



**HAL**  
open science

# Impact of small-scale conservation management methods on spider assemblages in xeric grassland

Tomas Hamřík, Ondrej Košulič

## ► To cite this version:

Tomas Hamřík, Ondrej Košulič. Impact of small-scale conservation management methods on spider assemblages in xeric grassland. *Agriculture, Ecosystems & Environment*, 2021, 307, pp.107225. 10.1016/j.agee.2020.107225 . hal-03040285

**HAL Id: hal-03040285**

**<https://hal.science/hal-03040285>**

Submitted on 21 Jan 2021

**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 **Impact of small-scale conservation management methods on spider**  
2 **assemblages in xeric grassland**

3

4 **Tomáš Hamřík<sup>a,b, c\*</sup>, Ondřej Košulič<sup>a</sup>**

5 <sup>a</sup>Department of Forest Protection and Wildlife Management, Faculty of Forestry and Wood  
6 Technology, Mendel University in Brno, Zemědělská 3, 613 00 Brno, Czech Republic

7 <sup>b</sup>Department of Zoology, Fisheries, Hydrobiology and Apiculture, Faculty of AgriSciences,  
8 Mendel University in Brno, Zemědělská 1, 613 00 Brno, Czech Republic

9 <sup>c</sup>UMR CNRS 6553 Ecobio, Université de Rennes 1, 263 Avenue du Gal Leclerc, 35042  
10 Rennes, France

11

12 Corresponding author\*

13 e-mail address: hamr.tom@seznam.cz (T. Hamřík)

14 **Abstract**

15 Seminatural grasslands are among the most valuable habitats for arthropod conservation in  
16 Central Europe. The abandonment of traditional farming has caused these areas to become  
17 overgrown and homogeneous, thereby resulting in loss of arthropod biodiversity. This  
18 traditional farming therefore needs to be complemented by active conservation management  
19 methods. An important question is whether small-scale conservation management may  
20 support arthropod diversity and habitat specialists inhabiting abandoned seminatural  
21 grasslands. We investigated the effects of mowing, prescribed burning, mechanical turf  
22 disturbance, and absence of active management on species richness, functional diversity and  
23 composition, conservation value, abundance of Red List species, and assemblage composition  
24 of spiders. The management methods were applied on small-scale (4 × 5 m) patches in  
25 protected xeric grassland. Spiders were sampled using pitfall traps and sweeping during 2017  
26 and 2018. A total of 11,634 specimens from 154 species were recorded, including many rare  
27 and threatened species. The temporal responses of spider assemblages to active conservation  
28 management depended on the particular group of spiders. Changes in assemblage composition  
29 and resulting shifts in the community weighted mean values were distinct in the second year  
30 of the experiment. Management had no significant effect on the species richness of ground-  
31 dwelling spiders. Mechanical turf disturbance shifted the assemblage composition of ground-  
32 dwelling spiders towards habitat generalist species. Vegetation-dwelling spiders had the  
33 highest species richness in unmanaged patches. Their functional diversity was lower in  
34 disturbed patches. Burnt patches supported species of conservation concern for both spider  
35 groups. Prescribed burning had positive results for most of the studied indicators.  
36 Nevertheless, mechanical turf disturbance had a rather negative effect on the spider  
37 assemblages. Our results indicate that even small, patch-like interventions with minimal costs

38 can considerably support the valuable spider assemblages of xeric grassland in the intensified  
39 landscape of Central Europe.

40 *Keywords:* Araneae; Central European xeric grasslands; Czech Republic; functional diversity;  
41 habitat management; prescribed burning

42

### 43 **1. Introduction**

44 In recent decades, huge worldwide biodiversity loss has been reported and intensified land  
45 use in agriculture has been a major driver of that loss (Tilman et al., 2001). During this time,  
46 humankind has changed ecosystems more profoundly and on a much larger scale than in any  
47 other period of human history. Due to changes in the land use complex, natural ecosystems  
48 have been converted to simplified, managed ecosystems (Tschardt et al., 2005; Henle et al.,  
49 2008). Species-rich meadows and pastures have been transformed into large blocks of fields  
50 for crop production or they have been abandoned and overgrown by scrublands or forests  
51 (Robinson and Sutherland, 2002; Stoate et al., 2009). These changes constitute the main  
52 reasons for loss of invertebrate biodiversity that plays an important role in the ecosystems'  
53 functioning (Ausden, 2007). Not only rare habitat specialists but common habitat generalists  
54 have also been greatly reduced in recent history (Conrad et al., 2006; Gaston and Fuller, 2007;  
55 Van Dyck et al., 2009; Cizek et al., 2012).

56 Grasslands have an important role within biodiversity protection in terms of providing  
57 valuable habitats and refuges for many organisms (Duelli, 1997). Most of the grasslands  
58 located in Central Europe are seminatural and are maintained by mowing and grazing regimes  
59 (Pech et al., 2015). The seminatural grasslands host high species richness of invertebrates  
60 (Poschlod and WallisDeVries, 2002; Duelli and Obrist, 2003; Littlewood et al., 2012).  
61 Socioeconomic changes at the end of the 20th century caused a reduction in livestock  
62 numbers and thus the abandonment of seminatural pastures (Isselstein et al., 2005).

63 Furthermore, traditional mowing is no longer suitable because of lower demand for forage in  
64 many areas (Valkó et al., 2014). The abandonment of traditional farming has resulted in shrub  
65 encroachment and litter accumulation that causes changes in xeric habitats and thus decreases  
66 their conservation value (Jongepierová et al., 2018; Valkó et al., 2018).

67 Spiders (Araneae) are present in high numbers across vegetation strata, from leaf litter to the  
68 top of the canopy. They are able to respond immediately to sudden changes in the  
69 environment (Samu et al., 2011; Lafage and Pétilon, 2014; Košulič et al., 2016). Variation in  
70 spider species composition between habitats is determined by the differences in vegetation  
71 structure and micro-climatic conditions such as humidity (Entling et al., 2007; Peres et al.,  
72 2007; Pinto-Leite et al., 2008; Buchholz, 2010; Nogueira and Pinto-da-Rocha, 2016). Distinct  
73 differences in spider communities can be obvious even on small patches (Samu et al., 1999;  
74 Pearce et al., 2004; Košulič et al., 2016). Most threatened and regionally extinct spider  
75 species are mainly linked to open habitats characterized by a lack of nutrients and proper  
76 conservation management (Tropek et al., 2010; Řezáč et al., 2015). Therefore, spiders are  
77 commonly used as ecological indicators in order to evaluate the nature conservation value of  
78 particular sites or to evaluate the effects of changes in habitat structure caused by  
79 conservation management (Maelfait and Hendrickx, 1998; Marc et al. 1999). Within the food  
80 chain, spiders have been found to significantly affect ecosystem processes through cascading  
81 effects, and numerous foraging strategies have been evolved in spiders that result in different  
82 functional roles within different ecosystems (Michalko and Pekár, 2016; Michalko et al.,  
83 2019). Knowledge of their foraging strategies and other functional traits is useful for  
84 assessing their functional diversity (Cardoso et al., 2011; Gallé and Batáry, 2019).

85 Compared to the traditional measure of species richness, functional diversity takes into  
86 account functional traits which may reveal changes in the community that would not be  
87 captured by species richness alone (Tilman 2001; Petchey and Gaston, 2006). Functional

88 traits can be any behavioral, morphological, phenological, and physiological characteristics of  
89 species that can be affected by environmental variables (Simon et al., 2016). Differences in  
90 species traits result in differences in their roles in the ecosystem, thus affecting the functions  
91 of the ecosystem (Ricotta and Moretti, 2011; Gallé and Batáry, 2019). Any changes in habitat  
92 that influence the distribution of these traits will consequently affect ecosystem functioning  
93 (Díaz and Cabido 2001). Therefore, evaluating the functional diversity rather than species  
94 richness, per se, provides more accurate insights into ecosystem processes (Díaz and Cabido,  
95 2001; Loreau, 2010).

96 Several studies have shown mowing and intensive disturbances such as grazing to have a  
97 detrimental effect on spider diversity (Gibson et al., 1992; Bell et al., 2001; Pétilion et al.,  
98 2007; Lafage and Pétilion, 2014; Řezáč and Heneberg, 2018; Řezáč et al., 2019). Intensive  
99 disturbances reduce vegetation structural complexity (Bell et al. 2011; Horváth et al. 2019;  
100 Lyons et al. 2018). Mowing immediately kills both the spiders and potential prey while  
101 altering vegetation structure in ways that result in rapid changes of microclimate conditions  
102 (Morris, 2000; Humbert et al., 2009, 2010; Cizek et al., 2012; Mazalová et al., 2015). On the  
103 one hand, prescribed burning reduce accumulated plant biomass and create open soil surface  
104 with sparse vegetation, thus provide microhabitats that support grassland specialist and  
105 xerothermic spider species (Polchaninova, 2015; Valkó et al., 2014; Valkó et al., 2016).  
106 Otherwise, fires at a large landscape scale cause significant spider mortality (Polchaninova,  
107 2015). Prescribed burning carried out on a small spatial scale adjacent to intact areas mitigates  
108 the detrimental effect of fire (Samu et al., 2010; Valkó et al., 2016).

109 In order to mitigate the negative effects of local management changes on invertebrates, it  
110 is important to implement an appropriate conservation management. The open landscape of  
111 Central Europe, however, lacks knowledge on the impacts of uncommon management  
112 methods like prescribed burning of xeric grasslands to spiders, when compared with the more

113 popular practices such as grazing and mowing (e.g., Valkó et al., 2016). Furthermore, most of  
114 the studies published to date were carried out in areas that were managed by mowing or  
115 affected by fire on large spatial scales (>1 ha) (e.g., Samu et al., 2010; Polchaninova, 2015;  
116 Torma et al., 2019). Little is known about the effect of small-scale conservation management  
117 methods on spider diversity. In addition, many studies do not take into account species of  
118 conservation concern that plays an important role in ecological and conservation studies (e.g.,  
119 Tropek et al., 2010; Košulič et al., 2014).

120 In order to fill the abovementioned knowledge gaps, our experiment compared the effects  
121 of different conservation management methods on spider assemblages on small spatial scales  
122 in xeric grassland. In particular, we evaluated (i) species richness, (ii) functional diversity and  
123 composition, (iii) conservation value, (iv) abundance of the Red List species, and (v)  
124 assemblage composition in xeric grassland. We hypothesized that (a) small-scale management  
125 methods will affect species richness, functional diversity and composition, and assemblage  
126 composition of spiders, (b) vegetation-dwelling spiders will be more sensitive to direct effects  
127 of management methods than ground-dwelling spiders, (c) small-scale prescribed burning will  
128 support grassland specialists, and (d) more intensive disturbances will have negative effects  
129 on both spider groups.

130

131

132 **2. Materials and methods**

133 *2.1 Research area and study sites*

134 The present research was carried out at Pláně Nature Monument, Czech Republic (Fig. 1).  
135 Pláně Nature Monument is a former pasture located in the South Moravian Region (Fig. 1;  
136 49.35139°N, 16.30361°E, altitude 400–458 m). The study area was traditionally used as a  
137 cow and goat pasture until half of the 1970s. After the abandonment of traditional farming,  
138 Pláně NM was preserved through conservation management like sheep grazing. Between  
139 2009 and 2016, the whole area of Pláně NM was left without any management (Matuška  
140 2016). The protected area covers 11.1 ha, where the most valuable parts include xeric  
141 grassland and southward, eastward, and westward oriented slopes with scattered rocks (1 ha).

142 The subsoil is characterized by orthogneiss with brown forest soils. This area belongs to  
143 the phytogeographical region of Mesophyticum along the border of the Czech Republic's  
144 South Moravian and Bohemian–Moravian Highlands. The mean annual precipitation is 579  
145 mm and the mean annual temperature is 8 °C (Hamřík and Košulič, 2019).

146 Three experimental sites consisting of xeric grasslands were selected within the protected  
147 area (Fig. 1): Site 1 (452–448 m a.s.l.), Site 2 (442–439 m a.s.l.), and Site 3 (435–421 m  
148 a.s.l.). There had been no management in the area since 2009, and thus the experimental sites  
149 had identical starting conditions. All sites are characterized by steep slopes with southerly  
150 (Site 1) or southeasterly (sites 2 and 3) orientations with sparse herbaceous vegetation  
151 consisting of *Festuca ovina*. In the small parts of the experimental sites, there are present  
152 scattered trees and shrubs including *Crataegus monogyna* and *Prunus spinosa*.

153 *2.2 Application of management methods*

154 Four management methods were applied in both years of the study (2017 and 2018).  
155 These were (i) mowing, (ii) prescribed burning, (iii) mechanical turf disturbance, and (iv) no-  
156 management. The mowing was carried out using a string trimmer in June (cutting height  
157 approx.. 6 cm). The mown material was raked and removed from the research plots.  
158 Prescribed burning was performed using torches with a flammable cloth and took place in  
159 early March, which was outside of the growing season and the time for main activity among  
160 invertebrates. Burning was done during a windless day when there had been frost on the  
161 previous night. Mechanical turf disturbance was carried out using a hoe, and the soil turf was  
162 vigorously disrupted. This management method can be considered as an imitation of very  
163 intensive livestock grazing and trampling. This was done at the beginning of April. In the  
164 unmanaged plots, no management actions were taken. Although we are aware that all the  
165 aforementioned active management methods are forms of disturbance, for simplicity's sake  
166 we will hereafter use the term "disturbance" in reference to the mechanical turf disturbance.  
167

168 *2.3 Study design and sampling*

169 The treatments were laid out in a checkerboard arrangement (four rows and four columns)  
170 of 4 × 5 m plots. Every method of management had four replications at each site. Therefore,  
171 16 experimental plots covering a total area of 572 m<sup>2</sup> (22 m × 26 m) were established per site  
172 (Fig. 1). The plots had 2 m unmanaged buffer zones between one another.

173 Pitfall trapping and sweep-netting were used as sampling methods to collect ground-  
174 dwelling and vegetation-dwelling spiders, respectively. Pitfall traps made of 500 ml plastic  
175 cups (diameter 9 cm, depth 15 cm) without funnels and roofs were sank into the soil surface  
176 and filled with a 3–4% solution of formaldehyde and detergent as a killing and preserving  
177 agent. Two pitfall traps were placed 1 m apart from the center of each plot. In total, 32 pitfall

178 traps were placed in each site. Sweeping was carried out inside each plot (20 sweeps along a  
179 line per plot) using a sweep net (40 cm in diameter) only during a sunny, windless day with a  
180 minimum of 17 °C (between 10:00 and 17:30).

