PUCRS

ESCOLA DE CIÊNCIAS DA SAÚDE E DA VIDA
PROGRAMA DE PÓS-GRADUAÇÃO EM ECOLOGIA E EVOLUÇÃO DA BIODIVERSIDADE
DOUTORADO EM ECOLOGIA E EVOLUÇÃO DA BIODIVERSIDADE

EDUARDO CHIARANI

VARIAÇÃO TEMPORAL NA COMUNIDADE E NA REPRODUÇÃO DE AVES CAMPESTRES EM ÁREAS QUEIMADAS NOS CAMPOS DE ALTITUDE NO SUL DO BRASIL

Porto Alegre
2020

PÓS-GRADUAÇÃO - *STRICTO SENSU*

Pontifícia Universidade Católica do Rio Grande do Sul

PONTIFÍCIA UNIVERSIDADE CATÓLICA DO RIO GRANDE DO SUL

ESCOLA DE CIÊNCIAS DA SAÚDE E DA VIDA

PROGRAMA DE PÓS-GRADUAÇÃO EM ECOLOGIA E EVOLUÇÃO DA BIODIVERSIDADE

VARIAÇÃO TEMPORAL NA COMUNIDADE E NA REPRODUÇÃO DE AVES CAMPESTRES EM ÁREAS QUEIMADAS NOS CAMPOS DE ALTITUDE NO SUL DO BRASIL

Eduardo Chiarani

Orientadora: Dra. Carla Suertegaray Fontana

TESE DE DOUTORADO

PORTO ALEGRE – RS - BRASIL

2020

SUMÁRIO

"*In the end,*

we will conserve only what we love,

we will love only what we understand,

and we will understand only what we are taught."

Baba Dioum

AGRADECIMENTOS

Em primeiro lugar agradeço aos meus pais, Alcides Chiarani e Ester Basegio Chiarani, que foram, sem dúvida, responsáveis pelo fato de hoje eu ser biólogo, ao me proporcionarem o contato com a Natureza desde muito cedo. Também sempre me apoiaram nas minhas escolhas e me ensinaram muito sobre a vida. Por isso, dedico esse trabalho a eles. Muito obrigado! Infelizmente meu pai não pode acompanhar a conclusão desse trabalho, mas sei que ficaria muito contente em ver mais essa etapa alcançada no estudo dos “passarinhos”. Saudades do meu grande companheiro de pescarias!

Agradeço muito à minha orientadora, Carla Suertegaray Fontana, pelos ensinamentos, pela compreensão e apoio nos momentos difíceis, pela confiança depositada e por ter me dado a oportunidade de trabalhar com ornitologia junto a sua equipe. Já se vão oito anos de amizade, projetos e aprendizado, e que venham outros oito.

Sou muito grato à Patrícia Behling Schafer pela ajuda crucial na correção do meu inglês “tupiniquim” e pelas sugestões dadas para a melhoria do texto. Com certeza, sem teu importante apoio, em tudo, seria muito mais difícil chegar até aqui. Muito obrigado!

Agradeço à minha colega Thaiane Weinert da Silva por todos esses anos de convívio muito agradável. Acabamos nos tornando grandes amigos. Foram inúmeras conversas sobre todos os assuntos possíveis ao longo dessa jornada. Obrigado pelas discussões ornitológicas e ajuda nas dúvidas que surgiam. Ah, e obrigado por rir das minhas piadas (sempre!) hehe.

Agradeço ao colega Maurício Bettio que contribuiu na coleta de dados que possibilitaram a elaboração do capítulo 1 dessa tese. Também agradeço ao colega Gabriel Guimarães Larre pela parceria em campo, compartilhando o alojamento em várias ocasiões, e aos colegas de laboratório pela amizade e parceria de sempre, Christian Andretti, Christian Beier, Ismael Franz, Jonas Rosoni, Márcio Repenning, Natália Porzio e Tiago Steffen. Nossas conversas contribuíram para o amadurecimento das ideias desse trabalho. Valeu, gurizada!

Obviamente, não poderia deixar de agradecer aos meus amigos, Alberto, Alceu, Jessé, Welinton e “seu Deca”, com quem convivi muito tempo durante o trabalho de campo no Parque Estadual do Tainhas. Obrigado pela agradável companhia e pelas inúmeras prosas nessa longa caminhada (e põe longa!). Sem a companhia de vocês não sei se teria sido possível ficar todo esse tempo em campo.

Obrigado à toda equipe administrativa do Parque Estadual do Tainhas, especialmente à Ketulyn Füster Marques e à Fabiana Pinto Bertuol, pela parceria e pela permissão de uso da base da UC, sem a qual não seria possível desenvolver um trabalho desses. Obrigado também à equipe do Parque Nacional de Aparados da Serra e Serra Geral, em especial ao Deonir Zimmermann e à Eridiane Lopes da Silva, que sempre deram apoio logístico ao trabalho.

Agradeço ao Sr. Paulo Machado, Sr. Valdir B. de Castilhos e Sr. Ernani por permitirem a realização de parte desse trabalho em suas propriedades e por toda hospitalidade serrana com que me receberam.

Agradeço aos professores Gerhard Overbeck (UFRGS), Miguel Ângelo Marini (UnB) e Pedro Maria de Abreu Ferreira (PUCRS) que gentilmente aceitaram o convite para serem membros da minha banca de defesa do doutorado. Agradeço também à professora Betina Blochtein e ao professor Julio Cesar Bicca-Marques pela contribuição dada ao trabalho como membros da banca de acompanhamento do doutorado. Sou grato, também, ao professor Felipe Zilio (UFV) que aceitou participar do processo de qualificação.

Gostaria também de agradecer à PUCRS que, através do PPG-Ecologia e Evolução da Biodiversidade, me deu a oportunidade de aperfeiçoar minha formação acadêmica. Agradeço a todos os professores desse PPG, pelo aprendizado que tive nesses anos todos. Também sou grato à Patricia Baptista, secretária do PPG, pela sempre pronta disposição em ajudar e esclarecer as dúvidas.

Por fim, agradeço ao CNPq pela bolsa de doutorado e pelo projeto que financiou boa parte desse trabalho, e à Neotropical Grassland Conservancy, que também contribui financeiramente para o desenvolvimento da pesquisa de campo. Sem esse apoio financeiro o presente doutorado não teria sido possível. Nesse sentido, agradeço a todos aqueles que apoiam e incentivam a pesquisa e a educação no Brasil e espero, sinceramente, que aqueles que ainda não enxergaram que só é possível melhorar o mundo através de avanços nessas áreas abram os olhos a tempo.

RESUMO

Os campos da América do Sul vêm sofrendo uma intensa transformação nas últimas décadas, principalmente devido à conversão do campo nativo em áreas de silvicultura ou lavouras. O uso correto do fogo, geralmente associado à pecuária, tem sido apontado como uma forma de manejo para conservação de áreas campestres em várias partes do mundo. O objetivo desse estudo foi avaliar as mudanças ocorridas nas comunidades, na reprodução e nos territórios de aves campestres ao longo do tempo, em áreas submetidas a diferentes históricos de distúrbio por fogo nos campos de altitude no sul do Brasil. As comunidades de aves foram amostradas através de pontos de contagem durante quatro temporadas (2015 – 2018), enquanto as buscas por ninhos e territórios de *Emberizoides ypiranganus* e *Sporophila melanogaster* ocorreram em três (2013, 2016 e 2017) e duas (2016 e 2017) temporadas reprodutivas, respectivamente. Fizemos comparações da riqueza, abundância e composição de espécies nos diferentes anos. As taxas de sobrevivência diária dos ninhos foram calculadas com o programa MARK e os territórios foram medidos com o estimador kernel 95%. A riqueza, abundância e composição de espécie variou significativamente na temporada em que ocorreu a queimada, retornando aos valores iniciais (ano anterior ao fogo) dois anos após o distúrbio. Em áreas queimadas anualmente ou em áreas sem fogo também houve diferenças entre as temporadas, mas sem um padrão claro. Ainda nas análises da comunidade, das seis espécies avaliadas individualmente quanto à densidade, três responderam significativamente ao tempo decorrido desde o fogo. Quanto à reprodução, nós monitoramos 237 ninhos (178 de *E. ypiranganus* e 59 de *S. melanogaster*). A probabilidade de sobrevivência e a produtividade dos ninhos de ambas espécies não mostraram diferenças significativas entre as temporadas reprodutivas, tanto na área queimada ocasionalmente como na área frequentemente queimada. No entanto, o tamanho e o número de territórios de *E. ypiranganus* variaram ao longo do tempo na área queimada ocasionalmente, havendo menos territórios e estes sendo menores na temporada do fogo. O número de territórios de *S. melanogaster* foi menor um ano após o fogo em comparação com a temporada em que ocorreu a queimada. Na área com fogo frequente não houve mudanças no número e no tamanho dos territórios medidos. Nosso estudo é o primeiro a abordar a dinâmica temporal dos efeitos do fogo sobre as comunidades e aspectos importantes da história de vida de aves campestres no sul do Brasil. Os resultados obtidos permitem saber o tempo que a comunidade de aves ou os parâmetros territoriais de aves campestres levam para se recuperar após um evento de fogo. Embora os resultados mereçam a devida cautela na sua extrapolação devido ao número de áreas que foi possível amostrar, informações sobre a frequência com que o fogo pode ser utilizado são importantes para o planejamento e a implementação de práticas que usam o fogo como uma ferramenta de manejo em áreas campestres. Estudos adicionais deveriam avaliar os efeitos de longo prazo causados pela mudança no regime de fogo, quando uma área onde não havia queimadas regulares passa a ser manejada com fogo ou quando esse distúrbio é excluído de uma área que era frequentemente queimada.

Palavras-chave: aves campestres, conservação, *Emberizoides ypiranganus*, manejo, sobrevivência de ninhos, *Sporophila melanogaster*, território

ABSTRACT

Temporal variation in bird community and breeding of grassland birds in burned areas in highland grasslands of southern Brazil

Grasslands of South America have been suffering an intense transformation in the last decades, mainly due to the conversion of native grasslands to agriculture or afforestation areas. The correct use of fire, often associated to cattle raising, has been pointed as a form of management to conserve grassland areas over the world. We aimed to assess changes in communities, breeding, and territories of grassland birds over the time, in areas under different histories of fire disturbance in highland grasslands of southern Brazil. Bird communities were sampled through point counts during four seasons (2015 – 2018), while search for nests and territories of *Emberizoides ypiranganus* and *Sporophila melanogaster* occurred in three (2013, 2016 e 2017) and two (2016 e 2017) breeding seasons, respectively. We made comparisons of richness, abundance and species composition in different years. Daily survival rates of nests were calculated with MARK program, and territories were measured using the 95% kernel estimator. Richness, abundance and species composition varied significantly in the season of the fire, returning to the initial values (year before the fire) two years after the disturbance. In areas occasionally burned or areas without fire there were differences among the seasons either, but not with an equally clear pattern. Still in the analysis on the communities, of the six species assessed individually for density, three responded significantly to time since the fire. Considering breeding, we monitored a total of 237 nests (178 of *E. ypiranganus* and 59 of *S. melanogaster*). Cumulative survival probability and productivity of the nests of both species did not show statistically significant differences among breeding seasons, both in the area burned occasionally and in the area frequently burned. However, the size and number of territories of *E. ypiranganus* varied over time in the area burned occasionally, with small and less territories in the season of the fire. The number of territories of *S. melanogaster* was lower one year after the fire in comparison with the season of the fire. In the area with frequent fire there were no changes either in the number or in the size of the measured territories. Our study is the first one to address the temporal dynamics of the effects of fire on communities and important aspects of natural history of grassland birds in southern Brazil. The results allow to know the time taken for the bird community or the territorial parameters of grassland birds to recover after a fire disturbance. Although the results deserve caution in their extrapolation due to the number of areas that it was possible to sample, information about how often fire can be used is important for planning and implementing practices that use fire as a management tool in grassland areas. Further studies should assess long-term effects caused by changes in fire regime, when an area where there was no regular fire becomes managed with fire or when this disturbance is excluded from an area that was often burned.

Keywords: conservation, *Emberizoides ypiranganus*, grassland birds, management, nest survival, *Sporophila melanogaster*, territory

APRESENTAÇÃO

No sul do Brasil, aproximadamente 25% dos campos foram perdidos em decorrência de mudanças no uso da terra nas últimas três décadas, causadas, principalmente, por atividades antrópicas como agricultura e silvicultura (Overbeck et al. 2007). Esta perda e degradação dos habitats têm consequências diretas sobre a avifauna associada a áreas abertas, levando a declínios populacionais, como já observado em estudos realizados em diferentes partes do mundo (e.g., Di Giacomo e Di Giacomo 2004, Donald et al. 2006, Askins *et al.* 2007). Apesar disso, pouca importância é dada aos campos, e nem sempre as leis atuais são cumpridas para reverter a perda dos ecossistemas não-florestais no Brasil, pois as políticas de conservação no país estão fortemente voltadas para os biomas florestais (Overbeck et al. 2015). Exemplo disso é o grau de proteção dos campos no sul do Brasil, onde menos de 0,5% de sua área está protegida em unidades de proteção integral (Overbeck et al. 2007, Pillar e Vélez 2010). Os campos de altitude do sul do Brasil, localizados no nordeste do Rio Grande do Sul e nos estados de Santa Catarina e Paraná muitas vezes são negligenciados por estarem localizados em encraves de vegetação campestre no bioma Mata Atlântica e, por esse motivo, também merecem atenção. Esses campos possuem uma relação histórica com distúrbios como o fogo e o pastejo (Behling e Pillar 2007), sendo tradicionalmente manejados com fogo no final do inverno, no intuito de queimar a biomassa acumulada e estimular o rebrote da vegetação para alimentação do gado (Overbeck e Pfadenhauer 2007, Andrade et al. 2019). Sabe-se que na ausência desses distúrbios as formações florestais tendem a avançar sobre as áreas de campo, levando à descaracterização e perda das formações campestres (Overbeck et al. 2005, Behling e Pillar 2007, Buisson et al. 2018).

Por outro lado, distúrbios podem impactar negativamente algumas espécies da fauna. O sobrepastejo, pisoteio pelo gado e queimadas anuais dos campos são apontados como ameaças a espécies campestres no Rio Grande do Sul (Fontana et al. 2003). O fogo também pode ter efeitos negativos diretos (destruindo ninhos) ou indiretos (aumentando as taxas de predação e parasitismo) na reprodução de aves campestres (Rohrbaugh et al. 1999, Reinking 2005).

Diante disso, entender os efeitos do fogo sobre a biodiversidade é fundamental para subsidiar ações de conservação que utilizem o fogo como uma ferramenta de manejo de áreas campestres. Abordagens que avaliem a dinâmica temporal dos efeitos do fogo sobre as aves são de grande necessidade, dada a carência desses estudos na América do Sul. Portanto, desenvolvemos o primeiro estudo abordando a questão temporal do fogo em relação à reprodução de aves nos campos no sul do Brasil. O trabalho também configura o primeiro estudo que avaliou comunidades de aves ao longo de quatro anos na região, com dados desde o período anterior ao fogo até dois anos após o distúrbio. Assim, nossos resultados contribuem para o conhecimento dos efeitos do fogo na reprodução e na comunidade de aves, auxiliando na tomada de decisões para o planejamento de ações de manejo em áreas campestres.

A tese está dividida em dois capítulos, os quais estão estruturados na forma de artigos científicos. No primeiro capítulo (artigo 1), avaliamos os parâmetros da comunidade de aves (riqueza, abundância e composição) em áreas com diferentes históricos de distúrbio provocado por fogo, buscando verificar quais mudanças ocorrem ao longo do tempo e qual o período necessário para recuperação desses parâmetros após uma queimada. Esse artigo foi submetido para publicação no periódico *PloS ONE*, Qualis Capes A1. Além disso, os resultados parciais deste artigo foram apresentados no XXVI Congresso Brasileiro de Ornitologia, em 2019. O segundo capítulo (artigo 2) avalia os efeitos temporais do fogo na reprodução e nos territórios de duas espécies de aves associadas a campos altos (*Emberizoides ypiranganus* e *Sporophila melanogaster*). Este artigo será submetido para publicação no periódico *Ibis*, Qualis Capes A1. Os resultados parciais foram apresentados no XXV Congresso Brasileiro de Ornitologia, em 2018, onde o trabalho recebeu o prêmio Helmut Sick (2º lugar) na categoria apresentação oral pós-graduação.

Literatura citada

Andrade, B. O., C. L. Bonilha, G. E. Overbeck, E. Vélez-Martin, R. G. Rolim, S. A. L. Bordignon, A. A. Schneider, C. Vogel Ely, D. B. Lucas, É. N. Garcia, E. D. dos Santos, et al. (2019). Classification of South Brazilian grasslands: Implications for conservation. Applied Vegetation Science.

Askins, R. A., F. Chávez-ramírez, B. C. Dale, C. A. Haas, J. R. Herkert, F. L. Knopf, e P. D. Vickery (2007). Conservation of Grassland Birds in North America: Understanding Ecological Processes in Different Regions. Ornithological Monographs 64:1–46.

Behling, H., e V. D. P. Pillar (2007). Late Quaternary vegetation, biodiversity and fire dynamics on the southern Brazilian highland and their implication for conservation and management of modern Araucaria forest and grassland ecosystems. Philosophical Transactions of the Royal Society B: Biological Sciences 362:243–251.

Buisson, E., S. Le Stradic, F. A. O. Silveira, G. Durigan, G. E. Overbeck, A. Fidelis, G. W. Fernandes, W. J. Bond, J. Hermann, G. Mahy, S. T. Alvarado, et al. (2018). Resilience and restoration of tropical and subtropical grasslands, savannas, and grassy woodlands.

biological reviews. https://doi.org/10.1111/brv.12470

Di Giacomo, A. S., e G. Di Giacomo (2004). Extinción, historia natural y conservación de las poblaciones del Yetapá de Collar (Alectrurus risora) en la Argentina. Ornitologia Neotropical 15:145–157.

Donald, P. F., F. J. Sanderson, I. J. Burfield, e F. P. J. Van Bommel (2006). Further evidence of continent-wide impacts of agricultural intensification on European farmland birds 1990–2000. Agriculture, Ecosystems and Environment 116:189–196.

Fontana, C. S., G. A. Bencke, e R. E. Reis (2003). Livro vermelho da fauna ameaçada de extinção no Rio Grande do Sul. Edipucrs, Porto Alegre, Brazil.

Overbeck, G. E., S. C. Müller, V. D. Pillar, e J. Pfadenhauer (2005). Fine-scale post-fire dynamics in southern Brazilian subtropical grassland. Journal of Vegetation Science 16:655–664.

Overbeck, G. E., e J. Pfadenhauer (2007). Adaptive strategies in burned subtropical grassland in southern Brazil. Flora: Morphology, Distribution, Functional Ecology of Plants 202:27–49.

Overbeck, G. E., E. Vélez-Martin, F. R. Scarano, T. M. Lewinsohn, C. R. Fonseca, S. T. Meyer, S. C. Müller, P. Ceotto, L. Dadalt, G. Durigan, G. Ganade, et al. (2015). Conservation in Brazil needs to include non-forest ecosystems. Diversity and Distributions 21:1455–1460.

Overbeck, G., S. Muller, A. Fidelis, J. Pfadenhauer, V. Pillar, C. Blanco, I. Boldrini, R. Both,e E. Forneck (2007). Brazil's neglected biome: The South Brazilian Campos. Perspectives in Plant Ecology, Evolution and Systematics 9:101–116.

Pillar, V. D. P., e E. Vélez (2010). Extinção dos Campos Sulinos em Unidades de Conservação: um Fenômeno Natural ou um Problema ético? Natureza & Conservação 08:84–86.

