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## Fire shapes mammal abundance at the Cerrado-Pantanal ecotone: Scale of effect, species traits and land-cover interaction

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1 Fire shapes mammal abundance at the Cerrado-Pantanal ecotone: Scale of effect, species traits and  
2 land-cover interaction

3

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18 **Abstract**

19 In tropical regions, which harbour the majority of the Earth's biodiversity, land-cover change emerges  
20 as the primary driver of biodiversity loss. However, despite the propensity for many tropical biomes to  
21 experience fires, and the rapidly accelerating pace of intensification in fire regimes, the role of fire in  
22 shaping ecological communities has largely been overlooked. We examined whether and how fire  
23 regime affected the abundance of mammal species, the scale at which these putative effects operated  
24 and how fire effect interacted with land cover. Using camera traps, we studied four mammal species  
25 with different ecological traits in the Kadiwéu indigenous territory, a Cerrado-Pantanal ecotone.  
26 Specifically, we used abundance models to analyse the response of mammal species to fire frequency,  
27 spatial extent, and time since fire. Our results showed that mammals responded to fire at scales that  
28 align with the scale of responses to land cover. We found that the type of response to fire, and the scale  
29 at which fire effects operate, depended on species and possibly on their traits. The smallest species  
30 *Sylvilagus brasiliensis* demonstrated a clear response to fire, as well as to the mean patch area of  
31 savanna, exclusively at the smallest scale studied (i.e., within 500-m radius buffers). The abundance of  
32 this species decreased with increasing proportion of land burned in the 12 months preceding the  
33 monitoring. In contrast, larger species, including *Mazama gouazoubira* and *Tapirus terrestris*, showed  
34 no response to fire, while *Pecari tajacu* would exhibit a marginal response, at the larger scale of 1000  
35 m. Our results emphasize the importance of adopting trait-based approaches that consider the multiple  
36 aspects of fire regimes, at multiple scales, to disentangle the mechanisms governing the effects of fire  
37 on biodiversity. This should promote effective and sustainable fire management compatible with  
38 systemic conservation of species and ecological traits in fire-prone biomes.

39

40 **Keywords:** Neotropical mammals – Camera-trap monitoring – Cerrado hotspot – Tropical savanna – Fire  
41 management – Biodiversity conservation

## 42 1. Introduction

43 Wildfires have been a natural phenomenon and fundamental ecological process in shaping landscapes  
44 for millennia (Pivello et al., 2021). In the recent years, fire has exhibited a marked increase in activity  
45 (i.e., occurrence, size and intensity; Garcia et al., 2021; Iglesias et al., 2022; Nolan et al., 2022), largely  
46 attributed to climate change (Williams et al., 2019) and additional human-induced environmental  
47 changes, including land-use change, fragmentation, and inappropriate fire suppression policies  
48 (Durigan, 2020; Pausas & Keeley, 2021). The emergence of megafires, defined as fire events whose  
49 spatial extent exceeds 10,000 ha (Linley et al., 2022), has become commonplace (Iglesias et al., 2022),  
50 and future projections further predict a global escalation of wildfire potential (Bowman et al., 2020).  
51 This situation is all the more worrying in the tropics (Liu et al., 2010) where many biomes are highly  
52 flammable, such as the fire-prone Pampa, Cerrado or Pantanal in South America, within which intricately  
53 associated fire-sensitive ecosystems, such as tropical rainforests of the Amazon and Atlantic Forest, are  
54 embedded (Pivello et al., 2021). As a result, megafires kill millions of vertebrates, including in fire-  
55 dependent ecosystems (Tomas et al., 2021). Therefore, understanding the compound effects of the  
56 dynamics of fire regimes and landscape features in space and time is not only key to anticipate the  
57 effects of changing fire regimes for biodiversity conservation, but also crucial to inform sustainable fire  
58 management by designing low fire risk landscapes that harmonize the coexistence of indigenous  
59 communities with wildfire (Doerr & Santín, 2016; Oliveira et al., 2022; Pausas & Keeley, 2021).

60 Fire regimes can be characterized through spatial and temporal aspects, including fire  
61 frequency, intensity, spatial extent (or size), and time since the last fire (Nolan et al., 2022; Pausas &  
62 Keeley, 2021; Turner, 2010). In the tropics, high fire frequency over the long term maintains a marked  
63 shift in plant communities at the forest-savanna interface by limiting the establishment, growth and  
64 recruitment of forest species seedlings, thereby preventing the woody encroachment of savannas  
65 (Hoffmann et al., 2009; Mourik et al., 2007). This results in a high degree of specialisation of plant  
66 communities, with low-diversity communities composed of fire-adapted species in open landscapes  
67 subject to a high fire frequency, and highly diverse plant communities in closed landscapes (Costa et al.,

68 2023; Dantas et al., 2013). In turn, such variation in plant communities adversely affects vegetation  
69 structure and resource availability (Bond & Keeley, 2005) for vertebrate communities (Fontaine et al.,  
70 2009). For instance, Barlow & Peres (2006) found that fire frequency negatively affected the availability  
71 of fleshy fruits in areas burned once within 10 years, causing the decline of large frugivorous vertebrates.  
72 This effect was exacerbated in areas burned twice in 10 years, causing the extirpation of most forest  
73 specialists. Increased fire frequency would jeopardise mammal recruitment and survival (Griffiths &  
74 Brook, 2015), and population viability and persistence more than fire spatial extent (Griffiths et al.,  
75 2015). However, increased spatial extent of wildfires can also affect vertebrate communities (Pastro et  
76 al., 2011), by reducing survival, abundances and species richness locally, particularly when the fire  
77 extent exceeds the animal's home range (Lawes et al., 2015). Finally, time since the last fire, that relates  
78 to post-fire regeneration of the vegetation, may affect vertebrate populations (Pastro et al., 2011).  
79 Recolonisation and recovery of local populations from burned areas may occur as the vegetation  
80 regenerates and becomes once again suitable to fulfill the ecological requirements of species (e.g., food,  
81 shelter; Chia et al., 2016). For example, Valentin-Silva et al. (2021) found that fruiting is reduced in the  
82 month following fire. Such reduction in the immediate post-fire availability of food resources could  
83 explain the decrease in vertebrate abundance during the early compared to the late post-fire period  
84 (Giorgis et al., 2021).

85         The effects of fire may be species dependent (Souza et al., 2023), varying with their traits  
86 (Pocknee et al., 2023), such as body size (Nieman et al., 2022) and specialisation in resource use,  
87 including habitat and food preferences (Barlow & Peres, 2006; Griffiths & Brook, 2014). First, although  
88 large-bodied species with large home ranges are likely to be exposed to landscape changes, their high  
89 movement capacity could allow them to escape fire and recolonise burned areas more easily than  
90 smaller species (Cardillo et al., 2005; Chia et al., 2016). Conversely, small-bodied species would be more  
91 vulnerable, particularly if the extent of fire encompasses their small home ranges and their dispersal  
92 distance does not allow them to reach unburned refugia (Lawes et al., 2015). Complex relationships may  
93 also exist when considering the time since fire and the successional stages of post-fire vegetation

94 regeneration (Briani et al., 2004). Indeed, small grazer species would occupy recently-burned areas  
95 faster than larger browsers (Nieman et al., 2022), probably because of the shorter time needed for  
96 vegetation regeneration in grass than in shrub species. Second, greater flexibility in resource use,  
97 including habitat and food, makes generalist species less sensitive to fire disturbance than specialists. In  
98 the Brazilian Cerrado, fires favour the invasion of generalist small-mammal species from open  
99 formations at the expense of more specialized forest species (Camargo et al., 2018).

