



HAL
open science

Regeneration after fire in campo rupestre: Short- and long-term vegetation dynamics

Soizig Le Stradic, Pauline Hernandez, Geraldo Wilson Fernandes, Elise Buisson

► **To cite this version:**

Soizig Le Stradic, Pauline Hernandez, Geraldo Wilson Fernandes, Elise Buisson. Regeneration after fire in campo rupestre: Short- and long-term vegetation dynamics. *Flora*, 2018, 238, pp.191-200. 10.1016/j.flora.2016.12.001 . hal-01795514

HAL Id: hal-01795514

<https://hal.science/hal-01795514v1>

Submitted on 18 May 2018

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

1 **Title**

2 Regeneration after fire in *campo rupestre*: short- and long-term vegetation dynamics.

3

4 **Authors**

5 Soizig Le Stradic^{1,2,3}, Pauline Hernandez¹, G. Wilson Fernandes^{3,4}, Elise Buisson¹

6

7 ¹ IMBE - Institut Méditerranéen de Biodiversité et d'Ecologie marine et continentale,
8 Université d'Avignon et des Pays de Vaucluse, CNRS, IRD, Aix Marseille Univ, Agroparc BP
9 61207, 84911, Avignon, France.

10 ² Laboratorio de Fenologia, Departamento de Botânica / Instituto de Biociências, UNESP –
11 Universidade Estadual Paulista, Rio Claro SP, CP 199, Brazil.

12 ³ Ecologia Evolutiva e Biodiversidade, Universidade Federal de Minas Gerais, Belo Horizonte,
13 MG, 30161-901, Brazil.

14 ⁴ Department of Biology, Stanford University, Stanford, CA 94305 USA.

15

16 **Abstract**

17 The Cerrado (Brazilian savanna) is the second largest biome in Brazil, recovering 22% of the
18 country, and *campo rupestre* is one of the most biodiverse ecosystem. *Campo rupestre* are
19 extremely old mountaintop tropical ecosystems, composed of a mosaic of herbaceous,
20 shrubland and savanna vegetation, generally located above 900 m above sea level characterized
21 by shallow, acidic and nutrient-poor soils. In the context of increased land-use changes,
22 effective conservation and management projects appear necessary to guarantee the
23 conservation of these ecosystems. Although fire is a natural disturbance in *campo rupestre*, the
24 effects of fire on vegetation dynamics remain poorly understood. Our objective was to assess
25 the effects of fire on vegetation recovery and plant composition at both the short- and long-
26 term on the main herbaceous vegetation types: the sandy and stony grasslands. We monitored
27 plant community composition before and after a wildfire in order to assess the short-term
28 vegetation recovery. Diachronic analyses of grasslands burnt at various dates were used to
29 understand the effects of fire on longer-term vegetation dynamics. Our results highlighted a
30 rapid recovery of *campo rupestre* vegetation after wildfires, suggesting a high adaptation to
31 fire of plant communities. We did not find a significant variation in species richness of sandy
32 grasslands according to time after fire, whereas higher species richness was observed in the
33 recently burnt stony grasslands. No changes in plant composition of *campo rupestre* in
34 response to fire was found, probably due to the high heterogeneity of this ecosystem. After fire,
35 biomass gradually increased over time in both vegetation types. High biomass accumulation
36 could lead to stronger fires. Further studies are necessary to understand the relationship
37 between biomass accumulation and fire intensity in *campo rupestre* in order to set up adapted
38 fire management strategies to conserve *campo rupestre* biodiversity.

39

40 **Keywords:** Biomass; *campo arenoso*; *campo pedregoso*; rupestrian grasslands; vegetation
41 recovery; wildfire

42

43 1. Introduction

44 The Brazilian Cerrado is the world's most species-rich savanna, which originally covered over
45 2 million km² (Oliveira and Marquis, 2002). It is composed of a mosaic of ecosystems
46 influenced by the interaction between climate, soil and fire (Oliveira and Marquis, 2002),
47 ranging from the most open ecosystems, such as *campo limpo* grasslands, to ecosystems that
48 show higher tree density, i.e. *cerradão* woodlands (Coutinho, 1978). The Cerrado has already
49 lost ca. 50% of its area to soybean and maize plantation and to cattle raising (Carvalho et al.,
50 2009; Fernandes, 2016), but only 2.85% of its total surface is located in fully protected
51 conservation units (Bustamante, 2015). The protection of the remaining land therefore appears
52 as a high conservation priority (Bitencourt et al., 2016; Mittermeier et al., 2011).

53 In savanna ecosystems, fire is a common endogenous disturbance (*sensu* McIntyre and
54 Hobbs, 1999), i.e. disturbances to which the system has been exposed repeatedly through
55 evolutionary time and that can maintain the biodiversity under an appropriate regime (Bond et
56 al., 2003; Hoffmann et al., 2012; Mistry, 2000; Ratnam et al., 2011; Sankaran et al., 2005). In
57 such fire-prone environments, plants species may present adaptive strategies that can improve
58 their resilience capacity (i.e. implying a notion of both resistance and recovery *sensu* Hodgson
59 et al., 2015) in the face of recurrent fires. These adaptive strategies include belowground
60 meristematic and storage organs, thick corky bark, sclerophyllous leaves, fire resistant seeds,
61 ability to resprout, and specialized flowering and fruiting phenologies (Bond and Keeley, 2005;
62 Miranda et al., 2002; Oliveira et al., 2016).

63 Since the Cerrado flora, like many other savannas, has been assembled along with
64 recurrent fires (Oliveira and Marquis, 2002; Simon et al., 2009), it seems essential to include
65 fire in conservation management plans (Figueira et al., 2016; Overbeck et al., 2005; Ramos-
66 Neto and Pivello, 2014; Ribeiro and Figueira, 2011). This requires the identification of
67 adequate fire regimes (e.g. season of fire, frequency) as variation in fire regimes can induce
68 changes in plant community composition or structure and/or in ecosystem functioning
69 (Gorgone-Barbosa et al., 2015; Lehmann et al., 2008). By removing biomass, fire opens gaps
70 necessary for germination and reduces competition for light and the dominance of few species,
71 allowing the establishment and persistence of numerous other species (Fidelis et al., 2010;
72 Grubb, 1977; Knox and Clarke, 2006; Overbeck et al., 2005). Fire ecology (i.e. fire effects and
73 fire regimes) of some Cerrado ecosystems has been widely studied (Hoffmann, 1996;
74 Hoffmann et al., 2012; Miranda et al., 2002; Mistry et al., 2005; Oliveras et al., 2012; Ramos-
75 Neto and Pivello, 2014). However, such precise information is not available for *campo rupestre*
76 grasslands. Understanding how fire impacts vegetation composition and structure can help to
77 improve environment policies related to fire and thus future actions within reserves and
78 national parks (Figueira et al., 2016; Ribeiro and Figueira, 2011).

79 *Campo rupestre* are part of the Cerrado biome and can be defined as a mosaic of
80 herbaceous and savanna vegetations with shrubland on rocky outcrops (Fig. S1). This
81 ecosystem establishes at altitudes above 900 m a.s.l., mainly along the Espinhaço range in
82 eastern Brazil, (Silveira et al., 2016) and originally covered 66,447 km² (Bitencourt et al., 2016;
83 Silveira et al., 2016). *Campo rupestre* are species-rich ecosystems, established on shallow and
84 oligotrophic, quartzite-derived soils with successive periods of water logging during the rainy
85 season and water deficit during the dry season (Le Stradic et al., 2015a; Oliveira et al., 2015).
86 Previous studies have already evaluated fire effects at the population-level for some species
87 (Alves and Kolbek, 2010; Conceição et al., 2013; Conceição and Orr, 2012; Ferreira and

88 Conceição, 2012; Miola et al., 2010; Neves et al., 2011; Simon and Amaral, 2003). Overall,
89 the effects of fire on *campo rupestre* grassland plant communities are still poorly understood
90 (Figueira et al., 2016; Ribeiro and Figueira, 2011). To our knowledge only two studies
91 described the effects of fire on *campo rupestre* plant communities. One of them, was based on
92 a floristic survey complemented by plant collection and field observation (Kolbek and Alves,
93 2008) whereas the other one discussed the strategies of regeneration and investigated the
94 composition and structure vegetation in a recently burnt area but did not present pre-fire data
95 (Neves and Conceição, 2010). These studies concluded that vegetation cover and plant family
96 composition in burnt *campo rupestre* recover rapidly after fire, mainly because many plant
97 species have various adaptations to resprout from underground storage organs after fire.
98 However, it appears that fire, in combination with grazing, can lead to invasion by an African
99 grass (*Melinis minutiflora*) (Gorgone-Barbosa et al., 2016, 2015).

100 In this study, we aim to provide a precise assessment of the effects of fire on vegetation
101 recovery and composition at both the short- and long-terms on the main herbaceous vegetation
102 types: the sandy and stony grasslands. On the short-term, plant community composition was
103 monitored before and two and five months after a wildfire to assess vegetation recovery. We
104 used diachronic analyses of grasslands burnt at various dates to study the effects of fire on
105 vegetation dynamics on the long-term (up to 25 years after the last fire).

106 **2. Material and methods**

107 **2.1. Study sites**

108 The study was conducted in a *campo rupestre* located along and within the National Park of
109 Serra do Cipó, about 100 km northeast of Belo Horizonte, southeastern Brazil (19° 15'–25'S
110 and 43° 30'–40'W). The climate of the area is tropical altitudinal climate with an average
111 annual rainfall of ca. 1622 mm mainly concentrated during the rainy season extending from
112 November to April; the dry season extends from May to October (Madeira and Fernandes,
113 1999). The annual average temperature is 21.2°C (Madeira and Fernandes, 1999). All study
114 sites are at altitudes between 1100 and 1420 m and 12 km separate the most distant sites. We
115 studied the two dominant herbaceous vegetations of *campo rupestre*: the sandy grasslands (Sa)
116 and the stony grasslands (St) (Fig. S1). These grasslands are subjected to frequent fires, leading
117 to a mosaic of grassland patches burnt at different dates. According to Figueira et al. (2016)
118 the peak frequency of fire return time in the Serra do Cipó National Park is seven years,
119 indicating that the area is moderately susceptible to fire.

120 **2.2. Plant community composition and aboveground biomass sampling**

121 In order to obtain accurate data on short-term fire effects on plant communities, we monitored
122 permanent plots in one stony and one sandy grassland site (Table 1), before and after a wildfire
123 which occurred in August 2011 (sites are highlighted by * in Table 1). In each grassland site,
124 vegetation surveys were carried out in ten 1-m² quadrats prior to fire, in February 2009, and
125 twice after the fire, in October 2011 (i.e. two months after fire) and January 2012 (i.e. five
126 months after fire).