Reinking, D. L. (2005). Fire regimes and avian responses in the central tallgrass prairie. Studies in Avian Biology 30:116–126.

Rohrbaugh, R. W., J. R. Dan, L. R. Donald, H. W. Steve, e M. A. Jenkins (1999). Effects of prescribed burning and grazing on nesting and reproductive success of three grassland passerine species in tallgrass prairie. Studies in Avian Biology No.:165–170.

CAPÍTULO 1

Temporal changes in bird communities in areas with different histories of fire disturbance in highland grasslands of Brazil

Artigo submetido para publicação no periódico *PLOS ONE*

Temporal changes in bird communities in areas with different histories of fire disturbance in highland grasslands of Brazil

Eduardo Chiarani[1*], Maurício Bettio [1,#a], Carla S. Fontana[1]

[1] Laboratório de Ornitologia, Museu de Ciências e Tecnologia, Programa de Pós-Graduação em Ecologia e Evolução da Biodiversidade, Pontifícia Universidade Católica do Rio Grande do Sul, Porto Alegre, Rio Grande do Sul, Brazil

[#a] Current Address: Biometria Consultoria e Projetos, Caxias do Sul, Rio Grande do Sul, Brazil

* Corresponding author

E-mail: educhiarani@gmail.com (EC)

Abstract

Despite the importance and ubiquity of grasslands, the degradation and loss of these habitats has negatively affected bird populations throughout the world. The use of fire to manage grassland areas in some regions of southern Brazil can help to maintain these areas but can also influence the bird community in different ways. We assessed temporal changes in richness, abundance, and composition of bird communities in areas with different histories of fire disturbance in highland grasslands of southern Brazil, the most extensive remnant of grassland of the Atlantic Forest biome. We censused birds during four breeding seasons (2015–2018), through point counts in areas burned only once in the last ten years (OF, n = 3), areas burned annually (AF, n = 2), and areas without fire in the last ten years (WF, n = 2). In OF the richness, abundance, and species composition changed in the year of the fire, compared to the previous year, and returned to the initial values two years later. In AF and WF we found some differences among the years, but not with an equally clear pattern. Three of the six grassland species assessed individually for density responded significantly to temporal habitat modification caused by fire. Our results show that two years without fire were enough time for the bird community to recover after a fire, but some responses are species-specific. Therefore, fire can be used as a management tool for grasslands and may help in the conservation of birds of southern Brazil, as long as with a minimum interval between fires in an area is guaranteed.

Keywords: abundance, bird community, composition, disturbance, fire, southern Brazilian grasslands

Introduction

Grasslands occupy about 13.7 million ha of South Brazil and have been undergoing extensive transformation in recent decades, mainly from conversion of native grassland areas to agriculture or afforestation [1,2]. The degradation and loss of grassland areas have negatively affected bird populations throughout the world [1]. The biodiversity in southern Brazilian highland grasslands, located in northeastern Rio Grande do Sul and the states of Santa Catarina and Paraná, has also been impacted by anthropic actions [3–5]. This region houses about 70% of the bird species associated with grassland landscapes in southeastern South America, including endemic, migratory and/or threatened species [1,6–8].

The two main ways of managing grasslands are fire and grazing, and these disturbances are usually associated with cattle raising, an activity that allows to combine production and conservation in southern Brazil [3,9,10]. In southern Brazilian highland grasslands, fire is traditionally used to manage them at the end of winter, in order to burn the accumulated biomass and stimulate the regrowth of vegetation for cattle feed [11,12]. In the last years laws have been introduced in order to allow the use of controlled fire in grasslands in some municipalities of northeastern Rio Grande do Sul state, but this decision is usually based more on political and cultural issues than on scientific studies that assess the impacts of fire on animal and plant communities. Although the use of fire is already recommended as a management tool for protected areas in Brazil, its application is not yet a reality in these areas, and this issue is still a taboo [13,14].

It is known that in the absence of any disturbance (e.g., grazing or fire), grasslands show a high dominance of a few species of caespitose grasses (that form tussocks) and a low diversity of forbs, resulting in a homogenization of the vegetation structure [2,15] and a consequent reduction of the bird diversity [16,17]. In the long term, in abandoned grasslands (i.e. long periods without grazing or fire), the floristic richness can be reduced and the grassland vegetation itself can be lost due to encroachment of shrubs [12,18,19]. A recent study in highlands of southern Brazil estimated in 30 years the time needed for shrubs to encroach into 99% of grasslands without management [20]. Fire exclusion also leads to the accumulation of flammable biomass and, consequently, can increase fire intensity and risk of catastrophic fire [13,19,21]. However, when in excess (e.g., overgrazing, cattle trampling and high frequencies of fire) these disturbances can cause the decline of threatened bird species, mainly those dependent of tall grasslands, as observed for Rio Grande do Sul state in Brazil [5,22]. Thus, the threshold between sustainable use and degradation seems to be subtle when we consider fire is a factor in maintaining the integrity of grasslands [9].

In North America, the probability of occurrence of several grassland bird species has decreased significantly in areas where the coverage of tall shrubs and trees has increased [23]. Some species increase in density in recently burned areas, and may be excluded from unburned areas [24–26]. For other species, the reductions in the abundance of individuals and the number of nests suggest that fire has a strong negative effect [26,27]. Since fire can directly (nest destruction) or indirectly (changes in vegetation structure) affect the bird community in different ways [28], studies that assess the effects of burning are necessary for the proper management of grassland areas.

In South America most of the studies about fire effects on birds in non-forest habitats have been conducted in Argentina (e.g., [29–32]) or in Central Brazil (e.g., [33–36]). Few studies have specifically assessed this issue in southern Brazilian highland grasslands, and most have only compared burned and unburned areas (e.g., [37–39]). Despite temporal scale strongly influences both the ecosystem responses to fire and the effects of fire [40], long-term studies are uncommon, and temporal aspect of the effects of certain variables (e.g., intensity, period and frequency of fire) on bird response to fire is seldom discussed [41], particularly in

Neotropical region. Therefore, approaches that evaluate temporal dynamics of fire effects, including pre- and post-fire periods, are needed to assess possible changes associated with fire in local populations and bird community. Our study is the first that considers the temporal dynamics of fire on bird community in highland grasslands of South Brazil.

Here, we assessed the parameters of the grassland bird community (richness, abundance and composition) over four breeding seasons, in areas with different histories of fire disturbance, aiming to determine the effects of fire on grassland birds over time. We hypothesized that: (1) Bird community structure in areas with a constant history of management over time (i.e. annual fire or without fire) does not change temporally, or may present a different variation pattern when compared with areas burned occasionally; (2) Richness, abundance and bird composition changes after an occasional fire and return to pre-disturbance levels as time goes by; and (3) Since we know that birds respond differently to variations in habitat structure caused by fire [28,30], and differences in vegetation height is an important driver of species sorting [42], the predict abundance of tall-grass species will reduce and the abundance of short-grass species will increase in grasslands in the year of the fire.

Materials and methods

Study area

The study areas are situated in the southern Brazilian highland grasslands (*sensu* [14]), in the Atlantic Forest biome. We selected seven areas of native grassland, in three protected areas (Parque Nacional de Aparados da Serra, Parque Nacional da Serra Geral and Parque Estadual do Tainhas) and on nearby farms, in northeastern Rio Grande do Sul state (Fig 1). We categorized areas according their histories of fire disturbance: (1) occasional fire (OF) – areas burned (accidentally) only once in the last ten years (n = 3); (2) annual fire (AF) – areas burned annually (n = 2); and (3) without fire (WF) – areas without fire in the last ten years (n = 2). The fires occurred between August and October in the study areas, even those burned accidentally. All areas (including the protected areas) are used for cattle raising and have similar low grazing pressure (0.3 to 0.5 animal unit per hectare). Our study areas are composed mainly by large extensions of native grasslands interspersed with marshes, rocky outcrops, and patches of *Araucaria* forests (*Araucaria angustifolia*). Surrounding areas also have these features and, in addition, usually have exotic pine (*Pinus* spp.) plantations and crops (mainly maize, potatoes, and soybean).

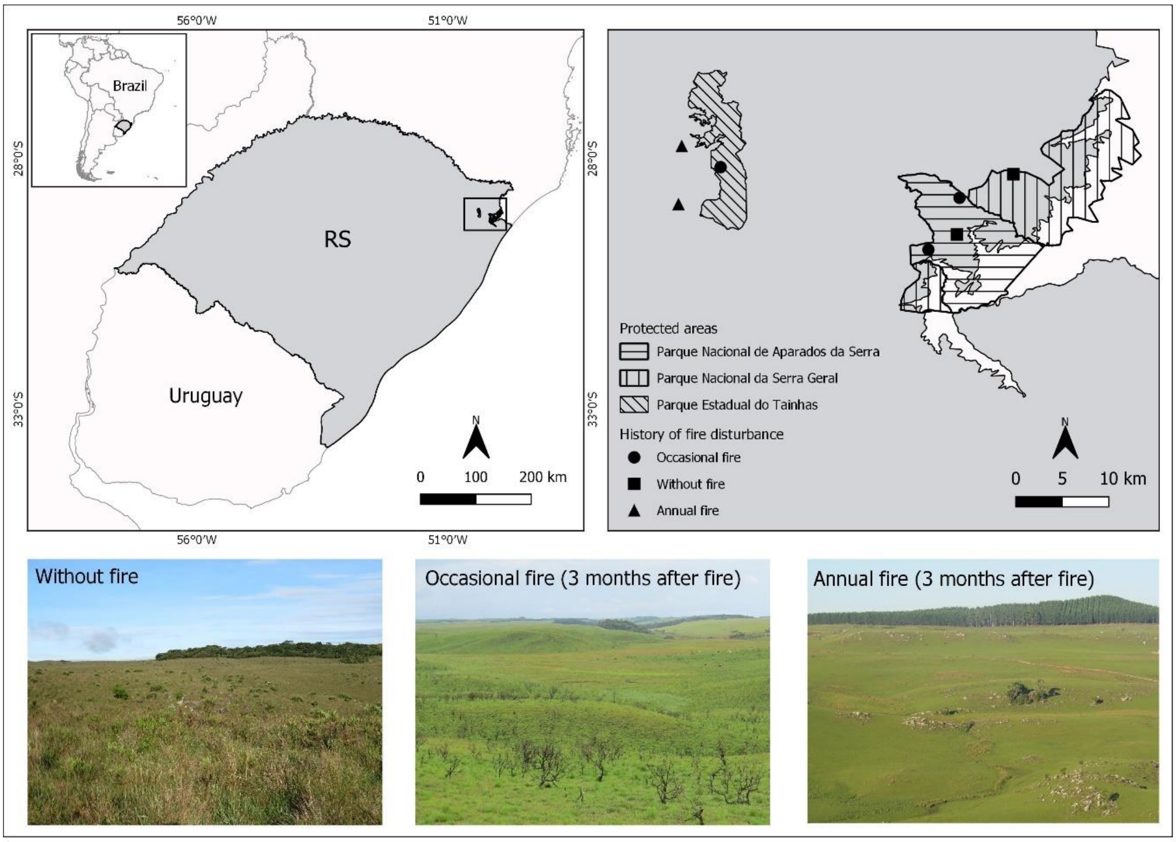


Fig 1. Locations of the study areas in highland grasslands in northeastern Rio Grande do Sul (RS) state, Brazil. Pictures are examples of landscapes of the three histories of fire disturbance in our study areas.

The landscape of the study areas resembles typically the grasslands in the Atlantic Forest biome of South Brazil. Southern Brazilian highland grasslands cover about 60,000 km^2 and consist of a mosaic of grasslands and *Araucaria* forest, together with other vegetation types in a minor contribution, such as wetlands and peat bogs, with an undulating relief and mean altitude of 900–1,000 m a.s.l. [1,5,12,14]. These grasslands are characterized by dominance of perennial grass species in terms of cover and by high levels of endemism and high overall species richness, where Poaceae (mostly represented by tussock species such as *Andropogon lateralis*, *Axonopus siccus*, and *Schizachyrium tenerum*), Asteraceae, Fabaceae and Cyperaceae are the main families in species number [14,43]. The mean annual temperature in the region ranges from 16 °C to 22 °C and the precipitation is evenly distributed throughout the year (1,500–2,000 mm), reaching up to 2,500 mm in certain subregions [44,45].

Data collection

Bird surveys were conducted during four consecutive breeding seasons (2015–2016 to 2018–2019), between November and February, the breeding period of most species in the region [46]. Birds were recorded in point counts of 10 min with an 80 m radius [47]. All individuals sighted and/or heard inside the circle were counted and their distances from the observer were estimated. Samplings were carried out from dawn to 10:00 a.m., in suitable weather conditions (without rain and with wind less than 10 km/h). The number of point counts was distributed according to the size and availability of the area in each breeding season (S1 Table), with a proportional number of points located in dry grasslands and near wetlands in each area. The minimum distance between two point-count centers was 300 m, and they were sited in open areas at a minimum distance of 150 m from the edges of other vegetation types (e.g., forests) or from fences. Each point was sampled twice per breeding season (see statistical analysis) in order to record the bird richness more accurately. The procedures adopted to avoid double counts and ensure independence among counts were: (1) sampling was conducted only in the breeding season, when individuals tend to keep their breeding territories; (2) minimum distance of 300 m between point counts; (3) counting only birds using the area within the point radius, excluding those merely flying over the area; (4) not counting groups of birds that move over large distances or are difficult to count (e.g., swallows, swifts, and birds of prey).

We sampled vegetation (height) and ground-cover variables (percentage of vegetation cover, bare ground, rocks, and water) of each point in four quadrats (1 × 1 m). These quadrats were placed at different distances from the point count: 10 m to the north, 25 m to the west, 50 m to the south, and 75 m to the east. Vegetation height was measured at five points in each quadrat (at the center and at the four vertices). These data were grouped as mean values at the point-count level. The vegetation was sampled at the end of each breeding season, immediately after the end of bird counts.

Statistical analysis

Differences in bird species richness between breeding seasons were assessed through the rarefaction and extrapolation method, based on sample coverage [48]. This method allows comparisons of richness based on samples with the same coverage (completeness) rather than the same size, which would be an advantage when comparing areas or years with very different degrees of diversity [48]. The species richness was calculated for each breeding season, based on the lowest sample coverage among the four values obtained in each history of disturbance. The 95% confidence intervals were obtained with 999 iterations by bootstrap resampling. Significant differences at the 5% level are guaranteed when the confidence intervals do not overlap [48]. This analysis was performed with the *iNEXT* package, using the *estimateD* function [49,50].

We used generalized linear mixed models (GLMM) with the Poisson error distribution to test for differences in bird abundance and vegetation height between breeding seasons. For the bird abundance analysis, we used only the maximum number of individuals recorded in the two samples taken in each point count per breeding season, to avoid overestimates caused by re-counting the same individual. We created models for each history of fire disturbance (occasional fire, annual fire, and without fire) separately, in order to assess only the temporal changes within each history, not among them. In all models the year was considered as fixed effect, while areas and point counts were treated as random effects, to control spatial and temporal variations, considering the dependence in our data. The model analyses were carried out in the *lme4* package using the *glmer* function [51]. The significance of the fixed effect (year) was assessed via likelihood-ratio tests, with an ANOVA between the model with the explanatory variable (fixed effect) and the model without this variable (null model) [51,52]. Differences among the four breeding seasons were evaluated via post-hoc Tukey pairwise comparisons, using the *multcomp* package [53].

In order to determine the responses of species to the time since fire (i.e. years since the last burn), we initially estimated the density (individuals/ha) of some species in occasional-fire areas, because only these areas had changes in their history of disturbance during our study. We used the Distance 7.3 program to adjust detectability issues [54]. Only species with more than 30 records were included in this analysis [55]. The estimates of density generated were tested for normality via a Shapiro-Wilk test, with the *RVAideMemoire* package [56], showing normal distribution. Therefore, we used linear mixed models (LMM) to assess the effect of the time since fire on the density of each species. In all models the year was considered as fixed effect and the areas as random effect, accounting for temporal dependence in the data (same areas sampled in four consecutive breeding seasons/years). The model analyses were carried out in the *lme4* package, using the *lmer* function [51]. Again, the significance of the fixed effect was evaluated using an ANOVA between the model with the explanatory variable and the null model (without this variable), and differences among breeding seasons were evaluated via post-hoc Tukey pairwise comparisons.

In order to test for differences in the composition of the bird community between breeding seasons, for each history of fire disturbance, we used permutational multivariate analysis of variance (PERMANOVA) with 999 iterations, using the *Adonis* function in the *vegan* package [57]. We used post-hoc Tukey pairwise comparisons to assess differences between seasons, using the *pairwiseAdonis* package [58]. We plotted only the results for occasional-fire areas in order to show associations between bird species and time since fire, through a non-metric multidimensional scaling (NMDS), using the Bray-Curtis index as a dissimilarity measure, with the *metaMDS* function in the *vegan* package. Environmental variables were then fitted in the NMDS plot and their statistical significance tested with 999 permutations, using the *envfit* function in *vegan*. For this analysis we used only grassland-dependent bird species (*sensu* [1]). All analyses were performed using R 3.4.0 [59].

Results

Bird richness and abundance

We recorded 73 bird species and 2,828 individuals during the study (S2 Table). Of this total, 59 species and 1,325 individuals were recorded in areas with occasional fire (OF), 58 species and 1,093 individuals in areas burned annually (AF), and 34 species and 410 individuals in areas without fire (WF). Eight species were exclusive to OF, nine to AF, and four to WF. Seven other species occurred in areas with all histories of fire disturbance and in all breeding seasons. The Ochre-breasted Pipit (*A. nattereri*) and the Saffron-cowled Blackbird (*X. flavus*), two species of conservation concern, did not occur in areas without fire or in the breeding season before the burn (in the case of OF), while in areas burned annually, these species were recorded in all four breeding seasons sampled.

The high values of estimated sample coverage (ranges from 0.94 in WF to 0.98 in OF and AF) indicate that the sampling was sufficient to detect most species. Considering the lowest sample coverage of each history of fire disturbance no overlap in confidence intervals showed a significant difference in richness over the breeding seasons in OF and AF (Table 1, Fig 2). In the OF areas, richness was higher in the year of the fire (47 species) and one year post-fire (46 species), compared to the year before the burn (31 species) and two years after the burn (34 species). In the AF areas, richness was higher in the third year (2017: 47 species) than in the others (2015: 33 species; 2016: 36 species; 2018: 37 species). In the WF areas, bird richness did not vary significantly during the four years.

Table 1. Bird richness in areas with different histories of fire disturbance during four breeding seasons (2015–2018) in highland grasslands in northeastern Rio Grande do Sul state, Brazil.

History / Breeding season	S.obs	SC	S	CI
Occasional fire / before	32	0.98	31	27.9 – 33.8 [A]
Occasional fire / fire	49	0.98	47	43.0 – 51.9 [a]
Occasional fire / 1 year post-fire	47	0.98	46	41.4 – 50.4 [a]
Occasional fire / 2 years post-fire	34	0.97	34	30.6 – 37.4 [A]
Annual fire / 2015	33	0.96	33	29.6 – 36.4 [B]
Annual fire / 2016	39	0.98	36	33.6 – 39.3 [B]

History / Breeding season	S.obs	SC	S	CI
Annual fire / 2017	47	0.97	47	42.3 – 51.0 [b]
Annual fire / 2018	38	0.97	37	33.4 – 41.2 [B]
Without fire / 2015	19	0.94	19	15.8 – 22.2 [C]
Without fire / 2016	17	0.97	16	13.5 – 18.0 [C]
Without fire / 2017	20	0.95	19	15.9 – 22.2 [C]
Without fire / 2018	16	0.96	15	12.3 – 16.9 [C]

S.obs = observed richness; SC = sample coverage; S = richness based on the lowest sample coverage for that history of disturbance; CI = 95% confidence interval. Differences between letters (upper and lower case) next to CI of a same history (A/a for occasional fire, B/b for annual fire, C/c for without fire) indicate significant differences between breeding seasons.