100         Using camera traps, the current study aimed to investigate whether and how effects of fire  
101 regimes shaped mammal abundances in addition to, or interaction with land cover in the Kadiwéu  
102 Indigenous Territory, Brazil. The study region is an ecotone between the Cerrado—a biodiversity  
103 hotspot and the most species-rich tropical savanna in the world (Myers et al., 2000)— and Pantanal  
104 wetland. Mammals are key species providing ecosystem services in Brazil (Vale et al., 2023), and are  
105 known to be sensitive to land-use/land-cover change (Harmange et al., 2023; Magioli et al., 2021) and  
106 fire (Souza et al., 2023). In the Cerrado, studies demonstrated that decreasing forest cover negatively  
107 affected the probability of occurrence of forest-specialist mammals (e.g., *Dasyprocta azarae* and *Pecari*  
108 *tajacu*; Amiot et al., 2021). Land cover around forest fragments plays a key role in modulating this effect  
109 and could contribute to maintain the diversity of mammal functional groups (Harmange et al., 2023).  
110 While the effects of the land cover on mammal communities are well documented, the influence of fire  
111 has remained understudied, resulting in uncertainty about its effects (but see Souza et al., 2023). Here,  
112 we examined how fire frequency, spatial extent, and time since fire, affected the abundance of mammal  
113 species, and determined the scale at which the effects of fire regime on species operates. Specifically,  
114 we focused on the four most abundant species in the camera-trap monitoring, which are common in  
115 the Brazilian Cerrado: *Sylvilagus brasiliensis*, a small generalist and charismatic endangered species that  
116 provides cultural ecosystem services (Ruedas & Smith, 2019); *Pecari tajacu*, a medium-sized forest  
117 specialist considered an ecosystem engineer that contributes to seed dispersal and nutrient transfer;  
118 *Mazama gouazoubira*, a medium-sized generalist species of cultural importance that is also used as a  
119 sentinel for disease prevalence, and *Tapirus terrestris*, a large generalist and charismatic vulnerable

120 species that contributes to seed dispersal and nutrient transfer (Amiot et al., 2021; Vale et al., 2023)  
121 (Appendix A, Table A.1).

122 In the Brazilian Cerrado, previous studies showed that mammal species respond to landscape  
123 processes at scales ranging from 500 to 1500 m, depending on their traits (Amiot et al., 2021; Harmange  
124 et al., 2023). We therefore expect mammal species to respond to fire at scales similar to land cover, as  
125 responses to fire also depend on morphological traits and ecological requirements (Pocknee et al., 2023;  
126 Souza et al., 2023). While the effects of land cover on mammal communities are well documented, we  
127 expected the abundance of the smallest species, *S. brasiliensis*, to be negatively affected by the increase  
128 in fire extent, as a result of a presumed limited movement capacity compared to larger species (Lawes  
129 et al., 2015). As a frugivorous forest specialist, *P. tajacu* is expected to be negatively affected by an  
130 increased fire frequency over a short period (e.g., areas burned twice; Barlow & Peres, 2006), and  
131 positively affected by time since fire, as a result of a presumed reduction in post-fire fruit availability  
132 (Valentin-Silva et al., 2021). Large generalist species, including *M. gouazoubira* and *T. terrestris*, are  
133 expected to be not much affected by fire (Chia et al., 2016), due to their greater movement capacity  
134 and flexibility in resource use (Camargo et al., 2018; Lawes et al., 2015).

135

## 136 2. Material and methods

### 137 2.1. Study area

138 The study was conducted from June to August 2021, in the Kadiwéu Indigenous Territory (from  
139 20°10'45.59" to 21°03'10.55" S and from 56°50'52.30" to 57°50'51.27" W), Mato Grosso do Sul, Brazil  
140 (Figure 1). The Kadiwéu IT comprises a region of about 5400 km<sup>2</sup> located in a transition area between  
141 two biomes: the Pantanal (wetlands) and Cerrado (savanna) (Oliveira et al., 2022). The local vegetation  
142 is mainly composed of savanna, deciduous seasonal forests, and periodically flooded mixed grasslands,  
143 composed of a mix of native and exotic species (Ferreira et al., 2021; Silva et al., 2014). The study area  
144 covers 114 km<sup>2</sup> and is situated in the eastern part of the Kadiwéu IT, within the Cerrado biome,  
145 recognized as a biodiversity hotspots (Myers et al., 2000).

146 In the Cerrado and Pantanal, fires are a common and recurring phenomenon, annually burning  
147 one to several tens of thousands of square kilometres (Araújo et al., 2012; Libonati et al., 2020). In the  
148 Kadiwéu IT, historically, fire has been used for hunting, agricultural and pastoral purposes, and to  
149 stimulate regrowth while limiting the encroachment of woody vegetation in pastures (Damasceno-  
150 Junior et al., 2021; Mistry et al., 2016). Wildfires are seasonal, with the highest frequencies occurring in  
151 August and September (Oliveira et al., 2022). Therefore, preventive measures, such as prescribed  
152 burning—fires deliberately set in strategic locations to reduce the likelihood of wildfires spreading—are  
153 conducted in May–July by indigenous brigades established in 2009.

154

## 155 2.2. Mammal data

156 In the study area, we set 36 camera traps (Bushnell Trophy Cam Aggressor Brown and Reconyx Hyperfire  
157 HC 500) (Figure 1). The monitoring was conducted in two different sessions of 30 consecutive days, from  
158 24 June to 24 July 2021 (n = 15 camera traps), and from 25 July to 24 August 2021 (n = 21), respectively.  
159 Camera traps were set at a height of 40 cm above the ground with a 10° inclination towards the ground.  
160 They were configured to take a series of three photographs when triggered, with a 3-second delay  
161 between two consecutive captures. At the end of the sampling period, photographs were downloaded  
162 and analysed visually using Wild.ID, a software developed by the Tropical Ecology Assessment and  
163 Monitoring Network (TEAM– [www.wildlifeinsights.org/team-network](http://www.wildlifeinsights.org/team-network)).

164 To run abundance analyses from photographs of unmarked individuals, we first needed to  
165 prevent pseudo-replication by filtering out repeated captures of the same individual in consecutive  
166 photographs (Zaman et al., 2022). Following common practices in studies using camera-traps (Peral et  
167 al., 2022), we considered for each species only consecutive photographs with at least a 30-minute  
168 interval as independent instances. Then, for each of the 36 camera traps and from the 31 sampled days,  
169 we built a matrix of 5 sampling occasions of 6 consecutive days each (see Ribeiro et al., 2019; see also  
170 Amiot et al., 2021). Over the 36 camera traps, 8 camera traps (i.e., 22%) detected the common tapeti

171 *Sylvilagus brasiliensis*, 8 (22%) detected the collared peccary *Pecari tajacu*, 20 (56%) detected the brown  
172 brocket deer *Mazama gouazoubira*, and 14 (39%) detected the South American tapir *Tapirus terrestris*.  
173

### 174 2.3. Calculation of fire metrics

175 To quantify the patterns of fire regimes at camera-trap locations, we calculated fire metrics based on  
176 two MapBiomas Fire products for the year 2021 (2<sup>nd</sup> collection), derived from Landsat images (30-m  
177 resolution): “Monthly burned coverage” and “Fire frequency” (Figure 1A–B and 1C–D, respectively). For  
178 each pixel that burned during the year 2021, the “Monthly burned coverage” value corresponds to the  
179 month in which fire scars were detected, ranging from 0 (no fire detected) to 12 (fire occurrence  
180 recorded in December). For each pixel, the “Fire frequency” value corresponds to the number of years  
181 a fire was recorded inside the pixel, from 1985 to 2021. Therefore, values range from 0 (no fire recorded  
182 in the period 1985–2021) to 37 (fire recorded in the pixel every year between 1985–2021).

183 The ability to detect effects of environmental variables (e.g., fire: Souza et al., 2023; land cover:  
184 Jackson & Fahrig, 2012) on animal populations may depend on the spatial extent used for metric  
185 calculations. Fire metrics were therefore computed at three spatial extents, defined as buffers with radii  
186 of 500, 750 and 1000 m from the camera-trap locations. Given the 30-m spatial resolution of the fire  
187 maps (based on Landsat images), we used a minimum radius of 500 m to ensure an adequate quantity  
188 of pixels offering the necessary variation for meaningful metric assessment (see Amiot et al., 2021).  
189 Since the mean distance between camera traps was 997 m ( $\pm$  189 SD), we used a maximum radius of  
190 1000 m to limit the overlap between neighbouring buffers (Appendix A, Table A.2) and prevent spatial  
191 autocorrelation (Amiot et al., 2021; Zuckerberg et al., 2012). Four types of fire metrics were derived  
192 from the MapBiomas products (see Table 1). The “frequency” metrics aimed to assess the recurrence  
193 of fire-induced disturbance. They include the mean and maximum frequencies recorded from 1985 to  
194 2021 in a given landscape (i.e., buffer area), and the proportion of land burned at least twice within the  
195 12 months preceding the monitoring (see Table 1 for details). The “spatial extent” metric aimed to  
196 characterize the spreading of the fire event, and is calculated as the proportion of the area of a

197 landscape (i.e., buffer area) burned within the 12 months preceding the monitoring. Finally, the  
198 “regeneration” metric aimed to quantify the delay since the last significant fire in the landscape, serving  
199 as a proxy for vegetation regeneration. It classifies the landscapes into two categories: “0-4 months” for  
200 those with a recorded fire within the 4 months preceding monitoring, and “≥ 11 months” for those in  
201 which the last recorded fire occurred at least 11 months before monitoring (see Table 1 for details).