127 To study long-term vegetation dynamics after fire, we sampled 12 stony and 12 sandy grassland
128 sites between January and March 2012. Three replicates sites of four different times since last
129 fire constituted the 12 sites of each vegetation type: (a) burnt less than a year ago, in 2011, (b)
130 burnt 5 years ago, in 2007, (c) burnt 11 years ago, in 2001 and (d) unburnt in the past 25 years,
131 last burn dated at least in 1987 (Table 1). The sites were selected using a map of fire occurrence
132 made from satellite images collected over the past 25 years and made available by the National

133 Park of Serra do Cipó. On each site, ten 1-m² randomly placed quadrats were used to assess the
134 plant community.

135 For both short-term and long term study, within each quadrat all angiosperm species were
136 identified and their percent cover visually assessed based on the vertical projection of all aerial
137 plant parts (Mueller-Dombois and Ellenberg, 1974).

138 Above-ground plant biomass was assessed in three 0.4 × 0.4 m randomly placed quadrats in
139 each site. All biomass above 5mm of the ground was clipped and bagged separately for each
140 quadrat. Living and dry biomass were separated in the laboratory. Biomass was weighed after
141 oven-drying at 60 °C for 48 h.

142 **2.3. Statistical analyses**

143 Statistical analyses were conducted separately for the sandy grasslands (Sa) and the stony
144 grasslands (St) considering the large difference in species composition between the two types
145 of *campo rupestre* (Le Stradic et al., 2015a). All statistical analyses were performed in software
146 R Version 3.2.1 (R Core Team, 2015), using stats, vegan (Oksanen et al., 2016) and lme4 (Bates
147 et al., 2015) packages.

148 *2.3.1 Short-term effect of a fire passage*

149 We ran generalized linear mixed models (glmer in the R package lme4) with a Poisson error
150 distribution, a log-link function and random effect for quadrats (Crawley, 2007), to assess
151 variation of species richness before and after fire (prior to fire in February 2009, two months
152 after fire in October 2011, and five months after fire in January 2012). For each grassland type,
153 analyses of variance (ANOVA) for repeated measures and pairwise comparisons with p values
154 adjustments according to Holm correction were used on the data of percentage cover over time,
155 including quadrats as random effect. Percentage cover data were arcsine transformed to meet
156 the assumptions of statistical analyses.

157 Dissimilarities between vegetation composition before and after fire were assessed using
158 ANOVA-style analysis Adonis. Adonis (i.e. Multivariate ANOVA based on dissimilarities) is
159 an analysis which implements a multivariate analysis of variances using distances matrices, to
160 study the differences in group means and performs permutation tests to inspect the
161 significances of the differences observed (Oksanen et al., 2016). Adonis function informs
162 whether the groups consistently differ in their community composition. Adonis analyses were
163 performed using Bray-Curtis dissimilarity to calculate distance matrices based on vegetation
164 data (quadrat × species abundance matrices); it ranges from 0 and 1 and tends towards 1 as the
165 dissimilarity between plant communities increases (Bray and Curtis, 1957).

166 Correspondence Analyses (CA) were performed to detect and visualize changes in plant
167 community composition, structure, and recovery for each vegetation type. The two CAs (i.e.
168 one for each vegetation type, with the matrix of species cover in sandy grasslands = 30
169 observations × 81 species and the matrix of stony grasslands = 30 observations × 69 species)
170 were performed using data before fire (February 2009) and after fire (October 2011 and January
171 2012).

172 *2.3.2 Long-term fire effects*

173 Effects of time since last fire on mean species richness (number of species/m²) and mean
174 vegetation cover were assessed using Generalized Linear Models (glm) with a Poisson
175 distribution for richness and a Gaussian distribution for vegetation cover. Time since last fire

176 ((a) burnt less than a year ago (2011), (b) burnt 5 years ago (2007), (c) burnt 11 years ago
177 (2001) and (d) unburnt in the past 25 years) was defined as the categorical explanatory variable.

178 Analyses of variance (ANOVA) and Tukey post-hoc tests were performed to compare biomass
179 (i.e. total and dry biomass) according to time since last fire. Parametric conditions were tested
180 for each grassland type using Shapiro-Wilk test of normality and Bartlett's test for homogeneity
181 of variance. Where necessary, data were log-transformed to meet the assumptions of statistical
182 analyses.

183 As previously explained, dissimilarities between vegetation composition according to time
184 since last fire were analyzed using ANOVA-style analysis Adonis. We performed Constrained
185 Correspondence Analyses (CCA, i.e. Correspondence Analyses with including a conditioning
186 matrix) to detect and visualize changes in plant community composition and structure with
187 time since last fire, including a conditioning matrix of altitude, the effect of which was removed
188 (Oksanen, 2011). The two CCAs (i.e. one for each grassland type, with the matrix of species
189 cover in sandy grasslands = 120 observations \times 229 species and the matrix of stony grasslands
190 = 120 observations \times 233 species) were thus performed on the vegetation data collected in 2012
191 to assess differences between sites according to time since last fire (i.e. burnt less than a year
192 ago (2011), burnt 5 years ago, burnt 10 years ago and unburnt in the past 25 years).

193

194 **3. Results**

195 ***3.1. Short-term effect of fire***

196 Species richness and vegetation cover recovered quickly after fire. For both vegetation types,
197 species richness showed a clear significant decrease just after fire from more than 30 species/m²
198 on average in February 2009 and less than 20 species/m² in October 2001 (Fig. S1). However,
199 five months after fire, in January 2012, species richness significantly increased to values close
200 to pre-fire ones (glmer $z_{Sa} = 59.69$, $p_{Sa} < 0.001$; glmer $z_{St} = 62.91$, $p_{St} < 0.001$, Fig. 1a, Fig. S1).
201 Vegetation cover was low two months after fire but increased significantly ($F_{Sa} = 68.59$, p_{Sa}
202 < 0.001 ; $F_{St} = 19.88$, $p_{St} < 0.001$, Fig. 1b, Fig. S1) to reached, five months after fire, values
203 similar to pre-disturbance level on the stony grassland and values close to the pre-disturbance
204 level on the sandy grassland (Fig. 1b, Fig. S1).

205 Five months after fire, vegetation composition and structure of both communities, did not differ
206 from the pre-disturbance state, and returned to be very similar to the pre-disturbance state
207 (Table 2, Fig. 2 & 3, Table S1). No significant difference in terms of composition was found
208 between plots sampled prior to fire, and two and five months after fire (Table 2). The CA
209 revealed that the species composition of each quadrat of both grassland types sampled before
210 fire and five months after fire were very similar (Fig. 2 & 3).

211 ***3.2. Long-term vegetation dynamics after fire***

212 Species richness remained stable in sandy grasslands over 25 years (glm procedure $z_{Sa} = 96.27$,
213 $p = 0.9$) but decreased with time since last fire on the stony grasslands (glm procedure $z_{St} =$
214 133.04 , $p < 0.001$), with a higher species richness on recently burnt grasslands (i.e. less than 5
215 years ago, in 2011 and 2007) and lower species richness on grasslands burnt more than 11 years
216 ago (Fig. 4a). Total vegetation cover strongly increased with time since last fire on sandy
217 grasslands (glm procedure $F_{Sa} = 97.27$, $p < 0.001$, Fig. 4b) while it slightly decreased on stony
218 grasslands (glm procedure $F_{St} = 34.76$, $p < 0.001$, Fig. 4b).

219 Species composition and community structure differed according to the date of the last fire
220 (Table 3). For both vegetation types, species composition in the most recently burnt sites (< 1
221 year) and sites burnt less than 5 years before were more similar (Table 3). In the two vegetation
222 types, main differences in terms of vegetation composition were found between grasslands
223 burnt 11 years prior to the study (2001) and grasslands unburnt in the past 25 years (Table 3).

224 CCA performed on sandy grassland data highlighted that recently burnt areas (2011 and 2007
225 i.e. burnt less than 5 years ago) were similar in terms of species composition (Fig. 5), and two
226 of the three sites that were unburnt in the past 25 years were grouped together (Fig. 5, see also
227 Table S2). On stony grasslands, only two of the three sites unburnt in the past 25 years showed
228 a divergent composition compared to the other sites (Fig. 6). All other sites, although different
229 in species composition, did not show divergent plant composition according to time since last
230 fire (Fig. 6, see also Table S2).

231 There was almost twice more biomass in sandy grasslands than on stony grasslands in sites
232 unburnt at last for 25 years (Fig. 7). Total biomass and dry biomass increased significantly with
233 time since the last fire in both vegetation types (Fig. 7). While total and dry biomass increased
234 gradually over time on stony grasslands ($F = 6.30$, $p < 0.001$ for total biomass, and $F = 5.69$, p
235 < 0.001 for dry biomass; Fig. 7), on sandy grasslands, both the total and dry biomass remained
236 stable for the first 11 years but doubled between 1987 and 2001, due to an important increase
237 of dry biomass ($F = 12.74$, $p < 0.001$ for total biomass and $F = 13.37$, $p < 0.001$ for dry biomass;
238 Fig. 7).

239

240 **4. Discussion**

241 The sandy and stony grasslands are two different habitat types within *campo rupestre*
242 ecosystem (Le Stradic et al., 2015a), and both of them recover quickly, c.a. of five months,
243 after wildfire in terms of species richness and community composition, which corroborates
244 previous studies (Kolbek and Alves, 2008; Neves and Conceição, 2010). The high resumption
245 capacity of both plant communities implies that most of the species are able to rapidly
246 regenerate after fire. Vegetation cover on stony grasslands is naturally lower than that on sandy
247 grasslands, because quartzitic rocks overlay a large share of the ground (Le Stradic et al.,
248 2015a), that is why, in stony grasslands, vegetation reaches pre-fire cover values in only five
249 months. On the other hand, vegetation cover is denser on sandy grasslands and, although our
250 results show a delay for vegetation recovery in sandy grasslands, vegetation cover reaches
251 values similar to pre-disturbance quite quickly as well.