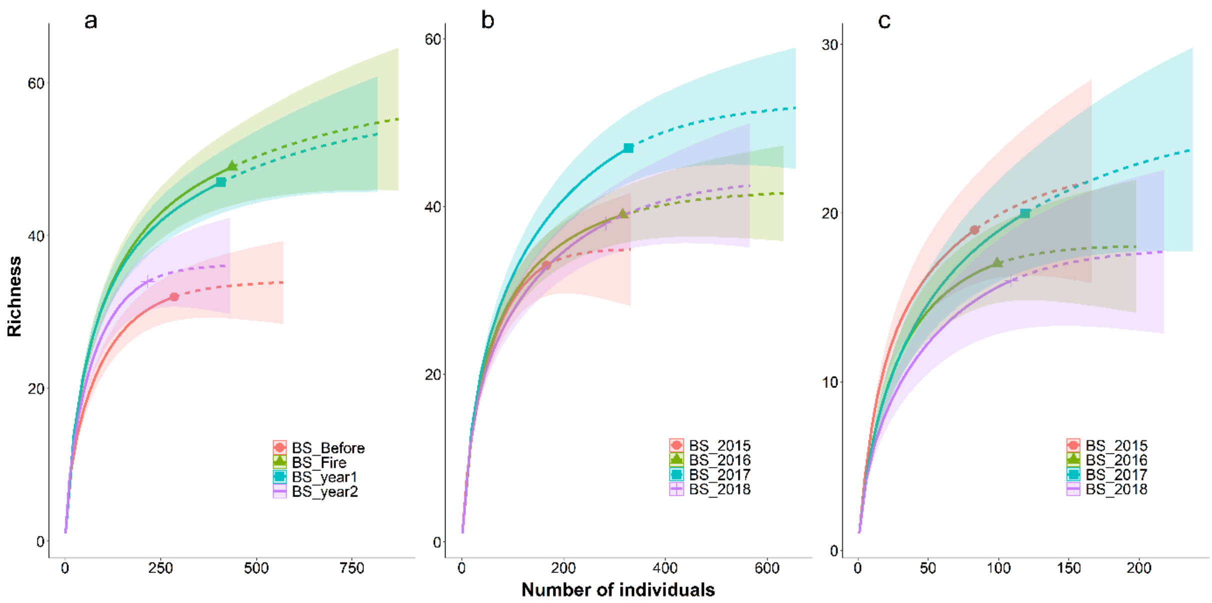


Fig 2. Species richness of birds recorded during four breeding seasons (BS) in grasslands with three histories of fire disturbance in northeastern Rio Grande do Sul state, Brazil. Histories of fire disturbance are occasional fire (a), annual fire (b), and without fire (c). In occasional fire breeding seasons refer to the year before the burn (before), in the year of the

burn (fire), and one and two years post-fire (year1 and year2, respectively). Solid and dashed lines are interpolated and extrapolated data, respectively, based on rarefaction and extrapolation method, with their associated 95% confidence intervals.

There was significant temporal variation in bird abundance in OF (χ^2 = 16.05; p = 0.001). Post-hoc tests showed that the number of individuals recorded increased in the year of the fire (Z = -3.76; p = 0.001), and did not differ in the following years (1 year post-fire: Z = -2.39; p = 0.07; 2 years post-fire: *Z* = -1.01; p = 0.74) in relation to the year before the disturbance (Fig 3). Bird abundance also changed significantly in AF (χ^2 = 8.89; p = 0.03) and WF (χ^2 = 10.31; p = 0.02). In AF, the number of records differed significantly only between the first and third years (Z = 2.79; p = 0.02), while in WF the number of records differed significantly between the second and third years (Z = -2.7; p = 0.03).

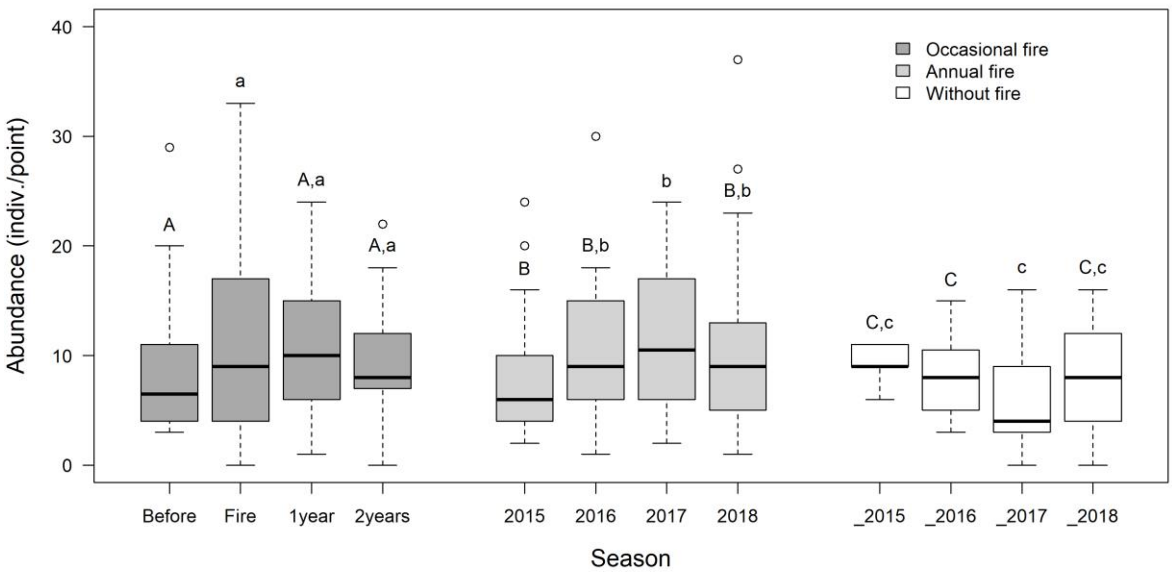


Fig 3. Abundance of birds recorded during four breeding seasons (2015 – 2018) in grasslands with three histories of fire disturbance in northeastern Rio Grande do Sul state, Brazil. In occasional fire breeding seasons refer to the year before the burn (before), in

the year of the burn (fire), and one and two years post-fire (1 year and 2 years, respectively). Differences between letters (upper and lower case: A/a for occasional fire, B/b for annual fire, C/c for without fire) indicate significant differences between breeding seasons, based on generalized linear mixed models. The line inside each box represents the median; top and bottom of each box represent upper and lower quartiles, respectively; whiskers represent maximum and minimum values; circles are outliers.

Considering the six species analyzed for density (for which the minimum number of records was obtained), three of them responded significantly in relation to time since fire (Fig 4). The time since fire positively affected the density of the Lesser Grass-Finch (*Emberizoides ypiranganus*; $\chi^2 = 8.03$; $p = 0.04$). Post-hoc tests showed that the density of the species was significantly higher one year post-fire ($Z = 2.89$; $p = 0.02$) and before the burn ($Z = 3.35$; $p = 0.004$) compared to the year of the disturbance. The density of the Lesser Grass-Finch did not vary between the year before the fire and one year ($Z = 0.45$; $p = 0.97$) or two years post-fire ($Z = 0.81$; $p = 0.85$). For Hellmayr's Pipit (*Anthus hellmayri*), the time since fire negatively affected its density ($\chi^2 = 8.04$; $p = 0.04$). The density of the species was significatively higher in the year of the fire than in the year before the disturbance ($Z = -3.24$; $p = 0.007$) and two years post-fire ($Z = -2.74$; $p = 0.03$). The density of the Hellmayr's Pipit did not differ between the year before the fire and one year ($Z = -1.02$; $p = 0.71$) or two years post-fire ($Z = -0.14$; $p = 0.99$). Similarly, the time since fire negatively affected the density of the Rufous-collared Sparrow (*Zonotrichia capensis*; $\chi^2 = 7.88$; $p = 0.04$). The density of the Rufous-collared Sparrow was significatively higher in the year of the fire than in the year before the disturbance ($Z = -3.3$; $p = 0.005$), while there was no difference between the year before the fire and one year ($Z = -0.98$; $p = 0.76$) or two years post-fire ($Z = -1.26$; $= 0.58$). For the other three species, their densities not changed with the time since fire (*S. melanogaster*: $\chi^2 = 1.98$; $p = 0.58$; *E. platensis*: $\chi^2 = 0.53$; $p = 0.91$; *S. luteola*: $\chi^2 = 4.23$; $p = 0.24$).

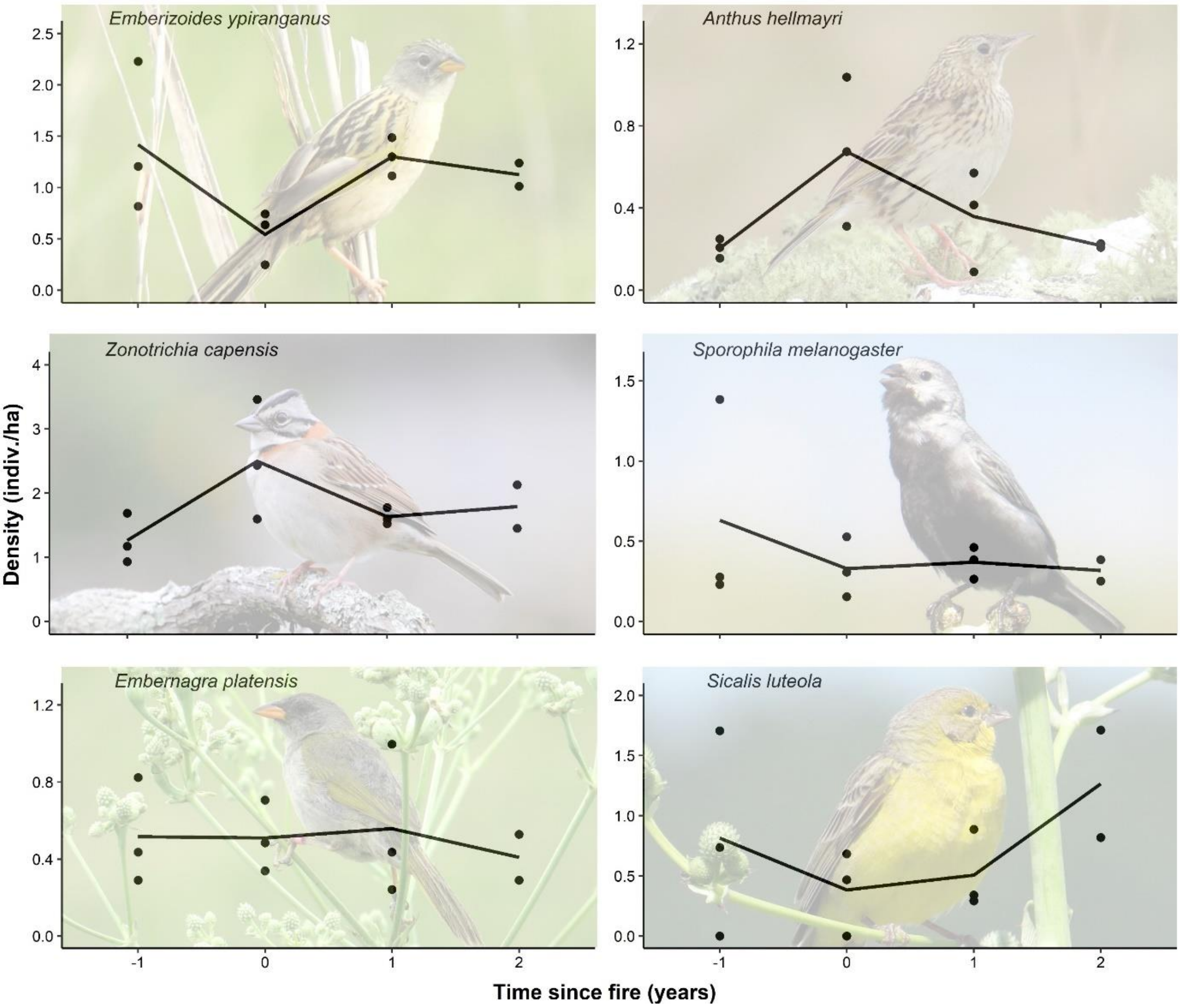


Fig 4. Density of six grassland birds one year before ("-1"), just following ("0"), and for two years after fire in highland grasslands in northeastern Rio Grande do Sul state, Brazil. Line represents the mean density of the species in each year. Only densities of *E. ypiranganus*, *A. hellmayri*, and *Z. capensis* varied significantly over time ($p < 0.05$), based on linear mixed models.

Species composition and environmental variables

The species composition of grassland birds changed over time in OF ($F = 3.99$, df = 3 and 121; $r^2 = 0.09$; $p = 0.001$). Post-hoc tests revealed significant differences between the year of the burn and the other three breeding seasons (before: $r^2 = 0.11$; $p_{adj} = 0.006$; one year post-fire: $r^2 = 0.05$; $p_{adj} = 0.03$; two years post-fire: $r^2 = 0.10$; $p_{adj} = 0.006$). Temporal changes also occurred in the species composition in AF ($F = 1.73$, df = 3 and 93; $r^2 = 0.05$; $p = 0.02$) and in WF ($F = 2.93$, df = 3 and 46; $r^2 = 0.16$; $p = 0.002$). In AF areas, the species composition in the first year was significantly different from the third ($r^2 = 0.06$; $p_{adj} = 0.02$) and fourth year ($r^2 = 0.07$; $p_{adj} = 0.01$). In the WF areas, there were significant differences between the first and third years ($r^2 = 0.13$; $p_{adj} = 0.04$) and between the third and fourth years ($r^2 = 0.12$; $p_{adj} = 0.02$).

As shown by the average of the breeding season scores (centroids), and reflecting the results of the tests in OF, the post-fire years (1 and 2 years) were more closely related to the year before the burn than to the year of the fire (Fig 5). Vegetation height ($r^2 = 0.24$; $p = 0.001$) and percentage of bare ground ($r^2 = 0.11$; $p = 0.005$) were the environmental variables significantly associated with the structure of the bird community in this history of fire disturbance. The other variables did not show significant values (rock: $r^2 = 0.04$; $p = 0.09$; water: $r^2 = 0.03$; $p = 0.12$). In AF areas, vegetation height ($r^2 = 0.24$; $p = 0.001$) and percentage of bare ground ($r^2 = 0.13$; $p = 0.002$) also were the environmental variables associated with the species composition, while in WF areas only vegetation height ($r^2 = 0.13$; $p = 0.03$) was significative. The percentage of plant cover was strongly correlated with vegetation height and was not evaluated in this analysis.

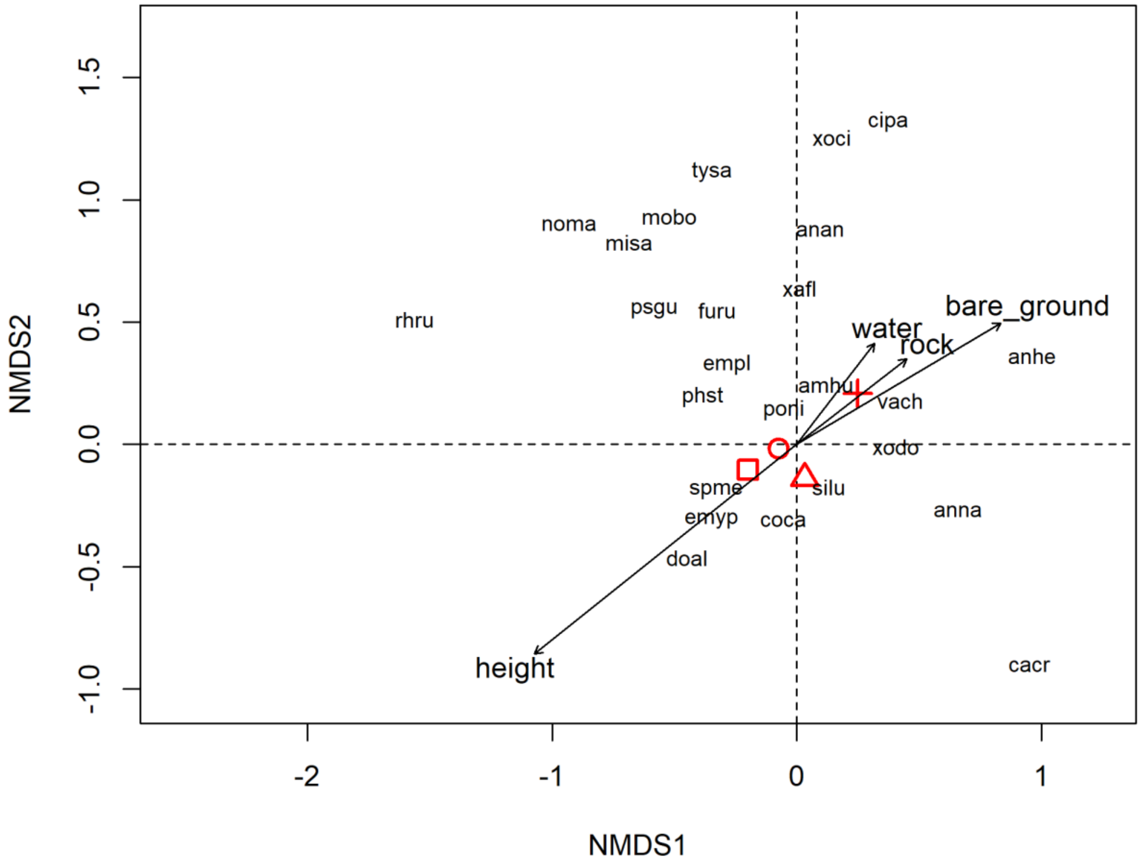


Fig 5. Non-metric multidimensional scaling (NMDS) plot illustrating the association among the grassland birds, four environmental variables (vegetation height, bare ground, rocks, and water) and a time gradient in relation to the fire disturbance. Results based on the species abundance (Bray-Curtis dissimilarity index) in areas with occasional fire (i.e. burned only once during the study). Stress = 0.22. Symbols represent the mean scores (centroids) of each breeding season (square = before the fire; cross = year of fire; circle = one year post-fire; triangle = two years post-fire). Acronyms are formed by the first two letters of the genus and species epithet of each species in S2 Table.

The vegetation height varied significantly over time in OF (χ^2 = 535.72; $p < 0.001$), decreasing after the burn and increasing in subsequent years (Table 2). Post-hoc comparisons indicated significant differences among all breeding seasons (in all combinations, $Z > 8.5$; $p < 0.001$), except between one year and two years post-fire ($Z = 2.45$; $p = 0.07$). There were also changes in vegetation height between the breeding seasons in AF ($\chi^2 = 56.17$; $p < 0.001$) and WF ($\chi^2 = 15.98$; $p = 0.001$). In AF areas, the vegetation height in the third year was significantly lower than in the others (2015: $Z = -6.73$; $p < 0.001$; 2016: $Z = -3.85$; $p < 0.001$; 2018: $Z = 5.98$; $p < 0.001$) and there were also differences between the first and second years ($Z = -3.36$; $p = 0.004$). In WF areas, the vegetation height in the first year was higher than in the second ($Z = -3.06$; $p = 0.01$) and third years ($Z = -3.71$; $p = 0.001$).

Table 2. Measurements of four environmental variables in areas with different histories of fire disturbance during four breeding seasons (2015–2018) in highland grasslands in northeastern Rio Grande do Sul state, Brazil.