202

#### 203 2.4. Landscape description and calculation of land-cover metrics

204 A land-cover map was prepared to characterize the landscape of the Kadiwéu IT for the year 2021. We  
205 used the 2021 annual land-cover map from MapBiomias (7<sup>th</sup> collection) (Souza et al., 2020), derived from  
206 Landsat images (30-m resolution) to produce a reclassified map with four land-cover classes (see Figure  
207 1E–F and Appendix A, Table A.3). ‘*Savanna*’ designated pixels categorized as savanna formation  
208 according to the MapBiomias classes. *Savanna* represented 60.7% of the study area’s land cover. ‘*Forest*’  
209 designated pixels categorized as forest formation according to the MapBiomias and represented 26.9%  
210 of the study area. ‘*Grassland*’ accounted for a total of 11.4% of the study area and mainly comprised  
211 natural grasslands (11.0% of the study area, while anthropogenic pastures represented 0.3%). Finally,  
212 ‘*Other*’ accounted for 1% of the study area and designated pixels of lesser-represented land-cover types  
213 (i.e., mosaic of farming uses, wetlands, and rivers and lakes) (Appendix A, Table A.3).

214 To characterize land cover around camera traps, we calculated land-cover metrics from the  
215 previously prepared land-cover map. As for fire metrics, land-cover metrics were calculated at three  
216 different spatial extents, specifically within buffers with radii of 500, 750 and 1000 m from the camera-  
217 trap locations. The calculated land-cover metrics encompass two class-level and two landscape-level  
218 metrics. First, class-level metrics include: (i) the proportion of land-cover class (composition metric),  
219 representing the percentage of the landscape occupied by the given land-cover class among *Savanna*,  
220 *Forest*, and *Grassland*; and (ii) the mean patch area (configuration metric), indicating the average size  
221 of patches of a given land-cover class within a landscape (represented as the buffer area). This metric  
222 was also applied to the land-cover classes *Savanna*, *Forest*, and *Grassland*. Secondly, landscape-level

223 metrics include: (i) the Shannon Diversity Index (composition metric) that measures landscape  
224 heterogeneity by considering both the number of classes and their respective amount within a  
225 landscape (i.e., a buffer area); and (ii) the edge density (configuration metric) which quantifies the total  
226 length of edges of land-cover classes within a landscape, divided by the total area of the given landscape  
227 (i.e., buffer area). Both landscape-level metrics were calculated considering the four land-cover classes  
228 of the map previously prepared, i.e., *Savanna*, *Forest*, *Grassland* and *Other*. All metrics were calculated  
229 using the R package *landscapemetrics* (Hesselbarth et al., 2019). All these metrics were used as  
230 covariates in the models described in the statistical analyses (see below), with the exception of Shannon  
231 Diversity Index (SHDI) which was strongly correlated, at each buffer extent, with savanna proportion  
232 (Pearson's correlation at 500-m buffer extent:  $r = -0.74$ ;  $p < 0.0001$ ), mean patch area of savanna ( $r = -$   
233  $0.81$ ;  $p < 0.0001$ ) and edge density ( $r = 0.87$ ;  $p < 0.0001$ ) (see Appendix B for details).

234

## 235 2.5. Statistical analyses

### 236 2.5.1. Scale of effects of fire and land-cover metrics on mammal abundance

237 To analyse the effects of fire and land cover on mammal species, we applied abundance modelling using  
238 the "*pcount*" function (N-mixture models) of the *unmarked* R package (Fiske & Chandler, 2011; see  
239 Harmange et al., 2023 for details). The modelling process can be described as follows: the abundance  $\lambda$   
240 of a mammal species (i.e., the number of filtered photographs of the given species taken by the camera  
241 traps aggregated into 6-day occasions) at each camera-trap location and the detection probability  $p_d$   
242 were modelled as a function of the covariates  $x_i$  and  $z_i$  using log and logit link functions, respectively  
243 (see Fiske & Chandler, 2011 for details).

244 First, for each species, the detection probability  $p_d$  was estimated using a stepwise process  
245 starting with model  $\lambda(.)p_d(.)$  (hereafter referred as the "null model"), where the abundance and  
246 detection probabilities are constant. The probability of an individual being captured at the sampling site  
247 can be influenced by several factors, including the visibility in front of the camera trap (i.e., depending  
248 on the openness of the vegetation), or the technical characteristics of the camera-trap model (e.g.,

249 sensitivity or format of the sensor). We therefore computed two covariates  $z_i$ : *visibility* (open vs. dense  
250 vegetation) and *camera-trap model* (Bushnell® and RECONYX®). The effect of detection covariates was  
251 estimated by running three models  $\lambda(\cdot)p_d(z_i)$  for each species, to test whether  $z_i$  covariates including  
252 *visibility*, *camera-trap model*, or both, affected the detection probability  $p_d$ , when the abundance  $\lambda$  was  
253 constant. These four models [i.e.,  $\lambda(\cdot)p_d(\cdot)$ ,  $\lambda(\cdot)p_d(\textit{visibility})$ ,  $\lambda(\cdot)p_d(\textit{camera-trap model})$ , and  
254  $\lambda(\cdot)p_d(\textit{visibility}, \textit{camera-trap model})$ ] were compared using Akaike Information Criterion (AIC) (Burnham  
255 & Anderson, 2002). For each species, we selected as the “null detection model” the model with (i) the  
256 lowest AIC value (i.e., the strongest statistical support), and (ii) a difference in AIC ( $\Delta\text{AIC}$ )  $> 2$  with the  
257 null model.

258 Finally, we tested the effect of fire and land cover on the abundance  $\lambda$  of the four mammal  
259 species, and identified the most appropriate scale at which study species response to fire and land  
260 cover. To do so, we ran models  $\lambda(x_i)p_d(z_i)$ , including as a single covariate  $x_i$  each of the five fire metrics  
261 and the seven land-cover metrics, calculated at each of the three buffer extents (i.e., 500, 750 and 1000  
262 m). For each metric, three models were thus run (i.e., one per buffer extent) and compared with the  
263 null detection model using AIC. A metric was considered significant when the AIC value of the respective  
264 univariate model was lower than the AIC of the null detection model, with an AIC difference ( $\Delta\text{AIC}$ )  $> 2$ .  
265 Competing models with a  $\Delta\text{AIC} < 2$  were considered with equivalent statistical support. For each  
266 significant metric, the buffer extent that yields the lowest AIC value among the three candidate models  
267 indicates the best scale for studying the effect of the given metric on the given mammal species.

268

### 269 2.5.2. Testing for additive and interactive effect of fire and land-cover metrics

270 For each species for which fire and land cover showed significant effects, we tested the additive or  
271 interactive effects of fire and land-cover metrics on relative abundance. This implied integrating several  
272 variables into a single model, which required inspection of correlations between predictors to avoid  
273 multicollinearity (see Appendix B for Pearson correlation matrices between all fire and land-cover  
274 metrics, prepared at each buffer extent).

275 For each species and based on the results of the previous analysis, we selected the species' main  
276 responsive scale (i.e., the scale for which the best statistical support from univariate fire and land-cover  
277 models converged). At this scale, due to the high correlation among most of the predictors, we  
278 separately compared fire metrics and land-cover metrics using AIC, to identify the metric within each  
279 category (i.e., fire and land cover) that best predicts the relative abundance of the mammal species.