252 Our study shows that fire is not related to an increase in species richness in *campo rupestre*,
253 and community composition remains very similar before and after fire, contrary to subtropical
254 grasslands in southern Brazil where fire can lead to a short-term increase in species richness,
255 mainly through a reduction in plant competition and an increase in recruitment right after the
256 fire (Overbeck et al., 2005). While some species seem to be slightly favored in the early post-
257 fire environment, such as Poaceae or Cyperaceae which produce flowers and fruits in the weeks
258 following a fire (personal observation), no post-fire colonization by opportunistic species has
259 been observed. In many fire-prone environments, the recruitment processes of many species
260 are restricted to the first post-fire year, as these processes are stimulated by factors related to
261 fire (Bond et al., 1984; Bond and van Wilgen, 1996; Hobbs and Atkins, 1990; Knox and Clarke,
262 2006). In Australia or South Africa, fire regimes increase seedling emergence of shrubs (Bond
263 et al., 1984; Bond, 1984; Hobbs and Atkins, 1990; Knox and Clarke, 2006). Although no study
264 has dealt so far with recruitment processes after fire in *campo rupestre*, potential increase in

265 recruitment of some *campo rupestre* species may not be due to fire-triggered germination
266 (Fidelis et al., 2016; Le Stradic et al., 2015b) but instead because some species present fire-
267 stimulated flowering and fruiting (Conceição and Orr, 2012; Fidelis and Blanco, 2014; Lamont
268 and Downes, 2011; Le Stradic et al., 2015b).

269 In the present case, the regeneration of the original community appears mostly driven by the
270 persistence of resprouters. In fire-prone environments, plant community assemblages contains
271 species that are able 1) to resprout from any plant structure such as rhizomes, root buds, stem
272 buds, etc. and persist via vegetative regeneration, i.e. resprouters, or 2) to thrive in and to
273 recolonize a site through sexual reproduction and to persist at species level by seeds, i.e. seeders
274 (Bell, 2001; Keeley and Fotheringham, 2000; Pausas et al., 2004). Whereas it has been usually
275 neglected in favor of the 'regeneration niche', our results support the idea of the 'persistence
276 niche' i.e. regeneration and persistence *in situ* is mainly due to species resprouting ability (Bond
277 and Midgley, 2001). Resprouting ability is a key trait for persistence at the population level,
278 enabling species to survive diverse disturbance regimes. It may result in a strong impact on
279 community re-assembly, especially in fire-prone environments (Bond and Midgley, 2001;
280 Clarke et al., 2013). Most of the species in sandy and stony grasslands are perennial species
281 that resprout quickly after fire (Le Stradic et al., 2015a), which could explain the limited
282 changes in terms of species richness observed in our plant communities; once established,
283 perennial species do not necessary rely on seed germination to be maintained in the system,
284 and therefore might be less impacted by biomass accumulation and shade.

285 No significant change in species richness over time was found in sandy grasslands, while only
286 a slight decrease in species richness was observed in stony grasslands unburnt for more than
287 25 years. A decrease in species richness with time since last fire was expected (Bond and
288 Keeley, 2005; Uys et al., 2004), especially because some plant species with high light
289 requirements should be negatively affected by shade produced by increasing biomass.
290 Competition generated by some dominant species and the accumulated biomass over time may
291 limit the germination of annual species (Overbeck et al., 2005), as well as the persistence and
292 growth of some herbaceous *campo rupestre* species that are not tolerant to shade. Stony
293 grassland species might be more sensitive to competition for light because stony grasslands are
294 naturally more open vegetation, inducing a negative correlation between species richness and
295 biomass production (Overbeck et al., 2005).

296 While vegetation composition varied between the different study sites, including between
297 recently burnt areas and unburnt areas, no clear pattern was observed in terms of fire impacts.
298 It can result from the fact that vegetation composition analysis highlighted first the natural
299 heterogeneity of *campo rupestre* plant communities (Le Stradic et al., 2015a) rather than the
300 effect of fire on community composition. *Campo rupestre* are patchy plant communities under
301 the control of local topography, nature of substrate, soil depth and fires (Fernandes, 2016; Le
302 Stradic et al., 2015a; Neves and Conceição, 2010; Silveira et al., 2016). *Campo rupestre* are
303 located on mountaintops and represent isolated islands where species endemism and narrow
304 species endemism are common (Alves and Kolbek, 1994; Barbosa et al., 2015), leading to very
305 high heterogeneity in plant community composition even at small spatial scale. Studies of long-
306 term fire impacts on vegetation dynamics is therefore challenging. Natural heterogeneity of
307 community composition and fire patchiness hampers greatly to detangle the effects of fire from
308 the geographic localization of the plots. Long-term experimental studies using controlled
309 burnings would be more appropriated to study the impact of fire on plant species composition
310 in *campo rupestre*.

311 Similarly to our results, studies have shown that fire occurs due to fine fuel accumulation and
312 many ignition sources (Figueira et al., 2016). As stony grasslands naturally present a higher
313 level of bare ground compared to sandy grasslands (Le Stradic et al., 2015a), biomass
314 accumulation in this type of vegetation is lower. Values of total biomass in *campo rupestre*
315 varied between 200 g.m⁻² and 400 g.m⁻², which are similar to those found in *campo sujo* (i.e.
316 open vegetation physiognomy) (Conceição and Pivello, 2011). However, without any fire for
317 a long time period (i.e. >25 years), total biomass in sandy grasslands doubles, reaching values
318 above 800 g.m⁻², which are similar to those in wet grasslands in Brazilian savanna without fire
319 occurrence for four years (Fidelis et al., 2013). This implies that fire suppression over a long
320 time period should lead to high biomass accumulation. Throughout the world, prescribed
321 burning policy in fire-prone environments have resulted in a large accumulation of biomass,
322 an increase of fuel load and in severe wildfires (Van Wilgen et al., 1998). In *campo rupestre*,
323 the absence of sustainable fire management will probably lead to biological impoverishment
324 (Figueira et al., 2016).

325 Whereas fire should be managed as a necessary conservation tool, prescribed burning is a
326 matter of debate in Brazil (Durigan and Ratter, 2016; Figueira et al., 2016). Fire can be used to
327 protect sensitive vegetation, some *campo rupestre* species (Conceição et al., 2013; Conceição
328 and Orr, 2012) and to maintain community dynamics (Oliveras et al., 2012). However, fire can
329 also lead to severe damages for some populations in case of inappropriate fire regimes (Silveira
330 et al., 1999). Oliveras et al. (2012) suggested that an interval of at least 3 to 4 years between
331 each burn can be a sustainable fire regime, allowing the reestablishment plant communities to
332 a pre-burn condition of some Cerrado ecosystems. In the case of *campo rupestre*, further
333 studies are needed to assess what fire regimes should be applied to maintain biodiversity and
334 ecosystem services and what fire management strategy should be taken to avoid deleterious
335 fires and conserve *campo rupestre* (but see Figueira et al., 2016). In order to guarantee
336 conservation of their biodiversity and ecosystems services that have been severely threatened
337 by fire suppression policy, the definition and implementation of a clear fire management policy
338 is urgently necessary, especially in conservation units of cerrado and associated vegetation as
339 the *campo rupestre* (Durigan and Ratter, 2016).

340

341 **Acknowledgments**

342 We thank the National Park of Serra do Cipó for having made their fire map available to us
343 and two anonymous reviewers and Fernando O. A. Silveira for improving greatly the
344 manuscript. This work was carried out with grants from the Conselho Nacional de Pesquisa
345 (CNPq) [CNPq 561883/2010-6], Fundação de Amparo à Pesquisa do Estado de Minas Gerais
346 (FAPEMIG), RTP CNRS-CEMAGREF “Ingénierie Ecologique” [Ecological Engineering]
347 SAVER – SAVanna Ecological Restoration, Natural Reserva Vellozia and the Ministère
348 Français des affaires étrangères et européennes (EGIDE)[EGIDE 2009/657176K].

349

350 **References**

- 351 Alves, R.J.V., Kolbek, J., 2010. Vegetation strategy of *Vellozia crinita* (Velloziaceae).
352 *Biologia (Bratisl)*. 65, 254–264. doi:10.2478/s11756-010-0005-y
- 353 Alves, R.J. V., Kolbek, J., 1994. Plant species endemism in savanna vegetation on table
354 mountains (Campo Rupestre) in Brazil. *Vegetatio* 113, 125–139.
355 doi:10.1007/BF00044230
- 356 Barbosa, N.P. de U., Fernandes, G.W., Sanchez-Azofeifa, A., 2015. A relict species restricted

- 357 to a quartzitic mountain in tropical America: an example of microrefugium? *Acta Bot.*
358 *Brasilica* 29, 299–309. doi:10.1590/0102-33062014abb3731
- 359 Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models
360 using lme4. *J. Stat. Softw.* 67. doi:10.18637/jss.v067.i01
- 361 Bell, D.T., 2001. Ecological response syndromes in the flora of southwestern Western
362 Australia: Fire resprouters versus reseederers. *Bot. Rev.* 67, 417–440.
363 doi:10.1007/BF02857891
- 364 Bitencourt, C., Rapini, A., Santos Damascena, L., De Marco Junior, P., 2016. The worrying
365 future of the endemic flora of a tropical mountain range under climate change. *Flora -*
366 *Morphol. Distrib. Funct. Ecol. Plants* 218, 1–10. doi:10.1016/j.flora.2015.11.001
- 367 Bond, W., van Wilgen, B., 1996. *Fire and plants*. Chapman & Hall, London.
- 368 Bond, W., Vlok, J., Viviers, M., 1984. Variation in seedling recruitment of Cape Proteaceae
369 after fire. *J. Ecol.* 72, 209–221.
- 370 Bond, W.J., 1984. Fire survival of Cape Proteaceae - influence of fire season and seed
371 predators. *Vegetatio* 56, 65–74.
- 372 Bond, W.J., Keeley, J.E., 2005. Fire as a global “herbivore”: the ecology and evolution of
373 flammable ecosystems. *Trends Ecol. Evol.* 20, 387–394. doi:10.1016/j.tree.2005.04.025
- 374 Bond, W.J., Midgley, G.F., Woodward, F.I., Hoffman, M.T., Cowling, R.M., 2003. What
375 controls South African vegetation — climate or fire? *South African J. Bot.* 69, 79–91.
376 doi:10.1016/S0254-6299(15)30362-8
- 377 Bond, W.J., Midgley, J.J., 2001. Ecology of sprouting in woody plants : the persistence niche.
378 *Trends Ecol. Evol.* 16, 45–51.
- 379 Bray, J.R., Curtis, J.T., 1957. An ordination of the upland forest communities of southern
380 Wisconsin. *Ecol. Monogr.* 27, 325–349. doi:10.2307/1942268
- 381 Bustamante, M.M.C., 2015. Política de clima negligencia o Cerrado – mais uma vez [WWW
382 Document]. *Obs. do Clima*. URL <http://www.observatoriodoclima.eco.br/politica-de-clima-negligencia-o-cerrado-mais-uma-vez/>
383
- 384 Carvalho, F.M.V., De Marco, P., Ferreira, L.G., 2009. The Cerrado into-pieces: habitat
385 fragmentation as a function of landscape use in the savannas of central Brazil. *Biol.*
386 *Conserv.* 142, 1392–1403. doi:10.1016/j.biocon.2009.01.031
- 387 Clarke, P.J., Lawes, M.J., Midgley, J.J., Lamont, B.B., Ojeda, F., Burrows, G.E., Enright,
388 N.J., Knox, K.J.E., 2013. Resprouting as a key functional trait: how buds, protection and
389 resources drive persistence after fire. *New Phytol.* 197, 19–35. doi:10.1111/nph.12001
- 390 Conceição, A.A., Alencar, T.G., Souza, J.M., Moura, A.D.C., Silva, G.A., 2013. Massive
391 post-fire flowering events in a tropical mountain region of Brazil: high episodic supply
392 of floral resources. *Acta Bot. Brasilica* 27, 847–850. doi:10.1590/S0102-
393 33062013000400025
- 394 Conceição, A.A., Orr, B.J., 2012. Post-fire flowering and fruiting in *Vellozia sincorana*, a
395 caulescent rosette plant endemic to Northeast Brazil. *Acta Bot. Brasilica* 26, 94–100.