History	Season	Vegetation height (cm)	Bare ground[a]	Rock[a]	Water[a]
Occasional fire	Before	63.2 ± 22.4 (31.2–109.6)	0.08 ± 0.45 (0–2.5)	2.04 ± 5.22 (0–25)	2 ± 6.34 (0–25)
	Fire	22.7 ± 12.2 (9.7–66.4)	15.6 ± 12.6 (1.5–47.5)	0.92 ± 2.93 (0–16.2)	1.57 ± 4.24 (0–21.2)
	1 year	34.4 ± 10.8 (15.3–54.2)	1.61 ± 3.43 (0–20.5)	0.67 ± 2.88 (0–17.5)	0.81 ± 3.50 (0–21.2)
	2 years	40.5 ± 11.2 (25.2–70.7)	0.2 ± 0.48 (0–2)	0.53 ± 1.98 (0–9.25)	0.29 ± 0.99 (0–4.5)
Annual fire	2015	25.5 ± 11.4 (7.1–46.2)	0 ± 0 (0–0)	7.51 ± 12.0 (0–40.6)	3.79 ± 8.36 (0–31.2)
	2016	20.9 ± 14.9 (9.5–75.1)	3.5 ± 3.51 (0.25–14.7)	2.74 ± 4.39 (0–20.5)	0.98 ± 3.09 (0–13.7)
	2017	16.2 ± 10.3 (5.6–38.7)	2.57 ± 3.04 (0–12.5)	3.68 ± 4.29 (0–16)	0.47 ± 0.90 (0–3.75)
	2018	24.4 ± 9.17 (9.1–40.8)	0.64 ± 1.06 (0–3.75)	2.72 ± 4.04 (0–16)	0.30 ± 0.89 (0–3.75)
Without fire	2015	59.6 ± 12.4 (41–78.8)	0 ± 0 (0–0)	1.38 ± 2.40 (0–6.25)	4.51 ± 8.43 (0–25)
	2016	46.6 ± 11.2 (26.1–61.9)	0 ± 0 (0–0)	1.14 ± 2.79 (0–8.75)	0 ± 0 (0–0)
	2017	44.7 ± 10.8 (30.8–67.8)	0.2 ± 0.41 (0–1.75)	0.14 ± 0.37 (0–1.25)	0.73 ± 1.55 (0–6.25)
	2018	50.3 ± 8.6 (31–67.5)	0 ± 0 (0–0)	0.07 ± 0.26 (0–1)	1.25 ± 4.01 (0–15)

Values correspond to mean ± SD (minimum–maximum).

[a] Mean percentage of cover in quadrats of 1 m^2 within bird point counts.

Discussion

We found that the richness, abundance, and species composition in the study areas changed over time in different ways. A more regular pattern of variation was found in areas that burned occasionally, where all parameters changed in the year of the fire and returned to the same levels as in the year before the fire one or two years after the disturbance. In areas burned annually or in areas without fire, changes did not occur (e.g., richness in areas without fire) or occurred in different years, without a definite temporal pattern. Lindenmayer et al. [60] also reported a different pattern of temporal changes between burned and unburned sites, where the rate of increase of bird richness was higher in burned sites.

The increase in bird richness observed shortly after the fire in areas that have gone through a long period without a burn may be related to a greater habitat heterogeneity. After a fire, the dry grassland has the shortest vegetation, while wetlands, depending on the fire intensity, are less impacted and have a different vegetation structure. The lower intensity of fire in wetlands than in dry grasslands observed in Brazilian *Cerrado* grasslands can be attributed to the higher soil water availability [61]. The heterogeneity of the vegetation structure increases the habitat variability and the diversity of the grassland bird community [16,62]. Thus, in our study areas, species associated with tall grasses (e.g., *Emberizoides ypiranganus*, *Sporophila melanogaster*, and *Phacellodomus striaticolis*, *sensu* [1]) did not disappear after the fire, but tended to occupy areas with tall vegetation such as *Eryngium* marshes, and were often restricted to these habitats. The dry grassland, in turn, provided habitat for species associated with low grasses (e.g., *Vanellus chilensis*, *Cinclodes pabsti*, *Anthus* spp.) that did not previously occur there or were less abundant before the fire. In fact, our results showed that some species are more associated with sites with taller vegetation, while other species occupy sites with shorter vegetation and more bare ground (Fig 5).

On the other hand, in the breeding season that the areas were unburned for long periods, we observed lower bird richness compared to years when these areas had been burned (Table 1). Probably in these cases, the greater homogenization of the habitat structure, due mainly to the presence of tall grasses, disadvantages species that occupy short grasslands. Sites without disturbances for long periods usually present an increase in vegetation height and plant biomass, which leads to a homogenization of vegetation structure [15,18]. The diversity of vegetation and structural differences, such as vegetation height, are important variables that determine the response of grassland birds [16,32,42,63]. We did not record some species in areas unburned for a long period, such as the Saffron-cowled Blackbird (*X. flavus*) and the Ochre-breasted Pipit (*A. nattereri*), two globally threatened species [64]. In the Brazilian Cerrado, *A. nattereri* has been recorded in native grasslands affected by fire [65,66]. Grasslands with more than two years of post-fire succession and no grazing, even in sites with favorable relief, do not seem to favor the occurrence of the Ochre-breasted Pipit [67]. In southern Brazil, the Saffron-cowled Blackbird used burned areas more frequently and avoided habitats with tall grasses and developed vegetation [37]. The species was absent from a protected area that has not experienced fires in nearly three decades [37], since the Saffron-cowled Blackbird depends on marshes to breed but also uses dry, short-grass areas to forage [37,68,69]. In Argentina, Isacch and Martinez [30] observed that areas with more tall-grass coverage had higher richness and abundance of birds. However, the authors noted that they did not sample sites with 100% tall-grass coverage (probably equivalent to WF areas in our study), and that in this situation, the richness is likely lower due to the loss of ground-feeding species.

In grasslands of the Serra da Canastra National Park, southeastern Brazil, burnings triggered profound and immediate changes in bird assemblages, increasing the number of species and individuals right after the fire [35]. Bahía and Zalba [32] found that the abundance and richness of birds were lower one year after a burn and increased significantly two years afterward. Besides the increase in richness, our data also showed an increase in abundance of individuals after fire. Probably the occurrence of some common species in our areas, such as the Rufous-collared Sparrow (*Z. capensis*), Hellmayr's Pipit (*A. hellmayri*), and Southern Lapwing (*V. chilensis*) contributes to this higher abundance. In North America, some species occurred in higher abundances in areas with fire disturbance [16,24,25].

Three species assessed individually showed responses to fire, in different ways. The densities of Rufous-collared Sparrow and the Hellmayr's Pipit (a non-grassland and a short-grass species, respectively, according to [1]) increased with a burn and later decreased over time after the fire. In contrast, the density of the Lesser Grass-Finch (a tall-grass species) decreased in the year of the fire, increasing again as the time since the fire lengthened (Fig 4). This is expected and occurs because some species are favored and others disadvantaged by fire disturbance, responding to variations in habitat structure [28,30]. Lesser Grass-Finch tends to be disadvantaged by fire disturbance due to the loss of suitable habitat, since it depends on a specific vegetation structure to nest, forage and seek refuge [70]. Some species of conservation concern that occur in the region of our study might be more frequent and abundant in areas burned frequently (e.g., *Cinclodes pabsti*, *Anthus nattereri* and *Xanthopsar flavus*), where short grasses are predominant, while others might be more frequent and abundant in areas with reduced fire management or without it (e.g., *Scytalopus iraiensis*, *Limnoctites rectirostris* and *Sporophila melanogaster*), where vegetation is higher [39]. Although we assumed that grazing pressure was constant at our study sites, it is important to note that grazing influences directly the vegetation structure and plant taxonomic diversity, promoting effects on plant and arthropod communities [15], and, consequently, may have additional effects on grassland birds. Several tallgrass-dependent birds are threatened in South America and are affected by the lack of tallgrass vegetation caused by the intensive disturbance of grassland due to cattle raising [1,42].

The present results also showed changes in grassland-bird species composition over the years, but the spatial dependence of the data used in the analysis requires caution in interpreting the effects of fire on community structure. Vegetation height was the main variable associated with species composition, corroborating other studies (e.g., [42,71]). Our data point to an effect of fire in areas that burned occasionally, but differences in species composition were also found among the years in areas burned annually and in areas without fire. Annual variation in areas without changes in the fire regime over the years may be the result of climatic variation. Temporal fluctuations of bird communities can be indirectly caused by both changes in temperature and precipitation, which determine the amount of resources available to birds [72,73]. Areas burned annually, for example, may have variable amounts of primary productivity, depending on rainfall and temperatures, affecting the stability of the grassland bird community [17].

Our results converge to the ideas that grasslands should be managed in a way that forms mosaics with a spatial and temporal arrangement of both short and tall grass, creating vegetation heterogeneity and promoting bird diversity [16,17,42]. Patch-burn management has been recommended for grassland bird conservation, because it creates the entire gradient of vegetation structure required to maintain grassland bird species that differ in habitat preferences [16,74]. Increasing heterogeneity at landscape scales also results in higher stability of bird communities over time [17]. Thus, management of grasslands that creates a shifting mosaic, using prescribed fires in areas with different times since burnings and areas with different histories of fire disturbance, can be useful in conservation of grassland birds and habitats. Fire in highland grasslands of southern Brazil has already been recommended as an important management tool to ally cattle raising with bird conservation [39].

This study is the first to use data covering the period before and after a burn in southern Brazilian grasslands, since we are not allowed to burn large areas for experiments, especially in Brazilian protected areas. We took advantage of three events of occasional (accidental) fire in the protected areas where we usually develop bird monitoring to answer relevant questions to grassland conservation in southern Brazil. Our results showed that two years without fire was a period long enough for the evaluated parameters of the bird community and the density of some species to return to levels estimated before the disturbance. The recovery of the vegetation after a fire and the response time of the birds, in one or two breeding seasons, depending on the species, were also observed in Argentina [32]. In southeastern Australia, an endangered bird species either remained continuously on burned sites or returned to previously occupied sites within two years after an unplanned fire [75]. Another study in Argentina, with a threatened grassland bird, found that the species did not show avoidance of the burned patch in the third breeding season after the prescribed fire, suggesting burning intervals longer than two years [31]. This kind of information is useful for planning the periodicity with which fire can be used to manage grassland areas.

We showed that the use of fire in highland grasslands of southern Brazil should consider a period of at least two years (or two breeding seasons) without burnings in the same grassland patch, to ensure the recovery of the bird community of the area. Another point of concern is that fire must occur before the breeding season of most grassland birds (which is the austral spring and summer in southern Brazil [76]) to avoid burning active nests, and do not affect wetlands to avoid losing important sites of refuge and/or nest sites for tall-grass birds, including migratory and philopatric species such as *Sporophila melanogaster*. However, it is needed to consider that management decisions and a better understanding of the effects of fire require an analysis that integrates different taxonomic groups of animals and plants.

Acknowledgments

We thank Gerhard E. Overbeck, Miguel Â. Marini, Pedro M. A. Ferreira, and two anonymous reviewers for important contributions to the manuscript. We thank Mariana B. Neves for assistance with field work and Thaiane W. da Silva for help in statistical analysis. We thank Janet W. Reid and Patrícia B. Schafer for improving the English writing of the manuscript. We are very grateful to the farm owners (Paulo Machado, Valdir B. de Castilhos, and Ernani) that allowed access to their properties. We thank administrative team of Parque Estadual do Tainhas and Parque Nacional de Aparados da Serra for the support during the field work. We thank Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio) and Secretaria do Meio Ambiente e Desenvolvimento Sustentável (SEMA) for research permission. We also thank Conselho Nacional do Desenvolvimento Científico e Tecnológico (CNPq) and Neotropical Grassland Conservancy for financial support to field work. E.C. received a scholarship from CNPq (process 141055/2015-4). C.S.F. received CNPq grants (processes 457475/2012-9, 309438/2016-0, and 402083/2016-4) through Ministério da Ciência, Tecnologia, Inovações e Comunicações (MCTIC). This research was conducted in compliance with the requirements of ICMBio (license number 49728) and SEMA (license number 13/2015).

References

1. Azpiroz AB, Isacch JP, Dias RA, Di Giacomo AS, Fontana CS, Palarea CM. Ecology and conservation of grassland birds in southeastern South America: a review. J F Ornithol. 2012;83: 217–246. doi:10.1111/j.1557-9263.2012.00372.x

2. Overbeck G, Muller S, Fidelis A, Pfadenhauer J, Pillar V, Blanco C, et al. Brazil's neglected biome: The South Brazilian Campos. Perspect Plant Ecol Evol Syst. 2007;9:

101–116. doi:10.1016/j.ppees.2007.07.005

3. Bencke GA. Diversidade e conservação da fauna dos Campos do Sul do Brasil. In: Pillar VP, Müller SC, Castilhos ZMS, Jacques AVA, editors. Campos Sulinos: Conservação e Uso Sustentável da Biodiversidade. Brasília, DF, Brazil: Ministério do Meio Ambiente; 2009.

4. Boldrini II. A flora dos campos do Rio Grande do Sul. In: Pillar VDP, Muller SC, Castilhos ZMS, Jacques AVA, editors. Campos Sulinos: Conservação e Uso Sustentável da Biodiversidade. Brasília, DF, Brazil: Ministério do Meio Ambiente; 2009.

5. Fontana CS, Dotta G, Marques CK, Repenning M, Agne CE, dos Santos RJ. Conservation of grassland birds in South Brazil: a land management perspective. Nat e Conserv. 2016;14: 83–87. doi:10.1016/j.ncon.2016.09.005

6. Fontana CS, Rovedder CE, Repenning M, Gonçalves ML. Estado atual do conhecimento e conservação da avifauna dos Campos de Cima da Serra do sul do Brasil, Rio Grande do Sul e Santa Catarina. Rev Bras Ornitol. 2008;16: 281–307.

7. Repenning M, Fontana CS. A new species of gray seedeater (Emberizidae: Sporophila) from upland grasslands of southern Brazil. Auk. 2013;130: 791–803. doi:10.1525/auk.2013.12167

8. Fontana CS, Bencke GA. Biodiversidade de aves. In: Pillar VDP, Langue O, editors. Os Campos do Sul. Porto Alegre, RS, Brazil: UFRGS; 2015.

9. Pillar VD, Boldrini II, Hasenack H, Jacques AVA, Both R, Müller SC, et al. Estado atual e desafios para a conservação dos campos. Work "Estado atual e desafios para a Conserv dos campos. 2006; 1–24. Available: http://ecoqua.ecologia.ufrgs.br

10. Castilhos ZMS, Machado MD, Pinto MF. Produção animal com conservação da flora

campestre do bioma Pampa. In: Pillar VDP, Muller SC, Castilhos ZMS, Jacques AVA, editors. Campos Sulinos: Conservação e Uso Sustentável da Biodiversidade. Brasília, DF, Brazil: Ministério do Meio Ambiente; 2009.

11. Overbeck GE, Pfadenhauer J. Adaptive strategies in burned subtropical grassland in southern Brazil. Flora Morphol Distrib Funct Ecol Plants. 2007;202: 27–49. doi:10.1016/j.flora.2005.11.004

12. Andrade BO, Bonilha CL, Overbeck GE, Vélez-Martin E, Rolim RG, Bordignon SAL, et al. Classification of South Brazilian grasslands: Implications for conservation. Appl Veg Sci. 2019;22: 168–184. doi:10.1111/avsc.12413

13. Pillar VDP, Vélez E. Extinção dos Campos Sulinos em Unidades de Conservação: um Fenômeno Natural ou um Problema ético? Nat Conserv. 2010;08: 84–86. doi:10.4322/natcon.00801014

14. Andrade BO, Bonilha CL, Ferreira PMA, Boldrini II, Overbeck GE. Highland grasslands at the southern tip of the atlantic forest biome: Management options and conservation challenges. Oecologia Aust. 2016;20: 37–61. doi:10.4257/oeco.2016.2002.04

15. Ferreira PMA, Andrade BO, Podgaiski LR, Dias AC, Pillar VD, Overbeck GE, et al. Long-term ecological research in southern Brazil grasslands: Effects of grazing exclusion and deferred grazing on plant and arthropod communities. Aldrich SP, editor. PLoS One. 2020;15: e0227706. doi:10.1371/journal.pone.0227706

16. Fuhlendorf SD, Harrell WC, Engle DM, Hamilton RG, Davis CA, Jr DML. Should heterogeneity be the basis for conservation? Grassland bird responses to fire and grazing. Ecol Appl. 2006;16: 1706–1716.

17. Hovick TJ, Elmore RD, Fuhlendorf SD, Engle DM, Hamilton RG. Spatial heterogeneity

increases diversity and stability in grassland bird communities. Ecol Appl. 2015;25: 662–672. doi:10.1890/14-1067.1

18. Overbeck GE, Müller SC, Pillar VD, Pfadenhauer J. Fine-scale post-fire dynamics in southern Brazilian subtropical grassland. J Veg Sci. 2005;16: 655–664. doi:10.1111/j.1654-1103.2005.tb02408.x

19. Buisson E, Stradic S Le, Silveira FAO, Durigan G, Overbeck GE, Fidelis A, et al. Resilience and restoration of tropical and subtropical grasslands , savannas , and grassy woodlands. Biol Rev. 2018. doi:10.1111/brv.12470

20. Sühs RB, Giehl ELH, Peroni N. Preventing traditional management can cause grassland loss within 30 years in southern Brazil. Sci Rep. 2020;10: 783. doi:10.1038/s41598-020-57564-z

21. Batista EKL, Russell-Smith J, França H, Figueira JEC. An evaluation of contemporary savanna fire regimes in the Canastra National Park, Brazil: Outcomes of fire suppression policies. J Environ Manage. 2018;205: 40–49. doi:10.1016/j.jenvman.2017.09.053

22. Fontana CS, Bencke GA, Reis RE. Livro vermelho da fauna ameaçada de extinção no Rio Grande do Sul. Porto Alegre, RS, Brazil: Edipucrs; 2003.

23. Grant TA, Madden E, Berkey GB. Tree and shrub invasion in northern mixed-grass prairie: implications for breeding grassland birds. Wildl Soc Bull. 2004;32: 807–818. doi:10.2193/0091-7648(2004)032[0807:tasiin]2.0.co;2

24. Winter M. Relationship of fire history to territory size, breeding density, and habitat of Baird's Sparrow in North Dakota. Stud Avian Biol. 1999;19: 171–177.

25. Shriver WG, Vickery PD. Response of Breeding Florida Grasshopper and Bachman's Sparrows to Winter Prescribed Burning. J Wildl Manage. 2001;65: 470–475.

doi:10.2307/3803099

26. Johnson DH. Effects of Fire on Bird Populations in Mixed-Grass Prairie. In: Knopf FL, Samson FB, editors. Ecology and conservation of Great Plains vertebrates. New York, NY, USA: Springer; 1997. pp. 181–206. doi:10.1007/978-1-4757-2703-6_8

27. Robel RJ, Hughes JP, Hull SD, Kemp KE, Klute DS. Spring burning: Resulting avian abundance and nesting in Kansas CRP. J Range Manag. 1998;51: 132–138. doi:10.2307/4003197

28. Reinking DL. Fire regimes and avian responses in the central tallgrass prairie. Stud Avian Biol. 2005;30: 116–126.

29. Comparatore VM, Martinez MM, Vassallo AI, Barg M, Isacch JP. Abundancia y relaciones con el hábitat de aves y mamíferos en pastizales de Paspalum quadrifarium (paja colorada) manejados con fuego (provincia de Buenos Aires, Argentina). Interciencia. 1996;21: 228–237.

30. Isacch JP, Martinez MM. Estacionalidad y relaciones con la estructura del habitat de la comunidad de aves de pastizales de paja colorada (Paspalum quadrifarium) manejados con fuego en la provicncia de Buenos Aires, Argentina. Ornitol Neotrop. 2001;12: 345–354.