181 In 2017, the pitfall traps were installed on 7 May and deactivated on 15 October. In 2018,  
182 the traps were installed on 19 April and deactivated on 21 September. Traps were emptied  
183 monthly. Sweeping was carried out on the following dates: 28 May, 14 July, and 17 August in  
184 2017; 21 May, 23 July, and 29 September in 2018. The sampled material was preserved in 70  
185 % ethanol, and stored in the laboratory at Mendel University in Brno, Faculty of Forestry and  
186 Wood Technology.

187

#### 188 *2.4 Environmental variables evaluation*

189 Spiders are highly sensitive to alterations in their physical environments (Buchholz,  
190 2010). In order to evaluate the influence of management methods on the environmental  
191 characteristics, we measured three simple environmental variables during July 2018: a)  
192 vegetation cover, b) vegetation height, and c) proportion of bare soil. The vegetation cover  
193 was estimated as a percentage (precision = 1%) of vegetation, and the height was measured  
194 using a tape measure (precision = 0.1 cm) at five randomly selected spots (mean for the plot  
195 was then calculated) within each of the 16 experimental plots. The proportion of bare soil was  
196 estimated by eye as the percentage (precision = 1%) of bare patches without vegetation on the  
197 surface of the research plots. Values of environmental variables in each study plot are shown  
198 in Table A.1.

199 *2.5 Species classification and ecological traits*

200 Adult and distinguishable subadult specimens were classified to species level using the  
201 identification keys of Heimer and Nentwig (1991) and of Nentwig et al. (2019). Nomenclature  
202 follows the latest version of World Spider Catalog (2019).

203 For the functional diversity and composition, three functional traits were chosen: body  
204 size that determines spider prey size (Gallé and Batáry, 2019), hunting strategy that reflects  
205 how spiders hunt their prey (Wise 1995; Jocqué and Dippenaar-Schoeman, 2006; Herberstein,  
206 2011), and humidity preference that describes microhabitat (Buchar and Růžička, 2002).

207 Body size was taken as a continuous variable and defined as the mean body length in mm  
208 averaged over females taken from Nentwig et al. (2019). Hunting strategies were categorized  
209 as hunters and web-builders (coded as 1 and 2, respectively) in accordance with Cardoso et al.  
210 (2011). Finally, each spider species was classified according to its humidity preference on the  
211 ordinal scale: 1 – very dry, 2 – dry, 3 – semi-humid, 4 – humid, 5 – very humid (Buchar and  
212 Růžička, 2002). Species belonging to more categories received an average value. If the  
213 average value was between two categories, then the value was rounded off according to the  
214 ecological data obtained from Arachnobase of the Czech Spiders (Kasal and Kaláb, 2020).

215 The conservation value of spider assemblages was evaluated based on the Red List of  
216 Czech Spiders conservation status (Řezáč et al., 2015), which are as follows: LC (least  
217 concern), VU (vulnerable), EN (endangered), and CR (critically endangered). The presence of  
218 individual species in a sample was weighted by ranked values according to the conservation  
219 status in the Red List: LC – 1, VU – 2, EN – 3, CR – 4. As a result, each individual patch was  
220 given a score that was based on a sum representing the conservation value of those species  
221 present. This approach has been used in many ecological and conservation studies (see  
222 Tropek et al., 2010, 2014; Košulič et al., 2014, 2016). In addition to the conservation value,  
223 the abundance of the Red List species in the given categories (LC–CR) was evaluated.

224 In order to evaluate the effect of the different management methods on spider assemblages  
225 in terms of their rarity, every species was classified according to degree of rareness in the  
226 Czech Republic (Buchar and Růžička, 2002) as follows: VA (very abundant), A (abundant), S  
227 (scarce), R (rare), or VR (very rare).

228 Ecological characteristics and functional traits of recorded spider species are shown in Table  
229 A.2.

230

### 231 *2.6 Statistical analyses*

232 Functional diversity was evaluated using Rao index that is based on the quadratic entropy  
233 of Rao (1982). This coefficient was proposed as a good candidate for an efficient index of  
234 functional diversity (Ricotta 2005; Botta-Dukat, 2005; Pavoine and Dolédec, 2005; Lepš et  
235 al., 2006). The Rao index (RaoQ) is a generalized form of the Simpson index of diversity  
236 representing a probability that two randomly picked individuals within a community are  
237 functionally different (Lepš et al., 2006). Furthermore, the community-weighted mean  
238 (CWM) was analyzed. CWM summarizes mean trait value weighted by the relative  
239 abundances of each species in a given community (Garnier et al., 2004). Since CWM  
240 represents the ‘mean’ of each trait and RaoQ represents the ‘dispersion’ of functional traits  
241 within a given community, these two measures provide complementary insight into the  
242 relationship between community structure and ecosystem functioning (Ricotta et al., 2011).

243 The Rao index (RaoQ) was computed using “melodic” function (de Bello et al., 2016) and  
244 the CWM was calculated using “functcomp” function in the “FD” package (Laliberté et al.,  
245 2014).

246 Comparison of species richness, abundance of Red List species, and conservation value  
247 among the four management methods were analyzed using generalized linear mixed models  
248 (GLMMs) with Poisson distribution and log link (GLMMs-p) in the package “lme4” (Bates et

249 al., 2020). Generalized linear mixed models (GLMMs) allow analyzing data with non-normal  
250 distribution and the presence of a random effect that quantifies variations between units  
251 (Bolker et al., 2009). In cases where overdispersion appeared, GLMMs with a negative  
252 binomial distribution (GLMMs-nb) within the “MASS” package were used (Ripley et al.,  
253 2020). This model is an alternative to the Poisson model with overdispersion (Pekár and  
254 Brabec, 2009). Dependent variables were species richness, abundance of Red List Species,  
255 and conservation value. The explanatory variables were the management methods, and the  
256 patch position was used as a random effect. The effects of management methods on functional  
257 diversity and composition were tested by linear mixed models (LMMs). This method is  
258 intended for analysis of normally distributed data that include a random effect (Bolker et al.,  
259 2009). Linear mixed models with RaoQ and CWM values as the dependent variables were  
260 used with management methods as explanatory variables, and patch position as a random  
261 effect. Post-hoc comparisons among management methods were conducted with “glht”  
262 function in the “multcomp” package (Hothorn et al., 2020).

263 Spearman rank correlation was used to analyze associations between environmental  
264 variables. A strong negative correlation was found between vegetation cover and proportion  
265 of bare soil (Spearman correlation coefficient;  $\rho = -0.906$ ,  $P < 0.001$ ). Thus, for further  
266 analysis, the proportion of bare soil was excluded. Effect of management methods on the  
267 vegetation cover and height, and species composition were examined by partial redundancy  
268 analysis (RDA) and canonical correspondence analysis (CCA), where the site served as a  
269 covariate. In order to exclude accidental species, only those species with  $> 3$  individuals were  
270 selected for the analysis. RDA was used if the initial detrended correspondence analysis  
271 revealed that the longest gradient was shorter than 3, while CCA was used if the longest  
272 gradient was greater than 4 (only vegetation-dwelling spiders in 2017) (Šmilauer and Lepš,  
273 2014). For the analysis of differences in species composition in relation to the management

274 methods, the data were  $\log(y + 1)$  transformed. The significance of the effect of management  
275 methods was tested using Monte Carlo permutation tests with 999 permutations. Permutations  
276 were restricted within the blocks represented by the sites. To investigate the effect of  
277 management on the degree of rareness, the degree of rareness of each species was passively  
278 projected into RDA biplots.

279 Univariate analyses were performed using R (R Development Core Team, 2018), and  
280 multivariate analyses were evaluated using CANOCO 5 (ter Braak and Šmilauer, 2012). Two  
281 traps collected in each plot throughout each year were pooled together. Subsequently, the data  
282 from both years and the two collection methods were analyzed separately.

283 To detect characteristic species of each management method, the indicator value analysis  
284 (IndVal; Dufrene and Legendre, 1997) was performed using “indval” function in the “labdsv”  
285 package (Roberts, 2019).

286

287

288 **3. Results**

289 *3.1 Overview*

290 A total of 11,634 specimens representing 25 families, 88 genera, and 154 species was  
291 collected and identified (Table A.2). Overall, 10,532 specimens (91%) including 133 species  
292 were sampled by pitfall traps and 1,102 specimens (9%) including 49 species were sampled  
293 by sweeping. The most abundant species collected by pitfall traps were *Alopecosa cuneata* (n  
294 = 2,603) and *Pardosa palustris* (n = 1,884) from family Lycosidae. From sweeping, the most  
295 abundant species were *Mangora acalypha* (n = 537) and *Aculepeira ceropegia* (n = 135) from  
296 family Araneidae. Generally, 35 mainly xerothermic species listed on the Red List of Czech  
297 spiders were found (Řezáč et al. 2015). We have discovered a valuable araneofauna  
298 composition encompassing 18 % of all species within the Czech Republic. Detailed faunistic  
299 information, including the list of species with their ecological characteristics, is available in  
300 Hamřík and Košulič (2019).

301 Three significant indicator species of ground-dwelling spiders were found (see Table A.3).  
302 *Agroeca cuprea*, species that inhabits broad range of open habitats, was associated with  
303 disturbed (2017), and unmanaged patches (2018). *Haplodrassus signifer*, a common habitat  
304 generalist, was found as an indicator species in mown patches (2017); and *Xerolycosa*  
305 *miniata*, typical for habitats with barren surface, was identified as a characteristic species in  
306 disturbed patches (2018).

307 For vegetation-dwelling spiders, three significant indicator species were also identified (see  
308 Table A.3): *Mangora acalypha* and *Aculepeira ceropegia* (2017); and *Mangora acalypha* and  
309 *Argiope bruennichi* (2018). These typical grassland orb-web spiders were found as an  
310 indicator species in unmanaged patches.

311

312 *3.2 Effect of management on environmental characteristics*

313 The redundancy analysis (RDA) model revealed a significant effect of management  
314 method on vegetation cover and height (RDA, pseudo-F = 54.0, P = 0.001; Fig. 2). The first  
315 two constrained axes explained 79.4% of the variability, while the other two unconstrained  
316 axes explained 16.3% and 4.3% of the variability in the environmental variables. Disturbed  
317 patches were characterized by a high proportion of bare substrate and relatively tall  
318 vegetation, while unmanaged patches were characterized by tall vegetation with extensive  
319 cover. Burnt patches contained taller vegetation generally, but the vegetation cover was not so  
320 extensive as was that in the unmanaged patches. This management method creates patches  
321 with sparse vegetation. Mown patches had very low vegetation with extensive coverage.

322

323 *3.3 Effect of management on ground-dwelling spiders in 2017*

324 The management method did not have a significant effect on the species richness  
325 (GLMM-nb,  $\chi^2_3 = 0.2$ , P = 0.976), functional diversity (LMM,  $\chi^2_3 = 2.9$ , P = 0.406), CWM of  
326 body size (LMM,  $\chi^2_3 = 5.1$ , P = 0.166), CWM of hunting strategy (LMM,  $\chi^2_3 = 2.0$ , P =  
327 0.580), conservation value (GLMM-p,  $\chi^2_3 = 2.5$ , P = 0.477), and abundance of Red List  
328 species (GLMM-nb,  $\chi^2_3 = 0.3$ , P = 0.965; Tables 1 and A.4).

329 The CWM of humidity differed significantly among management methods (LMM,  $\chi^2_3 =$   
330 8.5, P = 0.036; Tables 1 and A.4). Disturbed patches had significantly higher values than  
331 unmanaged patches, but comparable to that of burnt and mown patches. There was no  
332 significant difference in the CWM of humidity between unmanaged, burnt, and mown patches  
333 (Fig. 3a).

334 Management method did not significantly affect the spider assemblage composition  
335 (RDA, pseudo-F = 1.1, P = 0.316). The first three (constrained) axes explained 7.0 % and the  
336 remaining (unconstrained) axes explained 16.4 % of the variability in the spider species data.

337

338 *3.4 Effect of management on ground-dwelling spiders in 2018*

339 The species richness did not differ significantly among management methods (GLMM-p,  
340  $\chi^2_3 = 2.9$ ,  $P = 0.406$ ; Tables 1 and A.5).

341 The management method did have a significant effect on functional diversity (RaoQ)  
342 (LMM,  $\chi^2_3 = 10.2$ ,  $P = 0.017$ ; Tables 1 and A.5). The functional diversity in disturbed patches  
343 was significantly greater in comparison to the unmanaged patches, but it did not differ  
344 significantly from that in burnt and mown patches. There was no significant difference in  
345 functional diversity between unmanaged, burnt, and mown patches (Fig. 4a). There was no  
346 significant difference in CWM of body size (LMM,  $\chi^2_3 = 2.3$ ,  $P = 0.522$ ), and CWM of  
347 hunting strategy (LMM,  $\chi^2_3 = 0.2$ ,  $P = 0.974$ ; Tables 1 and A.5) among management methods.  
348 The management method had a significant effect on the CWM of humidity (LMM,  $\chi^2_3 = 14.5$ ,  
349  $P = 0.002$ ; Table 1 and A.5). The CWM of humidity was highest in disturbed patches. The  
350 burnt, mown, and unmanaged patches had comparable CWM of humidity (Fig. 3b).

351 Conservation value differed significantly among management methods (GLMM-p,  $\chi^2_3 =$   
352  $11.5$ ,  $P = 0.009$ ; Tables 1 and A.5). Burnt patches had significantly higher values than  
353 disturbed patches, but it did not differ significantly from that in unmanaged and mown  
354 patches. Mown patches had significantly higher conservation value than did disturbed  
355 patches, but their values did not differ significantly from those of unmanaged. There was no  
356 significant difference in conservation value between disturbed and unmanaged patches (Fig.  
357 4b). The abundance of Red List species varied significantly among management methods  
358 (GLMM-nb,  $\chi^2_3 = 17.4$ ,  $P < 0.001$ ; Tables 1 and A.5). The burnt and mown patches hosted  
359 significantly greater abundance of Red List species than did disturbed patches, but their  
360 abundance did not differ significantly from those of unmanaged patches. The burnt and

361 mown patches had comparable abundance of Red List species. Unmanaged patches had a  
362 comparable abundance of Red List species to that of disturbed patches (Fig. 4c).

