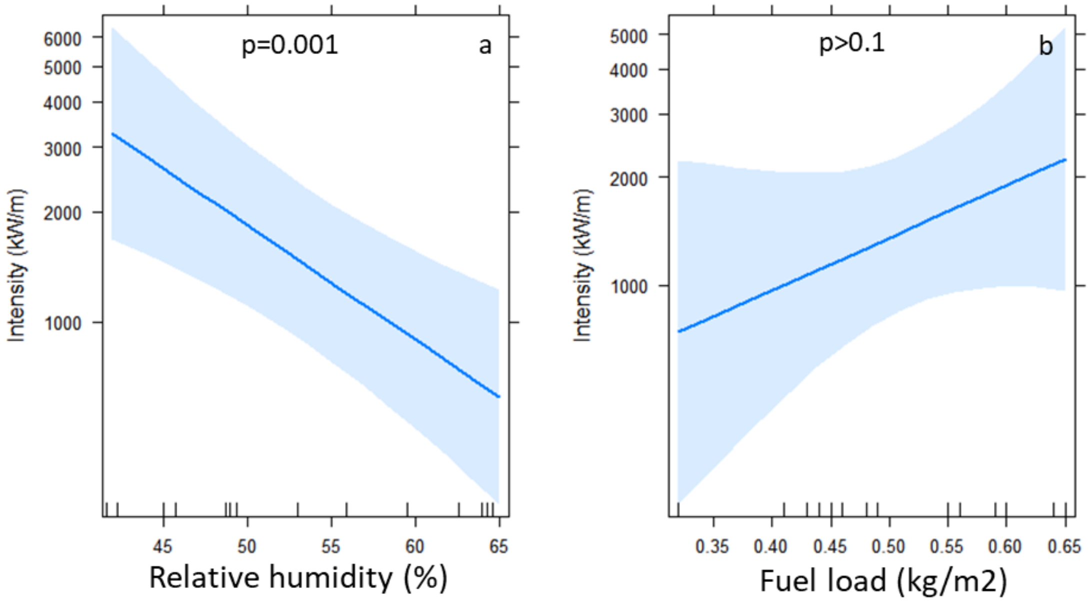

Fig. 6. Fire intensity in relation (a) to relative humidity and (b) fuel load, considering biennial and triennial, and High and Low-EF fires in 2015. Where p values were added on top of each parameter, given by the linear mixed-effects model.

Table 3. Weather conditions and fire behaviour in 2015 and 2017 for each treatment and fire frequency. TSF=time since fire; LF = high-intensity, LDS mid-day fires (September); High-EF = high-intensity, EDS mid-day fires (May); and Low-EF = low-intensity, EDS evening fires (May). Values represent the mean and ± standard error.

Fire treatment	TSF	Year	Air temperature (°C)	Air relative humidity (%)	Wind speed ($m\ s^{-1}$)	Annual precipitation (mm)	Fuel load ($kg\ m^{-2}$)	Fire rate of spread ($m\ s^{-1}$)	Fire intensity ($kW\ m^{-1}$)	Fine fuel consumption (%)	Heat released ($kW\ m^{-2}$)
LF	two	2015	35.6 ± 0.38	19.1 ± 0.65	1.7 ± 0.12	1,466	0.37 ± 0.01	0.53 ± 0.12	2,833 ± 721	88.5 ± 1.6	5,076±259
High-EF	two	2015	34.2 ± 0.62	48 ± 0.89	1.8 ± 0.4	1,466	0.437 ± 0.05	0.26 ± 0.07	1,570 ± 456	81.5 ± 3.75	5,619±780
Low-EF	two	2015	28.5 ± 0.12	64.2 ± 0.14	0.7 ± 0.26	1,466	0.46 ± 0.01	0.11 ± 0.04	640 ± 223	87 ± 2.3	6,200±305
High-EF	three	2015	34.1 ± 0.2	44.7 ± 1.55	1.5 ± 0.25	1,466	0.582 ± 0.02	0.57 ± 0.11	4,400 ± 942	84.5 ± 2.8	7,634±348
Low-EF	three	2015	29.9 ± 0.31	57.8 ± 1.81	0.5 ± 0.14	1,466	0.565 ± 0.04	0.18 ± 0.05	1,339 ± 294	85.5 ± 2.45	7,440±475
LF	two	2017	36.8 ± 0.27	29.2 ± 0.47	2.4 ± 0.05	1,395	0.357 ± 0.02	0.75 ± 0.14	3,717 ± 754	89.3 ± 1.91	4,840±297
High-EF	two	2017	33.8 ± 0.37	46.1 ± 0.57	1 ± 0.18	1,395	0.374 ± 0.02	0.33 ± 0.03	1,570 ± 141	80.7 ± 2.65	4,702±173
Low-EF	two	2017	30.2 ± 1.13	65.5 ± 2.24	0.6 ± 0.04	1,395	0.372 ± 0.02	0.22 ± 0.04	749 ± 220	54.5 ± 1.1	3,068±546

Within biennial fires, the LF in 2017 presented the highest fire intensity (3,717 ± 754 kW m^{-1}), rate of spread (0.75 ± 0.14 m s^{-1}), air temperature (36.8 ± 0.28 ºC) and wind speed (2.4 ± 0.05 m s^{-1}), and the lowest fuel load (0.357 ± 0.02 kg m^{-2}). Annual precipitation in 2016-2017 (1,395 mm) was lower than 2015 (1,466 mm, Tables 3 and 4). These LF were significantly more intense than Low-EF fires in 2015 (640 ± 223 kWm^{-1}, p=0.02), which presented the lowest fire intensity, rate of spread (0.11 ± 0.04 m s^{-1}) and air temperature (28.5 ± 0.12 ºC) and the highest fuel load (0.460 ± 0.01 kg m^{-2}). This difference was best explained by relative humidity (t value=-3.32, p=0.003) than by fuel load (t value=-0.34, p=0.7; Fig. 7). Fire intensity tended to decrease from Low-EF to LF, and High-EF was the intermediate intensity in both burned years. Differences between fire intensities in 2015 and 2017 were not significant within the same treatment (F=0.69, p=0.40), nor between treatments in the same year (F=0.02, p=0.97).

Table 4. Fire intensity, fuel consumption and heat released for each treatment within biennial fire frequency areas. Where: LF = high-intensity, LDS mid-day fires (September); High-EF = high-intensity, EDS mid-day fires (May); and Low-EF = low-intensity, EDS evening fires (May). Values represent the mean and ± standard error and lower case letters show significant difference (p ≤ 0.05), according to a posteriori contrasts between fire treatment and burned year.

Treatment	Time since fire	Year	Number of fires	Fire intensity (kW m^{-1})	Fine fuel consumption (%)	Heat released (kW m^{-2})
LF	two	2015	4	2,833 ± 721 ^{ab}	88.5 ± 1.6 ^{a}	5,076±259 ^{ab}
High EF	two	2015	4	1,570 ± 456 ^{ab}	81.5 ± 3.75 ^{a}	5,619±780 ^{a}
Low EF	two	2015	4	640 ± 223 ^{a}	87 ± 2.3 ^{a}	6,200±305 ^{a}
LF	two	2017	3	3,717 ± 754 ^{b}	89.3 ± 1.91 ^{a}	4,840±297 ^{ab}
High EF	two	2017	4	1,570 ± 141 ^{ab}	80.7 ± 2.65 ^{a}	4,702±173 ^{ab}
Low EF	two	2017	4	749 ± 220 ^{ab}	54.5 ± 1.1 ^{b}	3,068±546 ^{b}

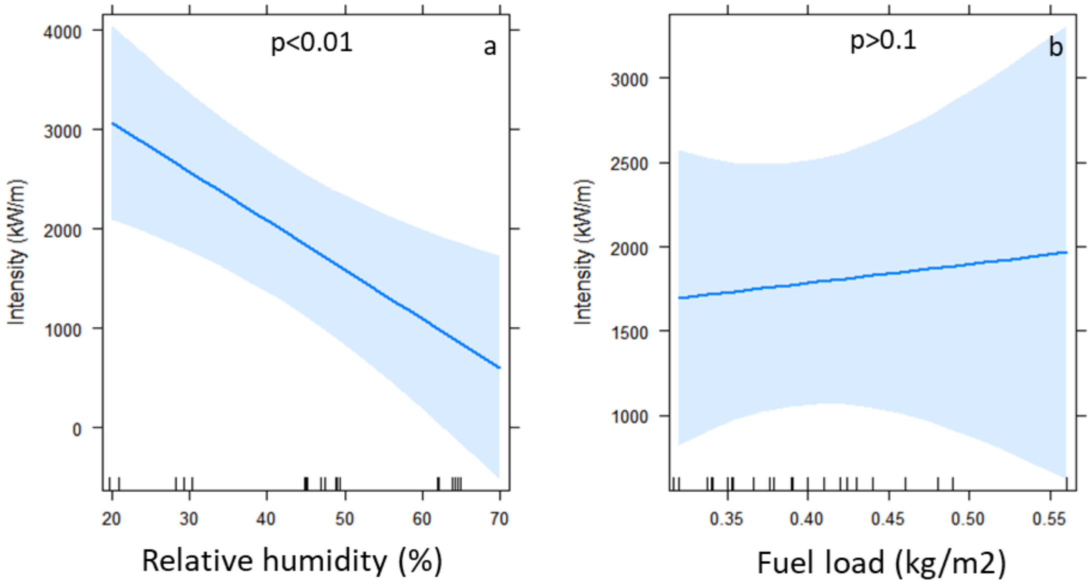

Fig. 7. (a) Fire intensity in function of relative humidity and (b) fire intensity in function of fuel load considering biennial and LF, High and Low-EF treatments in 2015 and 2017. Where p values were added on top of each parameter, given by the linear mixed-effects model.

3.2 Fine fuel consumption and heat released

Considering EDS experimental fires, there were no significant differences in fine fuel consumption and heat released between treatments (High-EF and Low-EF) and TSF (biennial and triennial) in 2015 (fuel consumption – F=1.32, p=0.2; heat released – F=0.49, p=0.5; Table 2). In 2015, the biennial Low-EF were carried out under the highest air relative humidity (64.2 ± 0.14 %) and lowest air temperatures (28.5 ± 0.12 ºC) and resulted in the largest proportion of fine fuel consumption (87 ± 2.3 %), and lower fire rate of spread (0.11 ± 0.04 m s^{-1}). The biennial High-EF were carried out under the highest wind speed (1.8 ± 0.4 m s^{-1}) and highest air temperature (34.2 ± 0.62 ºC), and presented the lowest amount of fuel load (0.437 ± 0.05 kg m^2), which resulted in the lowest proportion of fine fuel consumption (81.5 ± 3.75 %) and the lowest heat released (5,619 ± 780 kW m^{-2}).. The triennial High-EF areas had the highest fuel load (0.582 ± 0.02 kg m^{-2}) and presented the highest fire rate of spread (0.57 ± 0.11 m s^{-1}), heat released (7,634 ± 348 kW m^{-2}) and fire intensity (4,400 ± 942 kW m^{-1}).

As for the biennial fires in 2015 and 2017, the LF consumed the highest proportion of fine fuel in 2017 (89.3 ± 1.91 %, Fig. 8a, b) and in 2015 (88.5 ± 1.6 %), whereas the biennial Low-EF in 2017 consumed the lowest proportion (54.5 ± 1.1 %;

Table 3, Fig. 8c, d) , The consumption of fine fuel in the Low-EF in 2017 varied significantly from the other treatments (F=4.66, p=0.03) and year (F=9.68, p=0.008) , including measurements from the same treatment in 2015 (p = 0.006). Relative humidity (likelihood ratio test – LRT=1.73, p=0.1) and fuel load (LRT=0.03, p=0.8) did not explain the variations in fuel consumed. In 2016-2017, annual precipitation (1,395 mm) was lower than in 2014-2015 (1,466 mm), accordingly, fuel load in 2017 was lower in the Low-EF experimental areas (0.372 ± 0.02 kg m^{-2}) compared to 2015 (0.460 ± 0.01 kg m^{-2}), when fuel consumption was much higher (87 ± 2.3 %) for this experimental treatment.

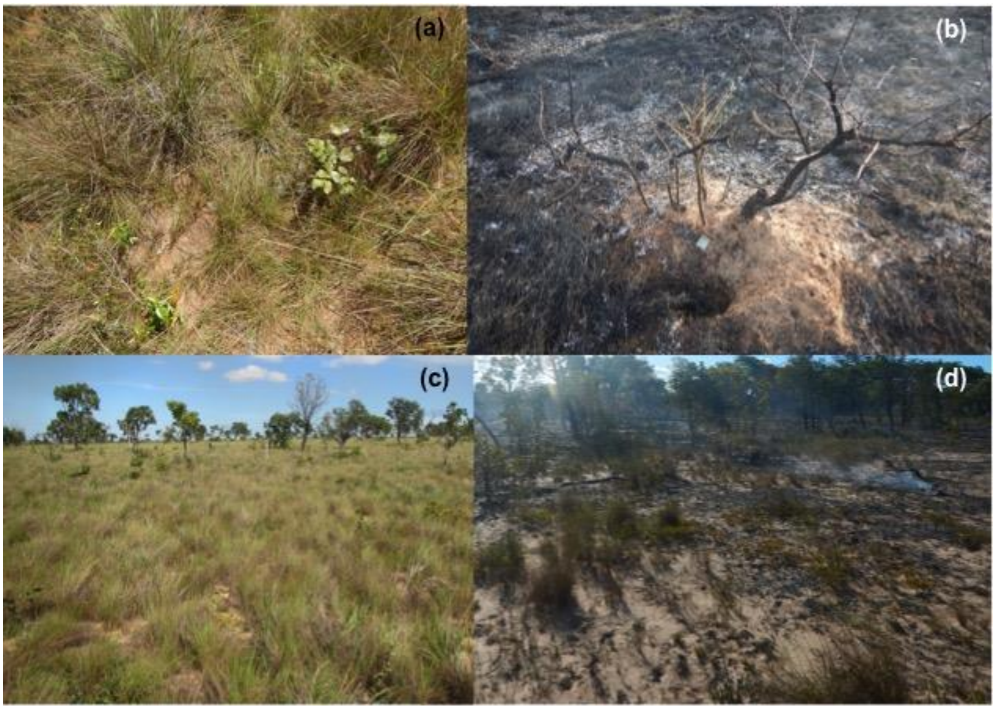

Fig. 8. (a) Pre-fire biomass in a LDS treatment; (b) Post-fire biomass in a LDS treatment; (c) Pre-fire biomass in a biennial, EDS evening treatment; and (d) Post-fire biomass in a biennial, EDS evening treatment. All pictures were taken in the survey of 2017.

The Low-EF treatment in 2015 presented the highest heat released within the biennial fires in 2015 and 2017 (6200 ± 305 kW m^{-2}) and Low-EF in 2017 presented the lowest (3068 ± 546 kW m^{-2}; Table 3). The heat released in the Low-EF in 2017 varied significantly from Low-EF and High-EF in 2015 (p=0.0004 and p=0.002, respectively). Relative humidity (t value=-2.8, p=0.01) and fuel load (t value=5.6, p<0.0001) explained the variations in heat released (Fig. 9). In 2017, the Low-EF was carried out in the highest air relative humidity (65.5 ± 2.24 %), one of the lowest air temperature

(30.2 ± 1.13 °C) and wind speed (0.6 ± 0.04 m s^{-1}) when compared to the other treatments. Such mild weather conditions at the time of burning, as well as the longer rainy season in 2017 have probably contributed to the low fire intensity, fine fuel consumption and heat released in the Low-EF experimental treatment.