280 Finally, five models were compared using AIC: the model incorporating the interaction between  
281 fire and land cover [i.e.,  $\lambda(\text{fire} * \text{land cover})p_d(z_i)$ ], the model with additive effect of fire and land cover  
282 [i.e.,  $\lambda(\text{fire} + \text{land cover})p_d(z_i)$ ], the two univariate models [i.e.,  $\lambda(\text{fire})p_d(z_i)$  and  $\lambda(\text{land cover})p_d(z_i)$ ], and the  
283 null detection model [ $\lambda(.)p_d(z_i)$ ]. Abundance estimates ( $\pm$ SE) were extracted from the best candidate  
284 models and predicted relative abundances ( $\pm$ 95% CI) were represented as a function of the covariate  $x_i$   
285 using the *predict* function for the *unmarked* package objects.

286 All statistical analyses were conducted using R (R Core Team, 2022).

287

### 288 3. Results

#### 289 3.1. Scales of effects

290 Fire metrics demonstrated scale-dependent effects on the relative abundance of *Sylvilagus brasiliensis*  
291 and *Pecari tajacu* (Figure 2A; Appendix C, Table C.1). We detected a significant effect of the proportion  
292 of land burned in the 12 months preceding monitoring on the relative abundance of the small species  
293 *S. brasiliensis*, exclusively at the smallest spatial extent (500 m) (Figure 2A, Table 2). For *P. tajacu*, a  
294 medium-sized forest specialist, the strongest statistical support was obtained at the largest scale (1000  
295 m), regarding the proportion of land burned twice in the 12 months preceding monitoring (Figure 2A,  
296 Table 2). However, this species was also affected by time since fire, but only at the spatial extent of 750  
297 m (Figure 2A; Table 2).

298 We also found scale of effects of land-cover metrics on the relative abundance of *S. brasiliensis*,  
299 *P. tajacu* and *M. gouazoubira* (Figure 2B; Appendix C, Table C.2). Consistently with the scale of the fire  
300 metric effects, we detected a significant effect of the mean patch area of savanna on the relative

301 abundance of *S. brasiliensis*, again exclusively at the smallest spatial extent (500 m) (Figure 2B; Table 3).  
302 Similarly, for *P. tajacu*, the strongest statistical support aligned with the scale of fire metric effects, i.e.,  
303 the largest scale of 1000 m for all significant land-cover metrics, including mean patch area of savanna,  
304 savanna proportion, mean patch area of grassland and grassland proportion (Figure 2B; Appendix C,  
305 Table C.2). At this scale, the mean patch area of savanna provided the best statistical support for  
306 explaining the relative abundance of the species (Table 3). Finally, the relative abundance of *M.*  
307 *gouazoubira*, a medium-sized generalist, was affected by land-cover metrics mainly at the largest scale  
308 of 1000 m (Figure 2B; Appendix C, Table C.2). At this scale, savanna proportion obtained the strongest  
309 statistical support for explaining the relative abundance of the species (Table 3). Finally, we did not  
310 detect any effect of fire (Figure 2A; Appendix C, Table C.1) and land cover (Figure 2B; Appendix C, Table  
311 C.2) on the large generalist species, *Tapirus terrestris*. Overall, there was no evidence of an effect of  
312 forest proportion, mean patch area of forest, edge density, and fire frequency (mean and max) on the  
313 species studied at any of the scales examined (Appendix C, Tables C.1 and C.2).

314

### 315 3.2. Interactive effects of fire and land cover on mammals

316 Model selection indicates that the univariate model that includes the fire metric proportion of land  
317 burned in the 12 months preceding monitoring had similar statistical support for explaining the relative  
318 abundance of *S. brasiliensis*, as the model incorporating the land-cover metric mean patch of savanna  
319 (Table 4). The additive model incorporating both metrics also received equivalent support (Table 4).  
320 According to the model predictions, the relative abundance of *S. brasiliensis* decreased with an  
321 increasing proportion of land burned within the same radius in 12 months preceding monitoring—a  
322 proxy for the fire spatial extent— (Figure 3B), while it increased with the mean patch area of savanna  
323 within a 500-m radius of camera-trap locations (Figure 3A).

324 At the most responsive scale of 1000 m, the relative abundance of *P. tajacu* was influenced by  
325 the additive effects of fire (proportion of land burned twice in the 12 months preceding monitoring) and  
326 land cover (mean patch area of savanna) (Table 4). The model which incorporates the interactive effect

327 between these metrics was not retained for reasons of parsimony, as it introduces complexity without  
328 significantly enhancing explanatory power (Table 4). The relative abundance of *P. tajacu* decreased with  
329 increasing proportion of land burned twice within the same radius in the 12 months preceding  
330 monitoring—a proxy for the frequency of fire-induced disturbance—, and increased with the mean  
331 patch area of savanna within a 1000-m radius of camera-trap locations. However, these results should  
332 be interpreted with caution, as the standard error values of the model estimates were high.

333 No combined effect of fire and land cover was detected on *M. gouazoubira*, the relative  
334 abundance of which was only positively affected by the proportion of savanna within the 1000-m-radius  
335 buffer of camera-trap locations (Figure 3C).

336

#### 337 4. Discussion

338 Our study revealed scale-dependent effects of fire on the relative abundance of mammal species,  
339 consistent with the scale of mammal responses to land cover. Our study further demonstrated that the  
340 response to fire was species-dependent, with no detected fire effect on the largest, habitat and  
341 herbivorous generalist species, *Mazama gouazoubira* and *Tapirus terrestris*.

342

##### 343 4.1. Scales of fire effects on mammal abundance

344 The effect of fire on the abundance of mammal species was found to be scale-dependent. Specifically,  
345 *Sylvilagus brasiliensis* and *Pecari tajacu* responded to fire at scales that align with the scale of responses  
346 to land cover. This result was expected due to two main mechanisms. Firstly, the Cerrado and Pantanal  
347 are fire-prone biomes that have adapted to, and been shaped by fire over millions of years (Pivello et  
348 al., 2021; Simon et al., 2009). For instance, fire regimes in the Cerrado are more intense and more  
349 frequent in savannas than in adjacent, less fire-prone forests (Hoffmann et al., 2009). Consequently, in  
350 these fire-adapted landscapes, landscape processes are linked to fire dynamics. It is therefore not  
351 surprising that the scales of wildlife responses to fire are closely related to those of responses to  
352 vegetation cover.

353           Secondly, similar scales of animal response to fire and land cover were expected, based on the  
354 hypothesis that these responses are determined by the ecological requirements and morphological  
355 traits of mammal species. Our results support this hypothesis, as we found (1) consistent response  
356 scales within species, and (2) variations between species with differing ecological and/or morphological  
357 traits. Indeed, the small species *S. brasiliensis* responded at the smallest scale of 500 m to both fire and  
358 land cover, while larger species such as *P. tajacu* and *Mazama gouazoubira* responded at a larger scale  
359 of 1000 m to fire and/or land cover. This is consistent with our predictions and previous studies  
360 indicating that body size and correlated movement capacity are key factors determining the scale of  
361 effects (Holland et al., 2005; Jackson & Fahrig, 2012). However, it is worth noting that the scales of  
362 effects of 1000 m identified for *P. tajacu* and *M. gouazoubira* in our current study are slightly larger than  
363 those found in the Bodoquena Plateau which ranges from about 15 to 150 km away from to our study  
364 area (i.e., respectively 500 m and 750 m for those species in Amiot et al., 2021). Three main mechanisms  
365 can explain the difference. First, increased fire regimes and associated disturbance in our study area  
366 (pristine environment) may contribute to an increase in the movement and home-range size of mammal  
367 species (Merrick et al., 2021), thereby expanding the scale at which they are influenced by landscape  
368 processes (including fire and land cover). Second, in human-modified landscapes, such as the  
369 surroundings of the Bodoquena Park studied in Amiot et al. (2021) which consist of a mixture native  
370 forest, savanna, pastures and crops (see Harmange et al., 2023), mammals may be less active and mobile  
371 due to increased human disturbance (Tucker et al., 2018, 2023), potentially leading to a reduction in  
372 the scale of effects. Additionally, in Amiot et al. (2021), the mountainous topography of the Bodoquena  
373 Plateau may have contributed to increased costs of terrestrial movement (Shepard et al., 2013),  
374 potentially resulting in constrained mammal movements.