- 396 Conceição, A.A., Pivello, V.R., 2011. Biomassa combustível em campo sujo no entorno do
397 Parque Nacional da Chapada Diamantina, Bahia, Brasil. *Biodiversidade Bras.* 2, 146–
398 160.
- 399 Coutinho, L.M., 1978. O conceito de cerrado. *Rev. Bras. Botânica* 1, 17–23.
- 400 Crawley, M.J., 2007. *The R Book.*, John Wiley. ed. Chichester.
- 401 Durigan, G., Ratter, J.A., 2016. The need for a consistent fire policy for Cerrado
402 conservation. *J. Appl. Ecol.* 53, 11–15. doi:10.1111/1365-2664.12559
- 403 Fernandes, G.W., 2016. *Ecology and conservation of mountaintop grasslands in Brazil.*
404 Springer International Publishing, Switzerland. doi:10.1007/978-3-319-29808-5
- 405 Ferreira, M.M.A. de A.S., Conceição, A.A., 2012. Alocação preferencial de recursos e
406 morfologia de órgãos subterrâneos em plantas resistentes ao fogo em vegetação
407 campestre. *Sitientibus série Ciências Biológicas* 12, 143–149.
- 408 Fidelis, A., Blanco, C., 2014. Does fire induce flowering in Brazilian subtropical grasslands?
409 *Appl. Veg. Sci.* 17, 690–699. doi:10.1111/avsc.12098
- 410 Fidelis, A., Daibes, L.F., Martins, A.R., 2016. To resist or to germinate? The effect of fire on
411 legume seeds in Brazilian subtropical grasslands. *Acta Bot. Brasilica* 30, 147–151.
412 doi:10.1590/0102-33062015abb0187
- 413 Fidelis, A., Lyra, M.F. di S., Pivello, V.R., 2013. Above- and below-ground biomass and
414 carbon dynamics in Brazilian Cerrado wet grasslands. *J. Veg. Sci.* 24, 356–364.
415 doi:10.1111/j.1654-1103.2012.01465.x
- 416 Fidelis, A., Müller, S.C., Pillar, V.D., Pfadenhauer, J., 2010. Population biology and
417 regeneration of forbs and shrubs after fire in Brazilian Campos grasslands. *Plant Ecol.*
418 211, 107–117. doi:10.1007/s11258-010-9776-z
- 419 Figueira, J.E.C., Ribeiro, K.T., Ribeiro, M.C., Jacobi, C.M., França, H., Neves, A.C. de O.,
420 Conceição, A.A., Mourão, F.A., Souza, J.M., Miranda, C.A. de K., 2016. Fire in
421 rupestrian grasslands: plant response and management, in: Fernandes, G.W. (Ed.),
422 *Ecology and Conservation of Mountaintop Grasslands in Brazil.* Springer International
423 Publishing, Switzerland, pp. 415–448.
- 424 Gorgone-Barbosa, E., Pivello, V.R., Baeza, M.J., Fidelis, A., 2016. Disturbance as a factor in
425 breaking dormancy and enhancing invasiveness of African grasses in a Neotropical
426 Savanna. *Acta Bot. Brasilica* 30, 131–137. doi:10.1590/0102-33062015abb0317
- 427 Gorgone-Barbosa, E., Pivello, V.R., Bautista, S., Zupo, T., Rissi, M.N., Fidelis, A., 2015.
428 How can an invasive grass affect fire behavior in a tropical savanna? A community and
429 individual plant level approach. *Biol. Invasions* 17, 423–431. doi:10.1007/s10530-014-
430 0740-z
- 431 Grubb, P.J., 1977. The maintenance of species-richness in plant communities: the importance
432 of the regeneration niche. *Biol. Rev.* 52, 107–145. doi:10.1111/j.1469-
433 185X.1977.tb01347.x
- 434 Hobbs, R.J., Atkins, L., 1990. Fire-related dynamics of a *Banksia* woodland in south-western
435 western Australia. *Aust. J. Bot.* 38, 97–110. doi:http://dx.doi.org/10.1071/BT9900097

- 436 Hodgson, D., McDonald, J.L., Hosken, D.J., 2015. What do you mean, “resilient”? Trends
437 Ecol. Evol. 30, 503–506. doi:10.1016/j.tree.2015.06.010
- 438 Hoffmann, W.A., 1996. The effects of fire and cover on seedling establishment in a
439 Neotropical Savanna. J. Ecol. 84, 383–393. doi:10.2307/2261200
- 440 Hoffmann, W.A., Geiger, E.L., Gotsch, S.G., Rossatto, D.R., Silva, L.C.R., Lau, O.L.,
441 Haridasan, M., Franco, A.C., 2012. Ecological thresholds at the savanna-forest
442 boundary: how plant traits, resources and fire govern the distribution of tropical biomes.
443 Ecol. Lett. 15, 759–768. doi:10.1111/j.1461-0248.2012.01789.x
- 444 Keeley, J.E., Fotheringham, C.J., 2000. Role of fire in regeneration from seed, in: Fenner, M.
445 (Ed.), Seeds: The Ecology of Regeneration in Plant Communities. CABI publishing,
446 Wallingford, pp. 311–330.
- 447 Knox, K.J.E., Clarke, P.J., 2006. Fire season and intensity affect shrub recruitment in
448 temperate sclerophyllous woodlands. Oecologia 149, 730–739. doi:10.1007/s00442-006-
449 0480-6
- 450 Kolbek, J., Alves, J.R.V., 2008. Impacts of cattle, fire and wind in rocky savannas,
451 southeastern Brazil. Acta Univ. Carolinae Environ. 22, 111–130.
- 452 Lamont, B.B., Downes, K.S., 2011. Fire-stimulated flowering among resprouters and
453 geophytes in Australia and South Africa. Plant Ecol. 212, 2111–2125.
454 doi:10.1007/s11258-011-9987-y
- 455 Le Stradic, S., Buisson, E., Fernandes, G.W., 2015a. Vegetation composition and structure of
456 some Neotropical mountain grasslands in Brazil. J. Mt. Sci. 12, 864–877.
457 doi:10.1007/s11629-013-2866-3
- 458 Le Stradic, S., Silveira, F.A.O., Buisson, E., Cazelles, K., Carvalho, V., Fernandes, G.W.,
459 2015b. Diversity of germination strategies and seed dormancy in herbaceous species of
460 campo rupestre grasslands. Austral Ecol. 40, 537–546. doi:10.1111/aec.12221
- 461 Lehmann, C.E.R., Prior, L.D., Williams, R.J., Bowman, D.M.J.S., 2008. Spatio-temporal
462 trends in tree cover of a tropical mesic savanna are driven by landscape disturbance. J.
463 Appl. Ecol. 45, 1304–1311. doi:10.1111/j.1365-2664.2008.01496.x
- 464 Madeira, J., Fernandes, G.W., 1999. Reproductive phenology of sympatric taxa of
465 Chamaecrista (Leguminosae) in Serra do Cipo. J. Trop. Ecol. 15, 463–479.
- 466 McIntyre, S., Hobbs, R., 1999. A framework for conceptualizing human effects on
467 landscapes and its relevance to management and research models. Conserv. Biol. 13,
468 1282–1292. doi:10.1046/j.1523-1739.1999.97509.x
- 469 Miola, D.T.B., Correia, H.V.L., Fernandes, G.W., Negreiros, D., 2010. The effect of fire on
470 phenology of *Syagrus glaucescens* Glaz. ex Becc. (Arecaceae). Neotrop. Biol. Conserv.
471 5, 146–153. doi:10.4013/4759
- 472 Miranda, H.S., Bustamante, M.M.C., Miranda, A.C., 2002. The fire factor, in: Oliveira, P.S.,
473 Marquis, R.J. (Eds.), The Cerrados of Brazil: Ecology and Natural History of a
474 Neotropical Savanna. Columbia University Press, pp. 51–68.
- 475 Mistry, J., 2000. World savannas: ecology and human use. Routledge, New York.