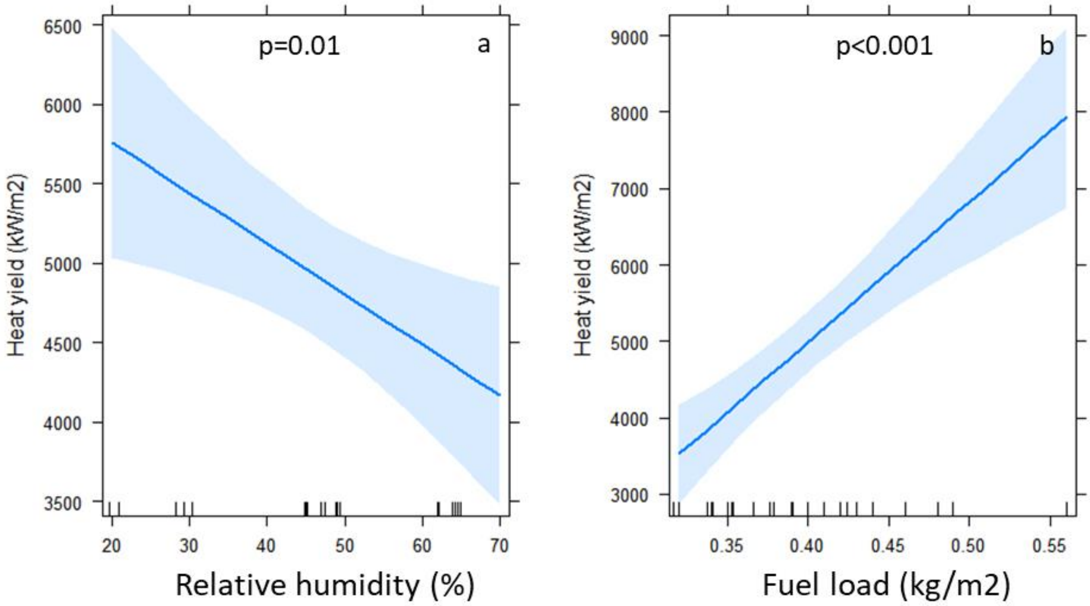

Fig. 9. Heat released in function of (a) relative humidity and (b) fuel load, considering biennial and LF, High and Low-EF treatments in 2015 and 2017. Where p values were added on top of each parameter, given by the linear mixed-effects model.

4. Discussion

4.1 Season, weather and fire interval

Many studies related to fire behaviour in savanna ecosystems focus on the influence of seasonality and frequency on fire intensity (Gomes et al., 2018; Govender et al., 2006; Miranda et al., 2010; Rissi et al., 2017; Schmidt et al., 2017; Williams et al., 1998).. Our results show that EDS fires with air relative humidity below 50%, have similar fire intensities to LDS fires, but they are both different from EDS fires with relative humidity above 50%. This pattern shows that fire intensity is mostly related to weather variations, such as air temperature, air relative humidity and wind speed, which may vary along time of the day and season of the year, as shown by our experimental records.

Other experiments in a *campo sujo* (open Cerrado physiognomy, dominated by grasslands and some spaced trees) in Central Brazil, based on fire seasonality - early, middle and late dry season - registered average fire intensities of 503 ± 119, 2,707 ± 1,899 and 3,009 ± 1,408 kW m^{-1} (mean ± standard error) , respectively, which were not found significantly different from each other (Rissi et al., 2017). Higher fire intensities are commonly reached in the LDS (August-October) in open Cerrado physiognomies, in central Brazil from 3,009 to 16,000 kW m^{-1} , and lower in denser Cerrado physiognomies 2,062 to 14,396 kW m^{-1} (Kauffman et al., 1994; Rocha-Silva, 1999; Sato, 2003; Sato & Miranda, 1996; Ward et al., 1992). Fire intensities in wet grasslands in central Brazil did not show significant differences between LDS and EDS fires varying from 240 to 1083 kW m^{-1} (Schmidt et al., 2017), where fine fuel load varied from 0.43 to 1.3 kg m^{-2}. In an open grassland physiognomy in southern Brazil ~2.2 kg m^{-2} of fine fuel was accounted before burning, where the mean relative water content was ~50%, and fire intensities in the LDS were below 900 kW m^{-1} (Seger et al., 2013). In northern Australian savanna fire intensities in the LDS varied from 2,700 to 7,700 $kWm^{-}1$ (Russell-Smith et al., 2003; Williams et al., 1998) and in southern African savanna from 2,566 to 12,912 kWm^{-1} (Govender et al., 2006; Trollope & Trollope, 2002), where the vegetation composition and flammability can be notably different from Cerrado physiognomies. However, the period of the year alone is not a reliable predictor for fire intensity (Bowman et al., 1988; Gomes et al., 2018; Russell-Smith & Edwards, 2006), especially because of its variation within types of fuel, fuel moisture, fine fuel load (Fidelis et al., 2010) and dead fuel percentage (Rissi et al., 2017; Trollope & Trollope, 2002), and more generally air temperature, relative humidity, wind and slope (Brown & Davis, 1973; Cheney & Sullivan, 2008; Luke & McArthur, 1978; Trollope & Trollope, 2002; Whelan, 1995; Wright & Bailey, 1982). Our study showed that fire season is not the only determinant of fire intensity, and that and weather conditions are important predictors. TheTSF is directly associated with fuel age, dead fuel percentage and fuel load (Batmanian & Haridasan, 1985; Gill et al., 1996; Wagner, 1978), influencing fuel consumption (Castro & Kauffman, 1998; Kauffman et al., 1994; Ward et al., 1992). While weather conditions determine the air relative humidity and fuel moisture of alive plants, air temperature and wind speed may influence fire rate of spread (Cheney & Sullivan, 2008; Luke & McArthur, 1978; Rothermel, 1983) and combustion rate (Brown & Davis, 1973). However, in our experiment fuel load did not explain differences in fire intensity, only relative humidity did.

Fine fuel consumption and heat released are other parameters used for characterizing fire behaviour, which give valuable information about the proportion of biomass consumed by fire and how much energy is being spent to consume it. From our experiment, low-intensity, EDS fires with relative humidity above 50% in a lower rainfall year (2017) registered lower fuel consumption (54.5%) and heat released (3,068 kW m^{-2}) compared to the same conditions in a higher rainfall year (2015). Aboveground fuel consumption varies between 54 and 97% in savanna physiognomies in central Brazil, under different fuel types, weather conditions and annual precipitations (Castro & Kauffman, 1998; Gomes et al., 2018; Kauffman et al., 1994; Miranda et al., 2010; Seger et al., 2013). In the LDS in graminous physiognomies, fuel consumption varied from 97 to 100%, whereas in typical Cerrado the consumption varied from 72 to 84% (Ward et al., 1992). Although fuel load was higher than 2.2 kg m^{-2} in a grassland area in southern Brazil, fuel consumption was 77% and heat released was ~17,000 kW m^{-2}, due to the high relative water content in fuel load (Seger et al., 2013). In a typical Cerrado in central Brazil, heat released tended to be higher in the EDS reaching 30,000 kW m^{-2} and lower in the LDS varying from 13,000 to 22,000 kW m^{-2} (Sato, 2003). In northern Australian savanna, 74% of the fine fuel is consumed within EDS fires and 86% in LDS fires (Russell-Smith et al., 2009a), and, accordingly, in southern African savanna between 88-95% is consumed in the LDS (Shea et al., 1996; Ward et al., 1996). Most fuel consumed is part of the graminous herbaceous stratum , which is influenced by variations in microclimate features to burn (Beerling & Osborne, 2006; Hoffmann et al., 2012; Miranda et al., 2002).

Even though total annual precipitation was lower in the period from August of 2016 to July of 2017 than in 2014-2015, and below the average annual precipitation for the region, the rainy season extended longer in 2017 and consequently, Low-EF were carried out in milder conditions in 2017 compared to 2015. These milder weather conditions (lower temperatures and higher relative humidity) at the time of burning, resulted in lower fuel consumption in the EDS evening fires for that calendar year. Fuel moisture is correlated to weather features, such as relative humidity, temperature, precipitation and wind speed (Hoffmann et al., 2012; Miranda et al., 2002). Previous investigations show that these characteristics influence fire behaviour in different fire-prone vegetation, such as the Cerrado (Castro & Kauffman, 1998; Miranda et al., 2002; Rissi et al., 2017), Australian savanna (Cheney & Sullivan, 2008; Gill et al., 1996), and African savanna (Govender et al., 2006; Savadogo, 2007). Dead fuel is especially influenced by local weather conditions that change along the day, e.g. in tropical savannas in Australia after 3 pm in the EDS air relative humidity is >60%, whereas in the LDS is <30% (Gill et al., 1996), with solar radiation reinforcing fuel drying process (Cheney & Sullivan, 2008), however fuel composition can change fuel availability to burn (Pausas & Moreira, 2012; Whelan, 1995). In our experimental areas, the EDS evening fires were lit under relative humidity higher than 53% and air temperature between 28 and31 °C; EDS mid-day fires relative humidity between 41.5 and 49 % and temperature between 33 and 36 °C; and LSD mid-day fires relative humidity under 30% and temperature above 35 °C.

Additionally, TSF is associated to progressive curing of grassy layers reaching the maximum dead fuel percentage two to three years after fire (Coutinho, 2002; Miranda et al., 2010). In fact, each year after fire the herbaceous layer can recover 70% of the total biomass it had before fire and can recover 100% two years after burned in Cerrado physiognomies (Batmanian & Haridasan, 1985; Neto et al., 1998). However, extreme fire weathers can override fuel characteristics, whereas in mild weather conditions fuel load and fuel moisture are usually the main drivers of fire spread (Moritz, 2003). In our experiment air relative humidity did not explain differences between heat released and fuel consumption, whereas fuel load only explained the significant differences between heat released within biennial fire frequencies.

4.2 Fire behaviour predictions for management systems

Wildfires are recurrent in the Cerrado PAs due to the combination of marked rainfall seasonality that favours biomass production during the wet season and dries this biomass along the dry season, increased anthropogenic source of ignition and unmanaged flammable fuels in large continuous areas(Barradas, 2017; Batista et al., 2018; França, 2010; França et al., 2007; Pereira Júnior et al., 2014; Pivello, 2006). In fire-prone environments, fire management should consider the natural evolution of fire adapted ecosystems, effect of weather, slope, fuel load, type of fuel, natural barriers, environmental valuation, socioeconomic sustainability and local communities' activities are acknowledged and are part of the planning and evaluation process.

As Cerrado PAs aim to conserve local biodiversity and landscapes, managing institutions are recently recognizing that using fire as a landscape management tool may helping achieving these goals, and prescribed fires are increasingly been applied within the IFM programme. In this context, EDS fires when relative humidity is above 50% can be applied to increase the safety of prescribed burnings in areas with high risks of fire spread, such as areas with high fuel load, with longer fire intervals (>3 years), since the mild weather conditions will help fires to extinguish themselves during the night (Gill et al., 1996; Schmidt et al., 2016; Williamson et al., 2016). These high humidity fires are less likely to run out of control and burn unwanted areas with fire-sensitive species, and at the same time they can work as firebreaks to avoid the spread of LDS wildfires at least for one year, since one year after burned up to 70-80% of the herbaceous biomass is recovered in Cerrado physiognomies (Andrade, 1998; Neto et al., 1998). Although at a small scale the fuel consumption might be low and not much efficient to stop wildfires from spreading, at the landscape scale, these EDS prescribed fires carried out in relatively large area (dozens or hundreds of hectares) have been efficient to avoid the spread of LDS wildfires in the managed PA (Schmidt et al. 2018).

The EDS fires with relative humidity below 50% can reach higher intensities and fuel consumption and, therefore, if compatible with management objectives can be implemented in strategic management zones, where biotic and abiotic allow for the safe use and control of such prescribed fires. In the study region, this corresponds to areas of open savanna physiognomies unburned for not more than two years. Prescribed EDS fires may help generating patch-mosaic burnings along the seasons that may promote heterogenic fire histories across the landscape and avoid wildfires in fire-sensitive areas. Considering weather forecast to carry out higher-intensity fires are indispensable, including measurements in the field just before burning takes place. Finally, LDS fires happen under the most hazardous fire weather and are usually avoided by managers, once they put at risk extensive areas with homogeneous and unmanaged flammable fuels (Krawchuk et al., 2009; Moritz et al., 2014; Yates et al., 2008). However, for natural, evolutionary environmental processes and related or unrelated management objectives, as well as for productivity reasons, LDS fires might be desirable and can be safely performed if adequate planning and EDS fires are undertaken to create barriers to fire spread (Johnson et al., 2001; Keeley, 2009; Laris et al., 2017; Stephenson et al., 1991). Systematic fire management, monitoring and mapping will likely reduce areas burned by wildfires (Price et al., 2012; Russell-Smith et al., 2013; van Wilgen & Biggs, 2011) and can eventually help creating landscape heterogeneity, including areas from which fire is excluded for longer periods of time(Bowman et al., 2016; Perry et al., 2011; van Wilgen et al., 2003).

Acknowledging these biotic and abiotic local and regional conditions can lower human and equipment resources required for fire management, as well as better achieve conservation goals (Dickinson & Ryan, 2010; Driscoll et al., 2010; Moritz et al., 2014; Price et al., 2012; Van Wilgen et al., 2007). However, long-term studies focused on the effects of the different fire regimes upon the biota will be much more complementary and qualified to inform and help evaluating the outcomes of the implementation of these management systems.

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Capítulo 3 - How Cerrado woody species respond to different fire management systems?

Como as espécies lenhosas do Cerrado respondem a diferentes sistemas de manejo do fogo?

Abstract

The savanna is one of the most fire-prone biome and the most extensively burnt on Earth. Accordingly, the dynamic of savanna vegetation, such as grasses and woody species interactions, is regulated and driven by fire regimes for millions of years. Systems subjected to frequent fires (annual-biennial) can homogenize the structure and regulate species composition of arboreal stratum, preventing regenerant individuals to escape fire trap and preventing the establishment of fire-sensitive species. In the Brazilian savanna (Cerrado region), attempts to exclude fire led to frequent (biennial-triennial), extensive, high-intensity LDS (August-September) wildfires or led to longer fire return intervals when the woody vegetation encroaches and the herbaceous stratum reduces, affecting especially protected areas (PA). Since 2014, EDS (May-July) prescribed fires are being implemented in some Cerrado PAs aiming to reduce the extent of wildfires to smaller fragments. However, changing fire regime to predominant EDS and uniform fire frequency can compromise some species life cycles and interrupt critical life stages. Hence, we aimed to compare woody stem responses – stem survival, topkill and recruitment, changes in stem density and basal area– in areas subjected to EDS fires under low (<50%) and high (>50%) relative humidity and LDS fires under biennial and triennial fire frequencies, as well as areas protected from fire for four and five years in CMNP. To assess how different fire regimes can affect the dynamics of woody species, we established 28 permanent plots with ~4,900 woody stems, where we characterized their responses (alive, topkilled, dead and recruitment) to fire and their diameter sizes from pre-fire condition (April, 2015) until two years after fire passage (May, 2017). All biennial EDS fires increased topkill and reduced survival of smaller stems (1-5 cm) in comparison to areas unburned for four years. LDS fires resulted in significantly less basal area growth when compared to areas protected from fire for four years and EDS fires with relative humidity above 50%. Our results indicate that EDS fires with relative humidity above 50% are likely to have the same impact on woody communities as the ones with relative humidity below 50%, especially within two years

fire intervals. Therefore, it is important to consider not only fire seasonality, but also the weather conditions and fire intervals as part of management plans to implement prescribed fires in Cerrado.

Keywords: fire seasonality, recruitment, savanna, topkill, weather conditions, woody density.

1. Introduction

The ecological patterns found in different landscapes are the result of the interaction between ecosystems and socio-temporal processes (Vellend, 2010). Disturbances, such as fire, combined with water and nutrient availability are important determinants of these ecological processes and influence the dynamics of populations, communities and ecosystems (Accatino et al., 2010; Bond, 2008; da Silva & Batalha, 2008; Silva et al., 2013). As a natural disturbance, fire has shaped landscapes in almost every continent for millions of years (Scott & Glasspool, 2006), where fire regime is responsible for major or minor ecological impacts. Fire regimes are established mainly by the combination of fire seasonality and frequency (Whelan, 1995); however, the influence of each parameter depends on local conditions and ecosystem features (Archibald et al., 2013; Bowman et al., 2009; Collins, 1992; Lehmann et al., 2014; Roos et al., 2014; Williamson et al., 2016).