363 Management method significantly affected the spider assemblage composition (RDA,  
364 pseudo-F = 1.8, P = 0.003; Fig. 5a). The first three (constrained) axes explained 11.1% and  
365 the remaining (unconstrained) axes explained 13.9% of the variability in the spider species  
366 data. Burnt patches were preferred by rare and threatened xerothermic species requiring early  
367 stages of succession. Patches created by other management methods were preferred mainly  
368 by very abundant species and only rarely by scarce species (Fig. 5b).

369

### 370 *3.5 Effect of management on vegetation-dwelling spiders in 2017*

371 Species richness differed significantly among management methods (GLMM-p,  $\chi^2_3 =$   
372 23.4, P < 0.001; Tables 1 and A.6). Species richness was highest in unmanaged patches. The  
373 burnt, disturbed, and mown patches had comparable species richness (Fig. 6a).

374 The functional diversity (RaoQ) differed significantly by management method (LMM,  $\chi^2_3$   
375 = 12.7, P = 0.005; Tables 1 and A.6). Functional diversity in unmanaged patches was  
376 significantly greater in comparison to the disturbed patches, but it did not differ from that of  
377 in burnt and mown patches. The burnt, disturbed and mown patches had comparable  
378 functional diversity (Fig. 6b). The management method did not have a significant effect on  
379 CWM of body size (LMM,  $\chi^2_3 = 3.1$ , P = 0.377), CWM of humidity (LMM,  $\chi^2_3 = 7.0$ , P =  
380 0.073), and CWM of hunting strategy (LMM,  $\chi^2_3 = 6.6$ , P = 0.087; Tables 1 and A.6).

381 There was a significant difference in conservation value among management methods  
382 (GLMM-nb,  $\chi^2_3 = 9.4$ , P = 0.024; Tables 1 and A.6). Unmanaged patches had significantly  
383 higher conservation value than did disturbed patches, but it did not differ from that of in burnt  
384 and mown patches. The conservation value in burnt, disturbed, and mown patches was  
385 comparable (Fig. 6c). Management method had a significant effect on the abundance of Red

386 List species (GLMM-p,  $\chi^2_3 = 12.1$ ,  $P = 0.007$ ; Tables 1 and A.6). In unmanaged patches, there  
387 was a significantly greater abundance of Red List species than in mown and disturbed  
388 patches. Burnt patches hosted a significantly greater abundance of Red List species than did  
389 disturbed patches, but their abundance was not significantly different from that of mown and  
390 unmanaged patches. The abundance of Red List species in mown patches was comparable to  
391 that in disturbed patches (Fig. 6d).

392 Management method did not significantly affect the spider assemblage composition  
393 (CCA, pseudo-F = 1.1,  $P = 0.277$ ). The first three (constrained) axes explained 7.3% and the  
394 remaining (unconstrained) axes explained 16.8% of the variability in the spider species data.

395

### 396 *3.6 Effect of management on vegetation-dwelling spiders in 2018*

397 The management method did not have a significant effect on the species richness  
398 (GLMM-p,  $\chi^2_3 = 5.9$ ,  $P = 0.118$ ), functional diversity (LMM,  $\chi^2_3 = 4.7$ ,  $P = 0.198$ ),  
399 conservation value (GLMM-p,  $\chi^2_3 = 6.4$ ,  $P = 0.095$ ), and abundance of Red List species  
400 (GLMM-nb,  $\chi^2_3 = 3.4$ ,  $P = 0.333$ ; Tables 1 and A.7).

401 There was a significant difference in CWM of body size among management methods  
402 (LMM,  $\chi^2_3 = 20.4$ ,  $P < 0.001$ ; Tables 1 and A.7). Unmanaged and disturbed patches had  
403 significantly higher values than did burnt and mown patches. The values in burnt and mown  
404 patches were comparable (Fig. 7a). The CWM of humidity differed significantly by  
405 management method (LMM,  $\chi^2_3 = 28.3$ ,  $P < 0.001$ ; Tables 1 and A.7). In mown patches, there  
406 was the lowest CWM of humidity. The values in unmanaged, burnt and disturbed patches  
407 were comparable (Fig. 7b). The management method did not have a significant effect on  
408 CWM of hunting strategy (LMM,  $\chi^2_3 = 7.3$ ,  $P = 0.062$ ; Tables 1 and A.7).

409 Management methods had significant effect on spider assemblage composition (RDA,  
410 pseudo-F = 3.5,  $P = 0.001$ ; Fig. 8a). The first three (constrained) axes explained 20.1% and

411 the remaining (unconstrained) axes explained 25.9% of the variability in the spider species  
412 data. Burnt patches were preferred by scarce *Evarcha laetabunda* that require sparse  
413 vegetation in xerothermic habitats. The presence of the species found in burnt patches also  
414 correlated positively with that of species inclining toward unmanaged patches (Fig. 8b).

415

416

417

418 **4. Discussion**

419 The study location had gone 7 years without a management regime, and this allowed us to  
420 apply selected management methods into plots with advanced stages of succession and  
421 accumulated plant biomass. Thus, all experimental plots were in an equivalent state at the  
422 start of the experiment. This provided an opportunity to study the responses of spider  
423 assemblages to ecological succession mediated by various conservation management methods  
424 in small experimental plots. Furthermore, due to the relatively high number of Red List  
425 species (N = 35), we were able to determine which management method supported the  
426 formation of microhabitats providing environmental conditions suitable for these rare and  
427 threatened species of spiders.

428

429 *4.1 Effect on environmental variables*

430 The individual management methods created differences in vegetation structural  
431 complexity, and this indirectly affected studied indicators. Unmanaged patches were  
432 characterized by tall, dense vegetation with accumulated biomass. These habitats include wet  
433 microhabitats and plant litter that together create habitat conditions providing a supply of food  
434 sources and shelters for invertebrates (Lepš, 1999). Contrary to unmanaged patches, disturbed  
435 patches contained a large proportion of bare substrate and very sparse vegetation cover. These  
436 patches can be compared to habitats maintained by very intensive livestock grazing and  
437 trampling. Mown patches contained low-height vegetation with relatively extensive cover.  
438 Mowing also causes changes in the temperature and humidity of a given habitat (Lepš, 1999).  
439 Prescribed burning created patches with early stages of succession, but there was minimal  
440 vegetation removal. Therefore, the burnt patches were characterized by microhabitats with  
441 taller vegetation and bare soil. In burnt habitats, high plant diversity and large proportion of  
442 patches with flowering vegetation are present (Lunt, 1993; Valkó et al., 2016). Less

443 vegetation cover and lower height likely allow greater light access, and that supports  
444 microhabitats suitable for germination of many plant species (Valkó et al., 2016).

445

#### 446 4.2 Effect on ground-dwelling spiders

447 In the first year, the experiment revealed no differences in spider assemblages across most  
448 response variables, the exception was CWM of humidity.

449 The species richness was comparable across individual management methods. In a study  
450 initiated 2.5 years after an extensive fire, Samu et al. (2010) also found comparable species  
451 richness in both large burnt blocks (1 ha) and unburnt blocks. In several studies, the species  
452 richness of ground-dwelling spiders has been found to be slightly higher in unmanaged  
453 grasslands, and this has been attributed to the greater presence of prey and habitat  
454 heterogeneity (e.g., Cizek et al., 2012).

455 Most species of conservation concern preferred the burnt patches. This was confirmed  
456 also by high conservation values and abundances of the Red List species. The burnt patches  
457 were preferred by rare xerothermic species (e.g., *Alopecosa striatipes* and *Thanatus*  
458 *arenarius*). These species require bare soil with sparse vegetation (Buchar and Růžička,  
459 2002). This corresponds with the findings of a large-scale study (1 ha blocks) wherein  
460 xerothermic species such as *Arctosa perita* and *Alopecosa cursor* were found more  
461 abundantly in burnt patches (Samu et al., 2010). Open habitat specialists are more abundant in  
462 medium-grown swards than in full-grown and recently mown vegetation (Mazalová et al.,  
463 2015). Polchaninova (2015) found that spider assemblages alter in the predominance of  
464 xerothermic species such as *Drassodes pubescens*, *Thanatus arenarius*, and *Zelotes electus* in  
465 habitats affected by fires within 3 years after the fires. Our results already revealed the same  
466 pattern in the second year of the experiment due to the small-scale approach and presence of  
467 source habitats in the vicinity. It must be noted that species inclining towards unmanaged

468 patches were mostly widespread generalists, as confirmed also by Cizek et al. (2012). In  
469 disturbed patches, the conservation value and abundance of the Red List species were low. In  
470 the first year of the experiment, the disturbed patches had higher CWM of humidity than  
471 unmanaged patches. In the second year of the experiment, the CWM of disturbed patches  
472 became highest among management methods. Disturbed patches were only preferred by a few  
473 generalists (e.g., *Cicurina cicur*, *Pisaura mirabilis*). Thus, this pattern was probably caused  
474 by assemblage composition changes towards habitat generalists in disturbed patches, whereas  
475 xerothermic grassland specialists preferred less intensively managed patches. It was also  
476 supported by the detection of *Agroeca cuprea* as the indicator species in disturbed patches in  
477 the first year of the experiment but in unmanaged patches in the second year. Intensive soil  
478 disturbances have a negative effect on grassland specialist spider species (Batáry et al., 2008;  
479 Pétilion et al., 2007; Horváth et al., 2019). Disturbed patches had also greater functional  
480 diversity than did unmanaged patches. In early successional stages, rapid colonization from  
481 surrounding habitats and mobility from adjacent plots bring a number of species with unique  
482 combinations of functional traits (Hodeček et al., 2015). In our study, it was obviously a result  
483 of colonization by various habitat generalists. Mean spider body size and proportion of web-  
484 builders were comparable among management methods. However, previous studies showed  
485 that ground-dwelling spiders with large body size favor less intensively managed patches  
486 (Bell et al., 2001; Birkhofer et al. 2015). Our result may be explained by the low presence of  
487 web-builders captured by pitfall traps and the dominance of ground-dwelling spiders with  
488 comparable body size.

489 4.3 Effect on vegetation-dwelling spiders

490 In contrast to ground-dwelling spiders, the vegetation-dwelling spiders already responded  
491 to the management in the first year of the experiment. Management causes immediate changes  
492 in vegetation structure and direct mortality of vegetation-dwelling arthropods (e.g., Torma et  
493 al., 2019). Nonetheless, assemblage composition changes and consequent shifts in CWM  
494 values were observed in the second year of the experiment.

495 The vegetation-dwelling spiders had the highest species richness in unmanaged patches.  
496 The web-building spiders are limited by the presence of vegetation structural complexity that  
497 allows them to attach their webs (Greenstone, 1984; Mcnett and Rypstra, 2000; Jiménez-  
498 Valverde and Lobo, 2007). Denser vegetation in unmanaged patches also provides shelters  
499 and microhabitats with suitable conditions for hibernation and female cocoon formation (De  
500 Keer and Maelfait, 1987; De Keer et al., 1989; Bayram and Luff, 1993). It is evident that the  
501 vegetation-dwelling arthropods are negatively affected by intensive disturbances (Torma et  
502 al., 2019). Conversely, less-intensive management creates more complex communities with  
503 niches suitable for vegetation-dwelling spiders (Bell et al., 2001).

504 Prescribed burning and no-management had a positive effect on the conservation value  
505 and abundance of Red List species. Conversely, disturbed patches showed significantly lower  
506 abundance of Red List species. Intensive management in xeric grasslands reduces the  
507 proportions of rare and threatened web-building spiders (Gibson et al., 1992). Mown patches  
508 hosted lower abundance of Red List species than unmanaged patches. Vegetation-dwelling  
509 spiders require heterogeneous vegetation structure that is low due to the direct effect of  
510 mowing (Bucher and Entling, 2011). *Evarcha laetabunda*, which belongs to rare and  
511 threatened species of spiders, were inclined to burnt and unmanaged patches. This species is  
512 dependent on the presence of vegetation, which is lacking in disturbed and mown patches.  
513 The negative impact of burning in terms of immediate mortality and habitat destruction is

514 generally known for large-scale fires (e.g., Nemkov and Sapiga, 2010; Polchaninova, 2015)  
515 but not for application on a small spatial scale as in our study.

516 The disturbed patches had lower functional diversity than unmanaged patches. These  
517 patches were populated by functionally more similar species, which led to the lower  
518 functional diversity. Disturbed and unmanaged patches hosted species with larger mean body  
519 size than burnt and mown patches. Also, the ordination analysis shows the preference of  
520 dominant large spider species *Aculepeira ceropegia* for unmanaged and disturbed patches.  
521 This species can be found in both intact and anthropogenically disturbed habitats (Buchar and  
522 Růžička, 2002). Furthermore, *Argiope bruennichi* was found as indicator species in  
523 unmanaged patches. However, the occurrence of larger-sized species in disturbed patches is in  
524 contrast to the previous finding that revealed a dominance of small species of web-building  
525 spiders in heavily grazed grasslands (Gibson et al., 1992). It may be due to changes in prey  
526 availability of Aphididae, Psylloidea, and Collembola which are among the most important  
527 prey of small-sized spiders (Alderweireldt, 1994; Lawrence and Wise, 2000; Pekár et al.,  
528 2015). Grassland management affects habitat structure and these changes can cause  
529 significant mortality of these insect groups (Purvis and Curry, 1981; Ausden, 2007).  
530 Furthermore, intraguild predation is typical for habitats with low structural complexity  
531 (Schmidt and Rypstra, 2010). However, complex-structured habitats provide a refuge for  
532 predators from intraguild predation (Finke and Denno, 2006). Therefore, we assume that the  
533 exclusion of potential small-sized prey or intraguild predation could lead to the exclusion of  
534 small spider species in disturbed patches. In conjunction with other studies (Simons et al.,  
535 2016; Torma et al., 2019) our results also confirm that mechanical mowing can harm large  
536 species of arthropods. Furthermore, larger species that have a longer life cycle can be easily  
537 disrupted by management methods such as mowing (Simons et al., 2006). Small species from  
538 the family Linyphiidae, such as *Agyneta rurestris* and *Tenuiphantes flavipes*, had a tendency

539 towards burnt patches. Increased bare soil in burnt patches is colonized by pioneer linyphiid  
540 species (Merrett et al., 1976). We can conclude that while mowing harmed large web-building  
541 spiders, prescribed burning offered suitable microhabitats for small spider species. Mowing  
542 causes significant changes in microclimatic conditions that result in lower humidity (Bell et  
543 al., 2001). The lowest CWM of humidity in mown patches may be the result of unsuitable  
544 conditions for species with high humidity preferences.