31. Di Giacomo AG, Di Giacomo AS, Reboreda JC. Effects of grassland burning on reproductive success of globally threatened Strange-tailed Tyrants Alectrurus risora. Bird Conserv Int. 2011;21: 411–422. doi:10.1017/S0959270910000584

32. Bahía R, Zalba S. Changes in grassland bird communities and breeding success after a fire in the Argentinian Pampas. Biodivers Conserv. 2019. doi:10.1007/s10531-019-01850-4

33. Arruda FV de, Sousa DG de, Teresa FB, Prado VHM do, Cunha HF da, Izzo TJ. Trends and gaps of the scientific literature about the effects of fire on Brazilian Cerrado. Biota Neotrop. 2018;18: 1–6. doi:10.1590/1676-0611-bn-2017-0426

34. Cavalcanti RB, Alves MAS. Effects of fire on savanna birds in Central Brazil. Ornitol Neotrop. 1997;8: 85–87. doi:10.1046/j.1365-2699.2000.00422.x

35. dos Reis MG, Fieker CZ, Dias MM. The influence of fire on the assemblage structure of foraging birds in grasslands of the Serra da Canastra National Park, Brazil. An Acad Bras Cienc. 2016;88: 891–901. doi:10.1590/0001-3765201620150177

36. Duca C, Yokomizo H, Marini MÂ, Possingham HP. Cost-efficient conservation for the white-banded tanager (Neothraupis fasciata) in the Cerrado, central Brazil. Biol Conserv. 2009;142: 563–574. doi:10.1016/j.biocon.2008.11.010

37. Petry MV, Krüger L. Frequent use of burned grasslands by the vulnerable Saffron-Cowled Blackbird Xanthopsar flavus: implications for the conservation of the species. J Ornithol. 2010;151: 599–605. doi:10.1007/s10336-009-0489-9

38. Petry MV, Piuco R da C, Brummelhaus J. Aves Associadas ao Manejo com Fogo em Áreas de Campo na Porção Sul do Bioma Mata Atlântica. Biodiversidade Bras. 2011;1: 110–119.

39. Bettio M. Resposta da comunidade de aves ao uso do fogo nos campos de altitude do sul do Brasil. M.Sc. Thesis, Pontifícia Universidade Católica do Rio Grande do Sul. 2017.

40. Fuhlendorf SD, Limb RF, Engle DM, Miller RF. Assessment of Prescribed Fire as a Conservation Practice. In: Briske DD, editor. Conservation Benefits of Rangeland Practices Assessment, Recommendations, and Knowledg Gaps. Washington, D.C., USA: USDA-NRCS; 2011.

41. Saab VA, Powell HDW. Fire and avian ecology in North America: Process influencing pattern. Stud Avian Biol. 2005;30: 1–13.

42. Dias RA, Gianuca AT, Vizentin-Bugoni J, Gonçalves MSS, Bencke GA, Bastazini VAG. Livestock disturbance in Brazilian grasslands influences avian species diversity via turnover. Biodivers Conserv. 2017;26: 2473–2490. doi:10.1007/s10531-017-1370-4

43. Boldrini II, Eggers L, Mentz LA, Miotto STS, Matzenbacher NI, Longhi-Wagner HM, et al. Flora. In: Boldrini II, editor. Biodiversidade dos Campos do Planalto das Araucárias. Brasília, DF, Brazil: Ministério do Meio Ambiente; 2009. pp. 40–94.

44. Nimer E. Clima. In: IBGE, editor. Geografia do Brasil: Região Sul. Rio de Janeiro, Brazil: IBGE; 1990.

45. Almeida JA. Fatores abióticos. In: Boldrini II, editor. Biodiversidade dos campos do Planalto das Araucárias. Brasília, DF, Brazil: Ministério do Meio Ambiente; 2009.

46. Maurício GN, Bencke GA, Repenning M, Machado DB, Dias RA, Bugoni L. Review of the breeding status of birds in Rio Grande do Sul , Brazil. Iheringia. 2013;103: 163–184.

47. Bibby CJ, Burgess ND, Hill DA. Bird census techniques. London, UK.: Academic Press; 1992.

48. Chao A, Jost L. Coverage-based rarefaction and extrapolation : standardizing samples by completeness rather than size. Ecology. 2012;93: 2533–2547. doi:10.2307/41739612

49. Hsieh TC, Ma KH, Chao A. iNEXT: iNterpolation and EXTrapolation for species diversity. 2019. Available: http://chao.stat.nthu.edu.tw/blog/software-download/

50. Chao A, Gotelli NJ, Hsieh TC, Sander EL, Ma KH, Colwell RK, et al. Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies. Ecol Monogr. 2014;84: 45–67. doi:10.1890/13-0133.1

51. Bates D, Maechler M, Bolker B, Walker S. lme4: linear mixed-effects models using ‘eigen’ and S4. 2016. Available: https://cran.r-project.org/package=lme4

52. Zuur AF, Ieno EN, Walker NJ, Saveliev AA, Smith GM. Mixed effects models and extensions in ecology with R. New York, NY, USA: Springer; 2009.

53. Hothorn T, Bretz F, Westfall P. multcomp: simultaneous inference in general parametric models. 2016. Available: https://cran.r-project.org/package=multcomp

54. Thomas L, Buckland ST, Rexstad EA, Laake JL, Strindberg S, Hedley SL, et al. Distance software: Design and analysis of distance sampling surveys for estimating population size. J Appl Ecol. 2010;47: 5–14. doi:10.1111/j.1365-2664.2009.01737.x

55. Fritcher SC, Rumble MA, Flake LD. Grassland bird densities in seral stages of mixed-grass prairie. J Range Manag. 2004;57: 351–357. doi:10.2307/4003858

56. Hervé M. RVAideMemoire: Testing and Plotting Procedures for Biostatistics. 2019. Available: https://cran.r-project.org/package=RVAideMemoire

57. Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, et al. Vegan: community ecology package. 2019. Available: https://cran.r-project.org/package=vegan

58. Arbizu PM. pairwiseAdonis: Pairwise Multilevel Comparison using Adonis. 2017. Available: https://github.com/pmartinezarbizu/pairwiseAdonis

59. R Development Core Team. R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing; 2019. Available: https://www.r-project.org/

60. Lindenmayer DB, Candy SG, MacGregor CI, Banks SC, Westgate M, Ikin K, et al. Do temporal changes in vegetation structure additional to time since fire predict changes in bird occurrence? Ecol Appl. 2016;26: 2267–2279. doi:10.1002/eap.1367

61. Schmidt IB, Fidelis A, Miranda S, Ticktin T. How do the wets burn ? Fire behavior and intensity in wet grasslands in the Brazilian savanna. Brazilian J Bot. 2017;40: 167–175. doi:10.1007/s40415-016-0330-7

62. Hovick TJ, Miller JR, Dinsmore SJ, Engle DM, Debinski DM, Fuhlendorf SD. Effects of fire and grazing on grasshopper sparrow nest survival. J Wildl Manage. 2012;76: 19–27. doi:10.1002/jwmg.243

63. Vickery PD, Zuckerberg B, Jones AL, Shriver WG, Weik AP. Influence of fire and other anthropogenic practices on grassland and shrubland birds in New England. USDA Forest Service General Technical Report PSW-GTR-191; 2005.

64. IUCN – International Union for the Conservation of Nature. The IUCN Red List of Threatened Species. Version 2019-3. 2019 [cited 10 Dec 2019]. Available: http://www.iucnredlist.org

65. Mazzoni LG, Perillo A. Range extension of Anthus nattereri Sclater, 1878 (Aves: Motacillidae) in Minas Gerais, southeastern Brazil. Check List. 2011;7: 589–591. doi:10.15560/7.5.598

66. Lombardi VT, Santos KK, D’Angelo Neto S, Mazzoni LG, Rennó B, Faetti RG, et al. Registros notáveis de aves para o Sul do Estado de Minas Gerais, Brasil. Cotinga. 2012;34: 104–117.

67. Lombardi VT. Vivendo Na Linha De Fogo: História Natural De Anthus Nattereri (Aves: Motacillidae) Nos Campos Do Alto Rio Grande, Sul De Minas Gerais, Brasil. M.Sc. Thesis, Universidade Federal de Viçosa. 2017.

68. Dias RA, Mauricio G. Natural history notes and conservation of a Saffron-cowled Blackbird Xanthopsar flavus population in the southern coastal plain of Rio Grande do

Sul, Brazil. Bird Conserv Int. 2002;12: 255–268.

69. Fontana CS, Dias RA, Maurício GN. Xanthopsar flavus. In: Serafini PP, editor. Plano de ação nacional para a conservação dos passeriformes ameaçados dos Campos Sulinos e Espinilho. Brasília, DF, Brazil: ICMBio; 2013.

70. Chiarani E, Fontana CS. Breeding biology of the Lesser Grass-Finch (Emberizoides ypiranganus) in southern Brazilian upland grasslands. Wilson J Ornithol. 2015;127: 441–456.

71. Fisher RJ, Davis SK. From Wiens to Robel: A Review of Grassland-Bird Habitat Selection. J Wildl Manage. 2010;74: 265–273. doi:10.2193/2009-020

72. O'Brien EM. Water-energy dynamics, climate, and prediction of woody plant species richness: an interim general model. J Biogeogr. 1998;25: 379–398.

73. Boyle WA. Short-distance partial migration of Neotropical birds: a community-level test of the foraging limitation hypothesis. Oikos. 2011;120: 1803–1816.

74. Churchwell RT, Davis CA, Fuhlendorf SD, Engle DM. Effects of Patch-Burn Management on Dickcissel Nest Success in a Tallgrass Prairie. J Wildl Manage. 2008;72: 1596–1604. doi:10.2193/2007-365

75. Lindenmayer DB, MacGregor C, Wood JT, Cunningham RB, Crane M, Michael D, et al. What factors influence rapid post-fire site re-occupancy? A case study of the endangered Eastern Bristlebird in eastern Australia. Int J Wildl Fire. 2009;18: 84. doi:10.1071/wf07048

76. Belton W. Aves do Rio Grande do Sul, distribuição e biologia. São Leopoldo, RS, Brazil: Unisinos; 1994.

Supporting information

S1 Table. Areas used to sample birds in highland grasslands in northeastern Rio Grande do Sul state, Brazil, and history of fire disturbance. Numbers in each breeding season represent the years since fire ("0" indicates the year when the area was burned) and the quantity of count points sampled (between parentheses). APA = area in Parque Nacional de Aparados da Serra or Parque Nacional da Serra Geral; TAI = area in the region of Parque Estadual do Tainhas.

Area	**Area size (ha)**	**History of disturbance**	**Breeding Season**			
			2015–16	**2016–17**	**2017–18**	**2018–19**
APA 1	250	Occasional fire	≥ 8 (8)	0 (12)	1 (12)	2 (12)
TAI 1	210	Occasional fire	≥ 8 (10)	0 (14)	1 (14)	2 (11)
APA 3	220	Occasional fire	-	>8 (12)	0 (12)	1 (12)
TAI 2	210	Annual fire	0 (11)	0 (14)	0 (14)	0 (14)
TAI 3	170	Annual fire	0 (10)	0 (12)	0 (12)	0 (12)
APA 2	180	Without fire	-	>10 (12)	>10 (12)	>10 (4)
APA 4	150	Without fire	>10 (9)	-	>10 (9)	>10 (10)

S2 Table. Relative abundance (number of individuals/point) of bird species recorded during four breeding seasons (2015–2018) in grasslands with three histories of fire disturbance (occasional fire, annual fire, and without fire) in northeastern Rio Grande do Sul state, Brazil. In occasional fire breeding seasons refer to the year before the burn (before), in the year of the burn (fire), and one and two years after a fire (1year and 2years, respectively). Species are in alphabetical order.

Species	Occasional fire				Annual fire				Without fire			
	before	fire	1year	2years	2015	2016	2017	2018	2015	2016	2017	2018
Agelaioides badius	-	-	-	-	0.10	-	-	-	-	-	-	-
Amazonetta brasiliensis	0.10	0.08	0.05	-	-	0.08	-	-	-	0.33	-	-
Ammodramus humeralis	-	0.29	0.18	0.09	0.24	0.23	0.19	0.08	-	-	-	-
Anas flavirostris	-	0.05	-	-	-	0.12	0.08	-	-	0.17	-	-
Anas georgica	-	-	0.05	-	-	0.19	-	-	-	-	0.05	-
Anthus hellmayri	0.33	1.11	0.55	0.35	0.57	1.12	0.92	0.85	1.11	0.42	0.33	0.14
Anthus nattereri	-	0.08	0.08	0.26	0.62	0.42	0.27	0.04	-	-	-	-
Anumbius annumbi	-	0.26	0.16	0.04	0.05	0.23	0.54	0.08	0.33	-	-	-
Ardea alba	-	-	-	-	-	-	-	-	-	-	0.05	-
Butorides striata	-	-	-	-	-	-	-	-	0.11	-	-	-
Cariama cristata	-	0.05	0.05	0.04	-	-	0.04	0.04	-	-	-	-
Chlorostilbon lucidus	0.10	0.18	0.18	0.09	-	0.08	0.04	-	0.22	0.17	0.05	0.21
Ciconia maguari	-	-	-	-	-	-	-	0.04	-	-	-	-
Cinclodes pabsti	-	-	0.03	-	0.10	0.27	0.19	-	-	-	-	-

Species	Occasional fire				Annual fire				Without fire			
	before	fire	1year	2years	2015	2016	2017	2018	2015	2016	2017	2018
Colaptes campestris	0.17	0.42	0.26	-	0.38	0.50	0.92	0.73	-	0.33	-	-
Colibri serrirostris	0.07	-	-	-	-	-	-	-	-	-	-	-
Donacospiza albifrons	0.37	0.11	0.21	0.17	0.10	-	0.08	-	0.22	0.25	0.19	0.36
Elaenia parvirostris	-	0.05	-	-	-	-	-	-	-	-	-	-
Emberizoides ypiranganus	1.77	0.74	1.76	1.52	0.52	0.88	0.73	0.69	1.11	2.92	1.76	2.14
Embernagra platensis	0.83	0.89	1.00	0.78	0.10	0.58	0.81	0.88	0.67	0.58	0.57	0.50
Furnarius rufus	-	-	0.03	0.04	-	-	0.19	-	-	-	-	-
Gallinago paraguaiae	-	0.03	-	0.09	-	0.04	0.15	0.19	0.11	0.08	0.05	0.07
Geothlypis aequinoctialis	0.27	0.13	0.18	-	0.05	0.04	0.15	0.15	-	-	-	0.14
Gnorimopsar chopi	-	-	-	-	1.48	0.42	0.62	0.46	-	-	-	-
Guira guira	-	-	-	-	-	-	0.04	-	-	-	-	-
Hylocharis chrysura	-	0.03	-	-	-	-	-	-	-	-	-	-
Jacana jacana	-	-	-	-	-	-	-	-	0.11	-	-	-
Knipolegus lophotes	-	-	-	-	-	-	0.08	-	-	-	-	0.07
Knipolegus nigerrimus	-	-	-	-	-	-	-	-	-	-	0.10	-
Laterallus leucopyrrhus	0.07	-	0.11	-	-	0.12	0.08	0.12	-	0.08	0.05	-
Leucochloris albicollis	-	0.08	0.08	-	-	-	-	-	-	0.17	-	-
Limnoctites rectirostris	0.20	0.16	0.21	-	0.05	0.15	0.19	0.08	0.22	-	0.10	-
Mimus saturninus	-	0.03	0.05	-	0.19	-	0.15	0.08	-	-	-	-

Species	Occasional fire				Annual fire				Without fire			
	before	fire	1year	2years	2015	2016	2017	2018	2015	2016	2017	2018
Molothrus bonariensis	-	0.05	0.03	0.13	-	-	-	-	-	-	-	-
Myiarchus swainsonii	0.03	0.03	-	-	-	-	-	-	-	-	-	-
Myiphobus fasciatus	0.03	0.08	0.08	0.04	-	-	-	0.04	-	-	-	-
Nothura maculosa	0.03	0.03	0.08	0.09	0.19	0.04	0.04	0.04	-	-	-	-
Pardirallus sanguinolentus	0.07	0.11	0.08	-	0.10	0.08	0.08	0.23	-	-	-	0.14
Patagioenas cayennensis	-	0.03	-	-	0.10	-	-	-	-	-	-	-
Patagioenas picazuro	-	-	0.03	-	0.05	-	-	0.04	-	-	-	-
Phacellodomus striaticollis	0.07	0.18	0.13	0.09	0.05	0.12	-	-	-	-	-	-
Pipraeidea bonariensis	-	0.03	0.03	-	-	-	-	-	-	-	-	-
Pitangus sulphuratus	0.10	0.16	0.11	0.13	0.10	0.23	0.04	0.27	0.33	0.08	-	-
Plegadis chihi	-	-	-	-	-	-	0.46	-	-	-	-	-
Poospiza nigrorufa	-	-	0.03	-	-	0.08	0.04	0.04	-	-	-	-
Pseudoleistes guirahuro	0.33	0.45	0.24	0.26	0.52	0.31	0.62	0.69	-	-	-	-
Rhynchotus rufescens	0.10	0.08	0.05	-	0.05	-	-	-	0.11	0.17	0.10	-
Satrapa icterophrys	-	0.03	0.03	-	-	0.04	0.08	0.08	-	-	-	-
Scytalopus iraiensis	0.13	0.13	0.08	0.13	-	-	-	-	0.33	-	0.14	0.07
Serpophaga subcristata	0.17	0.08	0.03	0.04	-	-	-	-	-	-	0.14	-
Sicalis flaveola	0.13	0.13	0.29	0.26	-	-	0.08	0.12	-	-	0.14	0.07
Sicalis luteola	1.03	0.58	0.61	1.52	0.19	0.50	0.19	0.69	1.22	0.33	0.19	1.93

Species	Occasional fire				Annual fire				Without fire			
	before	fire	1year	2years	2015	2016	2017	2018	2015	2016	2017	2018
Spinus magellanicus	-	0.08	-	0.17	-	0.19	0.04	0.12	-	-	-	-
Sporophila caerulescens	0.03	0.16	0.16	0.13	-	-	-	0.04	-	-	-	-
Sporophila melanogaster	0.67	0.37	0.39	0.35	0.24	0.31	0.23	0.35	0.67	0.92	0.76	1.00
Stephanophorus diadematus	-	-	-	-	-	-	0.08	-	-	-	-	-
Sturnella superciliaris	-	-	-	-	-	-	0.04	-	-	-	-	-
Synallaxis spixi	0.10	0.08	0.16	0.09	-	-	0.04	0.08	-	-	-	-
Syrigma sibilatrix	-	-	-	-	-	0.08	0.08	-	-	-	-	-
Tangara sayaca	-	-	-	0.17	-	-	-	-	-	-	-	-
Thamnophilus ruficapillus	0.07	0.08	-	-	-	0.04	-	-	-	-	-	0.14
Theristicus caudatus	-	-	-	-	0.10	0.12	0.15	0.15	-	-	-	-
Troglodytes musculus	0.03	0.32	0.39	0.13	0.10	0.08	0.31	0.08	-	-	0.05	-
Turdus amaurochalinus	-	0.03	0.03	-	-	-	0.04	0.04	-	-	-	-
Turdus rufiventris	0.03	0.03	-	-	-	-	-	-	-	-	-	-
Tyrannus melancholicus	-	-	0.11	0.22	0.14	0.04	0.04	-	0.11	-	-	-
Tyrannus savana	0.17	0.21	0.21	0.22	0.19	0.42	0.15	0.12	-	-	-	-
Vanellus chilensis	-	0.21	0.26	0.09	0.33	0.54	1.04	0.81	0.67	-	-	-
Xanthopsar flavus	-	0.21	0.13	-	0.14	2.00	0.88	0.88	-	-	-	-
Xolmis cinereus	-	0.05	-	0.09	0.14	0.23	0.08	0.27	-	-	-	-
Xolmis dominicanus	0.10	0.29	0.37	0.35	0.14	0.31	0.35	0.38	0.67	0.67	0.24	0.21

Species	Occasional fire				Annual fire				Without fire			
	before	fire	1year	2years	2015	2016	2017	2018	2015	2016	2017	2018
Zenaida auriculata	-	-	0.03	0.04	-	0.12	0.08	-	-	-	-	-
Zonotrichia capensis	1.13	2.45	1.37	1.17	0.52	0.85	0.92	0.85	0.89	0.58	0.62	0.57
Total	8.83	11.47	10.74	9.39	7.90	12.15	12.62	10.88	9.22	8.25	5.67	7.79

CAPÍTULO 2

Implications of fire on breeding of grassland-dependent birds in highland grasslands of southern Brazil

Artigo a ser submetido para publicação no periódico *Ibis*

Implications of fire on breeding of grassland-dependent birds in highland grasslands of southern Brazil

Short title: Response of grassland birds to fire

Eduardo Chiarani[1,2] and Carla Suertegaray Fontana[1]

[1] PUCRS- Pontifícia Universidade Católica do Rio Grande do Sul, Escola de Ciências, Programa de Pós-Graduação em Zoologia, Museu de Ciências e Tecnologia, Laboratório de Ornitologia.