The savanna is one of the most fire-prone biome and the most extensively burnt on Earth (Giglio et al., 2013; van der Werf et al., 2017). Accordingly, the dynamic of savanna vegetation, such as grasses and woody speciesinteractions (Scholes & Archer, 1997), is regulated and driven by fire regimes for millions of years (Bond & Scott, 2010; Dantas et al., 2013; Simon et al., 2009; Skarpe, 1992). Fire regime has proven to influence woody densities, biomass and fuel load, as well as species composition in tropical savannas (Bond, 2008; Bond et al., 2005; Higgins et al., 2007; Hoffmann & Moreira, 2002). The rate of survival, mortality, growth, recruitment and resprout are some of the short-term responses of woody species to fire (Hoffmann et al., 2009; Kavanagh et al., 2010; Medeiros & Miranda, 2005; Murphy et al., 2010; Ryan & Williams, 2011), and structure and composition traits are some of the long-term plant adaptations to fire (Dantas & Pausas, 2013; Keeley et al., 2011; Lawes et al., 2011;

Moreira et al., 2012; Pausas & Schwilk, 2012). Concurrently, fuel accumulation and woody biomass increase according to moisture and nutrient availability in different savannas (Bond et al., 2005; Lehmann et al., 2014; Sankaran et al., 2005). Systems subjected to frequent fires (annual-biennial) can homogenize the structure and regulate species composition of arboreal stratum , preventing regenerant individuals to escape fire trap and preventing the establishment of fire-sensitive species (Dantas et al., 2013; Grady & Hoffmann, 2012; Hoffmann et al., 2009; Werner & Prior, 2013; Williams et al., 1999). Whereas, grasses expand due to their faster regrowth rate competing with juvenile woody plants and increasing fine fuel load (Bond, 2008; Hoffmann, 1999; Scholes & Archer, 1997; Zimmermann et al., 2010). When frequent, fire hinders woody resprout by topkilling (complete loss of aboveground biomass) individuals in a critical range of stem diameter (Hoffmann & Solbrig, 2003) that prevent seedlings and saplings from escaping the grass stratum (Bond, 2008). Demographic parameters of woody plant individuals after fire events are also related to initial height in northern Australian (Werner & Prior, 2013) and Afrotropical savannas and bark-diameter correlation in Neotropical savannas (Dantas & Pausas, 2013).

The effects of fire on the vegetation also depends on fire behavior characterized by fire intensity, flame height, fuel quality and quantity, burn efficiency, heat released and weather conditions (Cheney & Sullivan, 2008; Kauffman et al., 1994; Miranda et al., 2010; Rothermel, 1983; Whelan, 1995). Seasonality has shown to affect fire intensity by changing fuel moisture content in southern African (Govender et al., 2006) and northern Australian savannas (Russell-Smith et al., 2003; Williams et al., 1998), where late dry season (LDS) fires in Eucalypt-dominated woodlands had significantly higher intensities than in Eucalypt-dominated open forests and early dry season (EDS) fires (Russell-Smith et al., 2003). In the central Cerrado region grassland physiognomies carry out more intense fires and higher combustion efficiency compared to woody vegetation (Castro & Kauffman, 1998; Miranda et al., 1996), which cause high rates of topkill by eliminating the aerial portion of woody species (Hoffmann et al., 2009; Kauffman et al., 1994). On the other hand, fire parameters (such as fire intensity) are not always significantly affected by fire season (EDS vs. LDS), being rather determined by the proportion of dead fuel and structure of fuel load in grassland and savanna vegetation in the Cerrado region (Castro & Kauffman, 1998; Kauffman et al., 1994; Rissi et al., 2017; Ward et al., 1996). Fire intensity has proven to affect stem survival showing a negative correlation among savanna woody species in eucalyptus savanna (Russell-Smith & Edwards, 2006; Ryan & Williams, 2011; Williams et al., 1999) and typical Cerrado vegetation (Sato, 2003; Sato et al., 1998). In contrast, there is a positive correlation between high intensity fires and tree regeneration in sequoia-mixed conifer forests (Bond & Keeley, 2005; Stephenson et al., 1991) and boreal forests (Johnson et al., 2001). Fire intensity is not a synonym of fire severity. Fire severity describes the implications of fire intensity in the ecosystem, where different fire behaviour in combination with other parameters, such as fire regime and environmental conditions, will have different effects on plant individuals (Keeley, 2009; Pérez & Moreno, 1998). Fires in the EDS (May-June) tend to result in lower intensity and severity (less damage to woody species) compared to LDS (September-October) fires in northern Australian savanna, where severity is classified from low to high according to leaf-scorch height, vegetation structure and burn patchiness (Russell-Smith & Edwards, 2006).

Fire management systems in some savanna regions, such as in Western Arnhem Land (Russell-Smith et al., 2013b) and southern Africa (Goldammer & de Ronde, 2004; Moore et al., 2002; Van Wilgen et al., 2014), have driven fire regimes, where the frequency and seasonality of fires are manipulated by different management policies. Many fire-prone environments have experienced fire suppression policies, commonly creating hazardous flammable surfaces with fuel accumulation that increased wildfire incidence (Bond & Parr, 2010; Steel et al., 2015; Varner et al., 2005), or, when soil conditions and water supplies allow for, promoting woody encroachment that reduced fire frequency (Bond et al., 2005; Honda & Durigan, 2016; O'Connor et al., 2014; Ryan et al., 2013). In the Brazilian savanna (Cerrado region), attempts to exclude fire lead to frequent (biennial-triennial), extensive, high-intensity LDS (August-September) wildfires (Batista et al., 2018; Pereira Júnior et al., 2014; Pivello, 2011) or lead to longer fire return intervals when the woody vegetation encroaches and the herbaceous stratum reduces(Abreu et al., 2017; Klink & Machado, 2005; Stevens et al., 2017), affecting especially protected areas (PA). Nevertheless, successful savanna fire management approaches have inspired the integrated fire management (IFM) pilot programme initially established in three Cerrado PAs since 2014. The implementation of EDS (May-July) prescribed fires is part of the programme, and aims to reduce the extent of wildfires to smaller fragments, protect fire sensitive ecosystems by burning adjacent fire adapted vegetation and create patchy, low-intensity, EDS fire regimes (Schmidt et al., 2016, 2018). Accordingly, these prescribed fires are being tested for feasibility, meeting conservation goals and at the same time the PA's local specificities. Positive outcomes, such as 40-57% reduction in LDS fires in the first three years, already have triggered the implementation of the IFM programme in other eight PAs and 11 Indigenous Territories, and in the next few years it is likely to expand to all Cerrado PAs (Schmidt et al., 2018). However, changing fire regime to predominant EDS and uniform fire frequency can compromise some species life cycles and interrupt critical life stages (Baudena et al., 2010; Kirkman et al., 1998), and sometimes lead local populations to extinction (Armstrong & Phillips, 2012; Werner & Prior, 2013; Whelan et al., 2002). Additionally, low-intensity fires usually have lower rate of spread that might result in higher fire residence time and heat released when compared to high-intensity fires, therefore, can increase woody species mortality or structure damages (Guedes, 1993; Miranda et al., 1993, 2010; Rothermel & Deeming, 1980).

As a flexible management strategy to better guide and inform decision-making processes, the IFM programme has been scientifically monitored in Chapada das Mesas National Park (CMNP) since 2014 to provide ecological feedbacks and better achieve management goals, following examples from Kruger and Kakadu National Parks (Edwards et al., 2003; Parr et al., 2009; van Wilgen & Biggs, 2011). We have established permanent plots in CMNP to assess potential changes in the structure of open savanna physiognomy as a result of different fire management systems. Previous experiments in Cerrado physiognomies show higher woody mortality and topkill and reduced gain in basal area among woody species under frequent (some testing biennial and others quadrennial fire frequencies) LDS fires than in EDS ones (Medeiros & Miranda, 2005; Pivello & Norton, 1996; Ramos, 1990; Rocha-Silva, 1999; Sato, 2003; Sato et al., 1998). Other experiments show that only shorter fire interval (two to three years) results in higher woody mortality (Garda, 2018; Ribeiro et al., 2012) or favour phenological (Rissi, 2016; Schmidt et al., 2005) and specific reproductive groups (Hoffmann, 1998). Although, woody species responses to fire seasonality (early, mid and late) and frequency have been well documented within the Cerrado (Arruda et al., 2018; Gomes et al., 2018; Miranda, 2010), different fire weather conditions under the same season have not yet been reported.

Thus, we aimed to compare woody stem responses – stem survival, topkill and recruitment, changes in stem density and basal area– in areas subjected to EDS fires under low (<50%) and high (>50%) relative humidity, LDS fires under low (<50%) relative humidity under biennial and triennial fire frequencies, as well as areas protected from fire for four and five years in CMNP. We hypothesize that areas protected from fire for longer periods would present higher rate of stem survival and recruitment, lower topkill, and positive and higher changes in stem density and growth. Among the areas subjected to LDS fires we predict lower rate of stem survival and recruitment and higher topkill compared to EDS fires. Among all experimental fires we predict losses in stem density and basal area in relation to pre-fire measurements, where these losses are lower in areas subjected to longer fire interval (triennial fire frequency) when compared to shorter interval (biennial fire frequency).

2. Material and method

2.1 Study areas

The CMNP was one of the first PAs to conduct prescribed burns for management purposes within the IFM pilot program. The park is located in northern Cerrado, northeast of Brazil in the state of Maranhão (7°19'00"S and 47°20'06"W, Fig 1). It was created in 2005 with 160,000ha, aiming to conserve the natural landscape - including its distinctive large vegetation range, water sources and characteristic mesa geological formations - and local biodiversity (Ab'Sáber, 2003; Castro & Martins, 1999; da Silva et al., 2017; ICMBio, 2018; Medeiros et al., 2008).

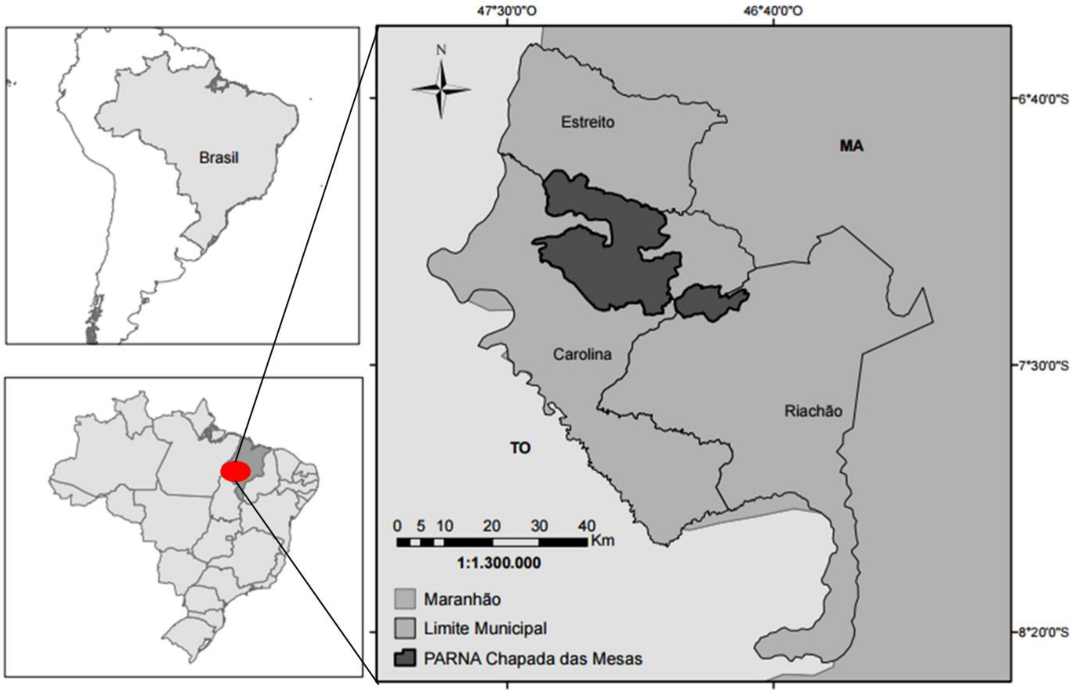

Fig. 1. Map with the location of Chapada das Mesas National Park in regional, national and continental scale, where the experiment for this study took place. Font: Victor Ferreira, 2015.

Most of CMNP is covered by well-drained sandy soils – yellow latosol (Marques, 2012)- and open savanna vegetation. The park is located in a transition zone between savanna, Amazonian tropical forest and Caatinga vegetation formations. Within the region, fire-sensitive (such as swampy forest, riparian and gallery forests) and fire-adapted vegetation (e.g. grasslands and savannas) types occur in mosaics (Aquino et al., 2007; Martins & Oliveira, 2011; Medeiros et al., 2008). The region receives annually from 1,500 to 1,900 mm of rainfall, with variations in mean annual air temperature from 25 to 31°C and relative humidity (RH) from 44 to 83% over the last ten years (INMET, 2017). The rainy season is concentrated between October and April, receiving from 1,300 to 1,600 mm rainfall, while the dry season (May-September) usually receives less than 200 mm (INMET, 2017). In this high rainfall tropical region, open savanna areas produce ~3.16 ± 0.06 t/ha (mean ± standard error) of biomass during the first year after a fire (Moura, *unpublished data*), which combined with fire suppression efforts commonly propagated extensive wildfires, burning up to 70% of the park (CMNP manager, personal communication, 2014, Fig. 2). Most of these wildfires were human caused and probably started by one of the 150 families that live inside the park or the communities around the PA. Rural communities use fire as a management tool for raising cattle upon native and exotic vegetation, hunting, slash and burn agriculture and as well as for firebreaks to protect watersheds, houses and agricultural areas (CMNP manager, personal communication, 2014).

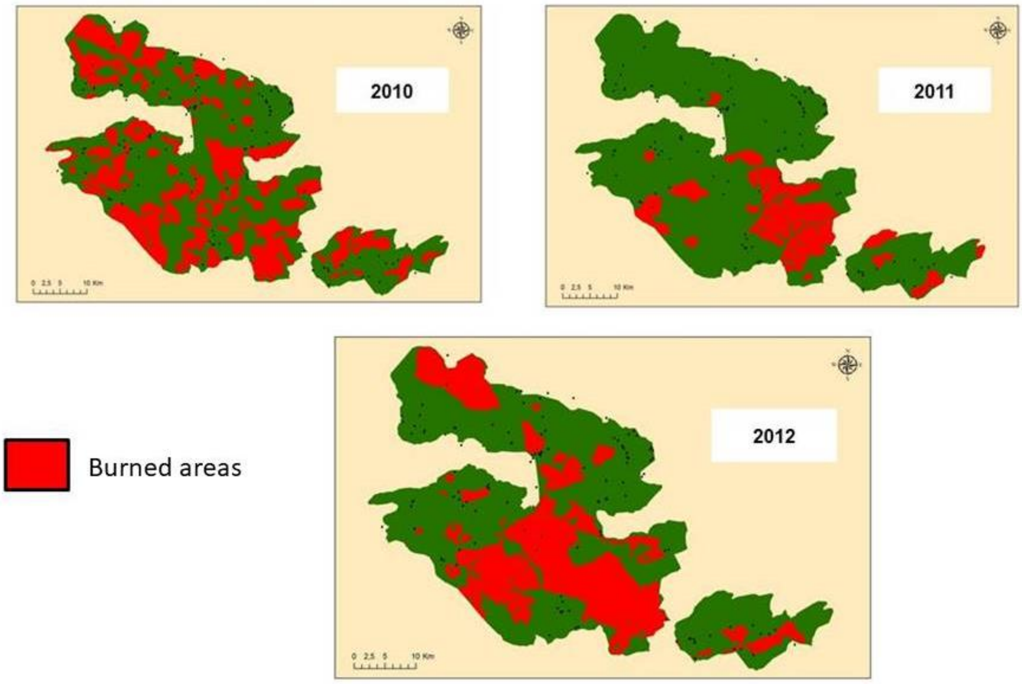

Fig. 2. Areas hit by wildfires in Chapada das Mesas National Park during the period of 2010-2012. Font: ICMBio, 2017.