545

#### 546 *4.4 Suggestions for management implications*

547 Based on our results, the no-management and small-scale prescribed burning are suitable  
548 conservation methods to maintain spider diversity. Therefore, their combination also appears  
549 to be appropriate. Prescribed burning creates patches that support species of conservation  
550 concern (Nemkov and Sapiga, 2010; Polchaninova, 2015; Valkó et al., 2016) while  
551 unmanaged patches offer habitats with high abundance of prey, shelters, and suitable  
552 microclimate (Bell et al., 2001; Cattin et al., 2003). Our experiment shows that small-scale  
553 prescribed burning is a management method that achieves comparable or better results than  
554 does traditional mowing in xeric grassland. Homogeneous mowing without hay-removal  
555 usually accumulates biomass (e.g., Noordijk et al., 2010; Cizek et al., 2013), which  
556 conversely can be reduced by prescribed burning (Niwa and Peck, 2002). On the other hand,  
557 there are studies showing adverse effects of burning on spider diversity (e.g., Polchaninova,  
558 2015). To avoid the adverse effect, it is important to carry out the prescribed burning during  
559 the dormant season on a small scale within patches adjacent to intact areas (Prishutov and  
560 Arzanova, 2008; Samu et al., 2010; Valkó et al., 2016). Unlike the mown patches, the burnt  
561 patches have relatively tall vegetation, which offers a refuge for vegetation-dwelling spiders.  
562 Our experiment is consistent with previous studies showing that intensive disturbance of  
563 habitats such as by soil disruption and/or intensive grazing has negative impacts on arthropods

564 (De Keer and Maelfait, 1988, Gibson et al., 1992; Belsky, 1992; Pétilion et al., 2007; Horváth  
565 et al., 2009; Van Klink et al., 2015). In accordance with our results, we suggest that one-time  
566 prescribed burning could be performed as a means of restoring recently homogeneous xeric  
567 grasslands that had been abandoned from former traditional farming (Reidsma et al., 2006). It  
568 is important to note that burning increases the dominance of expansive *Calamagrostis*  
569 *epigejos*, thus prescribed burning is inappropriate in grasslands where this plant species is  
570 present (Házi et al., 2011; Deák et al., 2014). In such context, it is more appropriate to  
571 combine prescribed burning with mowing in order to reduce patches with expansive plants  
572 such as *Calamagrostis epigejos* that in general threaten biodiversity of most xeric grasslands  
573 in Europe (Henning et al., 2017). Prescribed burning on a small spatial scale, together with  
574 mowing, has the potential to enhance the habitat heterogeneity, thus supporting the overall  
575 biodiversity (Ausden, 2007). It is necessary to leave intact some small parts within managed  
576 grasslands to maintain a habitat mosaic at various stages of succession in order to support  
577 arthropods for which regular disturbances are unsuitable (Batáry et al., 2010). We suggest  
578 diversification of conservation management methods to be conducted in parts of a given area  
579 throughout the year. However, it should be noted that in some regions, returning to traditional  
580 farming in grasslands is not feasible, or the implementation of such conservation management  
581 is high-cost (Poschlod and WallisDeVries, 2002; Valkó et al., 2014). For these reasons, we  
582 suggest that small-scale prescribed burning can be a suitable low-cost substitute for  
583 uneconomical management methods such as mowing. We also assume that such debatable  
584 conservation practices as small-scale prescribed burning in protected grasslands should be  
585 seen not as a threats to the environment, but as acceptable substitutes for traditional farming  
586 methods that are no longer in use, but which also have historically helped sustain a high level  
587 of biodiversity in the European landscape (Ausden, 2007).

588

589 **5. Conclusions**

590 Our results show that patches created by small-scale conservation management may  
591 enhance spider diversity in xeric grasslands. We observed high spider species turnover among  
592 small-scale (4 × 5 m) patches created by various management methods. These changes in  
593 such notably small areas subject to various management methods highlight the importance of  
594 habitat complexity even on limited patches of grassland habitats. Our results indicate that  
595 prescribed burning had a positive effect on the species of conservation concern. On the other  
596 hand, unmanaged patches showed a positive effect on species inhabiting the vegetation. Thus,  
597 it seems that no-management and prescribed burning are most appropriate for the  
598 management of xeric grasslands to support spider occurrence. Surprisingly, mowing had in  
599 some cases unfavorable values comparable to those when mechanical turf disturbance was  
600 carried out. Based on these results, properly conducted prescribed burning can achieve better  
601 results than does more costly mowing. Therefore, it could be a cost-effective alternative  
602 management tool for conserving grassland biodiversity. We suggest that small-scale  
603 management methods may be useful for supporting biodiversity in xeric grasslands within  
604 landscapes under intense human land use that formerly had been maintained by traditional  
605 means. Finally, the results of this study may be useful to conservation organizations and  
606 government institutions in considering whether prescribed burning is an effective  
607 conservation method for use on European xeric grasslands. Nevertheless, prescribed burning  
608 needs to be implemented by well-trained conservation managers who have proper knowledge  
609 of the given habitats, with the assistance of firefighters to avoid possible negative effects.

610

611

612

613

614

615 **Acknowledgements**

616 The authors would like to thank Radek Michalko, Jan Šipoš, Tomáš Kudláček, and Daniel  
617 Volařík for help with statistical analysis; Igor Malenovský and Julien Pétilion for useful  
618 comments and suggestions; Denise Paulina V. Doble for the English proofreading; and Petr  
619 Dolejš for determination and revision of spider taxa that were difficult to identify. We  
620 likewise extend our warm appreciation to Jaroslav Knotek (South Moravian Regional  
621 Authority), David Lacina (Kuřimská Nová Ves), Libor Sedlák and AGRO Skalka for enabling  
622 us to carry out our experiments at Pláně Nature Monument; friends and family and the  
623 members of ZO ČSOP Eresus and ZO ČSOP Náměšťské rybníky for their assistance during  
624 fieldwork; and Kamila Surovcová and Luboš Staněk for sorting materials. We also thank the  
625 academic editor Santiago Poggio and three anonymous reviewers for their useful comments  
626 and correction of the manuscript. The study was financially supported by the Specific  
627 University Research Fund of the Faculty of Forestry and Wood Technology, Mendel  
628 University in Brno (LDF\_PSV\_2017004/2017).

629

630

631 **Appendices**

632 **Table A.1.** Values of physical environmental characteristics in each study plot.

633 **Table A.2.** List of recorded species with ecological characteristics and functional traits.

634 Conservation status: CR (critically endangered), EN (endangered), VU (vulnerable), LC (least  
635 concern); Degree of rareness: VA (very abundant), A (abundant), S (scarce), R (rare), VR  
636 (very rare); Hunting strategy: active hunters (1), web-builders (2); Humidity preference: very  
637 dry (1), dry (2), semi-humid (3), humid (4), very humid (5); Body size (mm).

638 **Table A.3.** List of indicator species of ground-dwelling and vegetation-dwelling spiders in  
639 disturbed, mown and unmanaged patches.

640 **Table A.4.** Effect of prescribed burning (B), mechanical turf disturbance (D), mowing (M),  
641 no-management (N) on species richness, abundance of Red List species, conservation value,  
642 functional diversity and composition of ground-dwelling spiders in 2017. Effects of  
643 management methods were tested using GLMMs and LMMs. Significant results are marked  
644 with boldface.

645 **Table A.5.** Effect of prescribed burning (B), mechanical turf disturbance (D), mowing (M),  
646 no-management (N) on species richness, abundance of Red List species, conservation value,  
647 functional diversity and composition of ground-dwelling spiders in 2018. Effects of  
648 management methods were tested using GLMMs and LMMs. Significant results are marked  
649 with boldface.

650 **Table A.6.** Effect of prescribed burning (B), mechanical turf disturbance (D), mowing (M),  
651 no-management (N) on species richness, abundance of Red List species, conservation value,  
652 functional diversity and composition of vegetation-dwelling spiders in 2017. Effects of  
653 management methods were tested using GLMMs and LMMs. Significant results are marked  
654 with boldface.

655 **Table A.7.** Effect of prescribed burning (B), mechanical turf disturbance (D), mowing (M),  
656 no-management (N) on species richness, abundance of Red List species, conservation value,  
657 functional diversity and composition of vegetation-dwelling spiders in 2018. Effects of  
658 management methods were tested using GLMMs and LMMs. Significant results are marked  
659 with boldface.  
660

661 **References**

- 662 Ausden, M., 2007. *Habitat Management for Conservation: a Handbook of Techniques*,  
663 Oxford University Press, New York.  
664
- 665 Alderweireldt, M., 1994. Prey selection and prey capture strategies of linyphiid spiders in  
666 high-input agricultural fields. *Bull. Br. Arachnol. Soc.* 9, 300–308.
- 667  
668 Batáry, P., Báldi, A., Sárospataki, M., Kohler, F., Verhulst, J., Knop, E., Herzog, F., Kleijn,  
669 D., 2010. Effect of conservation management on bees and insect-pollinated grassland  
670 plant communities in three European countries. *Agric. Ecosyst. Environ.* 136, 35–39.  
671 <https://doi.org/10.1016/j.agee.2009.11.004>
- 672 Batáry, P., Báldi, A., Samu, F., Szűts, T., Erdős, S., 2008. Are spiders reacting to local or  
673 landscape scale effects in Hungarian pastures? *Biodivers. Conserv.* 141, 2062–2070.  
674 <https://doi.org/10.1016/j.biocon.2008.06.002>
- 675 Bates, D., Maechler, M., Bolker, B., Walker, S., Christensen, R. H. B., Singmann, H., Green,  
676 P., 2020. Package “lme4”: Linear Mixed-Effects Models using 'Eigen' and S4. R package  
677 version 1.1-23.
- 678 Bayram, A., Luff, M.L., 1993. Winter abundance and diversity of lycosids (Lycosidae,  
679 Araneae) and other spiders in grass tussocks in a feld margin. *Pedobiologia* 37. 357–364.
- 680 Bell, J.R., Wheater, C.P., Cullen, W.R., 2001. The implications of grassland and heathland  
681 management for the conservation of spider communities: a review. *J. Zool.* 255, 377–  
682 387. <https://doi.org/10.1017/S0952836901001479>
- 683 Belsky, A.J., 1992. Effects of grazing, competition, disturbance and fire on species  
684 composition and diversity in grassland communities. *J. Veg. Sci.* 3, 187–200.  
685 <https://doi.org/10.2307/3235679>
- 686 Birkhofer, K., Diekötter, T., Meub, C., Stötzel, K., Wolters, V., 2015. Optimizing arthropod  
687 predator conservation in permanent grasslands by considering diversity components  
688 beyond species richness. *Agric. Ecosyst. Environ.* 211, 65–72.  
689 <https://doi.org/10.1016/j.agee.2015.05.014>
- 690 Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H.,  
691 White, J. S. S., 2009. Generalized linear mixed models: a practical guide for ecology and  
692 evolution. *Trends Ecol. Evol.* 24, 127–135. <https://doi.org/10.1016/j.tree.2008.10.008>

- 693 Botta-Dukát, Z., 2005. Rao's quadratic entropy as a measure of functional diversity based on  
694 multiple traits. *J. Veg. Sci.* 16, 533–540. [https://doi.org/10.1111/j.1654-  
695 1103.2005.tb02393.x](https://doi.org/10.1111/j.1654-1103.2005.tb02393.x)
- 696 Buchar, J., Růžička, V., Merrett, P., 2002. Catalogue of spiders of the Czech Republic. Peres  
697 Publishers, Praha.
- 698 Bucher, R., Entling, M.H., 2011. Contrasting effects of habitat fragmentation, population  
699 density, and prey availability on body condition of two orb-weaving spiders. *Ecol.  
700 Entomol.* 36. 680–685. <https://doi.org/10.1111/j.1365-2311.2011.01317.x>
- 701 Buchholz, S., 2010. Ground spider assemblages as indicators for habitat structure in inland  
702 sand ecosystems. *Biodivers. Conserv.* 19, 2565–2595. [https://doi.org/10.1007/s10531-  
703 010-9860-7](https://doi.org/10.1007/s10531-010-9860-7)
- 704 Cardoso, P., Pekár, S., Jocqué, R., Coddington, J.A., 2011. Global Patterns of Guild  
705 Composition and Functional Diversity of Spiders. *PLoS ONE* 6, e21710.  
706 <https://doi.org/10.1371/journal.pone.0021710>
- 707 Cattin, M.-F., Blandenier, G., Banašek-Richter, C., Bersier, L.-F., 2003. The impact of  
708 mowing as a management strategy for wet meadows on spider (Araneae) communities.  
709 *Biol. Conserv.* 113, 179–188. [https://doi.org/10.1016/S0006-3207\(02\)00297-5](https://doi.org/10.1016/S0006-3207(02)00297-5)
- 710 Cizek, O., Zamecnik, J., Tropek, R., Kocarek, P., Konvicka, M., 2012. Diversification of  
711 mowing regime increases arthropods diversity in species-poor cultural hay meadows. *J.  
712 Insect. Conserv.* 16, 215–226. <https://doi.org/10.1007/s10841-011-9407-6>
- 713 Conrad, K.F., Warren, M.S., Fox, R., Parsons, M.S., Woiwod, I.P., 2006. Rapid declines of  
714 common, widespread British moths provide evidence of an insect biodiversity crisis.  
715 *Biol. Conserv.* 132, 279–291. <https://doi.org/10.1016/j.biocon.2006.04.020>
- 716 Deák, B., 2014. Grassland fires in Hungary—Experiences of nature conservationists on the  
717 effects of fire on biodiversity. *Appl. Ecol. Env. Res.* 12, 267–283.  
718 [https://doi.org/10.15666/aeer/1201\\_267283](https://doi.org/10.15666/aeer/1201_267283)
- 719 de Bello, F., Carmona, C. P., Lepš, J., Szava-Kovats, R., Pärtel, M., 2016. Functional  
720 diversity through the mean trait dissimilarity: resolving shortcomings with existing  
721 paradigms and algorithms. *Oecologia* 180, 933–940. [https://doi.org/10.1007/s00442-016-  
722 3546-0](https://doi.org/10.1007/s00442-016-3546-0)