375           Finally, our study found no effect of fire and land cover on the largest species, *Tapirus terrestris*,  
376 at any of the scales studied. This result is consistent with that of Amiot et al. (2021), who also found no  
377 effect of landscape features on this species in the Brazilian Cerrado, at scales ranging from 500 m to 10  
378 km. This result also confirms our predictions that larger species, including *M. gouazoubira* and *T.*

379 *terrestris*, would not be greatly affected by fire (Chia et al., 2016), due to their capacity to move over  
380 longer distances than small-sized terrestrial species (Cardillo et al., 2005). They might therefore escape  
381 fire, reach unburned refugia and recolonise burned areas after fire more easily than smaller species  
382 (Chia et al., 2016; Lawes et al., 2015). This is particularly relevant in our study area, where prescribed  
383 burning conducted by indigenous brigades has increased landscape pyrodiversity (in terms of fire  
384 frequency and time since last fire) and reduced the average and maximum extent of fires (Oliveira et  
385 al., 2022), likely decreasing the distance required to reach the nearest unburned refugia. Previous  
386 studies also found that *Mazama* spp. were not much affected by fire, supposedly because of their  
387 generalist herbivorous feeding habits, which make them less sensitive to fire disturbance (Souza et al.,  
388 2023). Camargo et al. (2018) also showed that fire facilitated the invasion of generalist small-mammal  
389 species from open habitats of the Cerrado, such as grassland and savanna.

390 It should be noted, however, that mammals may also respond to fire at scales smaller and/or  
391 larger than those studied, although our data did not allow us to test this hypothesis. Indeed, we focused  
392 on the landscape scale (i.e., using buffers with radius ranging from 500 to 1000 m, which equates to  
393 78.5 to 314.2 ha), while mammal species have also been documented to respond to fire at a more local  
394 scale (e.g., from 0.8 to 78.5 ha in Souza et al., 2023), and potentially at larger scales, as suggested by  
395 the reduced occupancy of *Mazama americana* observed after megafires (Bardales et al., 2024). Overall,  
396 our results demonstrate that the scale at which mammals best responded to fire is species-specific. This  
397 emphasizes the importance of systematically considering multiple scales when studying the effects of  
398 fire dynamics on biodiversity, as is common in analyses of landscape dynamics.

399

#### 400 **4.2. Effect of fire regime on mammal abundance**

401 Consistent with Souza et al. (2023), our results revealed species-specific responses of mammals to fire  
402 regimes. We found that the spatial extent of fire provided equivalent statistical support to that of mean  
403 patch size of savanna in explaining *Sylvilagus brasiliensis* abundance. This suggests that fire plays a key  
404 role in shaping mammal abundances and distribution, as important as land cover. Indeed, the relative

405 abundance of the small species *S. brasiliensis* decreased with increasing proportion of land that burned  
406 in 12 months preceding the camera-trap monitoring, within a 500-m radius. Specifically, 30% of the land  
407 burned in the 12 months preceding monitoring (i.e., 23.5 ha burned per 78.5 ha of land) reduced the  
408 population abundance by 50%. This result aligns with our expectations that smaller species are  
409 disproportionately affected by fire, particularly when fire extent exceed their home range (Lawes et al.,  
410 2015). *S. brasiliensis*, being a small species (i.e., 0.95–1.2 kg; Reis et al., 2006; Ruedas et al., 2017), is  
411 likely to have limited dispersal abilities. While data on its home range size are lacking (Pasqualotto et  
412 al., 2021), *S. brasiliensis* is likely to have a small home range, as observed in most *Sylvilagus* spp. species,  
413 characterized by home ranges  $\leq 3$  ha (Jones et al., 2009), equivalent to a buffer with a radius of 100 m.  
414 This result is corroborated by Lawes et al. (2015) who found that increased proportion of surrounding  
415 land burned and, to a lesser extent, mean distance to unburned vegetation—two proxies for fire extent—  
416 decreased the species richness and abundance of small mammals.

417         Furthermore, the increase in mean size of savanna patches (or savanna proportion in the  
418 landscape for *Mazama gouazoubira*) has contributed to an increase in the abundance of *S. brasiliensis*  
419 and *Pecari tajacu*. Considering that savannas in the Brazilian Cerrado are subject to frequent and intense  
420 fire regimes (Hoffmann et al., 2009), a combined effect of fire and land cover could potentially  
421 exacerbate the impact of fire and play a significant role in regulating the population dynamics of  
422 mammal species. Indeed, Griffith & Brook (2014) found that the reduction in small-mammal  
423 abundances and demographic parameters in burned compared to unburned areas was particularly  
424 pronounced for small-bodied species (101–1000 g) and those with fire-sensitive habitat preferences.  
425 Further studies are needed to quantify the combined effects of fire and land cover on mammal  
426 population growth rates and viability analysis in fire-prone landscapes. This is particular relevant for  
427 decreasing or low-density populations of long-lived, low reproductive rate species (de Silva &  
428 Leimgruber, 2019), as it can allow identify the return interval of perturbation they can cope with, and  
429 the time and extent to which populations recover after perturbation (Gerber & Hilborn, 2001).

430           Although the strength of the effect seems debatable, the frequency of fires within the 12  
431 months preceding monitoring may also affect mammal abundance. Indeed, although we detected, as  
432 expected, a negative effect of the proportion of the land burned twice in the 12 months preceding  
433 monitoring on the abundance of the frugivorous forest specialist *P. tajacu*, the high variability of  
434 coefficient estimates necessitates this result to be considered carefully. Previous studies indicate that  
435 short return intervals of fires may have negative impact on mammals (Pocknee et al., 2023). This result,  
436 if confirmed, would be consistent with Deus et al. (2023) who found a reduction in the occurrence of *P.*  
437 *tajacu* after fires in Pantanal, and with Barlow & Peres (2006) showing that a frequency of 1–2 fire events  
438 in 10 years contributed to the extirpation of most frugivorous and forest-specialist vertebrates.

439           In summary, our results suggest that fire, in addition to land cover, may exert selection pressure  
440 on mammals by selecting species based on specific ecological traits (see also Culhane et al., 2022). In  
441 particular, the effect of fire on species involved in key ecological functions and services, such as seed  
442 dispersal and nutrient transfer (e.g., the forest specialist *P. tajacu*), may raise concerns about the  
443 ecosystem integrity. However, other species that share similar ecological traits and in which no effect  
444 of fire has been detected (e.g., *T. terrestris*) also contribute to ecosystem functioning. Further studies  
445 at the community level are therefore needed to assess whether and how fire might affect the  
446 assemblage of species, ecological traits and functions available in landscapes. Adopting integrated fire  
447 and land-cover management strategies implemented at the landscape scale (Oliveira et al., 2022) would  
448 be necessary to reduce megafires, maintain landscape heterogeneity and effectively conserve the  
449 diversity of species and ecological functions in fire-prone landscapes.

450

#### 451           **4.3. Conservation implications and perspectives for fire management**

452 Faced with the rapidly accelerating changes in fire regimes (Iglesias et al., 2022; Nolan et al., 2022),  
453 identifying strategies that promote the coexistence of both human and biodiversity with fire has become  
454 an urgent challenge (Doerr & Santín, 2016; Garcia et al., 2021). Worldwide, fire management strategies  
455 and policies have increasingly moved towards prescribed burning (Costa & Thomaz, 2021; Hunter &

456 Robles, 2020), incorporating indigenous and local knowledge for increased efficiency (Bowman et al.,  
457 2020). Although our study did not specifically test the effects of prescribed burning on the abundance  
458 of mammals, previous research in the region has shown that prescribed burning reduced fire frequency  
459 by 80% and the size of the area burned by 53% (Oliveira et al., 2022), reducing the loss of species  
460 diversity to imperceptible levels in Cerrado and Pantanal (Durigan, 2020). However, some studies  
461 suggest that the outcomes of prescribed burning for biodiversity conservation could be more complex  
462 due to taxon-dependent and difficult-to-predict responses of species (Pastro et al., 2011). Further  
463 research would therefore be needed to confirm whether our results apply to mammal responses to  
464 prescribed burns, which would help refine fire management strategies.