2.2 Experimental design

To assess how different management strategies and fire regimes can affect the dynamics of woody species, we established permanent plots (100 x 50 m) and subplots (30 x 15 m) in open savannas that occupied 14 ha of the CMNP, distributed in 12 experimental sites (Tab. 1, Fig. 3 and 4). Within experimental sites we established plots subjected to the following treatments: (i) LDS fires (LF) - carried out at mid-day (12:00-14:30) with relative humidity <21% and average air temperature >35ºC in September, simulating LDS wildfires (Fig 5); (ii) high-intensity, EDS fires (High-EF) - carried out at mid-day (11:00-16:30) with relative humidity <50% and average air temperature >32ºC in May (Fig 6); (iii) low-intensity, EDS fires (Low-EF) - carried out during the evening (17:30-19:00) with relative humidity >50% and average air temperature <31ºC in May (Fig 7); and (iv) no fire/protected from fire - NF (Fig 8). The plots were distributed within experimental sites according to fire history (biennial and triennial fire frequencies). Experimental sites were located between 100 and 40,000 m from each other, and each plot was at least 50 m away from each other to allow for safe experimental fires and firebreaks.

Four experimental sites contained one plot and eight experimental sites contained three plots, where each plot (0.5 ha) was submitted to one treatment (Fig. 3) and each treatment was replicated four times. To test if shorter fire interval results in different rate of survival, growth and recruitment of woody species than longer fire interval, we established our plots in areas unburned for two (biennial fire frequency) and three years (triennial fire frequency) - since 2013 and 2012, respectively- which either burned in 2015 or were protected from fire for four or five years, according to the fire history of the plot. Prescribed fires were carried out in the months of May and September of 2015 to simulate fires in the EDS (management fires) and LDS (wildfires), respectively. Each experimental fire was at least one hectare in size and started at least 10m away from the permanent plot. For safety reasons, it was not possible to perform LDS experimental fires in areas unburnt for three years (triennial treatments).

Table 1. Number of experimental sites, plots and experimental fires within each treatment in the CMNP in 2015. Plots established in 2015 are 100 x 50 m; LF = LDS mid-day fires (September); High-EF = high-intensity, EDS mid-day fires (May); Low EF = low-intensity, EDS evening fires (May); and NF = no fire/protected from fire.

Fire type/ treatments	Time since fire (years)	Number of plots	Number of experimenatalfires in 2015
LF	2	4	4
High-EF	2	4	4
High-EF	3	4	4
Low-EF	2	4	4
Low-EF	3	4	4
NF	2	4	0
NF	3	4	0
Total	-	28	20

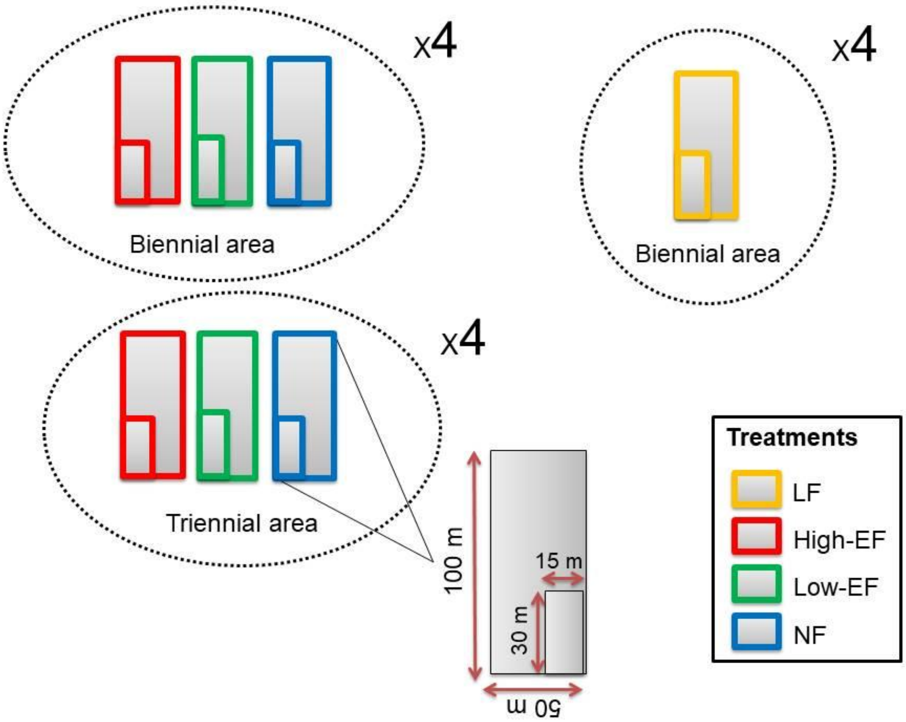

Fig. 3. Experimental design with treatments, time since fire, number of replicates and plot configuration in the CMNP. Where: LF = LDS, mid-day fires (September); High-EF = high-intensity, EDS mid-day fires (May); Low EF = low-intensity, EDS evening fires (May); and NF = no-fire/protected from fire.

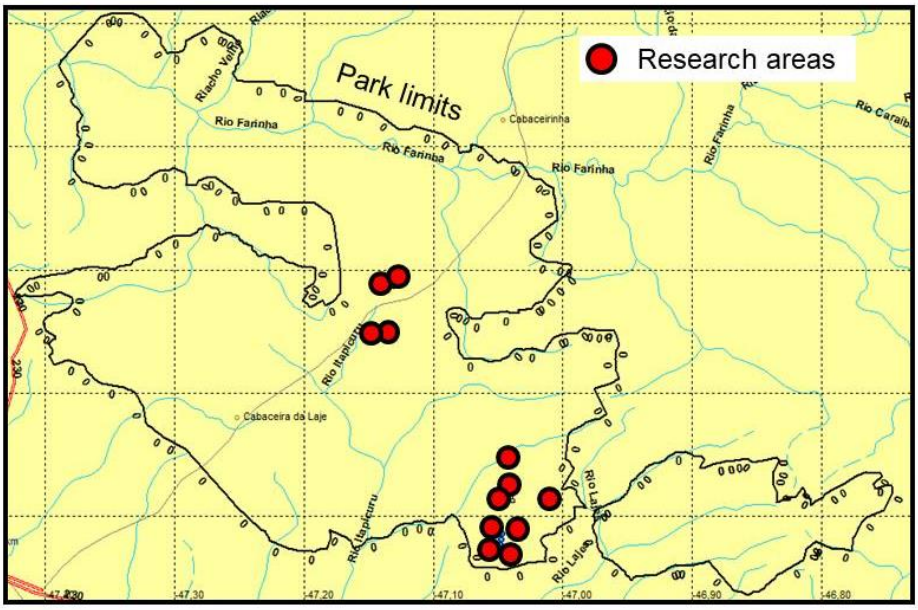

Fig. 4. Limits of the Chapada das Mesas National Park in Maranhão, Brazil with the 12 study experimental sites where we established 28 experimental plots in 2015.

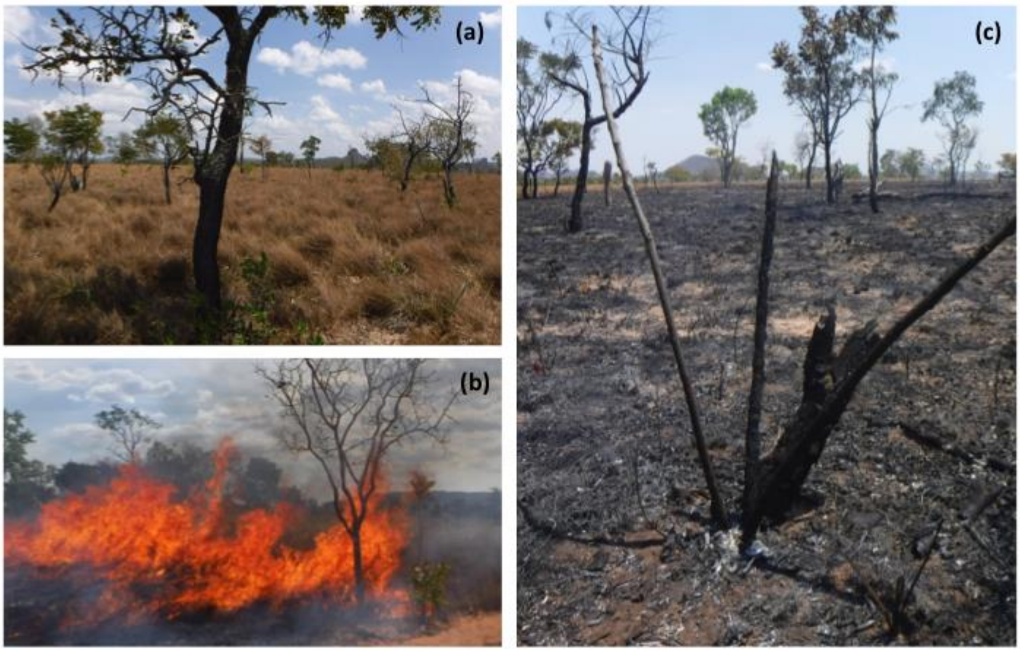

Fig. 5. The LDS plots (LF) (a) before experimental fire; (b) during experimental fire (September, LF), carried out at mid-day; and (c) after experimental fire.

Fig. 6. Triennial, high-intensity EDS plots (High-EF) (a) before experimental fire; (b) during experimental fire, carried out at mid-day (May); and biennial, high-intensity EDS plots (High-EF) (c) before experimental fire; (d) during experimental fire, carried out at mid-day (May).

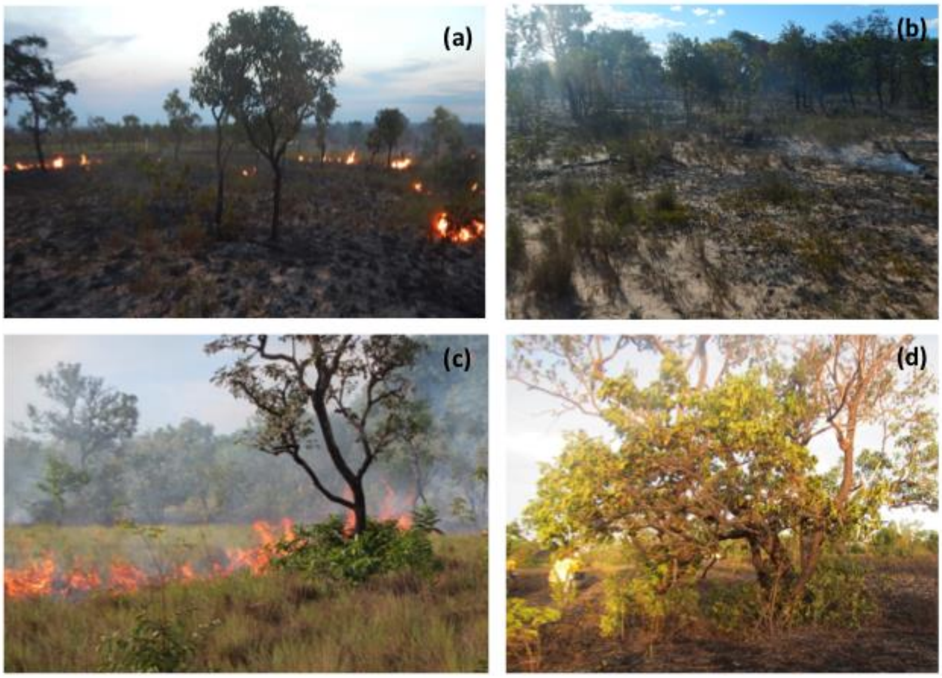

Fig. 7. Triennial, low-intensity EDS plots (Low-EF) (a)during experimental fire carried out in the evening (May,) and (b) after experimental fire; and biennial, low-intensity EDS plots (Low-EF) (c) during experimental fire carried out in the evening (May,) and (d) after experimental fire.

Fig. 8. Example of plots protected from fire (NF) for (a) five years and (b) four years .

In 2016, two experimental sites from the biennial NF treatment and one experimental site from the triennial NF treatment were accidentally burned, and in 2017 one experimental site from the LF treatment was burned

2.3 Woody response attributes

All woody species inside the plots (100 x 50 m) with diameter at 30 cm from the ground ≥5 cm were tagged, identified to species level and had all their stems measured (diameter), in 2015 (before experimental fire), 2016 and 2017, always between April-May (Fig. 9). We sampled approximately 3,250 stems per year from all our plots. Additionally, inside each plot we established a subplot (30 x 15 m) in one of the plot edges, where around 1,650 stems with diameter at ground ≥1 cm were measured (diameter), identified and tagged in the three study years (Fig. 3). Each year, we recorded stem response to the treatment into one of the following options: (1) alive without topkill, here refered as survival (2) dead (without any resprouting), (3) topkilled (complete loss of aboveground biomass with basal or underground resprouting). We recorded new recruits (stems that achieved measurement conditions) yearly in plots and subplots.

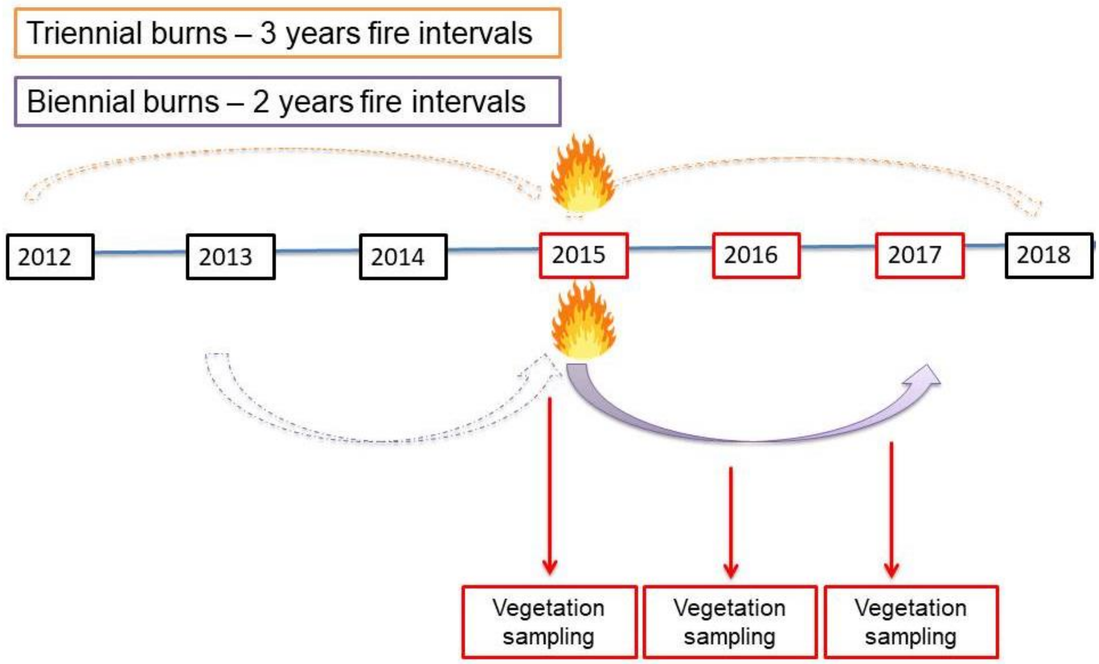

Fig. 9. Timeline showing fire history among the experimental sites, experimental fire in 2015 and vegetation sampling before burning took place in 2015 and in both consecutive years (April-May) in CMNP.