- 476 Mistry, J., Berardi, A., Andrade, V., Krahô, T., Krahô, P., Leonardos, O., 2005. Indigenous
477 fire management in the cerrado of Brazil: the case of the Krahô of Tocantins. *Hum.*
478 *Ecol.* 33, 365–386. doi:10.1007/s10745-005-4143-8
- 479 Mittermeier, R.A., Turner, W.R., Larsen, F.W., Brooks, T.M., Gascon, C., 2011. Global
480 biodiversity conservation: the critical role of hotspots, in: Zachos, F.E., Habel, J.C.
481 (Eds.), *Biodiversity Hotspots: Distribution and Protection of Conservation Priority*
482 *Areas*. Springer Science & Business Media, Berlin, pp. 3–22.
- 483 Mueller-Dombois, D., Ellenberg, H., 1974. *Aims and methods of vegetation ecology*. John
484 Wiley & Sons, New York.
- 485 Neves, A.C.O., Bedê, L.C., Martins, R.P., 2011. Revisão sobre os efeitos do fogo em
486 *Eriocaulaceae* como subsídio para a sua conservação. *Biodiversidade Bras.* 2, 50–66.
- 487 Neves, S.P.S., Conceição, A.A., 2010. Campo rupestre recém-queimado na Chapada
488 Diamantina, Bahia, Brasil : plantas de rebrota e sementes, com espécies endêmicas na
489 rocha. *Acta Bot. Brasilica* 24, 697–707.
- 490 Oksanen, J., 2011. *Multivariate Analysis of Ecological Communities in R : vegan tutorial*.
- 491 Oksanen, J., Blanchet, F.G., Kindt, R., Legendre, P., Minchin, P.R., O’Hara, R.B., Simpson,
492 G.L., Solymos, P., Wagner, M., Stevens, H.H., Wagner, H., 2016. *Vegan: community*
493 *ecology package [WWW Document]*. R Packag. version 2.3-3. URL [http://cran.r-](http://cran.r-project.org/package=vegan)
494 [project.org/package=vegan](http://cran.r-project.org/package=vegan)
- 495 Oliveira, P.S., Marquis, R.J., 2002. *The Cerrados of Brazil. Ecology and natural history of a*
496 *neotropical savanna*. Columbia University Press, New York.
- 497 Oliveira, R.S., Abrahão, A., Pereira, C., Teodoro, G.S., Brum, M., Alcantara, S., Lambers,
498 H., 2016. *Ecophysiology of Campos Rupestres Plants*, in: Fernandes, G.W. (Ed.),
499 *Ecology and Conservation of Mountaintop Grasslands in Brazil*. Springer International
500 Publishing, Switzerland, pp. 227–272.
- 501 Oliveira, R.S., Galvão, H.C., de Campos, M.C.R., Eller, C.B., Pearse, S.J., Lambers, H.,
502 2015. Mineral nutrition of campos rupestres plant species on contrasting nutrient-
503 impoverished soil types. *New Phytol.* 205, 1183–1194. doi:10.1111/nph.13175
- 504 Oliveras, I., Meirelles, S.T., Hirkuri, V.L., Freitas, C.R., Miranda, H.S., Pivello, V.R., 2012.
505 *Effects of fire regimes on herbaceous biomass and nutrient dynamics in the Brazilian*
506 *savanna*. *Int. J. Wildl. Fire* 22, 368–380. doi:10.1071/WF10136
- 507 Overbeck, G.E., Müller, S.C., Pillar, V.D., Pfadenhauer, J., 2005. Fine-scale post-fire
508 dynamics in southern Brazilian subtropical grassland. *J. Veg. Sci.* 16, 655–664.
509 doi:10.1658/1100-9233(2005)016[0655:FPDISB]2.0.CO;2
- 510 Pausas, J.G., Bradstock, R.A., Keith, D.A., Keeley, J.E., The GCTE (Global Change of
511 Terrestrial Ecosystems) Fire Network, 2004. *Plant functional traits in relation to fire in*
512 *crown-fire ecosystems*. *Ecology* 85, 1085–1100.
- 513 R Core Team, 2015. *R: A language and environment for statistical computing [WWW*
514 *Document]*. R Found. Stat. Comput. Austria. URL <http://www.r-project.org/>.
- 515 Ramos-Neto, M.B., Pivello, V.R., 2014. *Lightning fires in a Brazilian savanna national park:*

- 516 rethinking management strategies. *Environ. Manage.* 26, 675–684.
517 doi:10.1007/s002670010124
- 518 Ratnam, J., Bond, W.J., Fensham, R.J., Hoffmann, W.A., Archibald, S., Lehmann, C.E.R.,
519 Anderson, M.T., Higgins, S.I., Sankaran, M., 2011. When is a “forest” a savanna , and
520 why does it matter ? *Glob. Ecol. Biogeogr.* 20, 653–660. doi:10.1111/j.1466-
521 8238.2010.00634.x
- 522 Ribeiro, M.C., Figueira, J.E.C., 2011. Uma abordagem histórica do fogo no parque nacional
523 da Serra do Cipó, Minas Gerais – Brasil. *Biodiversidade Bras.* 1, 212–227.
- 524 Sankaran, M., Hanan, N.P., Scholes, R.J., Ratnam, J., Augustine, D.J., Cade, B.S., Gignoux,
525 J., Higgins, S.I., Le Roux, X., Ludwig, F., Ardo, J., Banyikwa, F., Bronn, A., Bucini, G.,
526 Caylor, K.K., Coughenour, M.B., Diouf, A., Ekaya, W., Feral, C.J., February, E.C.,
527 Frost, P.G.H., Hiernaux, P., Hrabar, H., Metzger, K.L., Prins, H.H.T., Ringrose, S., Sea,
528 W., Tews, J., Worden, J., Zambatis, N., 2005. Determinants of woody cover in African
529 savannas. *Nature* 438, 846–849. doi:10.1038/nature04070
- 530 Silveira, F.A.O., Negreiros, D., Barbosa, N.P.U., Buisson, E., Carmo, F.F., Carstensen, D.W.,
531 Conceição, A.A., Cornelissen, T.G., Echternacht, L., Fernandes, G.W., Garcia, Q.S.,
532 Guerra, T.J., Jacobi, C.M., Lemos-Filho, J.P., Le Stradic, S., Morellato, L.P.C., Neves,
533 F.S., Oliveira, R.S., Schaefer, C.E., Viana, P.L., Lambers, H., 2016. Ecology and
534 evolution of plant diversity in the endangered campo rupestre: a neglected conservation
535 priority. *Plant Soil* 403, 129–152. doi:10.1007/s11104-015-2637-8
- 536 Silveira, L., Rodrigues, F.H.G., Jácomo, A.T.A., Filho, J.A.D., 1999. Impact of wildfires on
537 the megafauna of Emas National Park, central Brazil. *Oryx* 33, 108–114.
- 538 Simon, M.F., Amaral, M.F., 2003. *Mimosa splendida* Barneby (Mimosoideae, Leguminosae)
539 rediscovered in Central Brazil: preliminary studies for conservation of a rare species.
540 *Brazilian J. Bot.* 26, 93–96. doi:10.1590/S0100-84042003000100010
- 541 Simon, M.F., Grether, R., Queiroz, L.P. De, Skema, C., Pennington, R.T., Hughes, C.E.,
542 2009. Recent assembly of the Cerrado, a neotropical plant diversity hotspot, by in situ
543 evolution of adaptations to fire. *Proc Natl Acad Sci U S A* 106, 20359–20364.
- 544 Uys, R.G., Bond, W.J., Everson, T.M., 2004. The effect of different fire regimes on plant
545 diversity in southern African grasslands. *Biol. Conserv.* 118, 489–499.
546 doi:10.1016/j.biocon.2003.09.024
- 547 Van Wilgen, B.W., Biggs, H.C., Potgieter, A.L.F., 1998. Fire management and research in
548 the Kruger National Park, with suggestions on the detection of thresholds of potential
549 concern. *Koedoe* 41, 69–87. doi:10.4102/koedoe.v41i1.248

550

551 Appendix A. Supplementary material

552 Supplementary Table S1: List of plant species occurring on sandy grassland (Sa-C) and stony
553 grassland (St-C) surveyed to study short-term fire effects on plant communities, including
554 family, species name, plant form and the presence of each species for the three sample date:
555 before fire (Feb 2009), two months after fire (Oct 2011) and five months after fire (Jan 2012).

556

557 Supplementary Table S2: List of plant species occurring on sandy grasslands and stony
558 grassland surveyed to study long-term fire effects on plant communities, including family,
559 species name, plant form and the presence of each species in sandy or stony grasslands.

560

561 Supplementary Figure S1: a) General view of *campo rupestre* at Serra do Cipó (Minas
562 Gerais, Brazil) with the mosaic of herbaceous and shrubland occurring in these particular
563 ecosystems, including sandy grasslands and stony grasslands in the foreground and shrubland
564 on rocky outcrops in the background. View of b) quadrats Ca1-5 in sandy grassland and c)
565 Cp1-11 in stony grassland in December 2009, before fire and view of the same quadrat d)
566 Ca1-5 and e) Cp1-11 in October 2011 and f) Ca1-5 and g) Cp1-11 in January 2012.

567

568 Table 1. Information about study sites including type of vegetation (sandy or stony grasslands),
569 sites name as presented in Fig. 5 & 6, altitude (m), year of the last fire and time since last fire
570 (yr): (<1) burnt less than a year ago, in 2011, (5) burnt 5 years ago in 2007, (11) burnt 11 years
571 ago in 2001 and (>25) unburnt in the past 25 years, with a last burn dated at least in 1987. *
572 indicates the two sites which were followed more precisely before (February, 2009) and after
573 the fire (October 2011 and January 2012).

Sandy grasslands				Stony grasslands			
Site name	Altitude (m)	Year of the last fire	Time since last fire (yr)	Site name	Altitude (m)	Year of the last fire	Time since last fire (yr)
Sa-A	1299	2011	<1	St-A	1313	2011	<1
Sa-C *	1178	2011	<1	St-C *	1174	2011	<1
Sa-D	1232	2011	<1	St -D	1220	2011	<1
Sa -B	1299	2007	5	St -B	1313	2007	5
Sa -F	1337	2007	5	St -F	1335	2007	5
Sa -L	1362	2007	5	St -L	1353	2007	5
Sa -G	1402	2001	11	St -G	1403	2001	11
Sa -H	1417	2001	11	St -H	1419	2001	11
Sa -I	1347	2001	11	St -I	1348	2001	11
Sa -E	1106	1987	> 25	St -E	1109	1987	> 25
Sa -J	1284	1987	> 25	St -J	1288	1987	> 25
Sa -K	1303	1987	> 25	St -K	1295	1987	> 25

574

575

576 Table 2: Mean Bray-Curtis dissimilarity based on floristic composition between plots before
 577 fire (February 2009) and after fire (October 2011 and January 2012), for the two kinds of
 578 vegetations, the sandy and the stony grasslands. Results of Adonis analysis (i.e. difference in
 579 group means) based on Bray-Curtis dissimilarity (vegetation data) in February 2009, October
 580 2011 and January 2012).