- 723 De Keer, R., Alderweireldt, M., Decler, K., Segers, H., Desender, K., Maelfait, J. P., 1989.  
724 Horizontal distribution of the spider fauna of intensively grazed pastures under the  
725 influence of diurnal activity and grass height. *J. Appl. Entomol.* 107, 455–473.  
726 <https://doi.org/10.1111/j.1439-0418.1989.tb00282.x>
- 727 De Keer, R., Maelfait, J. P., 1987. Life history of *Oedothorax fuscus* (Blackwall, 1834)  
728 Araneae, Linyphiidae in a heavily grazed pasture. *Rev. Ecol. Biol. Sol.* 24, 171–186.
- 729 De Keer, R., Maelfait, J. P., 1988. Observations on the life cycle of *Erigone atra* (Araneae,  
730 Erigoninae) in heavily grazed pasture. *Pedobiologia* 32, 201–212.
- 731 Díaz, S., Cabido, M., 2001. Vive la différence: plant functional diversity matters to ecosystem  
732 processes. *Trends Ecol. Evol.* 16, 646–655. [https://doi.org/10.1016/S0169-](https://doi.org/10.1016/S0169-5347(01)02283-2)  
733 [5347\(01\)02283-2](https://doi.org/10.1016/S0169-5347(01)02283-2)
- 734 Duelli, P., 1997. Biodiversity evaluation in agricultural landscapes: An approach at two  
735 different scales. *Agric. Ecosyst. Environ.* 62, 81–91. [https://doi.org/10.1016/S0167-](https://doi.org/10.1016/S0167-8809(96)01143-7)  
736 [8809\(96\)01143-7](https://doi.org/10.1016/S0167-8809(96)01143-7)
- 737 Duelli, P., Obrist, M.K., 2003. Regional biodiversity in an agricultural landscape: the  
738 contribution of seminatural habitat islands. *Basic Appl. Ecol.* 4, 129–138.  
739 <https://doi.org/10.1078/1439-1791-00140>
- 740 Dufrene, M., Legendre, P., 1997. Species assemblages and indicator species: the need for a  
741 flexible asymmetrical approach. *Ecol. Monogr.* 67, 345–367.  
742 [https://doi.org/10.1890/0012-9615\(1997\)067\[0345:SAAI\]2.0.CO;2](https://doi.org/10.1890/0012-9615(1997)067[0345:SAAI]2.0.CO;2)
- 743 Entling, W., Schmidt, M. H., Bacher, S., Brandl, R., Nentwig, W., 2007. Niche properties of  
744 Central European spiders: shading, moisture and the evolution of the habitat niche. *Glob.*  
745 *Ecol. Biogeogr.* 16, 440–448. <https://doi.org/10.1111/j.1466-8238.2006.00305.x>
- 746 ESRI (Environmental Systems Resource Institute), 2013. ArcMap 10.2 ESRI, Redlands,  
747 California. <https://www.esri.com/en-us/home>
- 748 Finke, D. L., Denno, R. F., 2006. Spatial refuge from intraguild predation: implications for  
749 prey suppression and trophic cascades. *Oecologia* 149, 265–275.  
750 <https://doi.org/10.1007/s00442-006-0443-y>
- 751 Gallé, R., Batáry, P., 2019. Trait-based paradise – about the importance of real functionality.  
752 *Commun. Ecol.* 20, 314–316. <https://doi.org/10.1556/168.2019.20.3.11>

- 753 Garnier, E., Cortez, J., Billes, G., Navas, M.L., Roumet, C., Debussche, M., Laurent, G.,  
754 Blanchard, A., Aubry, D., Bellmann, A., Neill, C., Toussaint, J.P., 2004. Plant functional  
755 markers capture ecosystem properties during secondary succession. *Ecology* 85, 2630–  
756 2637. <https://doi.org/10.1890/03-0799>.
- 757 Gaston, K.J., Fuller, R.A., 2007. Biodiversity and extinction: losing the common and the  
758 widespread. *Prog. Phys. Geog.* 31, 213–225. <https://doi.org/10.1177/0309133307076488>
- 759 Gibson, C.W.D., Hambler, C., Brown, V.K., 1992. Changes in Spider (Araneae) Assemblages  
760 in Relation to Succession and Grazing Management. *J. Appl. Ecol.* 29, 132.  
761 <https://doi.org/10.2307/2404356>
- 762 Greenstone, M.H., 1984. Determinants of web spider species diversity: vegetation structural  
763 diversity vs. prey availability. *Oecologia*, 62. 299–304.  
764 <https://doi.org/10.1007/BF00384260>
- 765 Házi, J., Bartha, S., Szentes, S.Z., Wichmann, B., Penksza, K., 2011. Seminatural grassland  
766 management by mowing of *Calamagrostis epigejos* in Hungary. *Plant Biosyst.* 145, 699–  
767 707. <https://doi.org/10.1080/11263504.2011.601339>
- 768 Hamřík, T., Košulič, O., 2019. Spiders from steppe habitats of Pláně Nature Monument  
769 (Czech Republic) with suggestions for the local conservation management. *Arachnol.*  
770 *Mitt.* 58, 85–96. <https://doi.org/10.30963/aramit5812>
- 771 Heimer, S., Nentwig, W., 1991. *Spinnen Mitteleuropas: ein Bestimmungsbuch*. Paul Parey,  
772 Berlin, Hamburg. [in German]
- 773 Henle, K., Alard, D., Clitherow, J., Cobb, P., Firbank, L., Kull, T., McCracken, D., Moritz,  
774 R.F.A., Niemelä, J., Rebane, M., Wascher, D., Watt, A., Young, J., 2008. Identifying and  
775 managing the conflicts between agriculture and biodiversity conservation in Europe—A  
776 review. *Agric. Ecosyst. Environ.* 124, 60–71. <https://doi.org/10.1016/j.agee.2007.09.005>
- 777 Henning, K., Lorenz, A., von Oheimb, G., Härdtle, W., Tischew, S., 2017. Year-round  
778 cattle and horse grazing supports the restoration of abandoned, dry sandy  
779 grassland and heathland communities by suppressing *Calamagrostis epigejos* and  
780 enhancing species richness. *J. Nat. Conserv.* 40, 120–130.  
781 <https://doi.org/10.1016/j.jnc.2017.10.009>

- 782 Herberstein, M. E., 2011. Spider behaviour: flexibility and versatility. Cambridge University  
783 Press, Cambridge, UK.
- 784 Hodeček, J., Kuras, T., Šipoš, J., Dolný, A., 2015. Post-industrial areas as successional  
785 habitats: Long-term changes of functional diversity in beetle communities. *Basic Appl.*  
786 *Ecol.* 16, 629–640. <https://doi.org/10.1016/j.baae.2015.06.004>
- 787 Horváth, R., Magura, T., Tóthmérész, B., Eichardt, J., Szinetár, C., 2018. Both local and  
788 landscape-level factors are important drivers in shaping ground-dwelling spider as-  
789 semblages of sandy grasslands. *Biodivers. Conserv.* 28, 297–313.  
790 <https://doi.org/10.1007/s10531-018-1654-3>
- 791 Horváth, R., Magura, T., Szinetár, C., Tóthmérész, B., 2009. Spiders are not less diverse in  
792 small and isolated grasslands, but less diverse in overgrazed grasslands: A field study  
793 (East Hungary, Nyírség). *Agric. Ecosyst. Environ.* 130, 16–22.  
794 <https://doi.org/10.1016/j.agee.2008.11.011>
- 795 Hothorn, T., Bretz, F., Westfall, P., Heiberger, R. M., Schuetzenmeister, A., Scheibe, S.,  
796 2016. Package “multcomp”: Simultaneous Inference in General Parametric Models. R  
797 package version 1.4-13.
- 798 Humbert, J.Y., Ghazoul, J., Walter, T., 2009. Meadow harvesting techniques and their impacts  
799 on field fauna. *Agric. Ecosyst. Environ.* 130, 1–8.  
800 <https://doi.org/10.1016/j.agee.2008.11.014Get>
- 801 Humbert, J.Y., Ghazoul, J., Sauter, G.J., Walter, T., 2010. Impact of different meadow  
802 mowing techniques on field invertebrates. *J. Appl. Entomol.* 134, 592–599.  
803 <https://doi.org/10.1111/j.1439-0418.2009.01503.x>
- 804 Isselstein, J., Jeangros, B., Pavlů, V., 2005. Agronomic aspects of extensive grassland farming  
805 and biodiversity management. *Agron. Res.* 10, 211–220
- 806 Jiménez-Valvedre, A., Lobo, J. M., 2007. Determinants of local spider (Araneidae and  
807 Thomisidae) species richness on a regional scale: climate and altitude vs. habitat  
808 structure. *Ecol. Entomol.* 32, 113–122. [https://doi.org/10.1111/j.1365-  
809 2311.2006.00848.x](https://doi.org/10.1111/j.1365-2311.2006.00848.x)
- 810 Jocqué, R., Dippenaar-Schoeman, A.S., 2006. Spider Families of the World. Royal Museum  
811 for Central Africa. Tervuren, Belgium.

- 812 Jongepierová, I., Pešout, P., Prach, K. (Eds.), 2018. Ecological restoration in the Czech  
813 Republic II, Nature Conservation Agency of the Czech Republic, Prague.
- 814 Kasal, P., Kaláb, V., 2020. Arachnobase of the Czech Spiders. Faculty of Biomedical  
815 Engineering, Czech Technical University, Prague. <http://www.arachnobaze.cz/en/info>
- 816 Košulič, O., Michalko, R., Hula, V., 2016. Impact of Canopy Openness on Spider  
817 Communities: Implications for Conservation Management of Formerly Coppiced Oak  
818 Forests. PLoS ONE 11, e0148585. <https://doi.org/10.1371/journal.pone.0148585>
- 819 Košulič, O., Michalko, R., Hula, V., 2014. Recent artificial vineyard terraces as a refuge for  
820 rare and endangered spiders in a modern agricultural landscape. Ecol. Eng. 68, 133–142.  
821 <https://doi.org/10.1016/j.ecoleng.2014.03.030>
- 822 Lafage, D., Pétilion, J., 2014. Impact of cutting date on carabids and spiders in a wet meadow.  
823 Agric. Ecosyst. Environ. 185, 1–8. <https://doi.org/10.1016/j.agee.2013.11.027>
- 824 Laliberté, E., Legendre, P., Shipley, B., 2014. FD: Measuring functional diversity from multiple  
825 traits, and other tools for functional ecology. R package version 1.0-12.
- 826 Lawrence, K. L., Wise, D. H., 2000. Spider predation on forest-floor Collembola and evidence for  
827 indirect effects on decomposition. Pedobiologia 44, 33–39. [https://doi.org/10.1078/S0031-4056\(04\)70026-8](https://doi.org/10.1078/S0031-4056(04)70026-8)  
828
- 829 Loreau, M., 2010. Linking biodiversity and ecosystems: Towards a unifying ecological theory.  
830 Philos. Trans. R. Soc. B: Biol. Sci. 365, 49–60. <https://doi.org/10.1098/rstb.2009.0155>
- 831 Lepš, J., de Bello, F., Lavorel, S., Berman, S., 2006. Quantifying and interpreting functional  
832 diversity of natural communities: practical considerations matter. Preslia 78, 481–501.
- 833 Lepš, J., 1999. Nutrient status disturbance and competition: an experimental test of  
834 relationships in a wet meadow. J. Veg. Sci. 10, 219–230.  
835 <https://doi.org/10.2307/3237143>
- 836 Littlewood, N.A., Stewart, A.J.A., Woodcock, B.A., 2012. Science into practice - how can  
837 fundamental science contribute to better management of grasslands for invertebrates?:  
838 Grassland invertebrate conservation. Insect Conserv. and Diver. 5, 1–8.  
839 <https://doi.org/10.1111/j.1752-4598.2011.00174.x>

- 840 Lunt, I.D., 1993. Variation in flower production of nine grassland species with time since fire,  
841 and implications for grassland management and restoration. *Pac. Conserv. Biol.* 1, 359–  
842 366. <https://doi.org/10.1071/PC940359>
- 843 Lyons, A., Ashton, P.A., Powell, I., Oxbrough, A., 2018. Epigeal spider assemblage responses  
844 to vegetation structure under contrasting grazing management in upland calcareous  
845 grasslands. *Insect Conserv. Divers.* 11, 383–395. <https://doi.org/10.1111/icad.12287>
- 846 Maelfait, J.P., Hendrickx, F., 1998. Spiders as bio-indicators of anthropogenic stress in  
847 natural and semi-natural habitats in Flanders (Belgium): some recent developments. In:  
848 Selden, P.A. (Ed.), *Proceedings of the 17th European Colloquium of Arachnology*,  
849 Edinburgh 1997. Dorset Press, Dorchester, pp. 293–300.
- 850 Marc, P., Canard, A., Ysnel, F., 1999. Spiders (Araneae) useful for pest limitation and  
851 bioindication. *Agric. Ecosyst. Environ.* 74, 229–273. [https://doi.org/10.1016/S0167-](https://doi.org/10.1016/S0167-8809(99)00038-9)  
852 [8809\(99\)00038-9](https://doi.org/10.1016/S0167-8809(99)00038-9)
- 853 Matuška, P., 2016 Plán péče o přírodní památku Pláně na období 2016–2025 [Conservation  
854 plan for Pláně Nature Monument in period 2016–2025]. Deposited in: Krajský úřad  
855 Jihomoravského kraje, Brno, pp. 17 [in Czech]
- 856 Mazalová, M., Šipoš, J., Rada, S., Kašák, J., Šarapatka, B., Kuras, T., 2015. Responses of  
857 grassland arthropods to various biodiversity-friendly management practices: Is there a  
858 compromise? *Eur. J. Entomol.* 112, 734–746. <https://doi.org/10.14411/eje.2015.076>
- 859 Mcnett, B.J., Rypstra, A.L., 2000. Habitat selection in a large orb-weaving spider:  
860 vegetational complexity determines site selection and distribution. *Ecol. Entomol.* 25,  
861 423–432. <https://doi.org/10.1046/j.1365-2311.2000.00279.x>
- 862 Merrett, P., 1976. Changes in the ground-living spider fauna after heathland fires in Dorset.  
863 *Bull. Br. Arachnol. Soc.*, 3, 214–221.
- 864 Michalko, R., Pekár, S., Entling, M.H., 2019. An updated perspective on spiders as generalist  
865 predators in biological control. *Oecologia*, 189, 21–36. [https://doi.org/10.1007/s00442-](https://doi.org/10.1007/s00442-018-4313-1)  
866 [018-4313-1](https://doi.org/10.1007/s00442-018-4313-1)
- 867 Michalko, R., Pekár, S., 2016. Different hunting strategies of generalist predators result in  
868 functional differences. *Oecologia* 181, 1187–1197. [https://doi.org/10.1007/s00442-016-](https://doi.org/10.1007/s00442-016-3631-4)  
869 [3631-4](https://doi.org/10.1007/s00442-016-3631-4)