2.4 Woody community assessment

We evaluated woody stem survival and topkill, number of recruitments and changes in live basal area and stem density. For stem survival and topkill and change in stem density we divided the stems in four diameter classes: 1-5 cm, 5-10 cm, 10-15 cm and >15 cm, since stem response is linked to stem diameter (Dantas & Pausas, 2013; Hoffmann & Solbrig, 2003; Lawes et al., 2011; Souchie et al., 2017). We aggregated these stems in the last diameter class (15-49 cm) because very few stems were larger than 15 cm in our plots.

Table 2. Number of stems per hectare (mean ± standard error) within the diameter classes, fire frequency and treatments. Values were calculated from the average of stem density per area in 2015, before experimental fires in the CMNP.

Diameter class (cm)	Two years without fire				Three years without fire		
	LF	High-EF	Low-EF	NF	High-EF	Low-EF	NF
1-5	1815±570	1194±451	967±245	1022±47	1100±332	1372±298	1104±293
5-10	83±11	112±15	117±20	84±13	76±8	103±11	121±21
10-15	24±4	39±6	49±7	40±3	67±10	58±9	71±6
15>	24±3	27±5	26±9	28±0	53±9	49±2	59±9

To compare stem survival and topkill we used relative frequency (%), calculating the proportion of stems that were considered alive or topkilled from all stems inside the same plot. We calculated basal area, stem density and number of recruits for each plot (0.5 ha) and subplot(0.045 ha) per treatment and extrapolated the results to 1 hectare to facilitate comparisons to other studies. We used 2016 data to calculate stem response to 2015 experiment fires (survival and topkill), since fire effects would be clearer to detect. We analysed the number of recruits, change in live basal area, and stem density from the data sampled in 2017, two years after the first survey, since a larger period of time would better indicate changes in plant growth. To calculate changes in stem density and basal area per hectare we considered all stems sampled in 2015 and in 2017 for each plot, excluding those that died, were topkilled or lost in that period. The areas that burned in 2016 were withdrawn from all stem response analysis, and the LF area burned in 2017 was withdrawn from the analysis that considered changes within two years after fire (2015-2017).

2.5 Data analysis

To identify differences among treatments considering stem survival, recruitment and topkill, as well as in stem density and growth, we applied linear mixed-effects models (Zurr et al., 2007). In our linear mixed-effects models we tested if there were differences in the rates of survival, recruitment and topkill, and changes in stem density and basal area between the biennial treatments - LF, High-EF, Low-EF and NF – for each diameter class, considering treatment as the fixed effect variable and experimental sites as random effect variable. We also applied linear mixed-effects models to identify differences in the same stem responses between biennial and triennial treatments - High-EF, Low-EF and NF – for each diameter class, considering treatment and TSF as the fixed effect variable and experimental sites as random effect variable. For the comparison between treatments from both fire frequencies (biennial and triennial) we did not include LF treatment because there was no equivalent treatment within triennial areas. To fit data to normal distribution we transformed topkill percentage in the inverse logistic, given by the formula: $log[p/(1 - p)]$, where p is the proportion value between 0 and 1 (Crawley, 2002).

From the results of the models we ran a posteriori test (Tukey), to check if the the response variables (survival, recruitment, topkill, density and basal area) were significantly different across treatments and TSF (when applicable) and to quantify these differences. We considered significantly different when p value was below 0.05. All analysis and graphics were run by the statistical software R (R-Core-Team, 2017). The results of stem density and recruitment, and survival and topkill percentages are given in median values, while growth rates, such as basal area, are given in mean and ± standard error.

3. Results

3.1 Stem density and recruitment

Among the biennial fire treatments there were no significant differences in stem recruitment to any size class (Fig. 10). When comparing different fire intervals, the recruitment of stems <5 cm was similar in all treatments and TSF (Fig 11a), however the recruitment to >5 cm stems was significantly higher in areas protected from fire for five years (NF=46 stems/ha) in comparison to areas subjected to biennial (High-EF=5 and Low-EF=11 stems/ha) and triennial (High-EF=6 and Low-EF=16 stems/ha) fires. The recruitment of stems >5 cm was similar between areas protected from fire for four and five years (Fig. 11b). Number of stem recruitment in smaller diameter size (<5 cm) was approximately ten times higher than the recruitment of stems of larger diameter sizes (>5 cm).

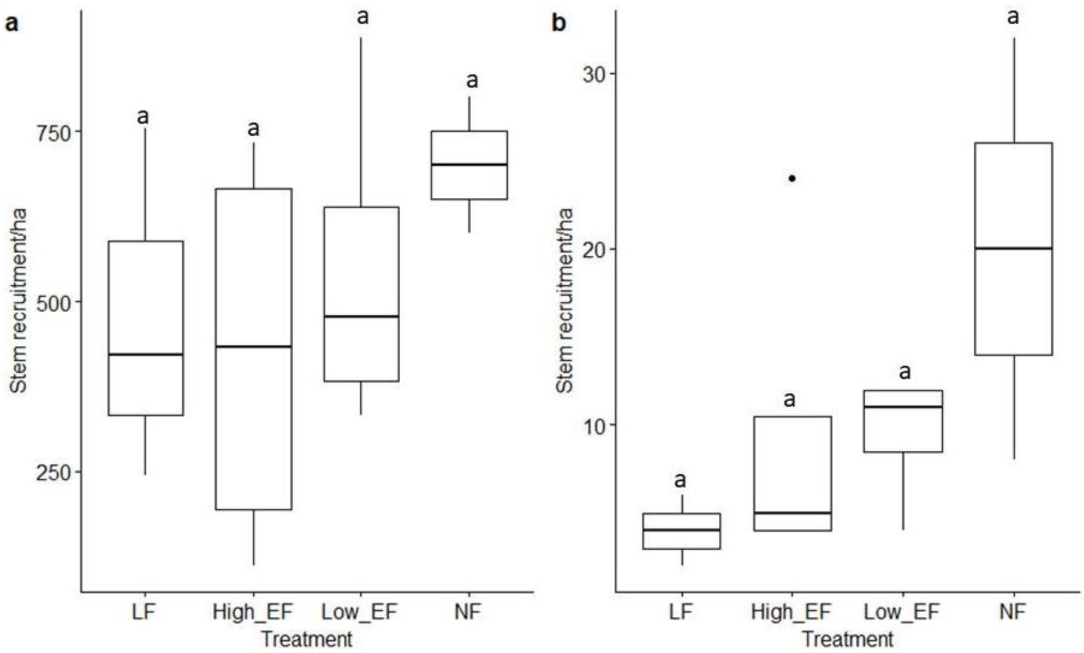

Fig. 10. Stem recruitment per hectare from 2015 to 2017 in experimental plots submitted to biennial fire frequencies and four fire treatments. (a) Stems with diameter ≥1 cm and <5 cm; and (b) Stems with diameter >5 cm. Treatments: LF = LDS fires (September); High-EF = high-intensity, EDS fires (May); Low-EF = low-intensity, EDS fires (May); and NF = no fire. The boxes encompass the first and third quartile, lines inside the boxes show the median, bars indicate minimum and maximum values, and the black circle is the outlier. The same lower case letter shows no significant difference ($p > 0.05$) per diameter class, according to *a posteriori* contrasts between treatments.

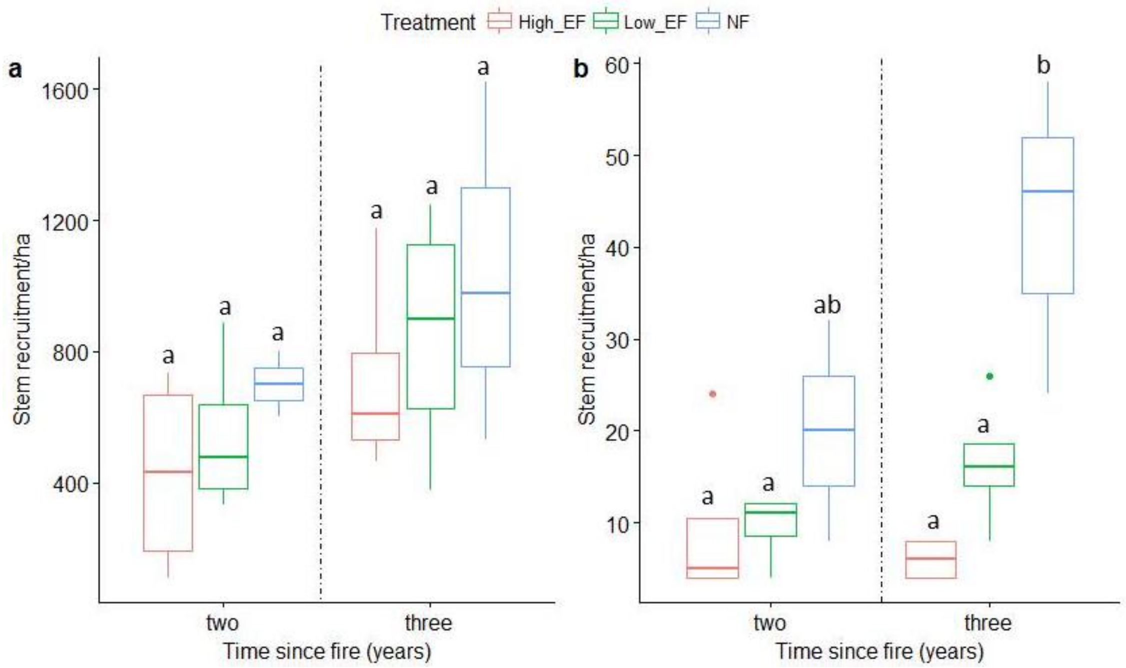

Fig. 11. Stem recruitment per hectare from 2015 to 2017 in experimental plots submitted to biennial and triennial fire frequencies – time since fire of two and three years, respectively - and three fire treatments. (a) Stems with diameter <5 cm; and (b) Stems with diameter >5 cm. Legend presents treatments, where: High-EF = high-intensity, EDS fires (May); Low-EF = low-intensity, EDS fires (May); and NF = no fire. The boxes encompass the first and third quartile, lines inside the boxes show the median, bars indicate minimum and maximum values, and the colored circles are the outliers. Different lower case letter shows significant difference ($p \leq 0.05$) per diameter class, according to *a posteriori* contrasts between treatments and TSF.

Change in stem density had the largest variation range (from -400 to 711 stems/ha) within the first diameter class (1-5 cm), and there were no significant changes of stems per hectare during the period of this study (three years, Fig. 12a) . However, stem density in the first diameter class did not present significant difference across all biennial experimental fire treatments. Even though stem recruitment to diameters below 5 cm was high (>422 stems/ha) in all experimental sites, it did not recover the initial number of stems in 2015, before the experimental fires. There was a smallincrease (5 stems/ha) in the second diameter class (5-10 cm) in areas protected from fires for four years (biennial NF), which was significantly different from the decrease of stems per hectare in LF (-22 stems/ha) and Low-EF (-13 stems/ha, Fig. 12b). . The density of stems reduced across all biennial treatments among larger individuals (>10 cm, encompassing our third and fourth diameter classes) (Fig 12c, d).

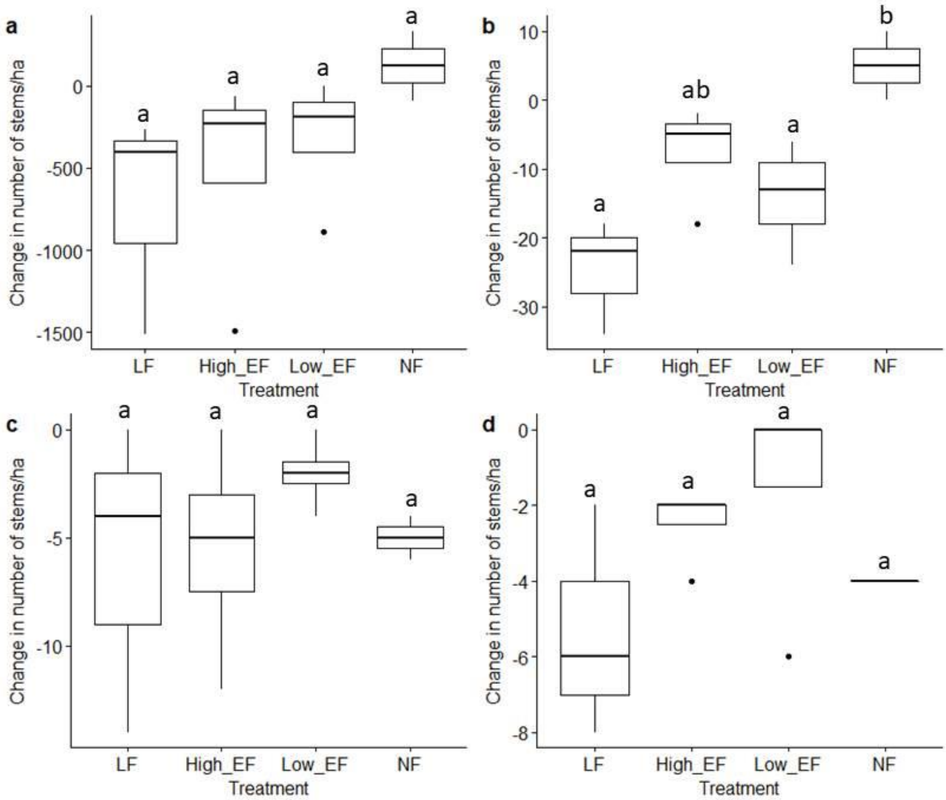

Fig. 12. Change in stem density per hectare in different diameter classes submitted to biennial fire frequencies and four treatments. Diameters <5 cm were measured at ground level and diameters >5 cm were measured 30 cm from the ground. (a) Stems from 1-5 cm; (b) stems from 5-10 cm; (c) stems from 10-15 cm; and (d) stems >15 cm in diameter. Treatments: LF = LDS fires (September); High-EF = high-intensity, EDS fires (May); Low-EF = low-intensity, EDS fires (May); and NF = no fire. The boxes encompass the first and third quartile, lines inside the boxes show the median, bars indicate minimum and maximum values, and black circles are the outliers. Different lower case letters show significant difference ($p \leq 0.05$) per diameter class, according to *a posteriori* contrasts between treatments.

Changes in stem density in diameters 1-5 cm in the biennial High-EF was negative (-233 stems/ha) and significantly different from areas protected from fire for 5 years (triennial NF=711 stems/ha, Fig. 13a). There was no significant differences in the stem density across all other treatments for this diameter class. In the second diameter class (5-10 cm), areas protected from fire for 5 years had the highest stem density increment per hectare (36 stems/ha), which was significantly different from all other treatments that were similar to each other and close to zero (Fig. 13b). Diameter classes >10 cm had similar change in stem density, with small deacreases in stem density over the study years, similar to zero and across all treatments from biennial and triennial fire frequencies (Fig. 13c,d).