		February 2009	October 2011	Adonis - F	p-value
Sandy grassland	October 2011	0.69		1.85	0.069
	January 2012	0.53	0.65		
		February 2009	October 2011	Adonis - F	p-value
Stony grassland	October 2011	0.47		0.62	0.700
	January 2012	0.40	0.46		

581

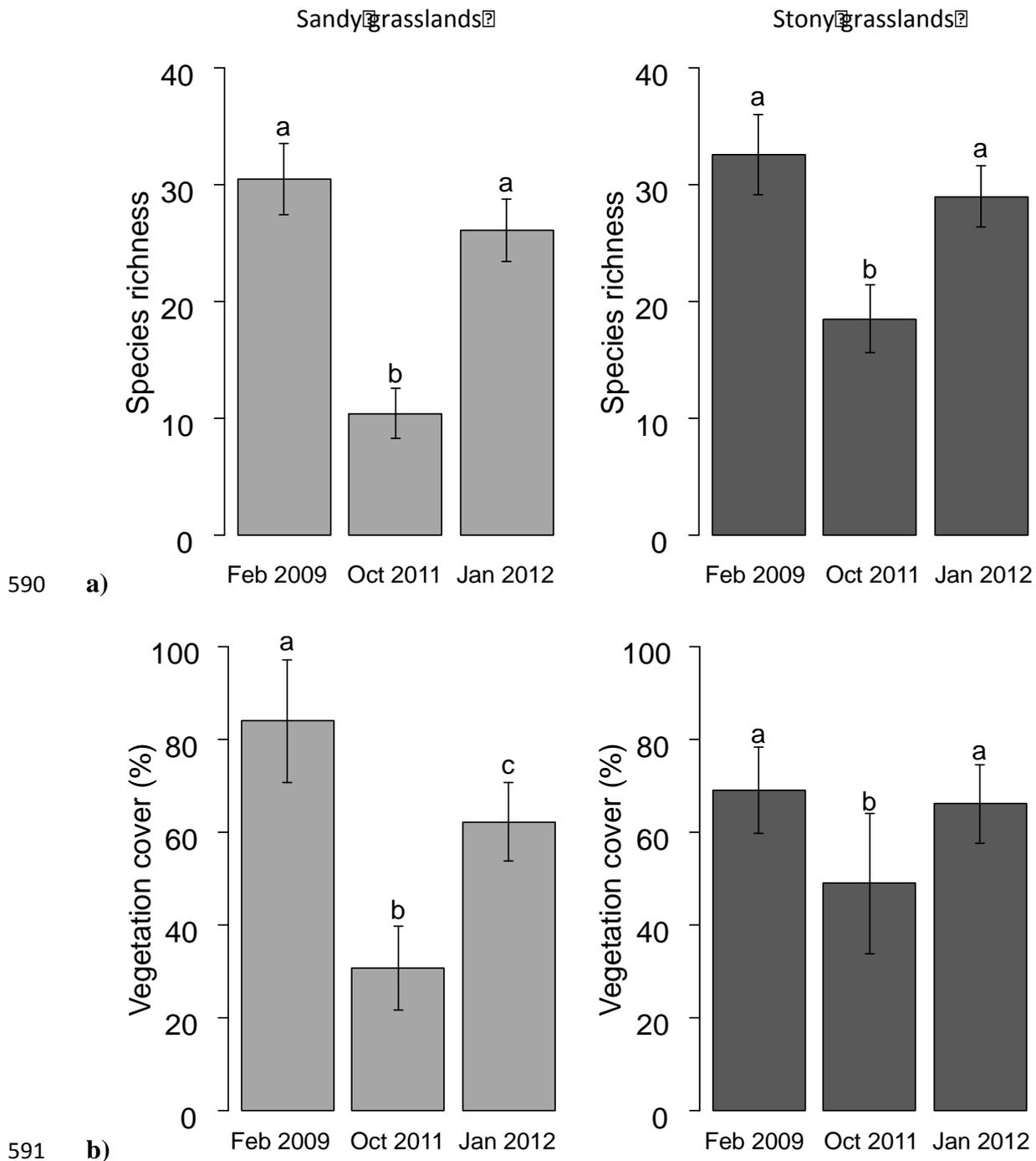
582

583 Table 3: Mean Bray-Curtis dissimilarity based on floristic composition between plots burnt in
 584 2011: burnt less than a year ago, 2007: burnt 5 years ago, 2001: burnt between 11 years ago
 585 and 1987: unburnt in the past 25 years, for the two kind of vegetations, the sandy and the stony
 586 grasslands. Results of Adonis analysis (i.e. difference in group means) were based on Bray-
 587 Curtis dissimilarity (vegetation data).

		1987	2001	2007	Adonis - F	p-value
Sandy grasslands	2001	0.83			10.32	0.0049 **
	2007	0.76	0.79			
	2011	0.71	0.79	0.68		
		1987	2001	2007	Adonis - F	p-value
Stony grasslands	2001	0.87			10.99	0.0049 **
	2007	0.85	0.83			
	2011	0.83	0.84	0.76		

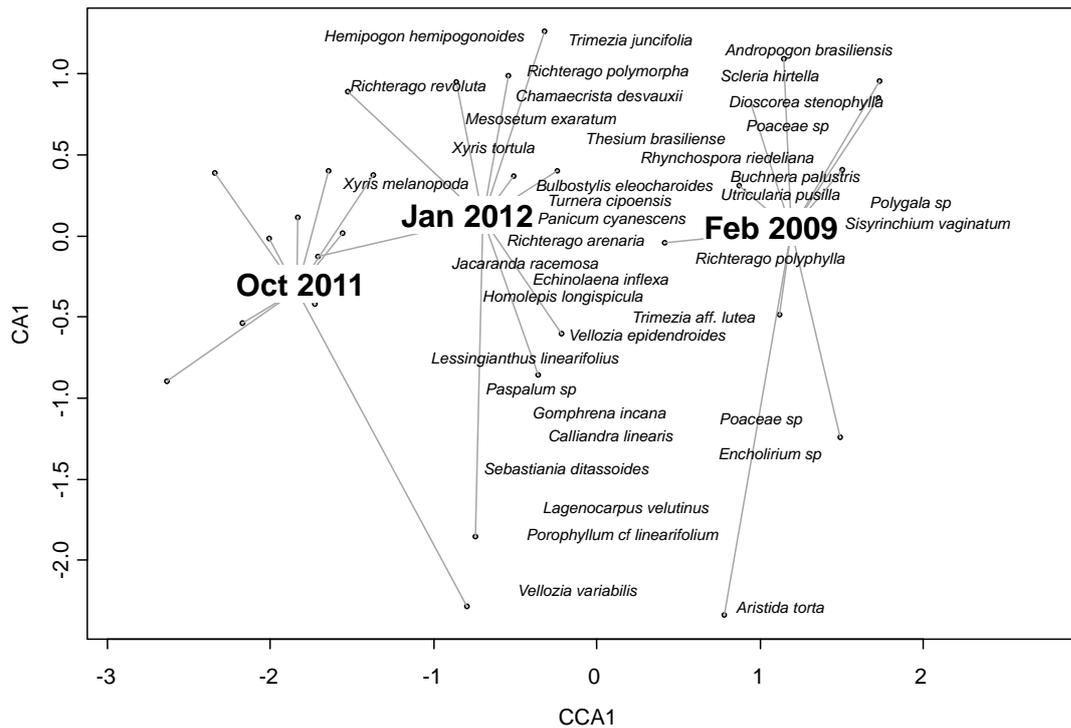
588

589



592 Fig. 1. Short-term impact of a wildfire on a) species richness (in 1m²) and b) vegetation cover
 593 (cover percentage in 1m²) in two types of *campo rupestre* vegetation at Serra do Cipó
 594 (Brazil): the sandy grasslands, represented with light grey bars, and the stony grasslands with
 595 dark grey bars. Quadrats were sampled before fire (Feb 2009), two months after fire (Oct
 596 2011) and five months after fire (Jan 2012). Means \pm standard errors are shown. Bars with no
 597 letter in common are significantly different according to the glmer procedures and pairwise
 598 comparisons ($p < 0.05$)

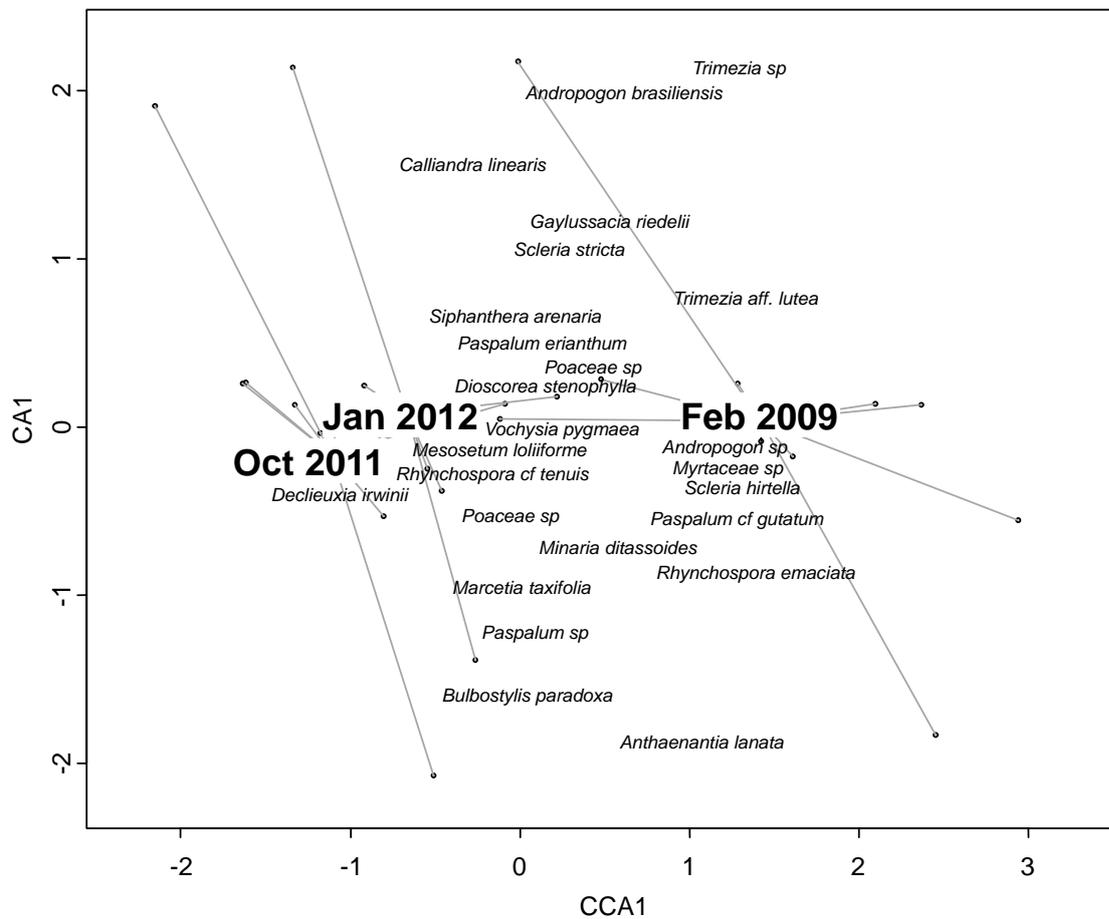
599



600

601 Fig. 2. Correspondence Analysis (CA) based on vegetation survey in ten 1 m² permanent plots
 602 on sandy grassland [30 observations × 81 species] before fire (Feb 2009), two months after fire
 603 (Oct 2011) and five months after fire (Jan 2012). For clarity, only the most correlated species
 604 are shown. Barycenters of each sampling date are represented.