- 870 Morris, M.G., 2000. The effects of structure and its dynamics on the ecology and conservation  
871 of arthropods in British grasslands. *Biol. Conserv.* 95, 129–142.  
872 [https://doi.org/10.1016/S0006-3207\(00\)00028-8](https://doi.org/10.1016/S0006-3207(00)00028-8)
- 873 Nemkov, V.A., Sapiga, E.V., 2010. Impact of fires on the fauna of terrestrial arthropods in  
874 protected steppe ecosystems. *Russ. J. Ecol.* 41, 173–179.  
875 <https://doi.org/10.1134/S1067413610020104>
- 876 Nentwig, W., Blick, T., Gloor, D., Hängi, A., Kropf, C., 2019. Spiders of Europe. Version  
877 03.2019. <https://araneae.nmbe.ch/> (accessed 15 March 2019)
- 878 Niwa, C.G., Peck, R.W., 2002. Influence of prescribed fire on carabid beetle (Carabidae) and  
879 spider (Araneae) assemblages in forest litter in southwestern Oregon. *Environ. Entomol.*  
880 31, 785–796. <https://doi.org/10.1603/0046-225X-31.5.785>
- 881 Noordijk, J., Schaffers, A.P., Heijerman, T., Boer, P., Gleichman, M., Sýkora, K.V., 2010.  
882 Effects of vegetation management by mowing on ground-dwelling arthropods. *Ecol.*  
883 *Eng.* 36, 740–750. <https://doi.org/10.1016/j.ecoleng.2010.01.003>
- 884 Nogueira, A.A., Pinto-Da-Rocha, R., 2016. The effects of habitat size and quality on the orb-  
885 weaving spider guild (Arachnida: Araneae) in an Atlantic Forest fragmented landscape.  
886 *Journal of Arachnology* 44, 36-45. <https://doi.org/10.1636/P15-19.1>
- 887 Pavoine, S., Dolédec, S., 2005. The apportionment of quadratic entropy: a useful alternative  
888 for partitioning diversity in ecological data. *Environ. Ecol. Stat.* 12, 125–138.  
889 <https://doi.org/10.1007/s10651-005-1037-2>
- 890 Pearce, J.L., Venier, L.A., Eccles, G., Pedlar, J., McKenney, D., 2004. Influence of habitat  
891 and microhabitat on epigeal spider (Araneae) assemblages in four stand types. *Biodivers.*  
892 *Conserv.* 13, 1305–1334. <https://doi.org/10.1023/B:BIOC.0000019403.26948.55>
- 893 Pech, P., Dolanský, J., Hrdlička, R., Lepš, J., 2015. Differential response of communities of  
894 plants, snails, ants and spiders to long-term mowing in a small-scale experiment.  
895 *Community Ecol.* 16, 115–124. <https://doi.org/10.1556/168.2015.16.1.13>
- 896 Pekár, S., Michalko, R., Loverre, P., Líznavá, E., Černecká, L., 2015. Biological control in  
897 winter: novel evidence for the importance of generalist predators. *J. Appl. Ecol.* 52, 270–  
898 279. <https://doi.org/10.1111/1365-2664.12363>

- 899 Pekár, S., Brabec, M., 2009. Moderní analýza biologických dat. 1. vydání. Zobecněné lineární  
900 modely v prostředí R. Scientia, Praha. [in Czech]
- 901 Peres, M.C.L., Cardoso Da Silva, J.M., Brescovit, A.D., 2007. The influence of treefall gaps  
902 on the distribution of web-building and ground hunter spiders in an Atlantic Forest  
903 remnant, northeastern Brazil. *Studies on Neotropical Fauna and Environment* 42, 49–60.  
904 <https://doi.org/10.1080/01650520600891889>
- 905 Pétilion, J., Georges, A., Canard, A., Ysnel, F., 2007. Impact of cutting and sheep grazing on  
906 ground-active spiders and carabids in intertidal salt marshes (Western France). *Anim.*  
907 *Biodivers. Conserv.* 30. 201–209.
- 908 Petchey, O.L., Gaston, K.J., 2006. Functional diversity: back to basics and looking forward.  
909 *Ecol. Lett.* 9, 741–758. <https://doi.org/10.1111/j.1461-0248.2006.00924.x>
- 910 Pinto-Leite, C.M., Guerrero, A.C., Brazil, T.K., 2008. Non-random patterns of spider species  
911 composition in an Atlantic rainforest. *Journal of Arachnology* 36, 448–452.  
912 <https://doi.org/10.1636/CT07-123.1>
- 913 Polchaninova, N., 2015. Recovery of Spider Communities After a Spontaneous Summer Fire  
914 in the Forb-Bunchgrass Steppe of Eastern Ukraine. *Hacquetia* 14, 79–96.  
915 <https://doi.org/10.1515/hacq-2015-0015>
- 916 Poschlod, P., WallisDeVries, M.F., M.F., 2002. The historical and socioeconomic perspective  
917 of calcareous grasslands – lessons from the distant and recent past. *Biol. Conserv.* 104,  
918 361–376. [https://doi.org/10.1016/S0006-3207\(01\)00201-4](https://doi.org/10.1016/S0006-3207(01)00201-4)
- 919 Prishutova, Z.G., Arzanova, Y.G., 2008: Influence of the local steppe fire on the  
920 gerpetobionts. *Trudy Gosudarstvennogo Prirodnogo Zapovednika ‘Rostovskiy’* Vypusk  
921 4, 192–208. [in Russian, with English summary].
- 922 Purvis, G., Curry, J. P., 1981. The influence of sward management on foliage arthropod  
923 communities in a ley grassland. *J. Appl. Ecol.* 18, 711–725.  
924 <https://doi.org/10.2307/2402363>
- 925 R Development Core Team, 2018. R: a Language and Environment for Statistical Computing.  
926 R Foundation for Statistical Computing, Vienna, Austria. <http://www.Rproject.org/>.
- 927 Rao, C.R., 1982. Diversity and dissimilarity coefficients: A unified approach. *Theor. Popul.*  
928 *Biol.* 21, 24–43. [https://doi.org/10.1016/0040-5809\(82\)90004-1](https://doi.org/10.1016/0040-5809(82)90004-1)

- 929 Reidsma, P., Tekelenburg, T., van den Berg, M., Alkemade, R., 2006. Impacts of land-use  
930 change on biodiversity: An assessment of agricultural biodiversity in the European  
931 Union. *Agric. Ecosyst. Environ.* 114, 86–102.  
932 <https://doi.org/10.1016/j.agee.2005.11.026>
- 933 Řezáč, M., Kůrka, A., Růžička, V., Heneberg, P., 2015. Red List of Czech spiders: 3rd  
934 edition, adjusted according to evidence-based national conservation priorities. *Biologia*  
935 70. 645–666. <https://doi.org/10.1515/biolog-2015-0079>
- 936 Řezáč, M., Heneberg, P., 2019. Grazing as a conservation management approach leads to a  
937 reduction in spider species richness and abundance in acidophilous steppic grasslands on  
938 andesite bedrock. *J. Insect. Conserv.* 23, 777–783. <https://doi.org/10.1007/s10841-019-00163-9>
- 939
- 940 Řezáč, M., Heneberg, P., 2018. Effects of uncut hay meadow strips on spiders. *Biologia* 73,  
941 43–51. <https://doi.org/10.1007/s10841-019-00163-9>
- 942 Ripley, B., Venables, B., Bates, D. M., Hornik, K., Gebhardt, A., Firth, D., Ripley, M. B.,  
943 2020. Package “MASS”: Functions and datasets to support Venables and Ripley,  
944 “Modern Applied Statistics with S”. R package version 7.3-51.6.
- 945 Roberts, D.W., 2019. Package “labdsv”: Ordination and Multivariate Analysis for Ecology. R  
946 package version 2.0-1.
- 947 Ricotta, C., Moretti, M., 2011. CWM and Rao’s quadratic diversity: a unified framework for  
948 functional ecology. *Oecologia* 167, 181–188. <https://doi.org/10.1007/s00442-011-1965-5>
- 949 Ricotta, C., 2005. A note on functional diversity measures. *Basic Appl. Ecol.* 6, 479–486.  
950 <https://doi.org/10.1016/j.baae.2005.02.008>
- 951 Robinson, R.A., Sutherland, W.J., 2002. Post-War Changes in Arable Farming and  
952 Biodiversity in Great Britain. *J. Appl. Ecol.* 39, 157–176.  
953 <http://dx.doi.org/10.1046/j.1365-2664.2002.00695.x>
- 954 Samu, F., Szinetár, C., Szita, É., Fetykó, K., Neidert, D., 2011. Regional variations in  
955 agrobiont composition and agrobiont life history of spiders (Araneae) within Hungary.  
956 *Arachnol. Mitt.* 40, 105–109. <https://doi.org/10.5431/aramit4012>
- 957 Samu, F., Kádár, F., Ónodi, G., Kertész, M., Szirányi, A., Szita, É., Fetykó, K., Neidert, D.,  
958 Botos, E., Altbäcker, V., 2010. Differential ecological responses of two generalist

959 arthropod groups, spiders and carabid beetles (Araneae, Carabidae), to the effects of  
960 wildfire. *Commun. Ecol.* 11, 129–139. <https://doi.org/10.1556/ComEc.11.2010.2.1>

961 Samu, F., Sunderland, K.D., Szinetár, C., 1999. Scale-dependent dispersal and distribution  
962 patterns of spiders in agricultural systems: a review. *J. Arachnol.* 27, 325–332.

963 Schmidt, J.M., Rypstra, A.L., 2010. Opportunistic predator prefers habitat complexity that  
964 exposes prey while reducing cannibalism and intraguild encounters. *Oecologia*, 164,  
965 899–910. <https://doi.org/10.1007/s00442-010-1785-z>

966 Simons, N.K., Weisser, W.W., Gossner, M.M., 2016. Multi-taxa approach shows consistent  
967 shifts in arthropod functional traits along grassland land-use intensity gradient. *Ecology*  
968 97, 754–764. <https://doi.org/10.1890/15-0616.1>

969 Šmilauer, P., Lepš, J., 2014. *Multivariate analysis of ecological data using Canoco 5*.  
970 Cambridge University Press, Cambridge, UK.

971 Stoate, C., Báldi, A., Beja, P., Boatman, N.D., Herzog, I., van Doorn, A., de Snoo, G.R.,  
972 Rakosy, L., Ramwell, C., 2009. Ecological impacts of early 21st century agricultural  
973 change in Europe – A review. *J. Environ. Manage.* 91, 22–46.  
974 <https://doi.org/10.1016/j.jenvman.2009.07.005>

975 ter Braak, C.J.F., Šmilauer, P., 2012. *Canoco 5*. Software for multivariate data exploration,  
976 testing, and summarization. Netherlands.

977 Tilman, D., 2001. Forecasting Agriculturally Driven Global Environmental Change. *Science*  
978 292, 281–284. <https://doi.org/10.1126/science.1057544>

979 Torma, A., Császár, P., Bozsó, M., Deák, B., Valkó, O., Kiss, O., Gallé, R., 2019. Species and  
980 functional diversity of arthropod assemblages (Araneae, Carabidae, Heteroptera and  
981 Orthoptera) in grazed and mown salt grasslands. *Agric. Ecosyst. Environ.* 273, 70–79.  
982 <https://doi.org/10.1016/j.agee.2018.12.004>

983 Tropek, R., Černá, I., Straka, J., Kadlec, T., Pech, P., Tichánek, F., Šebek, P., 2014.  
984 Restoration management of fly ash deposits crucially influence their conservation  
985 potential for terrestrial arthropods. *Ecol. Eng.* 73, 45–52.  
986 <https://doi.org/10.1016/j.ecoleng.2014.09.011>

987 Tropek, R., Kadlec, T., Karešová, P., Spitzer, L., Kočárek, P., Malenovský, I., Banař, P., Tuf,  
988 I.H., Hejda, M., Konvička, M., 2010. Spontaneous succession in limestone quarries as an

- 989 effective restoration tool for endangered arthropods and plants. *J. Appl. Eco.* 47, 139–  
990 147. <https://doi.org/10.1111/j.1365-2664.2009.01746.x>
- 991 Tschardtke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape  
992 perspectives on agricultural intensification and biodiversity – ecosystem service  
993 management. *Ecol. Lett.* 8, 857–874. <https://doi.org/10.1111/j.1461-0248.2005.00782.x>
- 994 Valkó, O., Venn, S., Żmihorski, M., Biurrun, I., Labadessa, R., Loos, J., 2018. The challenge  
995 of abandonment for the sustainable management of Palaearctic natural and semi-natural  
996 grasslands. *Hacquetia* 17, 5–16. <https://doi.org/10.1515/hacq-2017-0018>
- 997 Valkó, O., Deák, B., Magura, T., Török, P., Kelemen, A., Tóth, K., Horváth, R., Nagy, D.D.,  
998 Debnár, Z., Zsigrai, G., Kapocsi, I., Tóthmérész, B., 2016. Supporting biodiversity by  
999 prescribed burning in grasslands – A multi-taxa approach. *Sci. Total Environ.* 572,  
1000 1377–1384. <https://doi.org/10.1016/j.scitotenv.2016.01.184>
- 1001 Valkó, O., Török, P., Deák, B., Tóthmérész, B., 2014. Review: Prospects and limitations of  
1002 prescribed burning as a management tool in European grasslands. *Basic Appl. Ecol.* 15,  
1003 26–33. <https://doi.org/10.1016/j.baae.2013.11.002>
- 1004 Van Dyck, H., Van Strien, A.J., Maes, D., Van Swaay, C.A.M., 2009. Declines in Common,  
1005 Widespread Butterflies in a Landscape under Intense Human Use. *Conserv. Biol.* 23,  
1006 957–965. <https://doi.org/10.1111/j.1523-1739.2009.01175.x>
- 1007 Van Klink, R., Van der Plas, F., Van Noordwijk, C.G.E.T., Wallis DeVries, M.F., Olf, H.,  
1008 2015. Effects of large herbivores on grassland arthropod diversity. *Biol. Rev.* 90, 347–  
1009 366. <https://doi.org/10.1111/brv.12113>
- 1010 World Spider Catalog, 2019. World Spider Catalog, Version 20.0. Natural History Museum  
1011 Bern. <http://wsc.nmbe.ch/> (accessed 15 March 2019).
- 1012 Wise, D. H., 1995. Spiders in ecological webs. Cambridge University Press, Cambridge, UK.