[2] Corresponding author: educhiarani@gmail.com

ABSTRACT

Grasslands have been closely related to fire disturbances, which play an important role in their dynamics. Despite this, little is known about temporal dynamics of fire effects on breeding of birds, mainly in South America. We assessed the temporal changes on nesting success, productivity, and territories of grassland-dependent bird species in natural grassland ecosystems under different fire regimes. We based our study on two tall-grass species, Lesser Grass-Finch (*Emberizoides ypiranganus*) and Black-bellied Seedeater (*Sporophila melanogaster*). We searched for nests and territories during three breeding seasons (2013–2014, 2016–2017 and 2017–2018) in sites with occasional fire (OF) and frequent fire (FF), both located in a Conservation Unit at northeastern Rio Grande do Sul state, Brazil. We monitored a total of 237 nests (178 of Lesser Grass-Finch and 59 of Black-bellied Seedeater). Cumulative survival probability and productivity of the nests of the two species did not show statistically significant differences among breeding seasons both in OF and FF. Size and number of territories of Lesser Grass-Finch varied over breeding seasons only in OF, with fewer and smaller territories in the year of the fire. The number of territories of Black-bellied Seedeater in OF was lower one year after fire in comparison with the season of the fire, but did not differ in FF. Our study is the first to access temporal effects of fire on breeding and territories of grassland birds in southern Brazilian grasslands and provides information to support that controlled prescribed fire can be used as a tool to manage and conserve highland grasslands in South America.

Keywords: conservation, fledglings, *Emberizoides ypiranganus*, grassland birds, management, nest survival, *Sporophila melanogaster*

INTRODUCTION

Grassland ecosystems are largely spread around the world, covering about 40% of the Earth's surface (White *et al.* 2000), and their distribution has been related to regions where tree cover is limited by edaphic or climate conditions, or by disturbances regimes such as fire, herbivory, or flooding (Leys *et al.* 2018). Although the factors that control the distribution of grasslands around the world throughout history are varied and debated, it is recognized that fire is a consistently strong predictor (Staver *et al.* 2011, Leys *et al.* 2018). Fire plays an important role in the dynamics of open ecosystems like grasslands, with evolutionary effects on their biodiversity (Bond *et al.* 2005, Pausas & Keeley 2009).

Grassland bird populations have been declining in several parts of the world, and the main causes are loss and fragmentation of native grasslands, intensification of agricultural practices, and use of pesticides (Donald *et al.* 2001, Butchart, S.H.M., Stattersfield *et al.* 2004, Askins *et al.* 2007, Azpiroz *et al.* 2012b). Management of grasslands that promotes habitat uniformity instead of heterogeneity (e.g., annual burning and overgrazing) has also contributed to the decline in grassland bird diversity and abundance (Fuhlendorf *et al.* 2006). Despite widespread recognition of the importance of grasslands for global biodiversity, and varied attempts to protect these ecosystems, managing and conserving their essential processes remain a significant challenge (Leys *et al.* 2018).

In Southeastern South America Grasslands (SESA Grasslands sensu Azpiroz *et al.* 2012b) there is a natural portion of grasslands often neglected in open ecosystems bird studies, probably because they are located in the portion of Atlantic Forest Biome in Brazil (Fontana *et al.* 2008). Such portion is known as southern Brazilian highland grasslands (SBHG), located in northeastern Rio Grande do Sul state and part of Santa Catarina and Paraná states (Andrade *et al.* 2016). Grasslands of SBHG are usually burned at the end of winter, in order to renew and stimulate the regrowth of vegetation for cattle feed (Overbeck & Pfadenhauer 2007, Andrade *et al.* 2019). The traditional practice of fire can be one of the sustaining elements of these grasslands over the centuries (Behling & Pillar 2007). This statement is proved by studies showing that in the absence of fire or grazing for long periods in regions of grassland/forest mosaics like the ones found at SBHG, grassland vegetation can be replaced by shrubs (Overbeck *et al.* 2005, Pillar & Vélez 2010, Andrade *et al.* 2019, Sühs *et al.* 2020). Fire and grazing exclusion can also increase the risk of catastrophic fire due to the accumulation of flammable biomass (Pillar & Vélez 2010, Buisson *et al.* 2018). However, disturbances such as overgrazing, cattle trampling and annual grassland burning have been pointed as a cause of decline of populations of threatened grasslands bird species in Rio Grande do Sul (Fontana *et al.* 2003), what makes studies on the effects of fire on bird populations necessary for future conservation policies.

In contrast with North America, where there are several studies aiming specifically to assess the fire effects on breeding of grassland birds (e.g., Winter 1999, Churchwell *et al.* 2008, Hovick *et al.* 2012, Verheijen *et al.* 2019), little is known about this issue in South America (e.g., Di Giacomo *et al.* 2011, Bahía & Zalba 2019). In southern Brazilian grasslands none of the studies done on fire and birds (Petry & Krüger 2010, Petry *et al.* 2011, Bettio 2017) have analyzed in detail how fire affects the reproductive success and territories of grassland bird populations. These issues were only briefly discussed in few studies that accessed breeding biology of some grassland species in the region (Rovedder 2011, Franz 2013, Chiarani & Fontana 2015, Repenning & Fontana 2016). The lack of studies on breeding biology of several Neotropical bird species has been reported (Heming *et al.* 2013, Xiao *et al.* 2017), and a probable reason for this is the difficulty of collecting breeding data, because searching for nests takes a long time in the field, increasing the costs of the studies. The current deficiency in financial resources in Brazil, due to the political crisis on science funding (Magnusson *et al.* 2018), makes this type of research more difficult.

Another important issue little investigated worldwide is the temporal dynamics of fire effects. Fuhlendorf *et al.* (2011) in a review found that only 12.5% of the studies in North America assessed fire frequency, a temporal response to fire. Temporal scale strongly influences both the ecosystem responses to fire and the effects of fire (Fuhlendorf *et al.* 2011). Therefore, conservation purposes need to incorporate temporal dynamics to understand the effects of fire on territory size, local population density, and reproductive aspects of grassland birds, mainly if management based on fire is recommended as a strategy for grassland conservation (Churchwell *et al.* 2008, Fuhlendorf *et al.* 2011, Verheijen *et al.* 2019). In southern Brazil, the use of fire has already been recommended to manage grasslands even in protected areas, although this is still a taboo issue (Pillar & Vélez 2010, Andrade et al. 2016).

Fire may have negative impact on reproductive success of grassland birds, destroying their nests if burn occurs during the breeding period (Reinking 2005, Repenning & Fontana 2016), or increasing rates of nest predation and brood parasitism due to higher nest exposition caused by reduced vegetation cover in burned areas (Rohrbaugh *et al.* 1999, Churchwell *et al.* 2008, Davis *et al.* 2016). Fire may also affect size, habitat type, density, and choice of breeding territories of tall-grass species (Rovedder 2011, Chiarani & Fontana 2015). In SBHG, for a grassland-dependent bird species, burned sites showed fewer and smaller territories in comparison with unburned sites, due to the low availability of tall vegetation (which provides appropriate sites for shelter, foraging, and nest building) in burned dry grasslands (Chiarani & Fontana 2015). Fire effects on territory size and local population density should be better investigated, since these parameters are key drivers of population dynamics (Brown 1969, Winter 1999, Verheijen *et al.* 2019).

Here, we assessed temporal changes in nesting success, productivity, territory size, and number of breeding territories of two grassland-dependent species: the Lesser Grass-Finch (*Emberizoides ypiranganus*) and the Black-bellied Seedeater (*Sporophila melanogaster*), a least concern and a near-threatened species (regionally vulnerable), respectively. Our aim was to compare these reproductive and territory parameters over breeding seasons for each species in areas under different fire regimes (an area burned only once in the last years and an area frequently burned). We expected that in the area burned only once, the breeding parameters (e.g., nesting success and productivity) would decrease in the breeding season in which fire occurred in comparison with the breeding seasons before or after fire, while in the area frequently burned there would be no changes over the years. In relation to breeding territories, we expected that in the area burned only once, size and number of territories would decrease in the breeding season following the fire, while no changes among breeding seasons would be observed in the area frequently burned.

METHODS

Study area

Our study was conducted in Parque Estadual do Tainhas (PET), a protected area with 6,654 ha located in northeastern Rio Grande do Sul state, Brazil (29° 05’ 58” S, 50° 21’ 50” W). PET is a conservation unit that includes significant extensions of natural grasslands (Bencke & Duarte 2008), and recognized due to its role in the conservation of birds, including threatened, migratory and/or endemic species (Bencke *et al.* 2006, Chiarani & Fontana 2019). To date, 208 species of birds are reported for PET, of which ten are under some level of threat (Chiarani & Fontana 2019). The landscape of PET includes grasslands, marshes, peat bogs, rocky outcrops, araucaria forests, crops and exotic tree plantations (Bencke & Duarte 2008). The area is situated in the southern Brazilian highland grasslands, in the Atlantic Forest biome, with an undulating relief and mean altitude of 900–1,000 m (Fontana *et al.* 2016). The mean annual temperature ranges from 16 °C to 22 °C and the precipitation is evenly distributed throughout the year (1,500–2,000 mm), reaching up to 2,500 mm in certain subregions (Almeida 2009, Fontana *et al.* 2016).

We assessed temporal changes over three breeding seasons, 2013–2014, 2016–2017 and 2017–2018 (hereafter BS1, BS2, and BS3, respectively) in areas of native grassland with two different histories of fire disturbance: (1) occasional fire (OF) – a 250 ha area burned accidentally only once in the last ten years, in late August 2016; and (2) frequent fire (FF) – a 100 ha area where the use of fire is frequent, burned annually or biennially for cattle raising in the last decade (in all breeding seasons sampled during our study there was fire in this area). In both areas there is low grazing pressure (about 0.5 animal unit per hectare).

Study species

Lesser Grass-Finch is a grassland species that inhabits marshes and grasslands with dense vegetation in southeastern and southern Brazil, southeastern and northern Uruguay, northeastern Argentina, and eastern Paraguay (Ridgely & Tudor 1989, Sick 1997, Tobias *et al.* 1997, Claramunt & Cuello 2004). In our study area, Lesser Grass-Finch is a regular and common species (Chiarani & Fontana 2019) that uses the area to nest from early October to early March, building open-cup nests in grass clumps, mainly *Andropogon lateralis*, *Schizachyrium tenerum*, and *Sorghastrum setosum* (Chiarani & Fontana 2015). Little is known about its feeding habits, but invertebrates probably make up most of its diet (EC, pers. obs.). Lesser Grass-Finch is considered a species of Least Concern (BirdLife International 2019). In Brazil the species is non-threatened but in Argentina and Uruguay it is Vulnerable (Azpiroz *et al.* 2012a, MAyDS & AA 2017).

Black-bellied Seedeater is a tall-grass species often associated to marshy areas (Ridgely & Tudor 1989, Sick 1997). The species is endemic to Brazil and breeds only in a small portion of northeast of Rio Grande do Sul and southeast of Santa Catarina states from November to

March, and later migrates north as far as Minas Gerais and southern Goiás, in central Brazil (Rovedder 2011, Fontana & Repenning 2014, Malacco 2018). In our study area, Black-bellied Seedeater is a regular species during the breeding season, building its nests in small shrubs and grass clumps, both in marshes and dry grasslands (Chiarani & Fontana 2019). Its diet is mainly composed of grass seeds, but invertebrates may eventually be used to feed nestlings (Rovedder 2011). Although Black-bellied Seedeater is Vulnerable in Brazil (MMA 2014), where the entire world population occurs, it is considered a Near-Threatened species at global level (BirdLife 2019). Habitat loss and captures for cage bird trade are the main threats (Fontana & Repenning 2014).

We selected Lesser Grass-Finch and Black-bellied Seedeater because they are grassland-dependent species associated with tallgrass (Azpiroz *et al.* 2012a), with different feeding habits. The two species are abundant in our study area, but Black-bellied Seedeater is a migrant that only occurs there from November to March, and its local breeding population seems to be smaller than Lesser Grass-Finch, a resident species (Chiarani & Fontana 2019). Thus, understanding fire effects on these species may facilitate management initiatives that can also benefit other tall-grass species. The main group of threatened grassland birds in South America is formed by birds fully or partially associated with tall-grass habitats (Azpiroz *et al.* 2012a). Moreover, fire has important effects on breeding territories and nest site of Lesser Grass-Finch and Black-bellied Seedeater (Rovedder 2011, Chiarani & Fontana 2015).

Nest searching and monitoring

We systematically searched for nests of Lesser Grass-Finch in the two areas (OF and FF) during three breeding seasons (BS1, BS2, and BS3) from October to March, totaling 322 days of field work. In the area occasionally burned, the breeding seasons correspond to the period before the fire, the year of the fire, and one year after the fire, respectively. Nests of

Black-bellied Seedeater were searched in the same areas during two breeding seasons (BS2 and BS3) from November to March (because the species arrives at SBHG only in November for breeding), totaling 189 days of field work.

Nest search was conducted by direct observation of the adults' behavior in their breeding territories. Nests found were georeferenced with a handheld GPS device, and then marked with a colored tape placed about 5 m from the nest to facilitate its subsequent location and monitoring. We visited the nests at intervals of 1–5 days, varying according to the nest stage (i.e. shorter interval when events such as laying, hatching or fledging young were approaching). Nests were monitored from the time they were found until they became inactive in order to verify their contents (eggs or nestlings) and fate (successful or unsuccessful).

Nesting survival and productivity

Nests were considered successful when at least one nestling fledged. We considered a nest had been depredated when the contents (eggs or nestlings) disappeared between two consecutive visits, taking into account if nestlings would be old enough to fledge (Pretelli *et al.* 2015). A nest was considered abandoned if the female had not been seen on the nest and a small green leaf placed by us inside the nest had not been removed in more than two consecutive visits (Chiarani & Fontana 2015). As soon as a nest was successful, we followed the parents on feeding fledglings to confirm the nest fate and the number of young individuals, in order to calculate productivity.

Territories

We systematically searched for mated pairs and recorded the total number of territories found in each area during three breeding seasons for Lesser Grass-Finch and two breeding seasons for Black-bellied Seedeater. We captured adult individuals using mist nets and marked them with unique combinations of colored bands and one aluminum band (standard CEMAVE/ICMBio, Federal Brazilian Banding Agency). To delimit the breeding territories, we used a handheld GPS device to record all points where individuals manifested some territory-defense behavior or where processes associated with breeding occurred. Territory-defense behavior includes singing perches or boundaries defended by males in relation to other individuals of the same species, while mating, nest building, and feeding/rearing of young are breeding process, following the reproductive-territory definition type A proposed by Nice (1941) or type I according to Welty & Baptista (1988).

Data analysis

We estimated nesting success by the daily survival rate (DSR) generated by program MARK (White & Burnham 1999). We excluded from the survival analysis nests that did not provide all the basic information that the program requires to build an encounter history, such as abandoned nests. DSR and duration of the nesting cycle (i.e. from the laying of the first egg until birds fledged) were used to estimate cumulative probabilities for nest survival, that is the probability of a nest to survive for a complete nesting cycle. We defined nesting cycle as 26 days for Lesser Grass-Finch (Chiarani & Fontana 2015) and 23 days for the Black-bellied Seedeater (Rovedder 2011). We obtained the variance for cumulative survival probabilities using the delta method, that allows to approximate the variance when the daily nest survival rate is extrapolated to another temporal scale of survival estimate (Powell 2007, Pretelli *et al.* 2015). We used program CONTRAST (Hines & Sauer 1989) to compare cumulative survival probabilities among breeding seasons for each species in each area.

We calculated the annual production of fledglings by dividing the number of fledglings in each season by the total number of nests in the respective season (Ricklefs & Bloom 1977). Productivity of successful nests was obtained similarly, but in this case we divided the number of fledglings by the number of successful nests only. We tested if productivity differed among breeding seasons using the nonparametric Kruskal-Wallis tests.

We used the 95% fixed kernel estimator to calculate territory size of the Lesser Grass-Finch, using the least-squares cross-validation smoothing parameter (Worton 1989, Seaman & Powell 1996). Only territories for pairs with ≥30 locations were measured, to avoid bias in the size estimates (Seaman *et al.* 1999). This analysis was performed with the adehabitatHR package (Calenge 2006), using R 3.4.0 (R Development Core Team 2019). Then, we used Kruskal-Wallis tests to compare territory size among the three breeding seasons in each area (Of and FF). We measured only territories of Lesser Grass-Finch, for which we were able to define with precision the territory boundaries. In order to assess if the number of breeding territories was distributed with equal proportions among breeding seasons, we used a binomial test, considering only territories occupied by mated pairs (for both species). All comparisons were made among breeding seasons instead of between areas due to differences in sample size and because we were interested in assessing temporal changes. The level of significance considered in all tests was $P < 0.05$.

RESULTS

We found and monitored a total of 237 nests of the two species (Table 1). One hundred seventy-eight Lesser Grass-Finch nests were monitored during three breeding seasons: 130 in the area occasionally burned (51 in BS1, 30 in BS2, and 49 in BS3) and 48 in the area with frequent use of fire (12 in BS1, 13 in BS2, and 23 in BS3). Fifty-nine Black-bellied Seedeater nests were monitored in two breeding seasons: 36 in the area occasionally burned (30 in BS2, and six in BS3) and 23 in the area frequently burned (13 in BS2, and 10 in BS3). We banded 57 individuals of Lesser Grass-Finch, and 27 individuals of Black-bellied Seedeater.

Nest survival

Daily survival rates varied over the breeding seasons (Table 2), but the cumulative survival probability of the nests did not show significant differences among years for both species in the two monitored areas. Nest survival probability for Lesser Grass-Finch in OF was 0.390 in BS1 (before the fire), 0.365 in BS2 (year of the fire), and 0.278 in BS3 (one year after the fire), not differing significantly among breeding seasons with or without fire (χ^2_2 = 1.55, *P* = 0.46; Fig. 1). In FF, where fire regimes did not vary through the years, nest survival probability for the species was 0.485 in BS1, 0.312 in BS2, and 0.200 in BS3, not differing significantly among seasons (χ^2_2 = 2.65, *P* = 0.26; Fig. 1). For Black-bellied Seedeater, cumulative probability of nest survival in OF was 0.208 in BS2 (year of the fire), and 0.060 in BS3 (one year after the fire), not differing significantly between breeding seasons (χ^2_1 = 1.74, *P* = 0.19; Fig. 1). In FF, nest survival probability for Black-bellied Seedeater was 0.293 in BS2 and 0.441 in BS3, not differing significantly between breeding seasons either (χ^2_1 = 0.35, *P* = 0.55; Fig. 1).