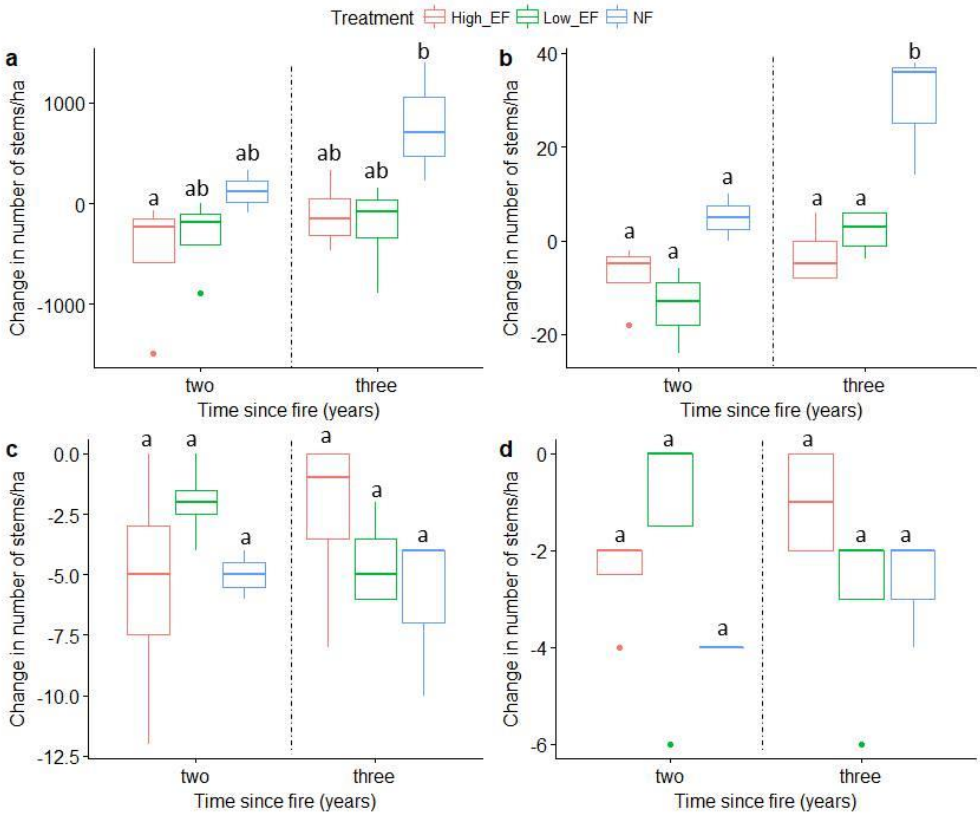

Fig. 13. Change in stem density per hectare in different diameter classes submitted to biennial and triennial fire frequencies and three treatments. Diameters <5 cm were measured at ground level and diameters >5 cm were measured 30 cm from the ground. (a) Stems from 1-5 cm; (b) stems from 5-10 cm; (c) stems from 10-15 cm; and (d) stems >15 cm in diameter. Treatments: High-EF = high-intensity, EDS fires (May); Low-EF = low-intensity, EDS fires (May); and NF = no fire. The boxes encompass the first and third quartile, lines inside the boxes show the median, bars indicate minimum and maximum values, and coloured circles are the outliers. Different lower case letters show significant difference ($p \leq 0.05$) per diameter class, according to *a posteriori* contrasts between TSF and treatments.

3.2 Stem survival and topkill

Individuals with diameters below 5 cm(first diameter class) presented significantly higher survival rates in areas protected from fire (NF=83%) in comparison to areas burned biennially (Fig. 14a). Fire treatments did not influence stem survival in this diameter class. In contrast, the survival of stems in the 5-10 cm diameter class (second class) was significantly lower in the biennial LF areas compared to both EDS fires and NF treatments (Fig. 14b). The survival of stems with diameters larger than 10 cm (third and fofth classes) was not affected by the treatments and tended to be higher than 80% (Fig. 14c, d).

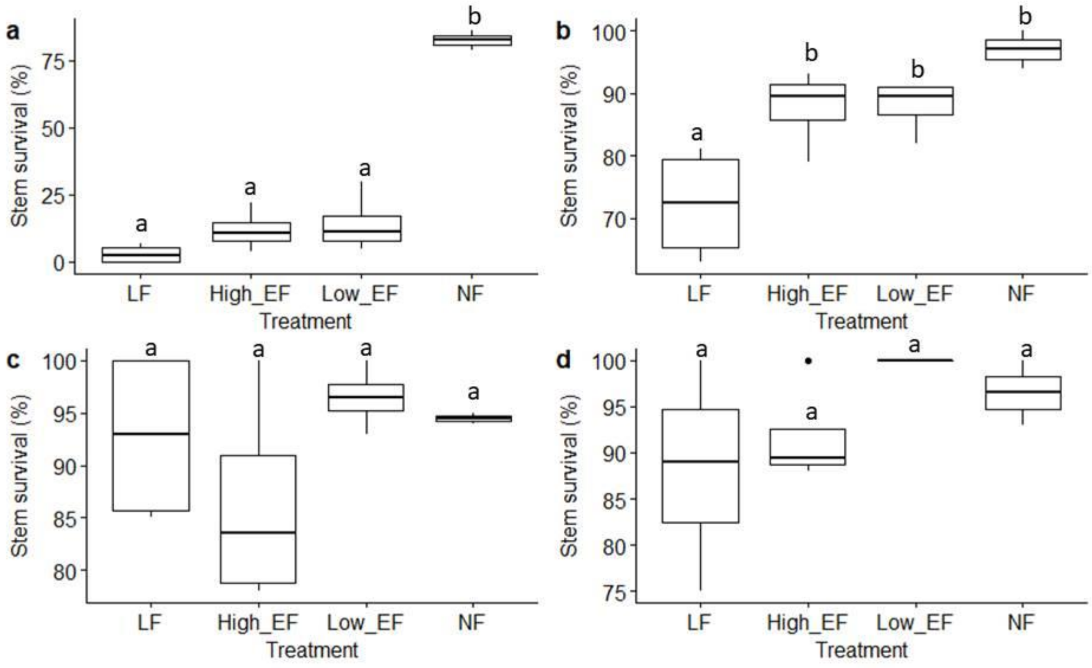

Fig. 14. Stem survival within different diameter classes submitted to biennial fire frequency and four treatments. Diameters <5 cm were measured at ground level and diameters >5 cm were measured 30 cm from the ground. (a) Stems from 1-5 cm; (b) stems from 5-10 cm; (c) stems from 10-15 cm; and (d) stems >15 cm in diameter. Treatments: LF = LDS fires (September); High-EF = high-intensity, EDS fires (May); Low-EF = low-intensity, EDS fires (May); and NF = no fire. The boxes encompass the first and third quartile, lines inside the boxes show the median, bars indicate minimum and maximum values, and black circles are the outliers. Different lower case letters show significant difference ($p \leq 0.05$) per diameter class, according to *a posteriori* contrasts between treatments.

When comparing TSF and treatments, stem survival in the first diameter class (1-5 cm) was significantly higher in biennial and triennial NF treatments (83% and 97%, respectively, Fig. 14a). The same pattern was kept for biennial and triennial areas, where only individuals between 1-5 cm that burned in the EDS had lower stem survival in relation to NF and individuals with diameters larger than 5 cm were not affected by treatments nor TSF (Fig. 15b, c, d).

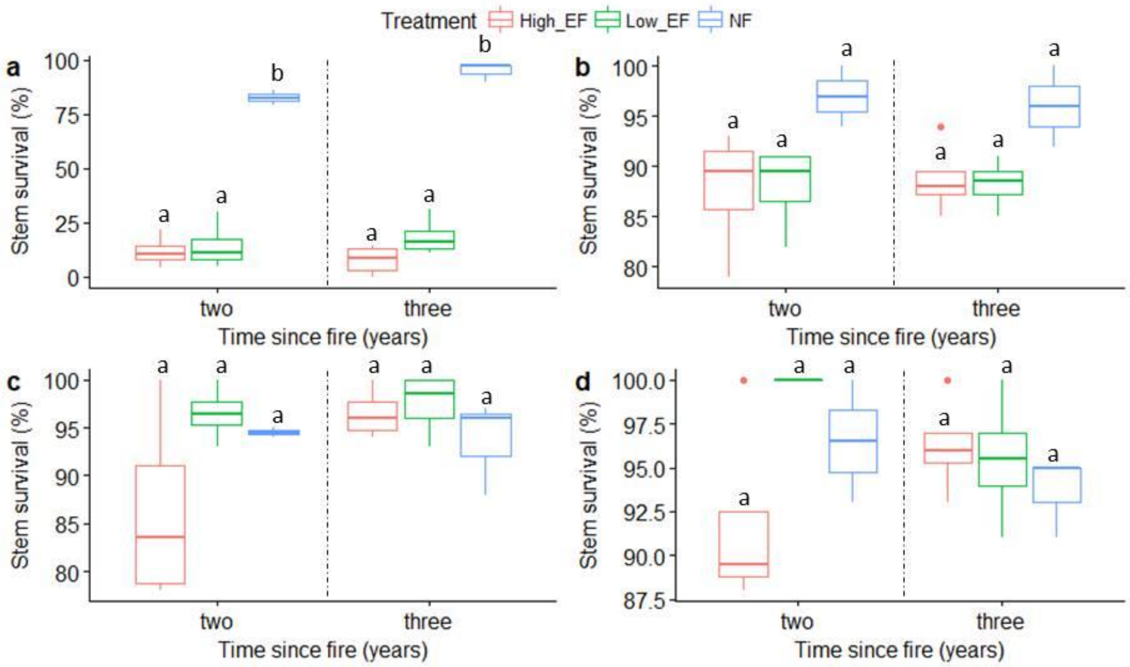

Fig. 15. Stem survival within different diameter classes submitted to biennial and triennial fire frequencies and three treatments. Diameters <5 cm were measured at ground level and diameters >5 cm were measured 30 cm from the ground. (a) Stems from 1-5 cm; (b) stems from 5-10 cm; (c) stems from 10-15 cm; and (d) stems >15 cm in diameter. Treatments: High-EF = high-intensity, EDS fires (May); Low-EF = low-intensity, EDS fires (May); and NF = no fire. The boxes encompass the first and third quartile, lines inside the boxes show the median, bars indicate minimum and maximum values, and colored circles are the outliers. Different lower case letters show significant difference ($p \leq 0.05$) per diameter class, according to *a posteriori* contrasts between TSF and treatments.

When comparing TSF and treatments, stem survival in the first diameter class (1-5 cm) was significantly higher in biennial and triennial NF treatments (83% and 97%, respectively, Fig. 14a). The same pattern was kept for biennial and triennial areas, where only individuals between 1-5 cm that burned in the EDS had lower stem survival in relation to NF and individuals with diameters larger than 5 cm were not affected by treatments nor TSF (Fig. 15b, c, d).

Following the same patterns as survival, Topkill (loss of all aerial tissue followed by resprout) in stems with diameter smaller than 5 cm was significantly lower in NF treatments (4%) when compared to biennial fire treatments (Fig. 16a); whereas stem topkill in diameters between 5-10 cm were significantly higher in biennial LF treatment (20%) when compared to the other biennial treatments (Fig. 16b). Stem topkill in individuals with diameters larger than 10 cm was close to zero in almost all treatments, except for stems between 10-15 cm that presented significantly higher topkill in NF treatment (5%) when compared to EDS fires (Fig. 16c, d).

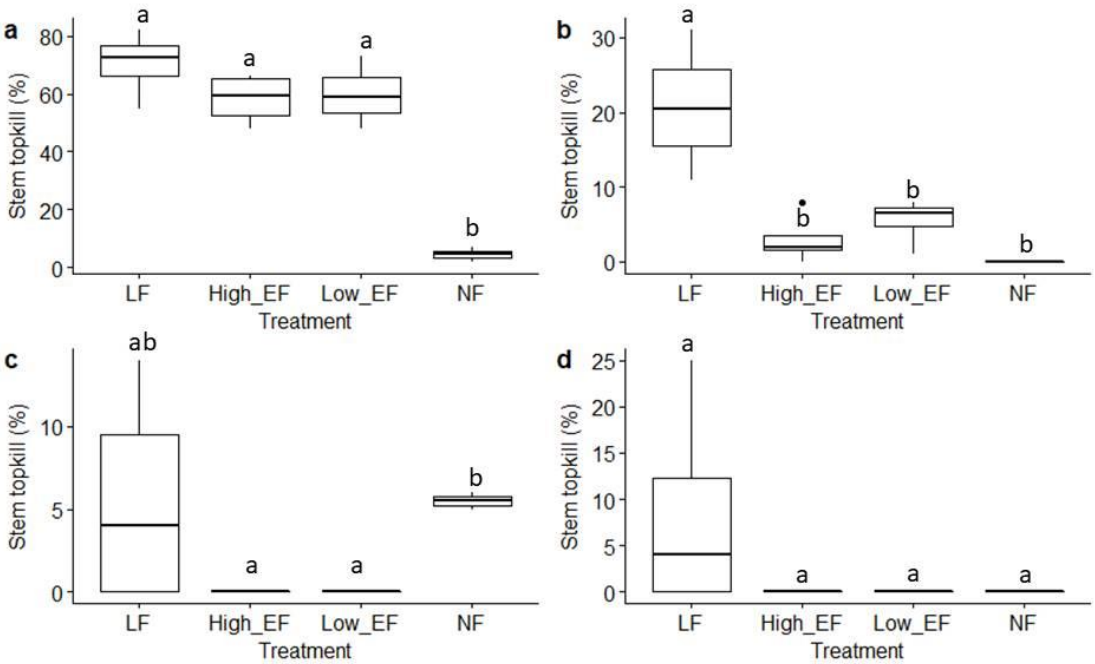

Fig. 16. Stem topkill within different diameter classes submitted to biennial fire frequencies and four treatments. Diameters <5 cm were measured at ground level and diameters >5 cm were measured 30 cm from the ground. (a) Stems from 1-5 cm; (b) stems from 5-10 cm; (c) stems from 10-15 cm; and (d) stems >15 cm in diameter. Treatments: LF = LDS fires (September); High-EF = high-intensity, EDS fires (May); Low-EF = low-intensity, EDS fires (May); and NF = no fire. The boxes encompass the first and third quartile, lines inside the boxes show the median, bars indicate minimum and maximum values, and black circles are the outliers. Different lower case letters show significant difference ($p \leq 0.05$) per diameter class, according to *a posteriori* contrasts between treatments.

Topkill between biennial and triennial fire frequency in diameters from 1-5 cm was significantly lower in areas protected from fire for three and four years (biennial NF= 4% and triennial NF=3%) compared to all fire treatments (Fig. 17a). There were no differences between TSF and treatments in stem topkill for diameters in the second and forth size classes (5-10 cm and >15 cm, Fig. 17b, d), however, topkills in biennial NF treatments (5%) were significantly higher than all other treatments and fire intervals (Fig. 17c).

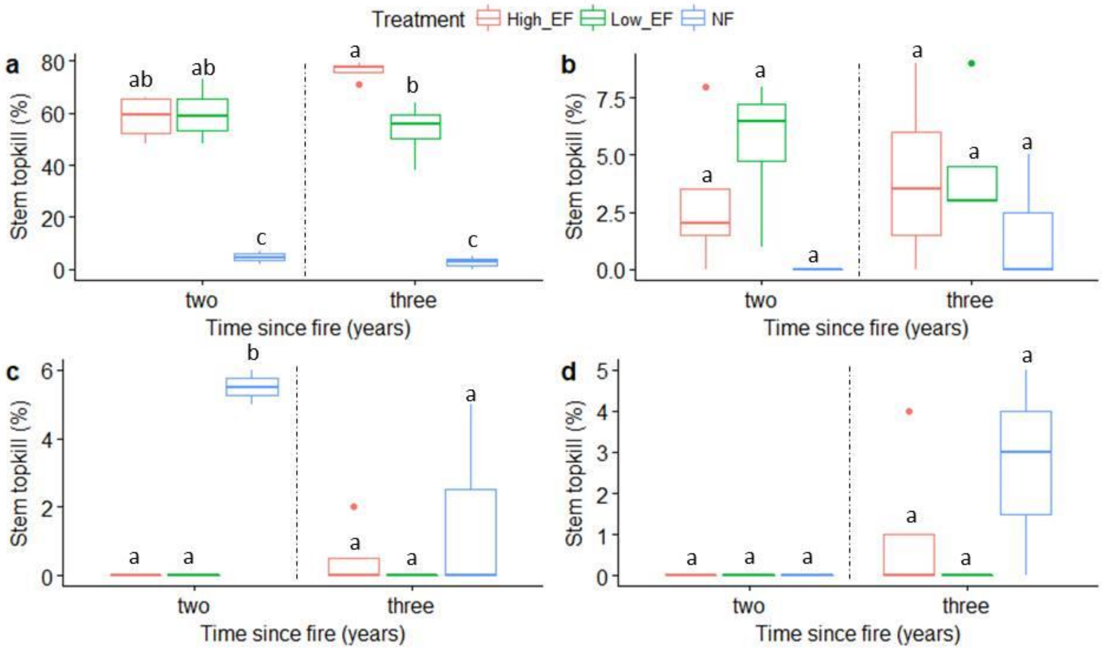

Fig. 17. Stem topkill within different diameter classes submitted to biennial and triennial fire frequencies and three treatments. Diameters <5 cm were measured at ground level and diameters >5 cm were measured 30 cm from the ground. (a) Stems from 1-5 cm; (b) stems from 5-10 cm; (c) stems from 10-15 cm; and (d) stems >15 cm in diameter. Treatments: High-EF = high-intensity, EDS fires (May); Low-EF = low-intensity, EDS fires (May); and NF = no fire. The boxes encompass the first and third quartile, lines inside the boxes show the median, bars indicate minimum and maximum values, and colored circles are the outliers. Different lower case letters show significant difference ($p \leq 0.05$) per diameter class, according to *a posteriori* contrasts between TSF and treatments.