1013 **Figure legends**

1014 **Fig. 1.** Map of the study area in the Czech Republic (red square on map) and sites with  
1015 rectangles ( $4 \times 5$  m) where each management method was applied (ESRI, 2013). The map  
1016 background was downloaded from free maps platform system (<http://www.freepik.com>) and  
1017 modified in Adobe Photoshop CS6.

1018 **Fig. 2.** Redundancy analysis ordination diagram (RDA) revealing relationship between  
1019 physical environmental characteristics and management methods. Conservation management  
1020 explained 79.4% of overall variability.

1021 **Fig. 3.** Effect of conservation management methods on CWM of humidity preference of  
1022 ground-dwelling spiders in a) 2017, and b) 2018. Horizontal lines on bars indicate median  
1023 values, box boundaries show quartiles, whiskers denote 1.5 times the interquartile range.  
1024 Different letters indicate statistically significant differences ( $P < 0.050$ ).

1025 **Fig. 4.** Effect of conservation management methods on diversity of ground-dwelling spiders  
1026 in 2018: a) functional diversity, b) conservation value, and c) abundance of Red List species.  
1027 Horizontal lines on bars indicate median values, box boundaries show quartiles, whiskers  
1028 denote 1.5 times the interquartile range. Different letters indicate statistically significant  
1029 differences ( $P < 0.050$ ).

1030 **Fig. 5.** Redundancy analysis ordination diagrams summarizing differences within ground-  
1031 dwelling spider assemblages in relation to conservation management methods in 2018: a) 25  
1032 spider species, best fitted by management method (conservation management methods explain  
1033 11.1% of overall variability), and b) degree of rareness. Species names are abbreviated by the  
1034 first four letters of the genus and species names in graph (a) (see Table A.2 for species list  
1035 with full names). Abbreviations in graph (b) distinguish degree of rareness: VA – very  
1036 abundant, A – abundant, S – scarce, R – rare, VR – very rare.

1037 **Fig. 6.** Effect of conservation management methods on diversity of vegetation-dwelling  
1038 spiders in 2017: a) species richness, b) functional diversity, c) conservation value, and d)  
1039 abundance of Red List species. Horizontal lines on bars indicate median values, box  
1040 boundaries show quartiles, whiskers denote 1.5 times the interquartile range. Different letters  
1041 indicate statistically significant differences ( $P < 0.050$ ).

1042 **Fig. 7.** Effect of conservation management methods on CWM of a) body size, and b)  
1043 humidity preference of vegetation-dwelling spiders in 2018. Horizontal lines on bars indicate  
1044 median values, box boundaries show quartiles, whiskers denote 1.5 times the interquartile  
1045 range. Different letters indicate statistically significant differences ( $P < 0.050$ ).

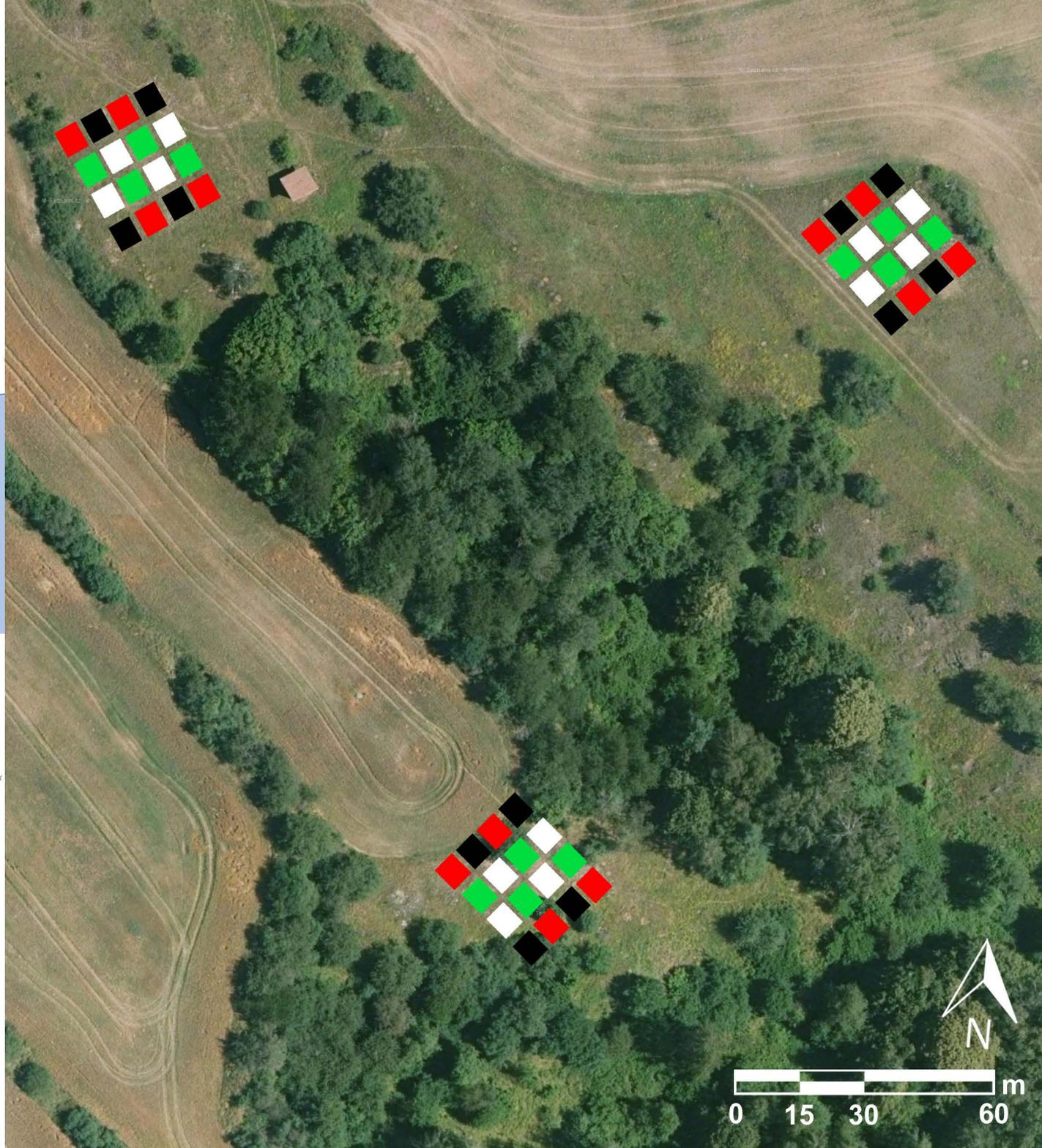
1046 **Fig. 8.** Redundancy analysis ordination diagrams summarizing differences within vegetation-  
1047 dwelling spider assemblages in relation to conservation management methods in 2018: a)  
1048 spider species composition (conservation management methods explain 20.1% of overall  
1049 variability), and b) degree of rareness. Species names are abbreviated by the first four letters  
1050 of the genus and species names in graph (a) (see Table A.2 for species list with full names).  
1051 Abbreviations in graph (b) distinguish degree of rareness: VA – very abundant, A – abundant,  
1052 S – scarce, R – rare.

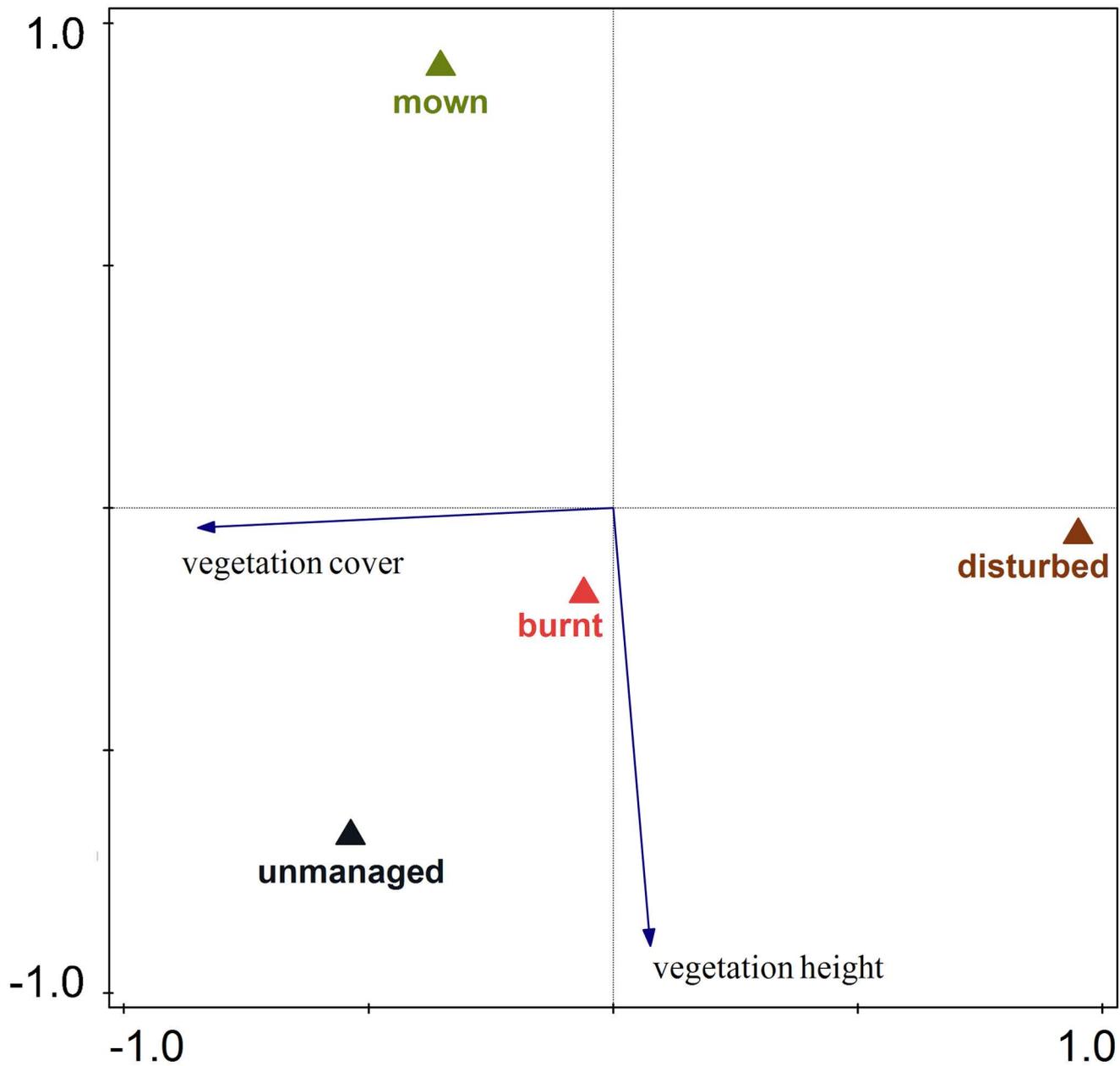
1053 **Graphical abstract caption:** Patterns of changes of spider assemblages in relation to various  
1054 conservation management methods in xeric grassland.

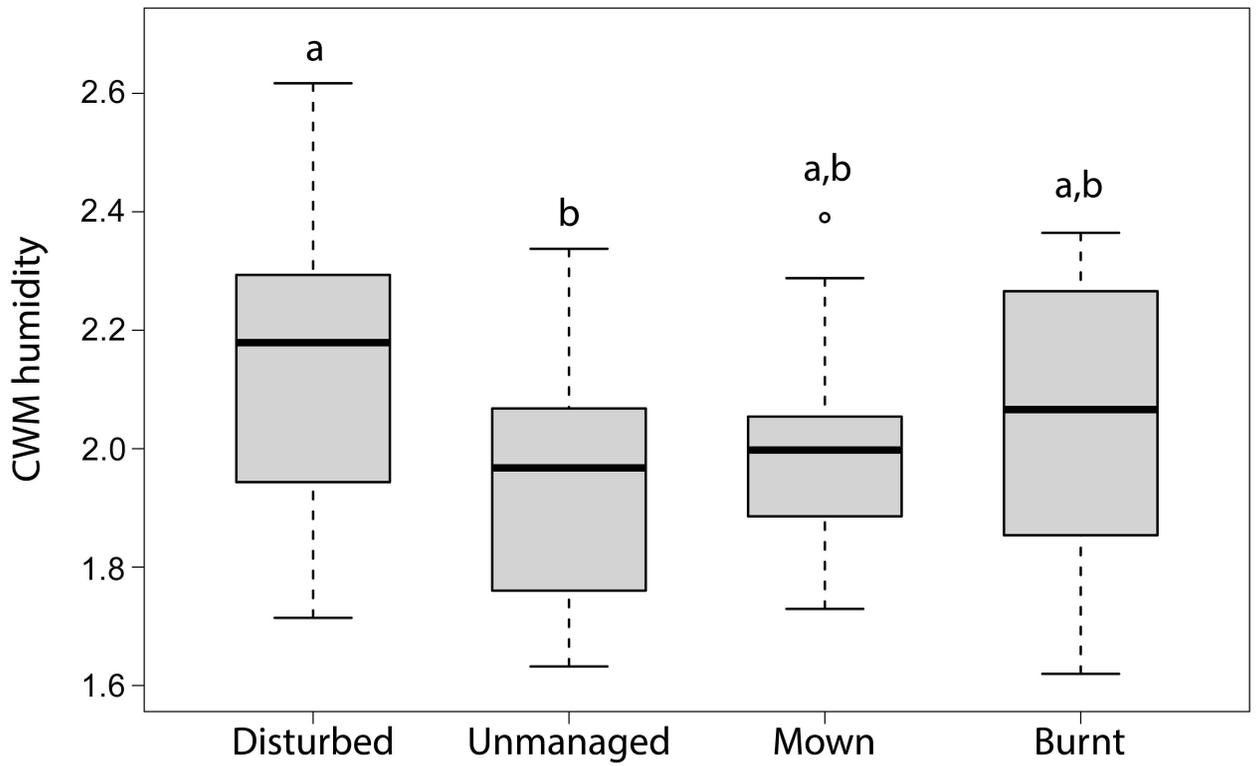
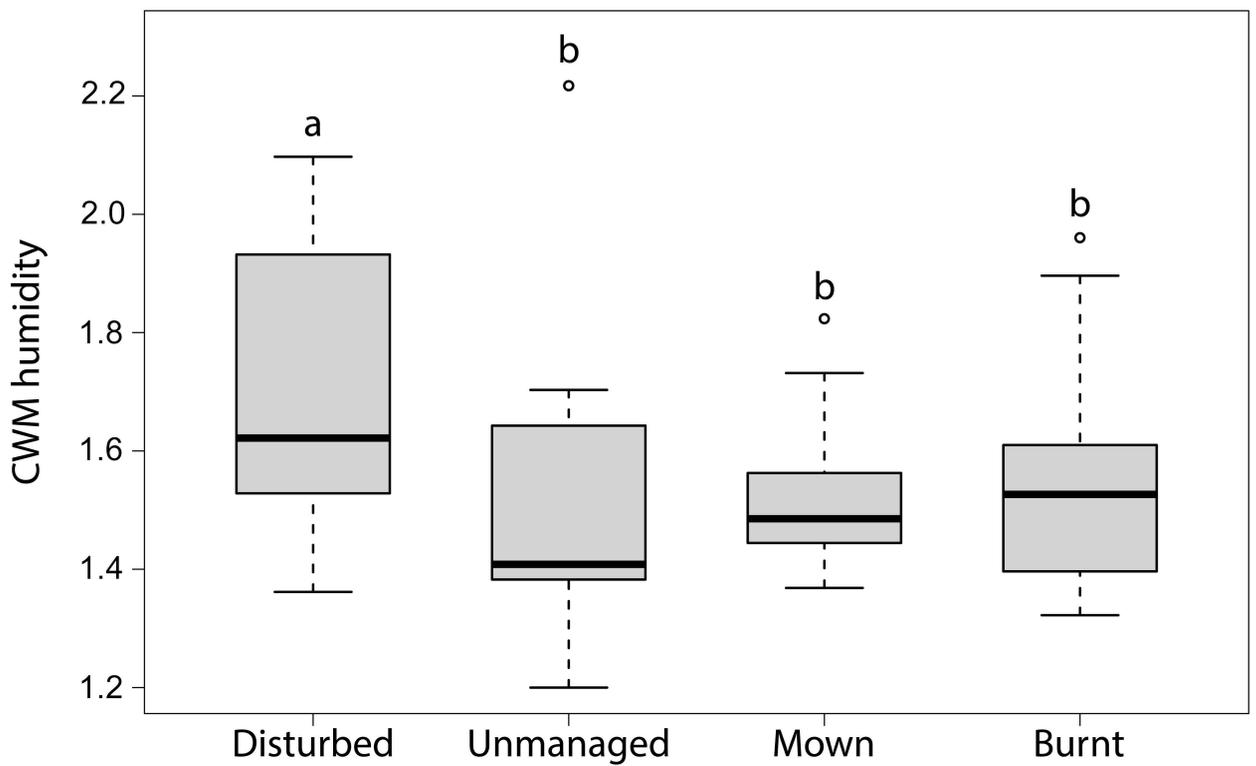