Productivity

Production of fledglings per nest did not differ among breeding seasons in the area burned only in 2016 (occasional fire), for both Lesser Grass-Finch (*H* = 0.66, *P* = 0.72 all nests; *H* = 0.31, *P* = 0.85 only successful nests) and Black-bellied Seedeater (*H* = 0.16, *P* = 0.69 all nests; *H* = 0.93, *P* = 0.34 only successful nests; see Table 1 for values). In the area with frequent fire there was no significant difference among breeding seasons either, for both Lesser Grass-Finch (*H* = 1.02, *P* = 0.60 all nests; *H* = 0.85, *P* = 0.65 only successful nests) and Black-bellied Seedeater (*H* = 0.41, *P* = 0.52 all nests; *H* = 1.00, *P* = 0.32 only successful nests).

Table 1. Number and productivity of nests of two grassland birds in areas with occasional fire (OF) and areas with frequent fire (FF) in Parque Estadual do Tainhas, northeastern of Rio Grande do Sul, Brazil. Nests were monitored during three breeding seasons (BS1 – BS3) for Lesser Grass-Finch, and two breeding seasons (BS2 and BS3) for Black-bellied Seedeater. Nests are defined as number of successful (S) or unsuccessful (U) nests (percentages between parenthesis), and productivity is the mean of fledglings per nest ± sd. The area with occasional fire was burned in 2016–2017 (BS2).

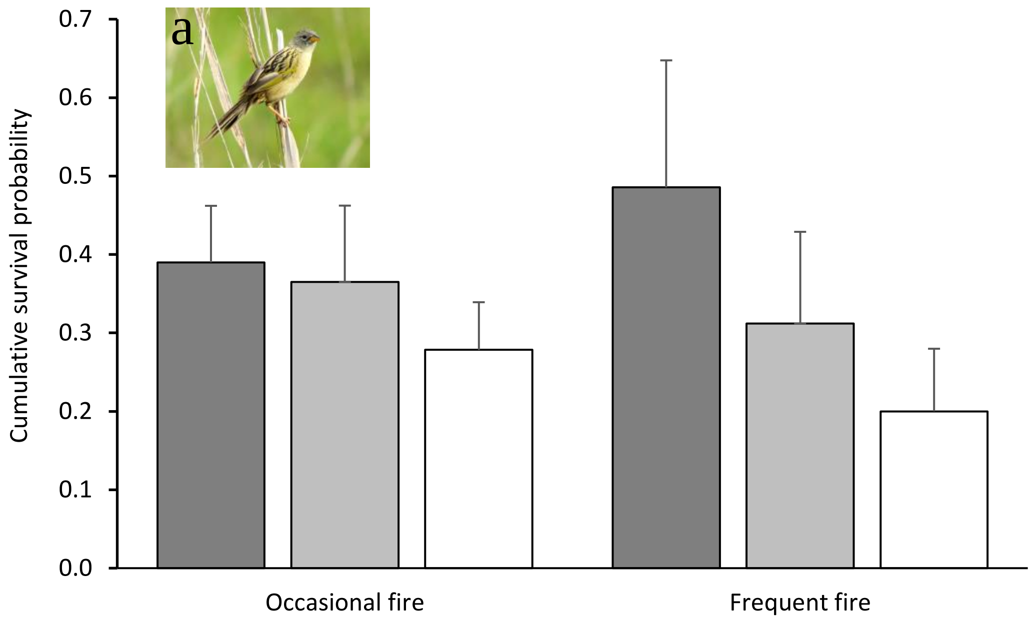
		2013-14 (BS1)			2016-17 (BS2)			2017-18 (BS3)		
Species	Area	S	U	Productivity[a]	S	U	Productivity[a]	S	U	Productivity[a]
Lesser Grass-Finch	OF	22 (43.1)	29 (56.9)	0.84 ± 1.1	11 (36.7)	19 (63.3)	0.77 ± 1.14	16 (32.7)	33 (67.3)	0.67 ± 1.05
				1.95 ± 0.79			2.09 ± 0.83			2.06 ± 0.68
	FF	6 (50)	6 (50)	1.33 ± 1.44	4 (30.8)	9 (69.2)	0.77 ± 1.3	8 (34.8)	15 (65.2)	0.78 ± 1.2
				2.67 ± 0.52			2.5 ± 1			2 ± 0.89
Black-bellied Seedeater	OF				6 (20)	24 (80)	0.4 ± 0.86	2 (33.3)	4 (66.7)	0.5 ± 0.84
							2 ± 0.63			1.5 ± 0.71
	FF				5 (38.5)	8 (61.5)	0.69 ± 0.95	5 (50)	5 (50)	1 ± 1.05
							1.8 ± 0.45			2 ± 0
Total nests		28	35		26	60		31	57	

[a] considering all nests (first value) and only successful nests (second value).

Table 2. Daily survival rates (± se) for Lesser Grass-Finch and Black-bellied Seedeater nests monitored in areas with two fire histories during three breeding seasons (only two for Black-bellied Seedeater), estimated using program MARK.

	Occasional fire			Frequent fire		
Species	2013-14	2016-17 [a]	2017-18	2013-14 [a]	2016-17 [a]	2017-18 [a]
Lesser Grass-Finch	0.964 (0.007)	0.962 (0.010)	0.952 (0.008)	0.972 (0.012)	0.956 (0.014)	0.940 (0.016)
Black-bellied Seedeater		0.934 (0.014)	0.885 (0.055)		0.948 (0.019)	0.965 (0.020)

[a] corresponds to breeding seasons in which fire occurred.



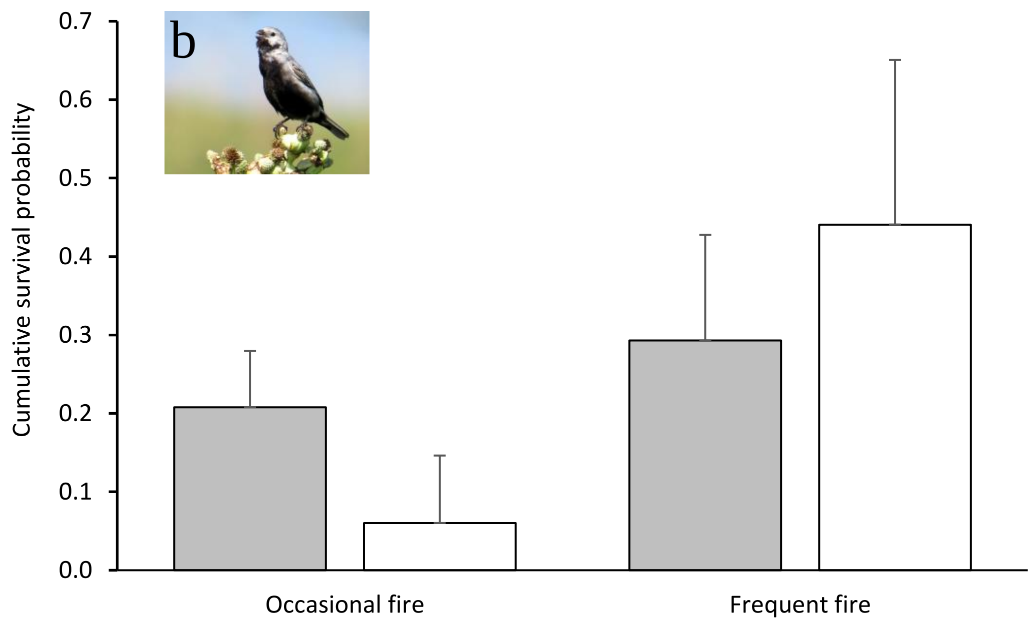


Figure 1. Cumulative survival probability for Lesser Grass-Finch (a) and Black-bellied Seedeater (b) nests monitored in different breeding seasons and areas with two fire histories in Parque Estadual do Tainhas, northeastern of Rio Grande do Sul, Brazil. Dark gray bar corresponds to 2013-2014 breeding season (BS1), light gray bar corresponds to 2016-2017 breeding season (BS2), and white bar corresponds to 2017-2018 breeding season (BS3).

Whiskers are standard errors, calculated using the delta method. All comparisons are non-significant at $P < 0.05$.

Territories

Territory size of Lesser Grass-Finch was significantly different over the breeding seasons in OF ($H = 27.76$, df = 2, $P < 0.001$; Table 3). Post-hoc tests showed that in the season before the fire (BS1) territories were larger than in the season of the fire (BS2; $z = 2.58$, $P = 0.008$) and one year after the fire (BS3; $z = 5.27$, $P < 0.001$) but territories did not differ between BS2 and BS3 ($z = 2.06$, $P = 0.055$). There were not differences in the territory size of Lesser Grass-Finch over the breeding seasons in FF ($H = 0.44$, df = 2, $P = 0.801$; Table 3).

The number of territories of Lesser Grass-Finch in OF was lower in the season of the fire in comparison with the season before the fire (BS2: 14 vs. BS1: 29, $P = 0.03$), and in relation to one year after the fire (BS2: 14 vs. BS3: 31, $P = 0.01$; Supporting Information Figure S1). The number of territories did not differ between the season before the fire and one year after fire (BS1: 29 vs. BS3: 31, $P = 0.90$). In FF the number of territories of Lesser Grass-Finch did not differ among breeding seasons (BS1: 11; BS2: 9; BS3: 11; all comparisons $P \geq 0.82$). The number of territories of Black-bellied Seedeater in OF was lower one year after the fire in comparison with the breeding season of the fire (BS3: 5 vs. BS2: 18, $P = 0.01$). In FF the number of territories of Black-bellied Seedeater did not differ among breeding seasons (BS2: 11 vs. BS3: 9, $P = 0.82$).

Table 3. Territory size (in ha) of Lesser Grass-Finch in areas with two fire histories during three breeding seasons (BS1 – BS3) in Parque Estadual do Tainhas, northeastern of Rio Grande do Sul, Brazil. Values are presented as mean ± sd (with sample size) and the results of Kruskal-

Wallis test among breeding seasons. The breeding season 2016–2017 corresponds to that in which the area with occasional fire was burned.

Area	2013-14 (BS1)	2016-17 (BS2)	2017-18 (BS3)	*H*	*P-value*
Occasional fire	3.77 ± 1.04 (19)	2.58 ± 0.88 (14)	1.9 ± 0.73 (26)	27.76	**<0.001**
Frequent fire	1.74 ± 0.5 (11)	2.03 ± 0.91 (9)	1.94 ± 1.09 (11)	0.44	0.8

DISCUSSION

We found no temporal changes on nest survival and production of fledglings of Lesser Grass-Finch and Black-bellied Seedeater, both in the area burned occasionally and in the area with frequent fire. Although the daily survival rates of Lesser Grass-Finch showed a decrease across the years in both areas it was not statistically different (Fig. 1). This refutes our hypothesis that nest success and productivity would be smaller in the breeding season following the fire in the area burned only once. In Argentina a prescribed fire did not affect the nest survival, clutch size, hatching success, and chick survival of Strange-tailed Tyrant *Alectrurus risora*, in comparison with the years before and after the fire (Di Giacomo *et al.* 2011). Interannual differences on nest survival were not detected between burned areas either, nor between unburned areas for grassland birds in the Argentine Pampas during two breeding seasons following a fire (Bahía & Zalba 2019). In North America, comparisons between two typical methods of grazing (patch-burn grazed pastures and traditionally managed pastures with annual fire) showed little difference in nest survival rates for grasshopper sparrows *Ammodramus savannarum* (Hovick *et al.* 2012). However, our results on nest survival of Black-bellied Seedeater must be interpreted with caution due to the small sample size (number of nests) in the breeding season one year after the fire.

The low number of Black-bellied Seedeater nests found one year after the fire is a result of the decrease of the number of territories in this breeding season. This may be related to a response of birds to the fire occurred one year before and possible changes in their territories, since this species has a philopatric behavior (Rovedder 2011). The potential benefits of returning to previously successful breeding sites could be denied if nesting habitat is variable and suffers disturbances, becoming unsuitable in a given time (Jones *et al.* 2007). Some grassland birds have adapted to this variability, often resulting in fluctuations in local population densities (Winter *et al.* 2005).

Territories of Lesser Grass-Finch were fewer and smaller in the breeding season following the fire in the area burned only once, corroborating our hypothesis. This probably occurred due to the loss of tall-grass vegetation in dry grasslands and, consequently, lower availability of suitable habitat for the species in this year. Fire affects both the composition and the structure of vegetation (Reinking 2005), and after fire, dry grasslands have low vegetation that does not provide appropriate sites for shelter, foraging, and nest building, restricting the territories of Lesser Grass-Finch to wetlands (Chiarani & Fontana 2015). Territories fully located on dry grassland were lost in the breeding season of the fire. One year after the fire some territories in this area occupied portions of dry habitats (some nests were even built in dry grasslands in this breeding season) but remained smaller than in the breeding season before fire, probably because the number of territories increased. In contrast, in the area frequently burned, fire seems not to affect territory stability of the Lesser Grass-Finch over years, because in this area there are few suitable habitats for the species outside the wetlands, being the territories in marshes less affected by fire. Lesser Grass-Finch occupies less dry grasslands in burned areas than in unburned areas (Chiarani & Fontana 2015). Thus, the smaller territory size in areas recently burned is probably related to habitat availability. An important issue in further studies would be to know if an area with frequent fire will present more territories and with larger size if fire disturbance is excluded for some years. Density and territory size are also often related to local habitat quality, which is usually determined by food availability (Marshall & Cooper

2004, Verheijen *et al.* 2019). The smaller territory size of Lesser Grass-Finch in areas recently burned may be also a consequence of the increased abundance of invertebrates. Although we do not have data on this assumption, studies show that the diversity and abundance of arthropods increase in the months following a fire (Swengel 2001, Podgaiski *et al.* 2013).

Our results show that temporal variations occurred in territories in the area burned occasionally, and no changes over years were observed on breeding in both occasional and frequent fire areas. For both species, the reduced number of territories in the year of the fire or one year after the fire shows that we should avoid burning the whole area (i.e. using patch-burn method) when it remains a long time without fire in order to prevent loss of breeding territories. Patch-burn management has been recommended as a useful tool for grassland bird conservation, because it creates the entire gradient of vegetation structure required to maintain grassland bird species that differ in habitat preferences (Fuhlendorf *et al.* 2006, Churchwell *et al.* 2008). In Argentina in the years when accidental fires did not affect the whole area the birds breed in the remaining unburned patches (Di Giacomo *et al.* 2011). Even with no effects on nest survival, a drastic loss of territories should be a major concern because it reduces the number of breeding birds in the population and consequently reduces the number of fledlings per year. For Strange-tailed Tyrant, Di Giacomo *et al.* (2011) consider that if the appropriate size and intensity of the fire are chosen, taking into consideration season and frequency of fire, the impact of prescribed fires on its reproduction can be minimized.

The number of territories of Lesser Grass-Finch returned to previous values (i.e. values recorded before the fire) one year after the fire, suggesting that for this species to recover its density of territories in an area with occasional fire the interval between burns should be at least two years. Interestingly for Black-bellied Seedeater the number of territories was reduced one year after the fire, and a longer study period is required to understand this relation of territory with time since fire. However, this result indicates that Black-bellied Seedeater probably needs a longer interval between fire disturbances than Lesser Grass-Finch. This information is of great concern since the species is threatened in Brazil, which encompasses its entire distribution. Results for Saffron-cowled Blackbird *Xanthopsar flavus*, another grassland threatened species that occurs in the same region of our study, showed that the species returned to a burned marshland to breed three breeding seasons after the fire (Petry & Krüger 2010). The same was observed in Argentina, where Strange-tailed Tyrant showed not to avoid a burned patch in the third breeding season after the prescribed fire (Di Giacomo *et al.* 2011).

Fire is an important structuring factor of natural communities in grasslands (Leys *et al.* 2018). However, management policies in protected natural areas of southern South America are frequently limited to preventing and extinguishing fire, which can lead to loss of biodiversity (Pillar & Vélez 2010, Bahía & Zalba 2019). Our study is the first about temporal effects of fire on breeding and territories of grassland birds in South Brazil and provides information to support management actions in grasslands. Although caution is needed for extrapolations of results due to the number of areas that it was possible to sample, our findings support that controlled prescribed fire can be used in some areas for managing grasslands without effects on breeding of Lesser Grass-Finch and Black-bellied Seedeater. However, the effects on territories of these species suggest that this management must consider the maintenance of portions of grasslands without fire and the interval between burns. Because responses of bird density and nesting success may vary among regions, years, and species, land managers need to provide grasslands with different types of vegetation structure (Winter *et al.* 2005). Another point of major concern is that the fire must occur before the breeding season of birds to avoid burning nests. Further studies could include areas without fire in analyzes, despite the difficulties in finding grassland areas that remain a long time without burn. Additionally, studies should consider long-term effects caused by changes in fire regime of an area, to know, for example, what happens when an area that was occasionally burned starts to be burned more often.

ACKNOWLEDGMENTS

We thank Gerhard Ernst Overbeck, Miguel Ângelo Marini, and Pedro Maria de Abreu Ferreira for important contributions to the manuscript. We thank Patrícia Behling Schafer for improving the English writing of the manuscript. We thank administrative team of Parque Estadual do Tainhas for the support during the field work. E.C. received a scholarship from Conselho Nacional do Desenvolvimento Científico e Tecnológico - CNPq (process 141055/2015-4) and C.S.F. received CNPq grants (processes 457475/2012-9, 303318/2013-6, 30 9438/2016-0, and 402083/2016-4) through Ministério da Ciência, Tecnologia, Inovações e Comunicações (MCTIC). This research also received financial support from Neotropical Grassland Conservancy (NGC).

REFERENCES

Almeida, J.A. 2009. Fatores abióticos. In: *Biodiversidade dos campos do Planalto das Araucárias* (I. I. Boldrini, ed). Ministério do Meio Ambiente, Brasília, DF, Brazil.

Andrade, B.O., Bonilha, C.L., Ferreira, P.M.A., Boldrini, I.I. & Overbeck, G.E. 2016. Highland grasslands at the southern tip of the atlantic forest biome: Management options and conservation challenges. *Oecologia Aust.* **20**: 37–61.

**Andrade, B.O., Bonilha, C.L., Overbeck, G.E., Vélez-Martin, E., Rolim, R.G., Bordignon, S.A.L., Schneider, A.A., Vogel Ely, C., Lucas, D.B., Garcia, É.N., dos Santos, E.D., Torchelsen, F.P., Vieira, M.S., Silva Filho, P.J.S., Ferreira, P.M.A.,

Trevisan, R., Hollas, R., Campestrini, S., Pillar, V.D. & Boldrini, I.I. 2019. Classification of South Brazilian grasslands: Implications for conservation. *Appl. Veg. Sci.* **22**: 168–184.

Askins, R.A., Chávez-ramírez, F., Dale, B.C., Haas, C.A., Herkert, J.R., Knopf, F.L. & Vickery, P.D. 2007. Conservation of Grassland Birds in North America : Understanding Ecological Processes in Different Regions. *Ornithol. Monogr.* **64**: 1–46.

Azpiroz, A.B., Alfaro, M. & Jiménez, S. 2012a. *Lista Roja de las Aves del Uruguay. Una evaluación del estado de conservación de la avifauna nacional con base en los criterios de la Unión Internacional para la Conservación de la Naturaleza.* Dirección Nacional de Medio Ambiente, Montevideo, Uruguay.

Azpiroz, A.B., Isacch, J.P., Dias, R.A., Di Giacomo, A.S., Fontana, C.S. & Palarea, C.M. 2012b. Ecology and conservation of grassland birds in southeastern South America: a review. *J. F. Ornithol.* **83**: 217–246.