3.3 Woody plants growth rates

The basal area increment was similar between biennial NF (0.544 ± 0.17 m^2ha^{-1}) and Low-EF (0.157 ± 0.09 m^2ha^{-1}), and significantly higher than LF (-0.375 ± 0.04 m^2ha^{-1}), which showed reductions in basal area (Fig. 18a). The basal area increment within High-EF plots had intermediate values between the other treatments. The plots protected from fires for five years (NF triennials) presented a significantly higher basal area increment (0.929 ± 0.18 m^2ha^{-1}) compared to both triennial EDS treatments, and similar to the basal area increment measured in the plots protected from fires for four years (NF biennial=0.544 ± 0.17 m^2ha^{-1}) (Fig 18b).

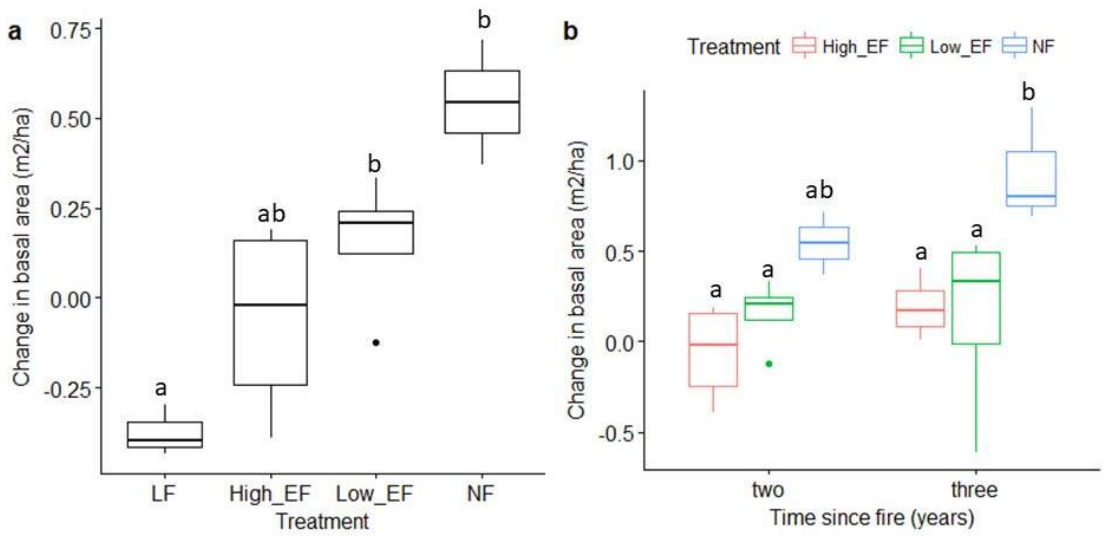

Fig. 17. Change in basal area under (a) biennial fire frequency and four treatments, and (b) biennial and triennial fire frequency and three treatments. Treatments: LF = LDS fires (September); High-EF = high-intensity, EDS fires (May); Low-EF = low-intensity, EDS fires (May); and NF = no fire. The boxes encompass the first and third quartile, lines inside the boxes show the median, bars indicate minimum and maximum values, and colored and black circles are the outliers. Different lower case letters show significant difference ($p \leq 0.05$) per diameter class, according to *a posteriori* contrasts between TSF and treatments.

4. Discussion

4.1 Short-term implications of fire management systems to woody species communities

In our experiment, stem recruitment was only different when comparing biennial and triennial fire frequencies, where areas protected from fire for five years presented higher recruitment of plants with diameters above 5 cm than all other treatments and fire frequencies, corroborating our initial hypothesis. Increment in stem density among diameters from 1-10 cm was also significantly higher in areas protected from fire for five years between biennial and triennial fire frequencies. Among biennial sites, plots protected from fire for four years presented significantly higher increment in stems with diameters from 5-10 cm when compared to the other treatments, except for the EDS mid-day treatment. Our results showed that LDS fires have similar results to EDS fires (independently from air relative humidity conditions) in reducing stem density of woody individuals from 1-10 cm in diameter.Woody plants recruitment responded to fire seasonality in the central Cerrado region, where within a period of 10 years there were 480, 148 and 60 recruitments per hectare (to diameters >5 cm) after burned by gradually increasing fire intensities in June, August and September, respectively (Sato et al., 2010). For Cerrado woody species, EDS fires have resulted in more woody species recruitment than LDS fires (Garda, 2018) and 15-18 years of fire protection increased the abundance of woody individuals in almost all Cerrado physiognomies (*cerradão, cerrado sentido restrito, campo cerrado and campo sujo*), including of fire-sensitive species (Moreira, 2000; Ramos, 1990). However, the seed bank and vegetative reproduction of several species can benefit from fire passage in different periods of the year and can stimulate the recruitment of less competitive species (Keeley & Fotheringham, 2000; Whelan, 1995) in savannas (Andrade & Miranda, 2010; Williams et al., 2005), woodlands (Enright & Lamont, 1989; Keeley, 1987; Roche et al., 1998), shrublands (Martin, 1995; Pierce & Cowling, 1991; Zammit & Zedler, 1988) and grassland vegetation (Benson & Hartnett, 2006; le Stradic et al., 2015).However, each of these vegetation present different environmental conditions for seedlings to establish. In general, shrub species in the Cerrado vegetation tend to be less affected by fires than woody species (Moreira, 2000). Several shrub species recruit more under frequent, high-intensity fires in the semi-arid woodland in eastern Australia (Hodgkinson & Hodgkinson, 2013; Knox & Clarke, 2006), as well as exotic fire-adapted species in fire-prone ecosystems (Brooks et al., 2004; Haidinger & Keeley, 2017).

Frequent LDS fire regimes have resulted in the reduction of woody species abundance in other Cerrado ecosystems (Hoffmann, 1999; Lima et al., 2009; Medeiros & Miranda, 2005; Ribeiro et al., 2012; Sato, 2003), northern Australian savannas (Braithwaite & Estbergs, 1985; Russell-Smith & Edwards, 2006; Russell-Smith et al., 2003; Williams et al., 1999; Yates et al., 2008), southern African savannas (Eckhardt et al., 2000; Higgins et al., 2007; Roques et al., 2001; van Wilgen et al., 2003) and oak savannas in North America (Peterson & Reich, 2001; Tester, 1989). Additionally, frequent fires may limit species composition (Anto & Aure, 2010; Durigan et al., 1994) and favor those with vegetative (Clarke et al., 2013; Hoffmann, 1998; Setterfield, 2002) and clonal reproduction. The combination of high-intensity and severity, frequent fires can lead to the dominance of low stature, multi-stemmed woody plants (Bellingham & Sparrow, 2000; Govender et al., 2006).

Severe fire weather conditions, as those presented in the LDS and triennial mid-day, EDS in the CMNP, combined with fine fuel load are likely to increase fire intensity (Cheney et al., 1993; Hoffmann et al., 2012; Whelan, 1995). Such combination may have increased woody stem mortality, especially in smaller diameter class stems (< 10 cm diameter), in Cerrado physiognomies (Garda, 2018; Kauffman et al., 1994; Medeiros & Miranda, 2005; Ribeiro et al., 2012; Sato & Miranda, 1996) and other savanna ecosystems that are characterized by different biomass quality, volume and flammability (Gill et al., 1990; Higgins et al., 2000; Williams et al., 2005, 1999). In our experiment, the survival - maintenance of aboveground biomass - of small stems (diameter <5 cm) after all types of fire was very low compared to stems that did not burn for four and five years, corroborating with our hypothesis; whereas small-medium stems (5-10 cm in diameter) were more tolerant to the experimental fires, and only stems submitted to biennial LDS fires survived significantly less than the ones unburned. Generally, plants with small stems did not die one year after fire passage but resprouted from the base or ground after losing their aboveground biomass, i.e. they were topkilled. Previous studies in Cerrado indicate that a significant number of small stems are able to recover to pre-fire size one year after burnt (Hoffmann & Solbrig, 2003; Medeiros & Miranda, 2005), but frequent fires keep them from growing into reproductive sizes (Hoffmann, 1998). More than 87% of our sampled stems were smaller than 5cm in diameter (Table 2), this is likely due to the frequent LDS fire regime the CMNP was subjected to in recent years (before 2014). The decrease in stem density among larger individuals (diameter >10 cm) in our plots, including the fire protected ones, is probably a consequence of this previous frequent fire regime. Other factors may have also influenced woody density in our experiment rather than fire, such as lower annual rainfall and prolonged drought, once water availability also influences savanna dynamics (Medina & Silva, 1990; Skarpe, 1992). In our study region. in the annual precipitation was lower in 2015 (1,243 mm) than the average of the five previous years (1,885 mm, INMET, 2017). Although, short-to-medium term outcomes of a single high-intensity wildfire can result in high mortality of medium-to-large diameter woody individuals in northern Australian savannas (Williams et al., 1999), larger stems (diameters >10 cm) subjected to LDS fires survived much better than the smaller ones (>93%) in the CMNP, as reported for other Cerrado regions in central Brazil (Medeiros

& Miranda, 2005; Ramos, 1990; Rocha-Silva, 1999; Sato & Miranda, 1996). Topkill observed in larger stems (>10 cm) in NF treatments may be a consequence of previous severe fires that could have weakened woody structure or other elements, such as low rainfall rates, storms that brought down woody individuals or parasites that could have also affected our plots rather than only our experimental fire.

Small Cerrado woody plants (diameter <5 cm) are more vulnerable to fire than larger ones because larger stems tend to present thicker barks that protect living tissues from heat or impacts (Dantas & Pausas, 2013; Guedes, 1993; Keeley et al., 2011; Miranda et al., 2002). However, fire-adapted species are capable of resprouting after repeated topkill by fires (Bond & Midgley, 2001; Hoffmann et al., 2009; Lawes et al., 2011). The small stems (diameters 1-5 cm) protected from fire for four and five years were the less topkilled (<4%), and the ones subjected to the triennial EDS fires under relative humidity below 50% were the most topkilled (77%), which was significantly different from the triennial EDS fires under relative humidity above 50% (56%).We did not predict that fires undertaken in different weather conditions in the EDS would result in different stem topkill rate, but we hypothesized that stems protected from fire for longer periods would be less topkilled. However, among biennial fire frequency only stems protected from fire for four years had lower topkill percentage than all fire treatments. A research in other typical Cerrado areashow that longer fire interval resulted in increased topkill (Rocha-Silva, 1999) and higher intensity fires caused more topkill than lower intensity fires, especially in small individuals (Sato, 2003). In all our experimental fires, the highest stem survival (97%) and lowest topkill percentages (3%) occurred among small individuals protected from fire for longer period (5 years). Other studies indicated that the combination of short fire interval and low-intensity fires, commonly presenting lower rate of spread and higher residence time at temperatures above 60ºC, can lead small stems (with bark thickness <6 mm) to higher wood damage than fires with higher rate of spread, such as high-intensity fires (Guedes, 1993; Uhl & Kauffman, 1990).

Furthermore, fire intensity has been correlated with scorch height in African and Australian savannas, where high-intensity fires are likely to produce greater flame heights and topkill stems and branches, reducing woody individuals to coppice from the stem collar region (Luke & McArthur, 1978; Trollope & Trollope, 2002; Werner & Prior, 2013; Williams et al., 1998, 1999). This correlation has not been reported to Cerrado vegetation yet. Frequently in fire-prone ecosystems, woody species present strategic traits that keeps them from being topkilled and protect their buds from fire (Bond & Keeley, 2005; Lehmann et al., 2014; Simon & Pennington, 2012). In Brazilian and Australian savannas bark thickness results in less mortality and biomass losses (Coutinho, 1990; Guedes, 1993; Hoffmann & Solbrig, 2003; Lawes et al., 2011; Medeiros & Miranda, 2005; Werner & Prior, 2013), while in African and north American savannas escaping the "fire trap" height presents a better strategy for woody species to overcome fire and animal predation (Dantas & Pausas, 2013; Grady & Hoffmann, 2012; Higgins et al., 2000; Staver et al., 2009).

Long-term site productivity is affected by frequent and widespread fires in Cerrado ecosystems (Batmanian & Haridasan, 1985; Ramos, 1990), especially when more volume of biomass is consumed by fire and greater quantities of nutrients (C, N and S) are lost (Kauffman et al., 1994). In our experiment, the total basal area of woody plants increased in areas protected from fire for four years and subjected to biennial and triennial EDS fires with relative humidity above 50% and reduced in areas subjected to more severe fire weather conditions (EDS with relative humidity below 50% and LDS) and shorter fire intervals (biennial frequency). Similar results are reported previously for other typical and grassland Cerrado areas (Batmanian & Haridasan, 1985; Hoffmann, 1999). In northern Australian savannas, the growth (in height and diameter) of eucalyptus adult woody species is limited by fires in the LDS (Murphy et al., 2010; Russell-Smith et al., 2003; Williams et al., 1999), and accelerated by EDS fires when compared to unburned individuals (Murphy et al., 2010; Prior et al., 2006; Werner, 2005). However, these studies were unable to account for slow-growing species (Werner & Prior, 2013) and local-scale variability in resources and ecological interactions (Lehmann et al., 2009), such as competition and predation, which could have masked wood species growth and survival in the experiment. Because Cerrado adult woody species individuals usually have thick barks that protect them from fire (Coutinho, 2002; Guedes, 1993; Ramos, 1990; Simon & Pennington, 2012; Walter & Ribeiro, 2010), high-intensity fires with high scorching height can remove or consume parts of the dead tissue and under-estimate diameter growths (Felfili et al., 2005), feature also reported in other savanna environments (Gill et al., 1986; Prior et al., 2006), where eucalyptus barks present higher flammability when compared to Cerrado woody species. Stem growth is remarkably different between species and among larger and smaller woody individuals (Felfili, 1995; Felfili et al., 2005; Miranda et al., 2007; Prior et al., 2006; Werner & Prior, 2013), as well as their capacity to survive fire events and keep their aboveground structures (Grady & Hoffmann, 2012; Hoffmann & Solbrig, 2003; Lawes et al., 2011). The overall increase in basal area in our experiment is probably explained by diameter growth among larger stems (>10 cm), since topkill percentage was <5% and survival percentage >83% among large woody individuals (third and fourth diameter classes) in all fire frequencies and treatments.