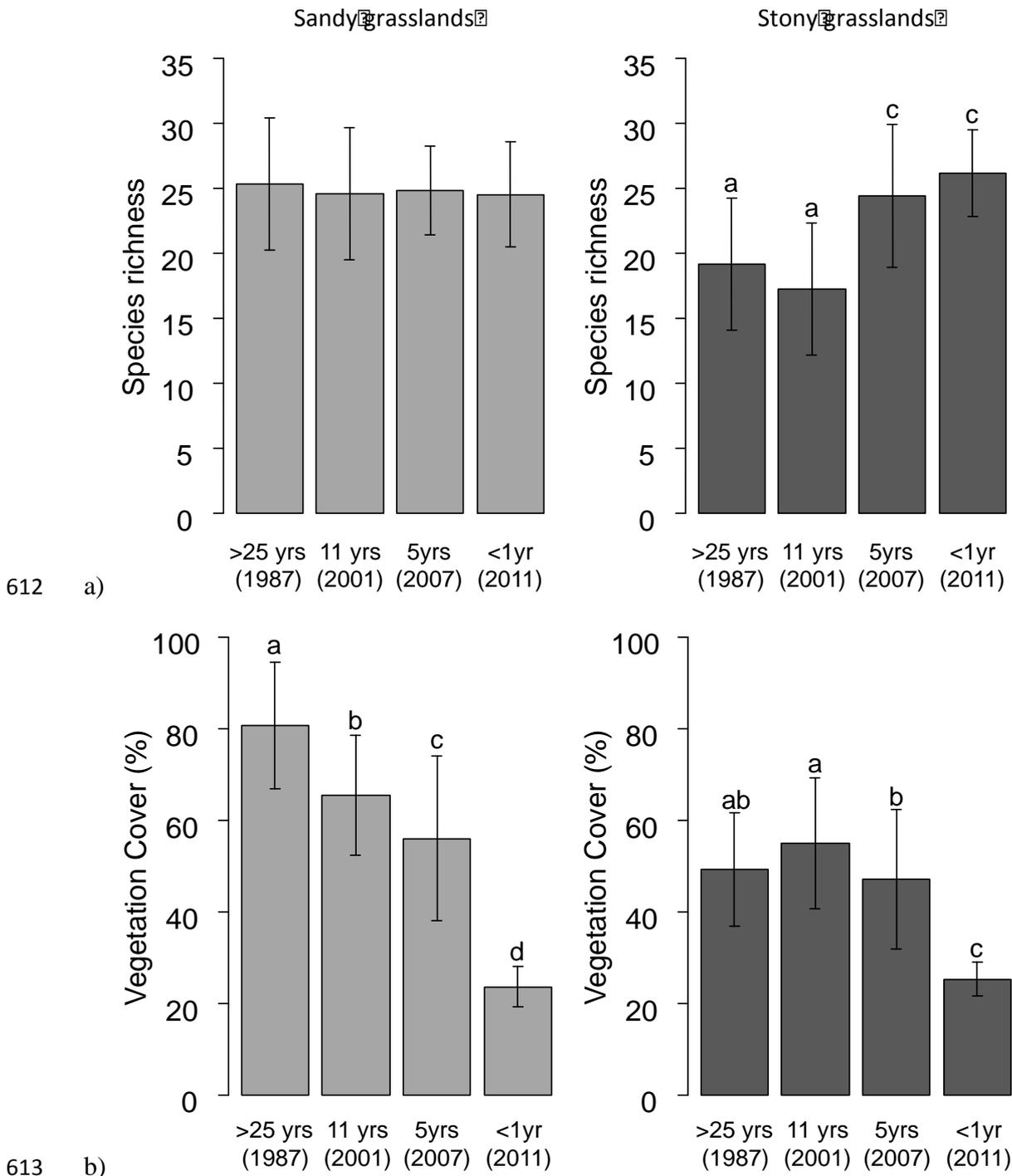
605



606

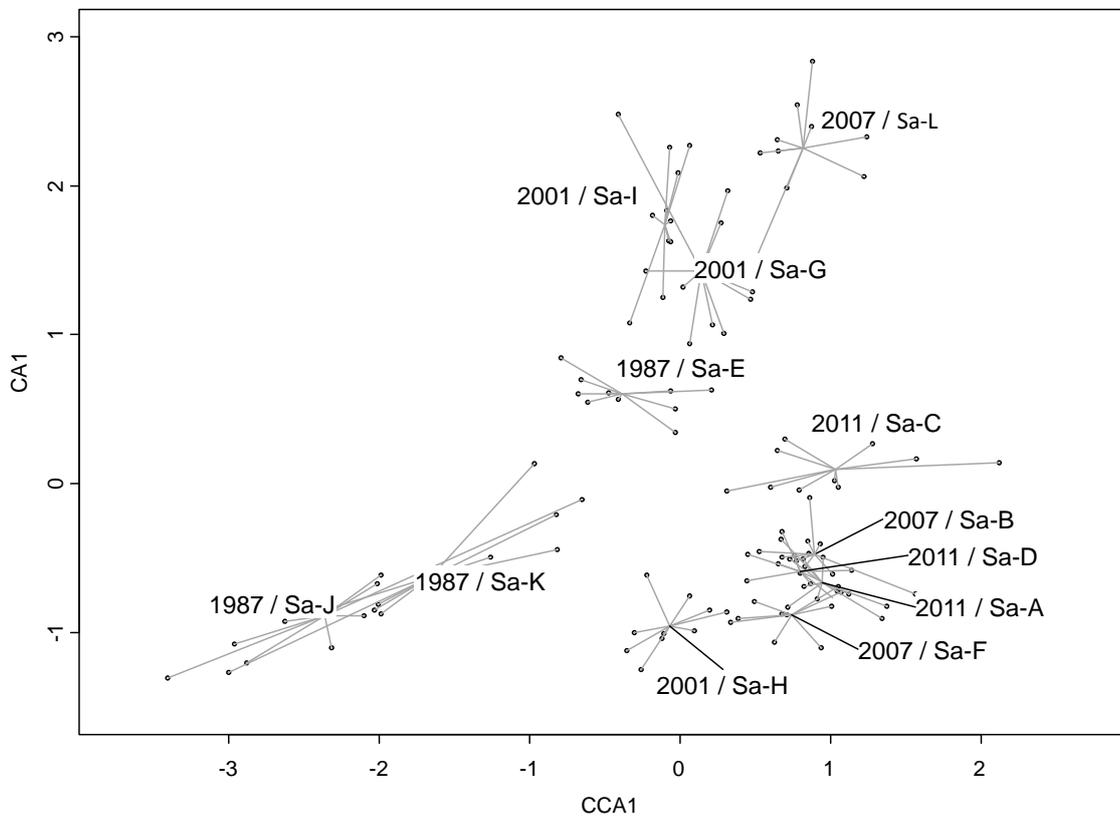
607 Fig. 3. Correspondence Analysis (CA) based on vegetation survey in ten 1m² permanent plots
 608 on stony grassland [30 observations × 69 species] before fire (Feb 2009), two months after fire
 609 (Oct 2011) and five months after fire (Jan 2012). For clarity, only the most correlated species
 610 are shown. Barycenters of each sampling date are represented.

611

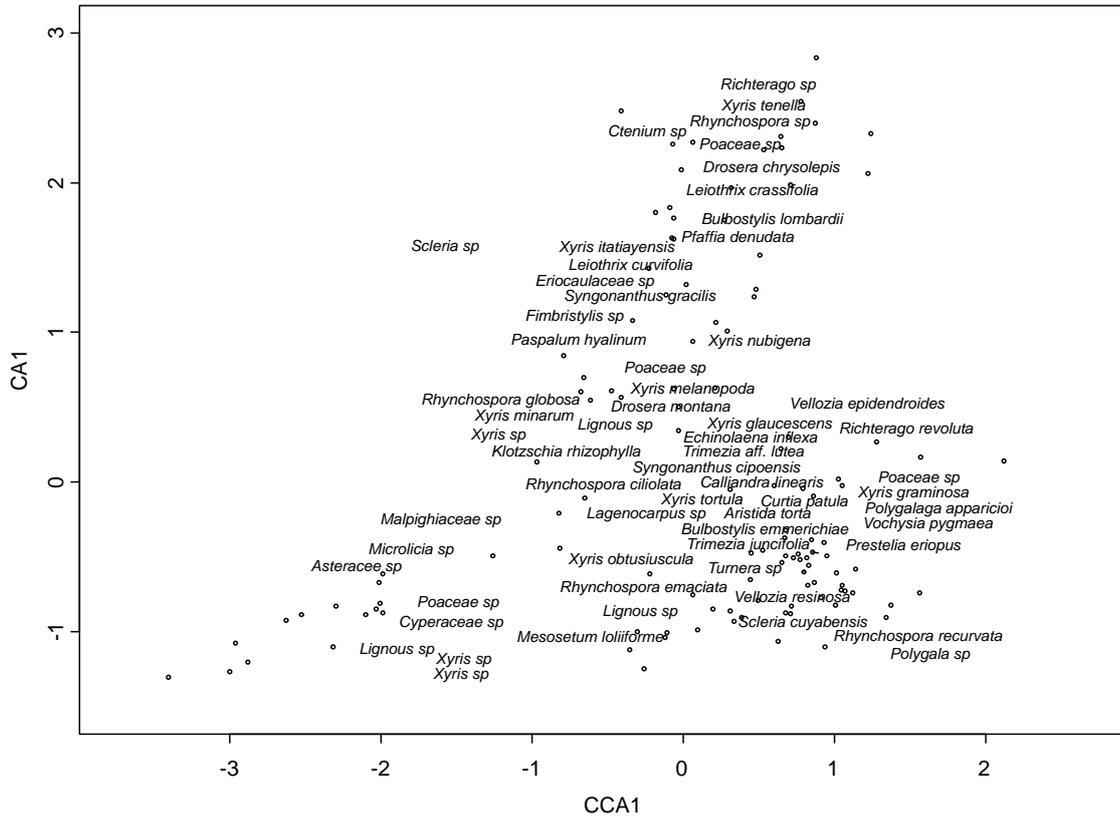


614 Fig. 4. Long-term impact of wildfires on a) species richness (in 1 m²) and b) vegetation cover
 615 (cover percentage in 1 m²). Two types of *campo rupestre* vegetation at Serra do Cipó (Brazil)
 616 are represented: the sandy grasslands, with light grey bars, and the stony grasslands with dark
 617 grey bars, according to time since last fire are: unburnt in the past 25 years (1987), burnt 11
 618 years ago (2001), burnt 5 years ago (2007) and burnt less than one year (2011). Means ±
 619 standard errors are shown. Bars with no letter in common are significantly different according
 620 to the glm procedures (p < 0.05).