Bahía, R. & Zalba, S. 2019. Changes in grassland bird communities and breeding success after a fire in the Argentinian Pampas. *Biodivers. Conserv.*, doi: 10.1007/s10531-019-01850-4.

Behling, H. & Pillar, V.D.P. 2007. Late Quaternary vegetation, biodiversity and fire dynamics on the southern Brazilian highland and their implication for conservation and management of modern Araucaria forest and grassland ecosystems. *Philos. Trans. R. Soc. B Biol. Sci.* **362**: 243–251.

Bencke, G.A. & Duarte, M.M. 2008. *Projeto conservação da Mata Atlântica no Rio Grande do Sul: plano de manejo do Parque Estadual do Tainhas.* Secretaria de Estado do Meio Ambiente do Rio Grande do Sul, Porto Alegre, RS, Brazil.

Bencke, G.A., Maurício, G.N., Develey, P.F. & Goerck, J.M. 2006. *Áreas Prioritárias para a Conservação de aves no Brasil. Parte I – estados do domínio da Mata Atlântica*. SAVE Brasil, São Paulo, SP, Brazil.

Bettio, M. 2017. Resposta da comunidade de aves ao uso do fogo nos campos de altitude do sul do Brasil. M.Sc. Thesis, Pontifícia Universidade Católica do Rio Grande do Sul.

BirdLife International. 2019. IUCN Red List for Birds. Downloaded from http://www.birdlife.org (accessed 08 December 2019).

Bond, W.J., Woodward, F.I. & Midgley, G.F. 2005. The global distribution of ecosystems in a world without fire. *New Phytol.* **165**: 525–538.

Brown, J.L. 1969. Territorial behavior and population regulation in birds: a review and reevaluation. *Wilson Bull.* **81**: 293–329.

Buisson, E., Stradic, S. Le, Silveira, F.A.O., Durigan, G., Overbeck, G.E., Fidelis, A., Fernandes, G.W., Bond, W.J., Hermann, J., Mahy, G., Alvarado, S.T., Zaloumis, N.P. & Veldman, J.W. 2018. Resilience and restoration of tropical and subtropical grasslands , savannas , and grassy woodlands. *Biol. Rev.*, doi: 10.1111/brv.12470.

Butchart, S.H.M., Stattersfield, A.J., Bennun, L.A., Shutes, S.M., Akçakaya, H.R., Baillie, J.E.M., Stuart, S.N., Hilton-Taylor, C. & Mace, G.M. 2004. Measuring global trends in the status of biodiversity: red list indices for birds. *Plos Biol.* **2**: 2294–2304.

Calenge, C. 2006. The package adehabitat for the R software: a tool for the analysis of space and habitat use by animals. *Ecol. Modell.* **197**: 516–519.

Chiarani, E. & Fontana, C.S. 2015. Breeding biology of the Lesser Grass-Finch (Emberizoides ypiranganus) in southern Brazilian upland grasslands. *Wilson J. Ornithol.*

127: 441–456.

Chiarani, E. & Fontana, C.S. 2019. Birds of parque estadual do Tainhas, an important protected area of the highland grasslands of Rio Grande do Sul, Brazil. *Pap. Avulsos Zool.* **59**: 0–4.

Churchwell, R.T., Davis, C.A., Fuhlendorf, S.D. & Engle, D.M. 2008. Effects of Patch-Burn Management on Dickcissel Nest Success in a Tallgrass Prairie. *J. Wildl. Manage.* **72**: 1596–1604.

Claramunt, S. & Cuello, J.. 2004. Diversidad de la biota Uruguaya: Aves. *An. del Mus. Nac. Hist. Nat. y Antropol.* **10**: 1–76.

Davis, A.D., Churchwell, R.T., Fuhlendorf, S.D., Engle, D.M. & Hovick, T.J. 2016. Effect of pyric herbivory on source-sink dynamics in grassland birds. *J. Appl. Ecol.* **53**: 1004–1012.

Di Giacomo, A.G., Di Giacomo, A.S. & Reboreda, J.C. 2011. Effects of grassland burning on reproductive success of globally threatened Strange-tailed Tyrants Alectrurus risora. *Bird Conserv. Int.* **21**: 411–422.

Donald, P.., Green, R.E. & Heath, M.F.. 2001. Agricultural intensification and the collapse of Europe’s farmland bird populations. *Proc. R. Soc. B* **268**: 25–29.

Fontana, C.S., Bencke, G.A. & Reis, R.E. 2003. *Livro vermelho da fauna ameaçada de extinção no Rio Grande do Sul*. Edipucrs, Porto Alegre, RS, Brazil.

Fontana, C.S., Dotta, G., Marques, C.K., Repenning, M., Agne, C.E. & dos Santos, R.J. 2016. Conservation of grassland birds in South Brazil: a land management perspective. *Nat. e Conserv.* **14**: 83–87.

Fontana, C.S., Rovedder, C.E., Repenning, M. & Gonçalves, M.L. 2008. Estado atual do conhecimento e conservação da avifauna dos Campos de Cima da Serra do sul do Brasil, Rio Grande do Sul e Santa Catarina. *Rev. Bras. Ornitol.* **16**: 281–307.

Franz, I. 2013. História natural de Sporophila hypoxantha Cabanis, 1851 (AVES: EMBERIZIDAE) em campos de altitude no Sul do Brasil. M.Sc. Thesis, Pontifícia Universidade Católica do Rio Grande do Sul.

Fuhlendorf, S.D., Harrell, W.C., Engle, D.M., Hamilton, R.G., Davis, C.A. & Jr, D.M.L. 2006. Should heterogeneity be the basis for conservation? Grassland bird responses to fire and grazing. *Ecol. Appl.* **16**: 1706–1716.

Fuhlendorf, S.D., Limb, R.F., Engle, D.M. & Miller, R.F. 2011. Assessment of Prescribed Fire as a Conservation Practice. In: *Conservation Benefits of Rangeland Practices Assessment, Recommendations, and Knowledg Gaps* (D. D. Briske, ed). USDA-NRCS, Washington, D.C., USA.

Heming, N.M., Greeney, H.F. & Marini, M.Â. 2013. Breeding Biology Research and Data Availability for New World Flycatchers. *Nat. Conserv.* **11**: 54–58.

Hines, J.H. & Sauer, J.R. 1989. *Program CONTRAST: a General Program for the Analysis of Several Survival or Recovery Rate Estimates*. Department of the Interior, Fish and Wildlife Service, Washington, DC, USA.

Hovick, T.J., Miller, J.R., Dinsmore, S.J., Engle, D.M., Debinski, D.M. & Fuhlendorf, S.D. 2012. Effects of fire and grazing on grasshopper sparrow nest survival. *J. Wildl. Manage.* **76**: 19–27.

Jones, S.L., Dieni, J.S., Green, M.T. & Gouse, P.J. 2007. Annual Return Rates of Breeding

Grassland Songbirds. *Wilson J. Ornithol.* **119**: 89–94.

Leys, B.A., Marlon, J.R., Umbanhowar, C. & Vannière, B. 2018. Global fire history of grassland biomes. *Ecol. Evol.* **8**: 8831–8852.

Magnusson, W.E., Grelle, C.E.V., Marques, M.C.M., Rocha, C.F.D., Dias, B., Fontana, C.S., Bergallo, H., Overbeck, G.E., Vale, M.M., Tomas, W.M., Cerqueira, R., Collevatti, R., Pillar, V.D., Malabarba, L.R., Lins-e-Silva, A.C., Neckel-Oliveira, S., Martinelli, B., Akama, A., Rodrigues, D., Silveira, L.F., Scariot, A. & Fernandes, G.W. 2018. Effects of Brazil's political crisis on the science needed for biodiversity conservation. *Front. Ecol. Evol.* **6**: 1–5.

Marshall, M.R. & Cooper, R.J. 2004. Territory size of a migratory songbird in response to caterpillar density and foliage structure. *Ecology* **85**: 432–445.

MAyDS & AA. 2017. *Categorización de las Aves de la Argentina*. Ministerio de Ambiente y Desarrollo Sustentable y Aves Argentina, Buenos Aires, Argentina.

Nice, M.M. 1941. The Role of Territory in Bird Life. *Am. Midl. Nat.* **26**: 441–487.

Overbeck, G.E., Müller, S.C., Pillar, V.D. & Pfadenhauer, J. 2005. Fine-scale post-fire dynamics in southern Brazilian subtropical grassland. *J. Veg. Sci.* **16**: 655–664.

Overbeck, G.E. & Pfadenhauer, J. 2007. Adaptive strategies in burned subtropical grassland in southern Brazil. *Flora Morphol. Distrib. Funct. Ecol. Plants* **202**: 27–49.

Pausas, J.G. & Keeley, J.E. 2009. A Burning Story: The Role of Fire in the History of Life. *Bioscience* **59**: 593–601.

Petry, M.V. & Krüger, L. 2010. Frequent use of burned grasslands by the vulnerable

Saffron-Cowled Blackbird Xanthopsar flavus: implications for the conservation of the species. *J. Ornithol.* **151**: 599–605.

Petry, M.V., Piuco, R. da C. & Brummelhaus, J. 2011. Aves Associadas ao Manejo com Fogo em Áreas de Campo na Porção Sul do Bioma Mata Atlântica. *Biodiversidade Bras.* **1**: 110–119.

Pillar, V.D.P. & Vélez, E. 2010. Extinção dos Campos Sulinos em Unidades de Conservação: um Fenômeno Natural ou um Problema ético? *Nat. Conserv.* **08**: 84–86.

Podgaiski, L.R., Joner, F., Lavorel, S., Moretti, M., Ibanez, S., Mendonça, M.D.S. & Pillar, V.D. 2013. Spider Trait Assembly Patterns and Resilience under Fire-Induced Vegetation Change in South Brazilian Grasslands. *PLoS One* **8**: e60207.

Powell, L.A. 2007. Approximating Variance of Demographic Parameters Using the Delta Method: a Reference for Avian Biologists. *Condor* **109**: 949.

Pretelli, M.G., Isacch, J.P. & Cardoni, D.A. 2015. Effects of fragmentation and landscape matrix on the nesting success of grassland birds in the Pampas grasslands of Argentina. *Ibis (Lond. 1859).* **157**: 688–699.

R Development Core Team. 2019. R: A language and environment for statistical computing.

Reinking, D.L. 2005. Fire regimes and avian responses in the central tallgrass prairie. *Stud. Avian Biol.* **30**: 116–126.

Repenning, M. & Fontana, C.S. 2016. Breeding biology of the Tropeiro Seedeater (Sporophila beltoni). *Auk* **133**: 484–496.

Ricklefs, R.E. & Bloom, G. 1977. Components of avian breeding productivity. *Auk* **94**: 86–

96.

Ridgely, R.S. & Tudor, G. 1989. *The Birds of South America, vol. I: The Oscine Passerines*. University of Texas Press, Austin, TX, USA.

Rohrbaugh, R.W., Dan, J.R., Donald, L.R., Steve, H.W. & Jenkins, M.A. 1999. Effects of prescribed burning and grazing on nesting and reproductive success of three grassland passerine species in tallgrass prairie. *Stud. Avian Biol.* **19**: 165–170.

Rovedder, C.E. 2011. História natural de Sporophila melanogaster (Pelzen 1870) (Aves: Emberizidae) com ênfase em sua biologia reprodutiva. M.Sc. Thesis, Pontifícia Universidade Católica do Rio Grande do Sul.

Seaman, D.E., Millspaugh, J.J., Kernohan, B.J., Brundige, G.C., Raedeke, K.J. & Gitzen, R.A. 1999. Effects of sample size on kernel home range estimates. *J. Wildl. Manage.* **63**: 739–747.

Seaman, D.E. & Powell, R.A. 1996. An evaluation of the accuracy of kernel density estimators for home range analysis. *Ecology* **77**: 2075–2085.

Sick, H. 1997. *Ornitologia Brasileira*. Editora Nova Fronteira, Rio de Janeiro, RJ, Brazil.

Staver, A.C., Archibald, S. & Levin, S.A. 2011. The global extent and de- terminants of savanna and forest as alternative biome states. *Science (80-.).* **334**: 230–232.

Sühs, R.B., Giehl, E.L.H. & Peroni, N. 2020. Preventing traditional management can cause grassland loss within 30 years in southern Brazil. *Sci. Rep.* **10**: 783.

Swengel, A.B. 2001. A literature review of insect responses to fire, compared to other conservation managements of open habitat. *Biodivers. Conserv.* **10**: 1141–1169.

Tobias, J.A., Clay, R.P. & Lowen, J.C. 1997. Field identification of Lesser Grass-finch Emberizoides ypiranganus. *Cotinga* **8**: 75–78.

Verheijen, B.H.F., Clipp, H.L., Bartolo, A.J., Jensen, W.E. & Sandercock, B.K. 2019. Effects of patch-burn grazing on breeding density and territory size of Dickcissels. *Avian Conserv. Ecol.* **14**.

Welty, J.C. & Baptista, L. 1988. *The life of birds*. W. B. Sauders, New York, NY, USA.

White, G.C. & Burnham, K.P. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study* **46**: 120–138.

White, R., Murray, S. & Rohweder, M. 2000. *Pilot Analysis of Global Ecosystems: Grassland Ecosystems*. World Resources Institute, Washington, DC, USA.

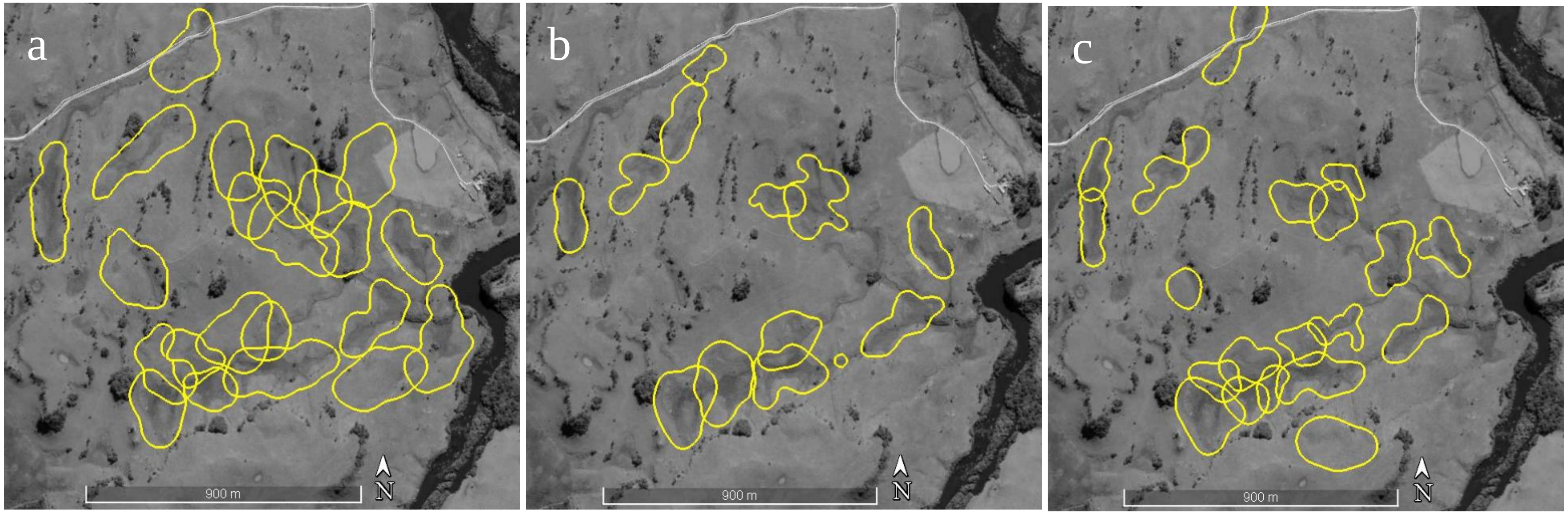
Winter, M. 1999. Relationship of fire history to territory size, breeding density, and habitat of Baird's Sparrow in North Dakota. *Stud. Avian Biol.* **19**: 171–177.

Winter, M., Johnson, D.H. & Shaffer, J.A. 2005. Variability in Vegetation Effects on Density and Nesting Success of Grassland Birds. *J. Wildl. Manage.* **69**: 185–197.

Worton, B.J. 1989. Kernel methods for estimating the utilization distribution in home range studies. *Ecology* **70**: 164–168.

Xiao, H., Hu, Y., Lang, Z., Fang, B., Guo, W., Zhang, Q., Pan, X. & Lu, X. 2017. How much do we know about the breeding biology of bird species in the world? *J. Avian Biol.* **48**: 513–518.

SUPPORTING INFORMATION



Supporting Information Figure S1. Territories of Lesser Grass-Finch during three breeding seasons (a, season before the fire; b, season following the fire; c, season one year after the fire) in an area occasionally burned in Parque Estadual do Tainhas, northeastern of Rio Grande do Sul, Brazil. Polygons represent some territories monitored in each breeding season and delimited using the 95% kernel estimator.

CONCLUSÕES GERAIS

Esse trabalho permitiu o entendimento de como o fogo influencia a comunidade e a reprodução de aves campestres em uma escala temporal. A resiliência da avifauna mostrou que o fogo pode ser uma opção de manejo dos campos, desde que feito de forma correta, ou seja, respeitando o período e a periodicidade das queimadas. O período é importante para a queima não coincidir com a época de reprodução das aves (i.e., deve ocorrer anteriormente), e a periodicidade implica dar o tempo necessário para uma área se recuperar após uma queimada. Durante o estudo, algumas espécies de aves (inclusive ameaçadas) só foram registradas em áreas recentemente queimadas, enquanto outras espécies tiveram sua abundância reduzida após o fogo. Isso mostra que o fogo deveria ser usado de forma a criar mosaicos de áreas com diferentes tempos desde a última queimada, gerando, assim, uma paisagem heterogênea de locais com alturas de vegetação variadas para favorecer um maior número de espécies de aves.

A utilização do fogo como ferramenta de manejo de áreas campestres é muito debatida em todo o mundo, embora essa questão seja mais abordada em relação aos efeitos das queimadas sobre a vegetação, como ocorre no Brasil. Uma melhor compreensão dos efeitos do fogo passa por uma análise que integre diferentes grupos taxonômicos da fauna e da flora. Apesar de muitos estudos demonstrem que distúrbios como o fogo são importantes para a manutenção de áreas campestres em algumas regiões do Brasil, essa temática ainda é vista como um tabu para a conservação, especialmente dentro de áreas protegidas. Após uma regulamentação, queimadas prescritas para fins científicos e/ou econômicos do campo deveriam ser mais utilizadas e estudadas no Brasil, pois uma das principais dificuldades de trabalhos com fogo é o encontro de áreas para desenvolver estudos de longo prazo. É frequente a ocorrência de queimadas acidentais (ou criminosas) que geram incêndios de grandes proporções, comprometendo projetos de pesquisa.

Nos campos de altitude do sul do Brasil, a pecuária possui uma estreita ligação com o uso do fogo. Além de questões econômicas, a proibição do uso do fogo contribuiu para mudanças significativas no uso da terra nas últimas décadas na região, transformando campos nativos que eram usados para criação de gado em áreas de lavouras ou silvicultura. Por isso, estudos que elucidem os efeitos do fogo sobre o campo são fundamentais para permitir que uma atividade econômica capaz de conciliar o uso do campo com sua conservação não continue sendo substituída por outras que resultam na perda desse importante ecossistema.

MARISTA PUCRS

Pontifícia Universidade Católica do Rio Grande do Sul
Pró-Reitoria de Graduação
Av. Ipiranga, 6681 - Prédio 1 - 3º. andar
Porto Alegre - RS - Brasil
Fone: (51) 3320-3500 - Fax: (51) 3339-1564
E-mail: prograd@pucrs.br
Site: www.pucrs.br