4.2 Fire management applicability and ecological precautions

The coexistence of woody and graminous species in savanna ecosystems is regulated by fire, climate, water and soil conditions (Accatino et al., 2010; Bond et al., 2005; Bowman et al., 2009; Gardner, 2006; Higgins et al., 2000; Lehmann et al., 2014; Sankaran et al., 2005; Simon et al., 2009). To guarantee the conservation of landscapes, fire sensitive ecosystems and local biodiversity in savanna PAs, managers must permanently assess the implemented management systems and, when necessary incorporate new strategies. The Kakadu and Kruger National Parks are great examples of successful fire implementation in savanna PAs that were subjected to long-term research monitoring and evaluation to help improve their management systems (Bond & Archibald, 2003; Edwards et al., 2003; McGregor et al., 2010; Parr et al., 2009; van Wilgen et al., 2004; Werner, 2005; Williams, 1995). Although IFM is a pilot programme and pioneer in using fire in Cerrado PAs (Schmidt et al., 2018), fire researchers have long been struggling to experimentally burn study areas and advance in ecological knowledge (Coutinho, 1981; Gomes et al., 2018; Miranda et al., 1993; Miranda, 2010; Pivello & Coutinho, 1996). In our study, we evaluated Cerrado woody plant communities subjected to different management strategies, including the burning protocols established by the IFM that consists in EDS prescribed fires under different weather conditions (<50%relative humidity and >50% relative humidity).

Overall in the CMNP, biennial, LDS experimental fires that mimic the frequent LDS wildfire regime most Cerrado PAs are subjected to, reduced woody density, recruitment, survival and growth, as well as increased stem topkills in the open savanna vegetation. In the same experiments, we found that all biennial EDS fires increased topkill and reduced survival of smaller stems (1-5 cm) in comparison to areas unburned for four years . However, LDS fires resulted in significantly less basal area growth when compared to areas protected from fire for four years and EDS fires with relative humidity above 50%. Our results indicate that EDS fires with relative humidity above 50% are likely to have the same impact on woody communities as the ones with relative humidity below 50%, especially within two years fire intervals. Therefore, it is important to consider not only fire seasonality, but also the weather conditions and fire intervals as part of management plans to implement prescribed fires in Cerrado (Cheney et al., 1993; Gill et al., 1996; Govender et al., 2006; Hoffmann et al., 2012; Price et al., 2014). Fire season may also have consequences related to the reproductive cycle of plants (Schmidt et al., 2005) and animals (Falleiro, 2011). .

Importantly, management of PAs is not exclusively dedicated in conserving woody plant communities, but also to keep landscapes and local biodiversity of other plant and animal communities, which may include opening dense woody physiognomies or fragmenting the fuel load in highly flammable unburned areas. To fragment fuel load in safe conditions, , evening fires in the EDS (when relative humidity is above 50%) are recommended since they will help reducing woody densities, specially biennial fire frequency (Fig. 11 and 12), and the weather conditions - air relative humidity above 50% and cooler temperatures in mesic savannas - will keep fire intensity and rate of spread at lower levels and will probably self-extinguish when fuel moisture increase during the night or when reaching moisturized environments, such as gallery forests or denser vegetation fragments (Gill et al., 1996; McGuffog et al., 2001; Schmidt et al., 2018; Williamson et al., 2016). Thus this type of fire: (i) offers more safety in hazardous flammable areas (Edwards et al., 2015; Govender et al., 2006; Haney & Power, 1996); (ii) lower management costs and losses (Moritz et al., 2014; Penman et al., 2011; Schmidt et al., 2018); (iii) reduces GHG emissions by avoiding frequent wildfire incidents (Murphy et al., 2015; Russell-Smith et al., 2009, 2013a; UNU, 2015; van der Werf et al., 2017); (iv) can help creating spatial mosaics of patchy burned vegetation (Bliege Bird et al., 2008; Laris, 2002; Oliveira et al., 2015; Weir et al., 2000); and (v) help protect fire-sensitive ecosystems from wildfires (Bradstock et al., 2005; Maravalhas & Vasconcelos, 2014; Price et al., 2012; Russell-Smith et al., 1998; Schmidt et al., 2018; Trauernicht et al., 2015).

Where grassland fuel load imposes high risk to fire spread, management actions with prescribed fires should be prioritized and burning under low relative humidity (<50%), high air temperatures and windy weather conditions should be avoided (Cheney et al., 1993; Just et al., 2016; Rothermel, 1972; Trollope & Trollope, 2002). Nonetheless, high-intensity fires also plays an important role in the natural dynamics of ecosystems (Bond & Keeley, 2005; França et al., 2007; Johnson et al., 2001; Meyn et al., 2007; Ramos-Neto & Pivello, 2000; Stephenson et al., 1991; Turner & Dale, 1998). High intensity fires can be strategic to certain management goals, such as to fragment homogeneous landscapes (Govender et al., 2006; Marcoux et al., 2015; Roques et al., 2001; Scott et al., 2012), opening woody encroached areas or maintaining grassland dominated landscapes (Bond & Midgley, 1995; Pausas & Moreira, 2012), eliminate invasive species (Brooks et al., 2004) and favour several plants' life stage or trait (Armstrong & Phillips, 2012; Bellingham & Sparrow, 2000; Keeley & Zedler, 2009; Werner, 2005; Whelan et al., 2002). In these scenarios, the interaction between LDS fires or fires carried out in more severe weather conditions during the dry season and flammable vegetation structure can create higher intensity fires and result in these desired, environmental outcomes. Notably, this is only applicable to fragmented areas, where natural or anthropogenic barriers will prevent fires from spreading to larger extents turning into uncontrolled wildfires.

Finally, every management system is susceptible to environmental changes and failing to acknowledge them will rise the risks of initiating wildfires (Fernandes, 2013; Moritz et al., 2012; Roos et al., 2016), therefore, managers and researchers' feedbacks will constantly enhance the systems' reliability (Christensen, 2005; Driscoll et al., 2010; Van Wilgen et al., 2007). We point out that these guidelines have been addressed to inform how woody plant communities respond to several fire management arrangements, whereas local and regional characteristics may require readjustments or even different evaluative protocols to deliver better results (Dickinson & Ryan, 2010; Driscoll et al., 2010; Gomes et al., 2018; Groffman et al., 2006). Cerrado woody species present slow-growing patterns, therefore, our results are limited to a short-term monitoring period and further lengthier studies and continuous monitoring in PAs will probably provide a more complete assessment.

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Conclusão

A partir desta tese foi possível: (i) sintetizar políticas de manejo do fogo nas principais regiões savânicas do hemisfério sul, e identificar os desafios associados a recentemente implementada política de manejo do fogo no Cerrado – especialmente aqueles relacionados a abordagem adaptativa, que considere os diferentes históricos de regimes de fogo (natural e antropogênico), e a sobreposição de interesses econômicos em detrimento da conservação ambiental; (ii) caracterizar o comportamento de queimas de manejo no início da estação seca, sob condições meteorológicas de maior e menor umidade relativa do ar, e de queimas tardias, que simulam os frequentes incêndios no Cerrado e (iii) quantificar os efeitos desses diferentes regimes de queima em comunidades lenhosas em áreas de cerrado sentido restrito no Parque Nacional da Chapada das Mesas. A recente institucionalização do uso do fogo no manejo de unidades de conservação (UCs) do Cerrado possibilitou a investigação científica em escala de paisagem deste trabalho, e a realização dos experimentos de acordo com parâmetros utilizados em campo para se estabelecer as queimas controladas do manejo integrado do fogo (MIF) desde 2014. Com o entendimento do processo de mudança da política do fogo, bem como dos regimes de queima estabelecidos por cada sistema, foi possível elaborar experimentos de pesquisa que subsidiem todas as etapas do manejo (planejamento, implementação e avaliação; Schmidt et al., 2016), incluindo alguns parâmetros que podem ser utilizados nesta transição.

Uma das metas de manejo mais frequente em formações savânicas é diminuir o número de ocorrência e a extensão de incêndios através da redução de material combustível (Goldammer & de Ronde, 2004; Govender et al., 2006; Russell-Smith, 2016; Schmidt et al., 2018). Quando esta meta é exclusiva os resultados geralmente são limitados e questionáveis, por levar a simplificações da dinâmica ecológica em diferentes fisionomias e, consequentemente, perdas locais de biodiversidade em longo prazo. Um dos maiores incentivos para a expansão do programa de MIF no Cerrado tem sido reduzir o risco de incêndios por meio de queimas controladas no início da estação seca. Porém implementar queimas em outras épocas do ano, que beneficiem espécies chave da fauna ou flora, ou mesmo que permitam melhores resultados de manejo de combustível ou da redução do adensamento da vegetação lenhosa, pode ser o próximo passo do programa para corresponder melhor aos objetivos de conservação estabelecidos pelas UCs. . Outros países também priorizaram o manejo do combustível com queimas controladas em ecossistemas pirofíticos ao quebrarem o paradigma de exclusão do fogo em atividades de manejo, e com o tempo conseguiram aprimorar as técnicas e incluir novas metas de conservação, que incluem queimas em diferentes épocas do ano (Finney, 2001; Laris, 2002; Penman et al., 2011; Russell-Smith et al., 2009; Stephens & Ruth, 2005; Wardell et al., 2004) e até gerar renda para populações rurais (Murphy et al., 2015; Russell-Smith et al., 2007), especialmente os programas que não priorizam exclusivamente as vantagens econômicas e sim a sustentabilidade da interação socioambiental. Neste sentido, na fase piloto do MIF as instituições ambientais, como o Instituto Chico Mendes da Biodiversidade (ICMBio) e o Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais (IBAMA), estão priorizando as ações de manejo que tenham o menor impacto possível na conservação e funcionamento dos ecossistemas manejados e contribuam para a redução de conflitos socioambientais que frequentemente resultam em incêndios. Dentro dessas ações de manejo, as UCs participantes do MIF estão pela primeira vez conduzindo queimas controladas no início da estação seca em horários estratégicos para que o combustível seja fragmentado e a extensão de áreas queimadas por incêndios anualmente seja limitada, assim diminuindo o risco de perdas ambientais – evitando a queima de áreas indesejadas, especialmente aquelas de vegetação sensíveis ao fogo como as matas de galeria – as e perdas de bens das comunidades locais presentes no interior ou arredor dessas áreas.

Queimas no início da estação seca no final da tarde, em que as condições meteorológicas são mais amenas (temperatura do ar e velocidade do vento mais baixa e umidade relativa mais alta), tendem a ter velocidade de propagação e intensidade do fogo mais baixas e a consumir menor quantidade de combustível fino (< 6 cm de diâmetro) do que queimas de meio-dia no início da estação seca ou queimas no final da estação seca. Neste trabalho, as queimas de menor intensidade além de serem mais fáceis de controlar, muitas vezes se auto extinguindo durante a noite, causaram o mesmo impacto nas comunidades lenhosas do cerrado típico que queimas de meio-dia com umidade relativa menor que 50%. As áreas protegidas do fogo por pelo menos quatro anos foram as únicas capazes de aumentar o adensamento da vegetação lenhosa e permitir o recrutamento de indivíduos que atinjam maiores diâmetros (classe de tamanho >5 cm). De maneira geral, a intensidade do fogo tendeu a ser maior em intervalos de queima maiores (três anos) do que intervalos menores (dois anos), em que a intensidade de queimas em áreas com histórico de fogo de três anos no início da estação seca ao meio-dia com umidade relativa menor que 50% resultou em queimas de intensidade significativamente maior do que queimas em áreas com histórico de fogo de dois anos na mesma época do ano no final da tarde com umidade relativa maior que 50%. As queimas de maior intensidade no início da estação seca em condições meteorológicas mais severas (umidade relativa <50%) resultaram em taxa de sobrevivência e topkill das espécies arbóreas similares em ambos os intervalos de queima (dois e três anos). Já as queimas trienais no início da estação seca no final da tarde (umidade relativa >50%) resultaram em impactos significativamente menores do que as áreas queimadas bienalmente no final da estação seca. Porém, as áreas protegidas do fogo por mais de quatro anos foram as que apresentaram impacto significativamente menor na vegetação lenhosa do que as áreas submetidas a fogos mais frequentes.

De maneira preliminar e a curto-prazo, foram observadas potenciais mudanças na dinâmica e estrutura da vegetação arbórea do Cerrado no PNCM, como: a diminuição do número de indivíduos de lenhosos, o atraso no crescimento de plantas, especialmente das plantas menores (diâmetro <5 cm) como resultado de queimas no final da estação seca, que caracterizam o regime de fogo predominante na maioria das UCs do Cerrado (França, 2010; Pereira Júnior et al., 2014; Pivello, 2011). As áreas protegidas do fogo por mais de quatro anos foram as que apresentaram menor mortalidade de indivíduos arbóreos entre 1-5 cm de diâmetro em comparação a todos os tratamentos submetidos a queimas. Dentre os indivíduos arbóreos entre 5-10 cm de diâmetro, queimas bienais no final da estação seca apresentaram maior mortalidade e menor crescimento de indivíduos arbóreos do que as queimas bienais no início da estação seca e áreas protegidas do fogo por quatro anos. . Assim, este sistema de fogo de fim de tarde no início da estação seca, embora fracamente descrito na literatura voltada para o Cerrado (Hoffmann et al., 2012; Schmidt et al., 2016, 2018), mostrou-se, pelo menos em curto prazo, como uma alternativa para atingir os objetivos iniciais do MIF em UCs, como fragmentar o combustível usando queimas com velocidade de propagação menores de mais fácil controle e que causem os menores dano a vegetação lenhosa. Porém, estas práticas precisam ser adaptadas para atender também outros objetivos de conservação e manejo.

Uma vez que as queimas controladas são comumente implementadas por moradores locais em zonas rurais brasileiras ao longo da estação seca em diferentes horários (Batista et al., 2018; Borges et al., 2016; Eloy et al., 2018; Mistry et al., 2018; Moura, 2013), os resultados obtidos com esta pesquisa também podem contribuir na investigação sobre o comportamento e consequências deste diferentes tipos de fogo na vegetação arbórea. Esta tese foi desenvolvida para melhorar as informações disponíveis na academia, nas instituições ambientais e comunidades que utilizam o fogo como instrumento para o manejo da terra. Desta forma, esta pesquisa contribuiu: (i) para informar moradores do PNCM, gestores e brigadistas sobre a ecologia do fogo; (ii) na formação de alunos de graduação e pós-graduação que acompanharam o desenvolvimento do projeto em laboratório e nas práticas de campo; e (iii) no fomento de discussões voltadas tanto para o conhecimento técnico-científico como para o conhecimento tradicional/local-empírico.

Pesquisas voltadas para o manejo podem auxiliar na tomada de decisão, e na avaliação e adaptação de sistemas implementados por órgãos ambientais, desde que: sejam guiadas pelos praticantes; levem em consideração o conhecimento tradicional e científico e a experiência profissional dos gestores envolvidos; e considerem a realidade e especificidades da região estudada (Christensen, 2005; Driscoll et al., 2010; Van Wilgen et al., 2007). As recomendações sobre os tipos de queimas que devem ser implementadas pelo MIF dependem integralmente dos objetivos de cada UC e, consequentemente, das metas estabelecidas para cumpri-los. Após definir as metas e os alvos de conservação, é importante ressaltar que fatores como material combustível, intervalo de queima, precipitação anual, condições meteorológicas (determinadas também pelo horário do dia), época do ano e aceiros e barreiras naturais ou artificiais podem ser usados como parâmetros para realizar as queimas controladas e atingir os resultados desejados.

Embora esta pesquisa tenha sido desenvolvida em apenas uma UC, foi possível acompanhar vários tipos de queimas em escala de paisagem, abrangendo diversas comunidades vegetais e suas interações com as queimas, bem como dar retorno à equipe do PNCM e produzir informações valiosas para a continuidade e expansão do programa de MIF em UCs. Os resultados aqui presentes estão limitados a um período de monitoramento curto (três anos) e ao acompanhamento exclusivo de espécies arbóreas como os indicadores ecológicos dos experimentos. Portanto, pesquisas de longo-prazo que incluam outros indicadores ambientais e envolvam maiores escalas espaciais, embora difíceis, poderão contribuir ainda mais para aumentar a eficiência do manejo do fogo no Cerrado.

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