621



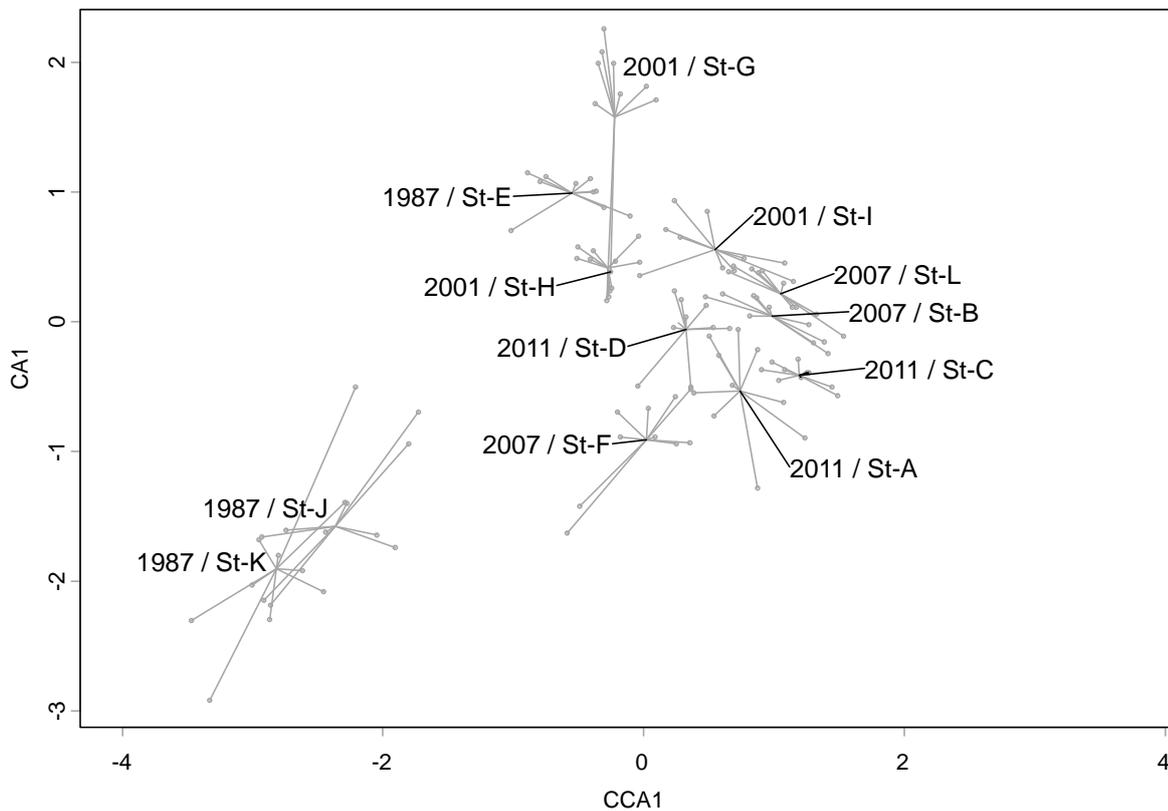
622



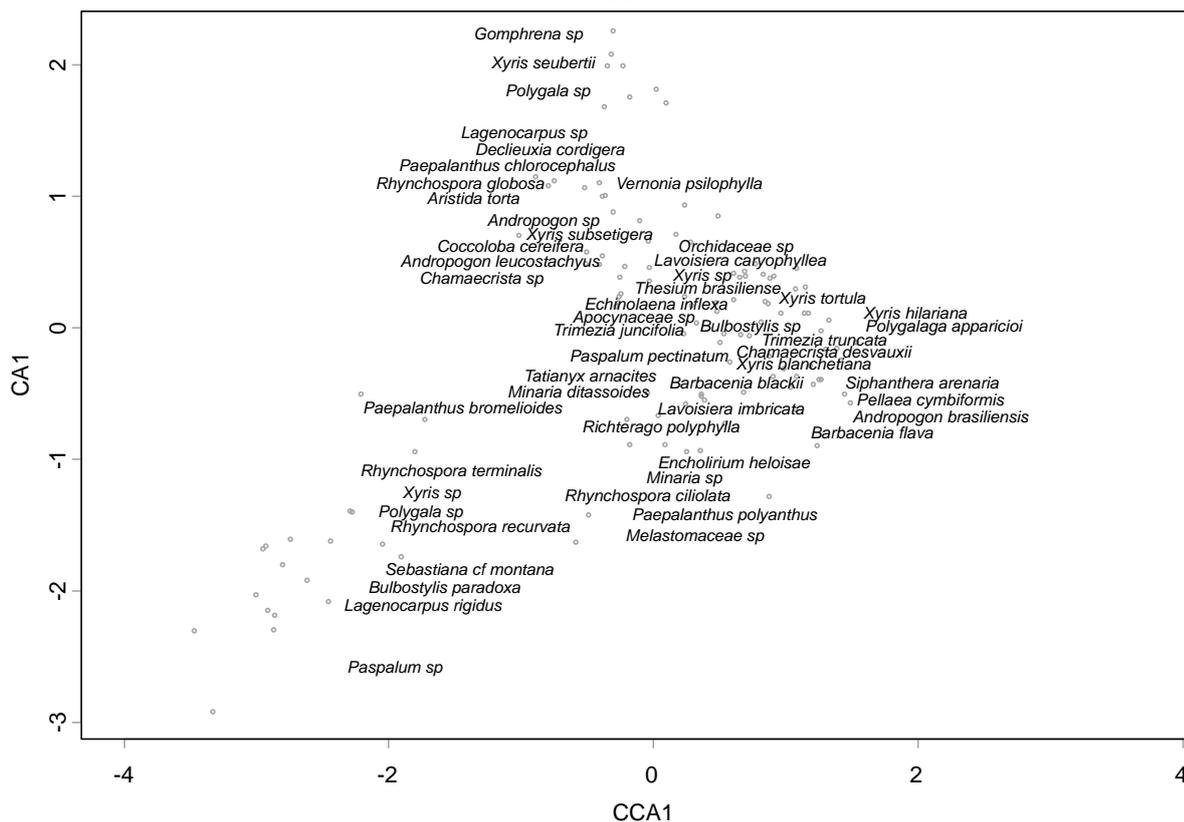
623

624 Fig. 5. Constrained Correspondence Analysis (CCA) based on vegetation survey performed on 120 1-
 625 m² vegetation plots on sandy grasslands [120 observations × 229 species] in sites unburnt in the past
 626 25 years (1987), burnt 11 years ago (2001), burnt 5 years ago (2007) and burnt less than one year
 627 (2011). For clarity, barycenters of each sampling site are represented on above graph and only the most
 628 correlated species are shown on below graph. Abbreviations (i.e. St-A, B, C, D, E, F, G, H, I, J, K)
 629 refer to site name (see table 1).

630

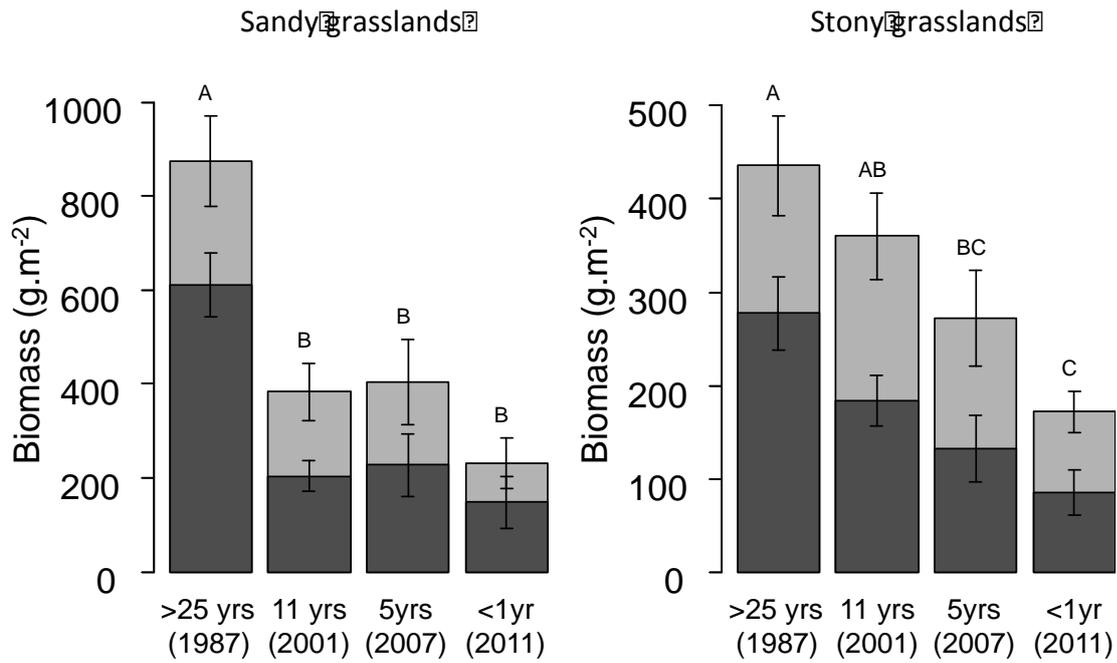


631



632

633 Fig. 6. Constrained Correspondence Analysis (CCA) based on vegetation survey performed on 120 1-
 634 m² vegetation plots on stony grasslands [120 observations × 233 species] in sites unburnt in the past 25
 635 years (1987), burnt 11 years ago (2001), burnt 5 years ago (2007) and burnt less than one year (2011).
 636 For clarity, barycenters of each sampling site are represented on above graph and only the most
 637 correlated species are shown on below graph. Abbreviations (i.e. St-A, B, C, D, E, F, G, H, I, J, K)
 638 refer to site name (see table 1).



639
640
641
642
643
644

Fig. 7. Total plant biomass (light grey) and dry plant biomass (dark grey) in sandy and stony grasslands at Serra do Cipó (Brazil) in sites unburnt in the past 25 years (1987), burnt 11 years ago (2001), burnt 5 years ago (2007) and burnt less than one year (2011). Means \pm standard errors are shown. Results of Tukey post-hoc tests performed on total biomass are indicated with letters and were equal than Tukey post-hoc tests performed on dry biomass.