465         Our study reveals that in mammals, the type of response to fire, and the scale at which it  
466 operates, are species-specific and possibly driven by trait-related processes (see also Pocknee et al.,  
467 2023; Souza et al., 2023). Fire management aimed at designing low fire risk landscapes for biodiversity  
468 conservation should carefully consider the scale used to determine the proportion of land targeted by  
469 prescribed burning, particularly within scales ranging from 500 to  $\geq 1000$  (78.5–314.1 ha) for mammals  
470  $\geq 1$  kg. We believe that finding a trade-off between the extent of land burned and return interval of fire,  
471 allowing for balanced proportions of recently burned to long-unburned areas (Souza et al., 2023), would  
472 create conditions favourable to the coexistence of biodiversity with fire. Further studies integrating  
473 analyses of multiple fire metrics to test different aspects of fire regimes, at multiple scales, and using  
474 trait-based approaches for all taxa, are critically needed to disentangle the mechanisms governing the  
475 effects of fire regimes on biodiversity. This research should inform sustainable fire management  
476 practices, compatible with the conservation of species, ecological traits and functions, particularly in  
477 protected areas and indigenous territories which constitutes biodiversity-rich, pristine environments.  
478 Faced with the unprecedented and rapid intensification of fire regimes, implementing such integrated  
479 management has become an urgent challenge to promote more sustainable coexistence between  
480 wildfire, indigenous communities and biodiversity, on which they depend for food security (Doerr &  
481 Santín, 2016; Oliveira et al., 2022).

482

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486

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504

505 **Appendix A.** Supplementary information

506 **Appendix B.** Correlation matrix of fire and land-cover metrics calculated at varying spatial extents

507 **Appendix C.** Supporting information and statistical details for the scale of effect analysis and the  
508 selection of fire and land-cover metrics in explaining mammal abundances

509

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760 **Table 1** Comprehensive description of the calculation of fire metrics, along with the primary data  
 761 source (i.e., MapBiomas products).

762

Metric type	Metric name	Calculation	Source
Fire frequency	Fire frequency (mean)	Mean of the fire frequency values of all pixels contained within a given buffer	Fire frequency 1985-2021
	Fire frequency (max)	Maximum value of fire frequency recorded across all pixels contained within a given buffer	Fire frequency 1985-2021
	Proportion burned twice	Proportion of pixels contained within the buffer (i.e., proportion of the buffer area) that burned twice during the 12 months preceding the monitoring: once within the period July/August 2020–December 2020, and once within the period January 2021–July/August 2021 (depending on the monitoring session)	Months burned coverage 2020, 2021
Fire spatial extent	Proportion burned	Proportion of pixels contained within the buffer (i.e., proportion of the buffer area) that burned during the 12 months preceding the monitoring, i.e., between July/August 2020 and July/August 2021	Months burned coverage 2020, 2021
Regeneration	Time since fire	First, the delay (in months) between the monitoring period and the month since the largest fire recorded within the buffer in 2021, was calculated. If no fire was recorded in 2021, the same process was applied considering the data of 2020. Secondly the distribution of months since fire identified two distinct groups: buffers for which fire was recorded within the 4 months preceding monitoring (“0-4 months”), and buffers for which the last recorded fire occurred at least 11 months before monitoring (“≥ 11 months”)	Months burned coverage 2020, 2021

763

764 **Table 2** Statistical support for univariate models testing the effects of fire metrics on the relative  
765 abundance of the four mammal species. The scale used for metrics calculation is indicated. The  
766 selection process was based on the Akaike’s information criterion (AIC). The number of parameters (K)  
767 is presented together with model weights ( $\omega_i$ ), log likelihood values (LL) and differences in AIC ( $\Delta$ AIC)  
768 in respect of the model with the highest statistical support (i.e., the lowest AIC value). Grey envelops  
769 within the table indicate the candidate models with a  $\Delta$ AIC < 2 compared with the null detection  
770 model (NULL). Only the best candidate models (i.e., those with a  $\Delta$ AIC < 2 with the candidate model  
771 with the lowest AIC value) and the null detection model are indicated (see Appendix C, Table C.3 in for  
772 statistical details on all candidate models). Models selected are represented in bold. Estimates  $\pm$  SE are  
773 indicated for significant metrics. Estimate of the categorical variable *Time since fire* are provided using  
774 as reference the level “ $\leq$ 4 months”.

775

Species	Scale	Candidate models	K	AIC	$\Delta$ AIC	$\omega_i$	LL	Evidence ratio	Estimate $\pm$ SE
<i>S. brasiliensis</i>	500 m	<b>Proportion burned</b>	<b>4</b>	<b>111.53</b>	<b>0.00</b>	<b>0.47</b>	<b>-51.76</b>		<b>-0.021 <math>\pm</math> 0.010</b>
		Time since fire	4	113.21	1.68	0.20	-52.60	2.31	
		NULL	3	114.54	3.01	0.10	-54.27	4.50	
<i>P. tajacu</i>	1000 m	<b>Proportion burned twice</b>	<b>3</b>	<b>106.95</b>	<b>0.00</b>	<b>0.66</b>	<b>-50.48</b>		<b>-1.577 <math>\pm</math> 1.477</b>
		NULL	2	111.21	4.26	0.08	-53.60	8.41	
	750 m	<b>Proportion burned twice</b>	<b>3</b>	<b>107.43</b>	<b>0.00</b>	<b>0.44</b>	<b>-50.71</b>		<b>-1.613 <math>\pm</math> 1.726</b>
		<b>Time since fire</b>	<b>3</b>	<b>108.03</b>	<b>0.61</b>	<b>0.33</b>	<b>-51.02</b>	<b>1.35</b>	<b>1.381 <math>\pm</math> 0.594</b>
		NULL	2	111.21	3.78	0.07	-53.60	6.62	

776

777 **Table 3** Statistical support for univariate models testing the effects of land-cover metrics on the  
778 relative abundance of the four mammal species. The scale used for metrics calculation is mentioned.  
779 The selection process was based on the Akaike's information criterion (AIC). The number of  
780 parameters (K) is presented together with model weights ( $\omega_i$ ), log likelihood values (LL) and  
781 differences in AIC ( $\Delta AIC$ ) in respect of the model with the highest statistical support (i.e., the lowest  
782 AIC value). Grey envelopes within the table indicate the candidate models with a  $\Delta AIC < 2$  compared  
783 with the null detection model (NULL). Only the best candidate models (i.e., those with a  $\Delta AIC < 2$  with  
784 the candidate model with the lowest AIC value) and the null detection model are indicated (see  
785 Appendix C, Table C.4 in for statistical details on all candidate models). Models selected are  
786 represented in bold. Estimates  $\pm$  SE are indicated for significant metrics.

787

Species	Scale	Candidate models	K	AIC	$\Delta AIC$	$\omega_i$	LL	Evidence ratio	Estimate $\pm$ SE
<i>S. brasiliensis</i>	500 m	<b>Savanna patch area</b>	<b>4</b>	<b>112.21</b>	<b>0.00</b>	<b>0.32</b>	<b>-52.11</b>		<b>0.021 <math>\pm</math> 0.010</b>
		Grassland patch area	4	112.79	0.58	0.24	-52.40	1.34	
		NULL	3	114.54	2.33	0.10	-54.27	3.20	
<i>P. tajacu</i>	1000 m	<b>Savanna patch area</b>	<b>3</b>	<b>101.69</b>	<b>0.00</b>	<b>0.75</b>	<b>-47.85</b>		<b>0.009 <math>\pm</math> 0.002</b>
		NULL	2	111.21	9.52	0.01	-53.60	116.48	
<i>M. gouazoubira</i>	1000 m	<b>Savanna proportion</b>							<b>0.032 <math>\pm</math> 0.011</b>
		NULL	5	234.45	0.00	0.60	-112.23		
		NULL	4	241.21	6.76	0.02	-116.60	29.33	

788

789 **Table 4** Statistical support for models testing univariate, additive and interactive effect of land-cover  
790 and fire metrics on the relative abundance of *Sylvilagus brasiliensis* and *Pecari tajacu*. The scale used  
791 for metrics calculation is indicated. The selection process was based on the Akaike's information  
792 criterion (AIC). The number of parameters (K) is presented together with model weights ( $\omega_i$ ), log  
793 likelihood values (LL) and differences in AIC ( $\Delta$ AIC) in respect of the model with the highest statistical  
794 support (i.e., the lowest AIC value). Grey envelopes within the table indicate the candidate models with  
795 a  $\Delta$ AIC < 2 compared with the null detection model (NULL). Models selected are represented in bold.  
796 Estimates  $\pm$  SE are indicated for significant effects. Models with  $\Delta$ AIC < 2 are considered statistically  
797 equivalent.

798

Species	Scale	Candidate models	K	AIC	$\Delta$ AIC	$\omega_i$	LL	Evidence ratio	Estimate $\pm$ SE
<i>S. brasiliensis</i>	500 m	<b>Proportion burned</b>	<b>4</b>	<b>111.53</b>	<b>0.00</b>	<b>0.38</b>	<b>-51.76</b>		<b>-0.021 <math>\pm</math> 0.010</b>
		<b>Savanna patch area</b>	<b>4</b>	<b>112.21</b>	<b>0.68</b>	<b>0.27</b>	<b>-52.11</b>	<b>1.41</b>	<b>0.021 <math>\pm</math> 0.010</b>
		<b>Proportion burned + Savanna patch area</b>	<b>5</b>	<b>112.22</b>	<b>0.69</b>	<b>0.27</b>	<b>-51.11</b>	<b>1.41</b>	<b>-0.014 <math>\pm</math> 0.012</b> <b>0.014 <math>\pm</math> 0.011</b>
		NULL	3	114.54	3.01	0.08	-54.27	4.50	
		Proportion burned * Savanna patch area	6	126.49	14.96	0.00	-57.24	1771.38	
<i>P. tajacu</i>	1000 m	Proportion burned twice * Savanna patch area	5	96.45	0.00	0.53	-43.23		
		<b>Proportion burned twice + Savanna patch area</b>	<b>4</b>	<b>96.88</b>	<b>0.42</b>	<b>0.43</b>	<b>-44.44</b>	<b>1.23</b>	<b>-2.54 <math>\pm</math> 2.15</b> <b>0.009 <math>\pm</math> 0.002</b>
		Savanna patch area	3	101.69	5.24	0.04	-47.85	13.74	
		Proportion burned twice	3	106.95	10.50	0.00	-50.48	190.57	
		NULL	2	111.21	14.76	0.00	-53.60	1603.59	

799

800 **Figure caption**

801

802 **Figure 1** Characterization of the Kadiwéu Indigenous Territory (Mato Grosso do Sul, Brazil; left-side  
803 panels) and the study area (right-side panels) in terms of: (A, B) monthly burned coverage in 2021; (C,  
804 D) fire frequency expressed as the number of years with fire from 1985 to 2021 (MapBiomias, 2<sup>nd</sup> fire  
805 collection); and (E, F) land cover in 2021 (MapBiomias, 7<sup>th</sup> land-cover collection). Blue dots indicate the  
806 locations of the camera traps (n = 36).

807

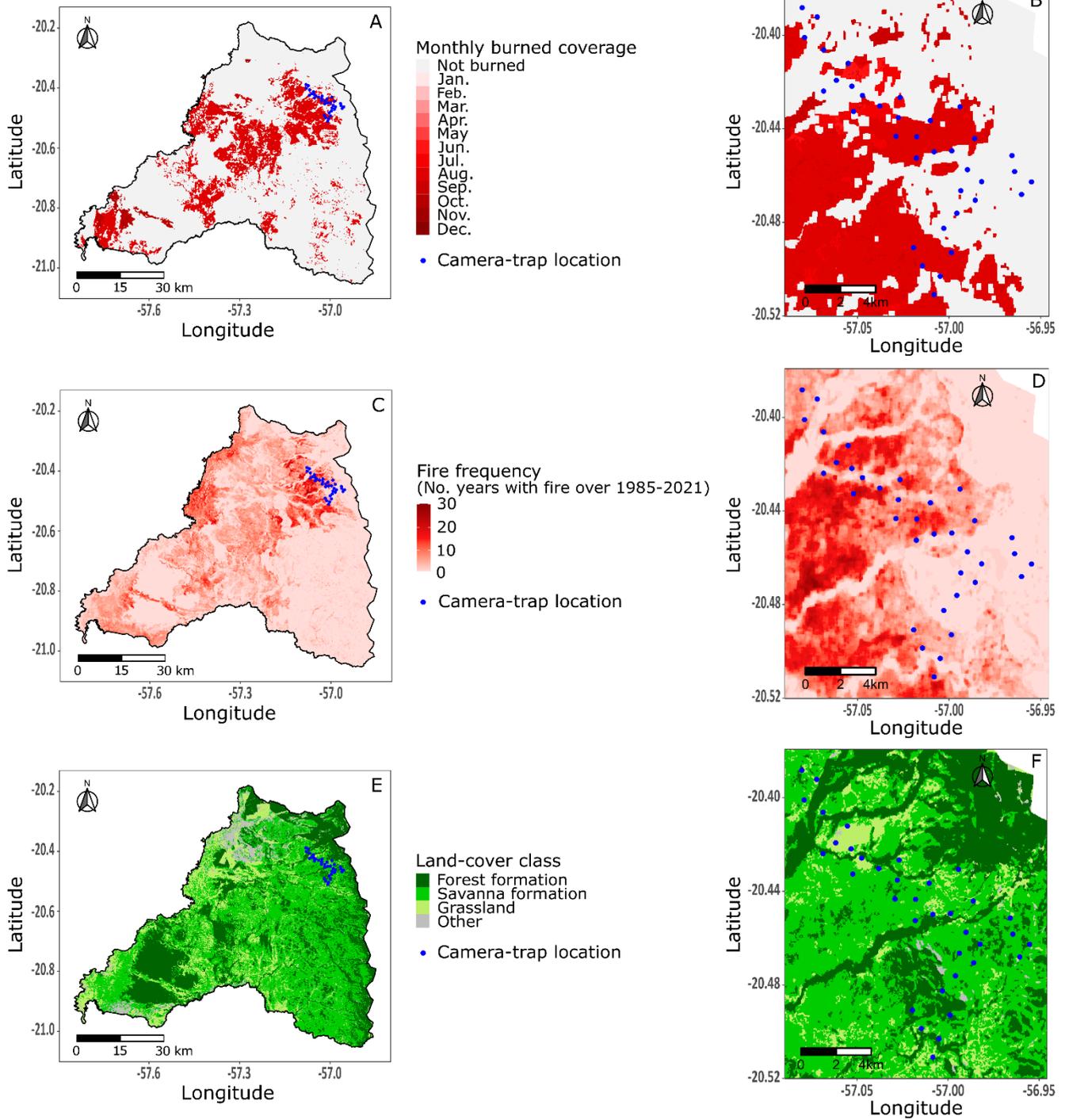
808 **Figure 2** Statistical support of univariate models testing the effects of fire metrics (A) and land-cover  
809 metrics (B) calculated at three buffer extent (i.e., within 500-m, 750-m, and 1000-m-radius buffers) on  
810 the relative abundance of the four mammal species. The dashed line indicates the AIC value of the null  
811 detection model. The grey envelope materializes the difference in AIC ( $\Delta AIC \leq 2$ ) relative to the null  
812 detection model. White dots refer to models with  $\Delta AIC \leq 2$  relative to the null detection model,  
813 therefore considered as equivalent in terms of statistical support. Black dots indicate significant  
814 models, i.e., models whose AIC value was lower than the AIC value of the null detection model, with a  
815  $\Delta AIC > 2$ . See Appendix C, Figures C.1 and C.2, and Tables C.1 and C.2 for statistical details on all the  
816 metrics analysed. Mammal silhouettes were extracted using the R package *rphylopic* (Gearty & Jones,  
817 2023).

818

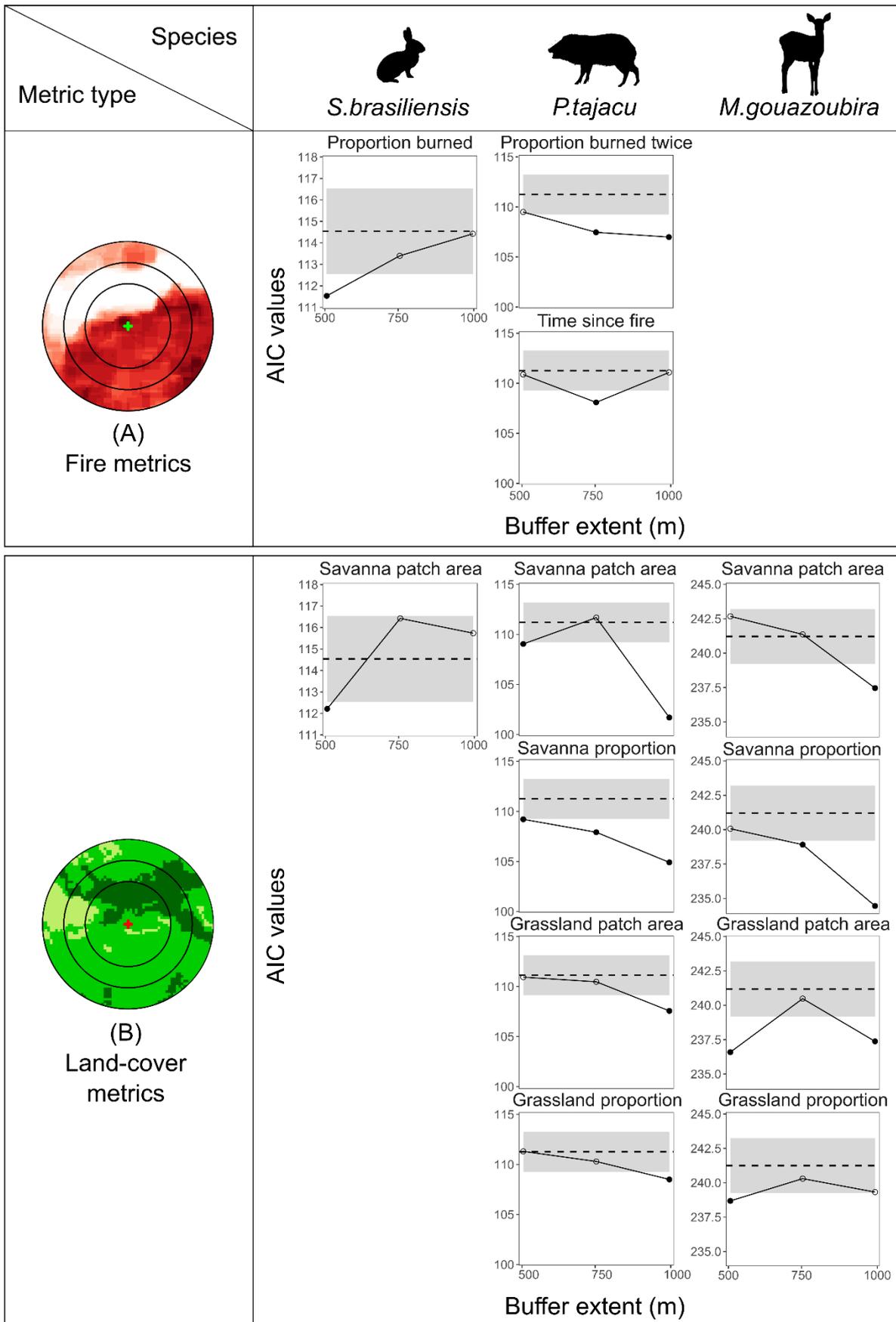
819 **Figure 3** Predicted relationships (solid lines) with 95% confidence interval (grey envelope) between fire  
820 or land-cover metrics and the relative abundances of *Sylvilagus brasiliensis* (A, B) and *Mazama*  
821 *gouazoubira* (C). Metrics were calculated within buffers of 500-m radius (*S. brasiliensis*) or 1000-m  
822 radius (*M. gouazoubira*) from camera-trap locations. Values of fire and land-cover metrics of each  
823 buffer are shown as vertical lines on the x-axes. See Table 4 for statistical details.

## Kadiwéu Indigenous Territory

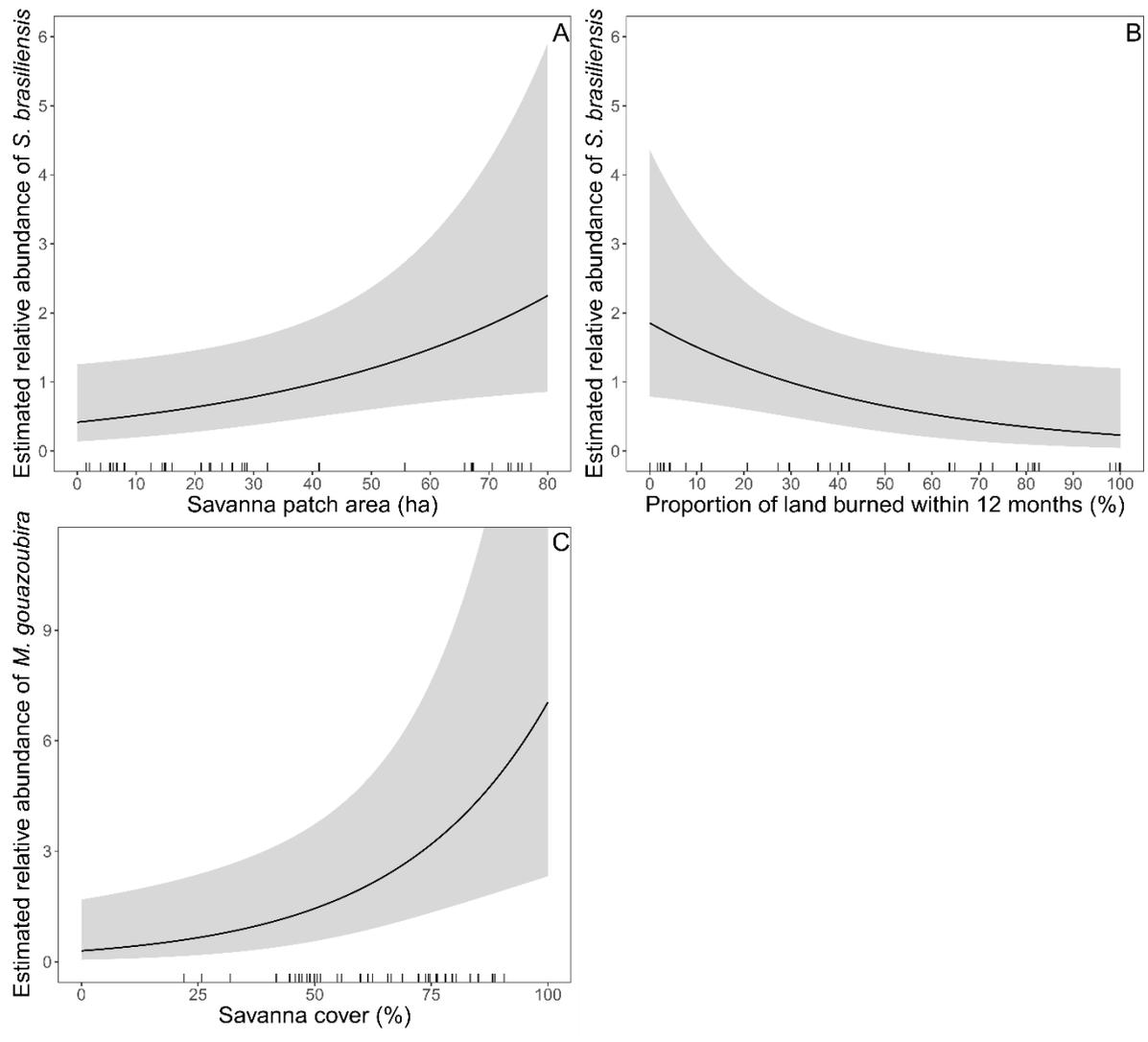
## Study area



824  
825 Figure 1



826  
827 Figure 2



828  
829 **Figure 3**