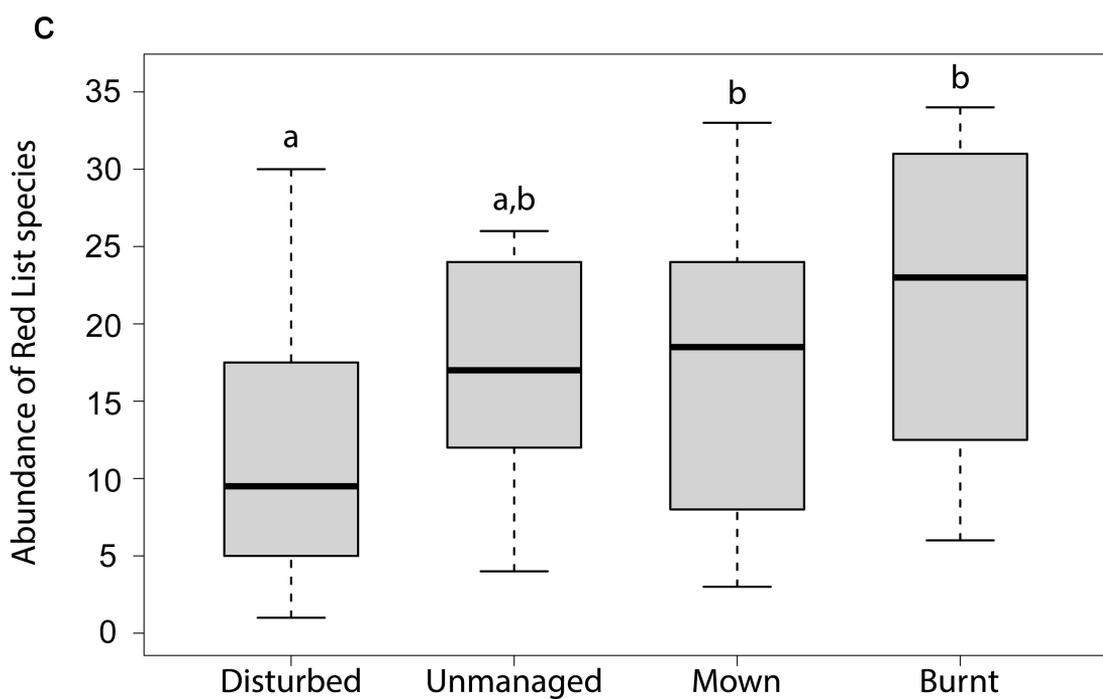
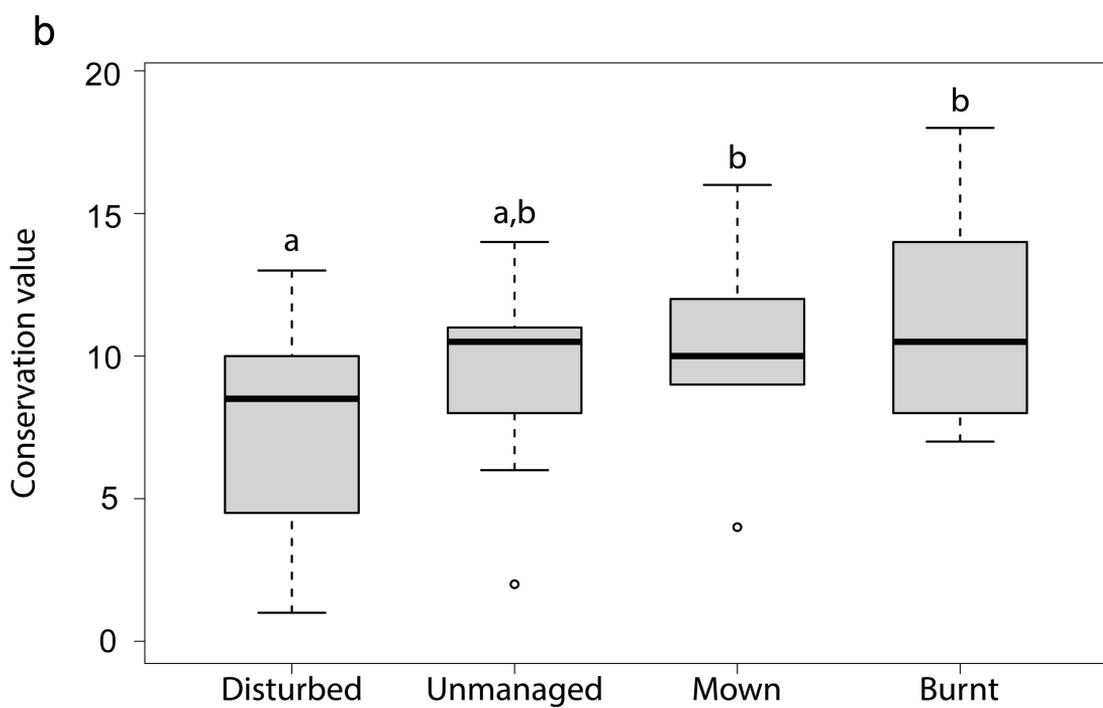
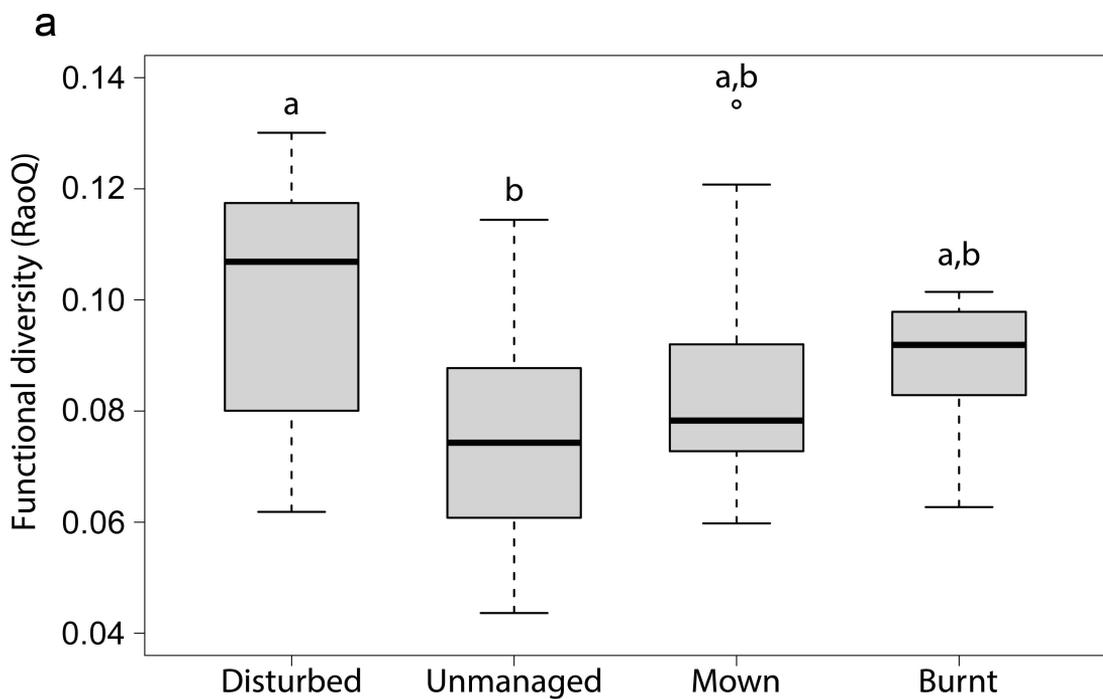
### Managements

-  prescribed burning
-  no-management
-  disturbance
-  mowing

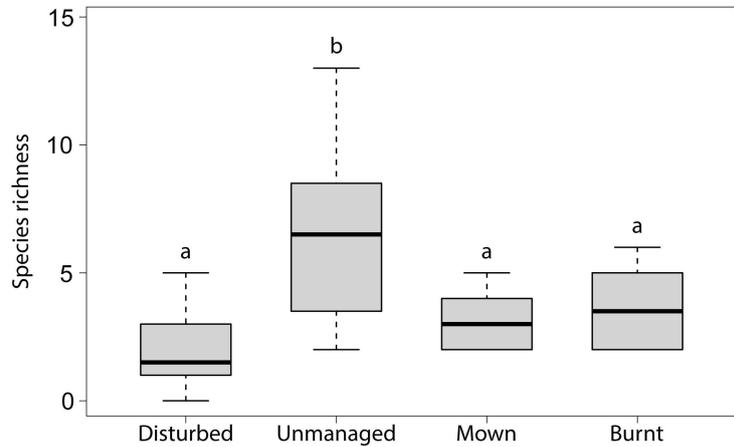
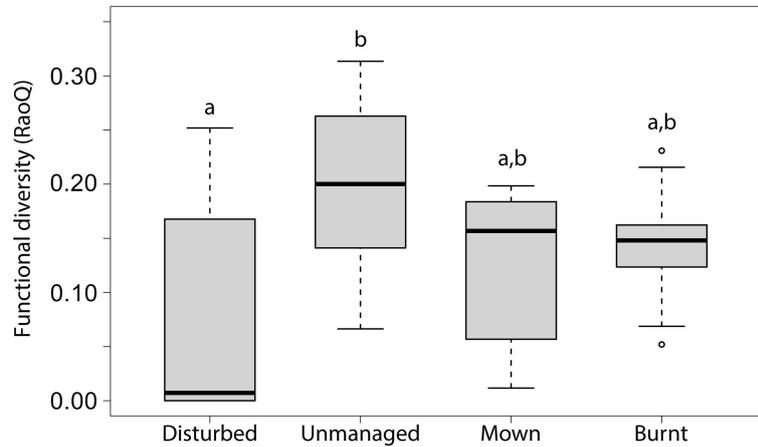
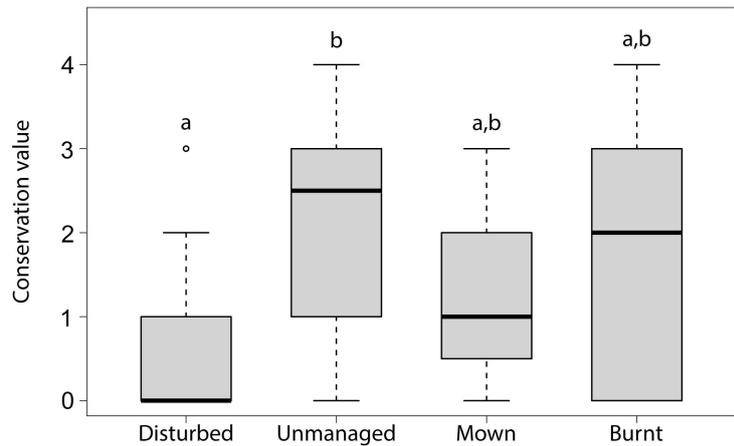
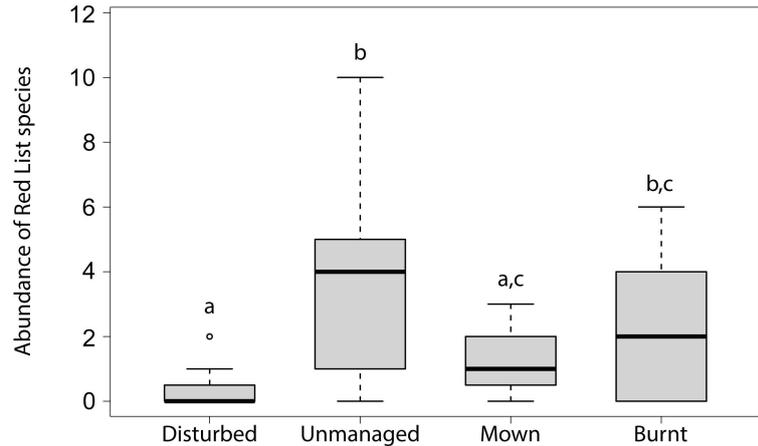


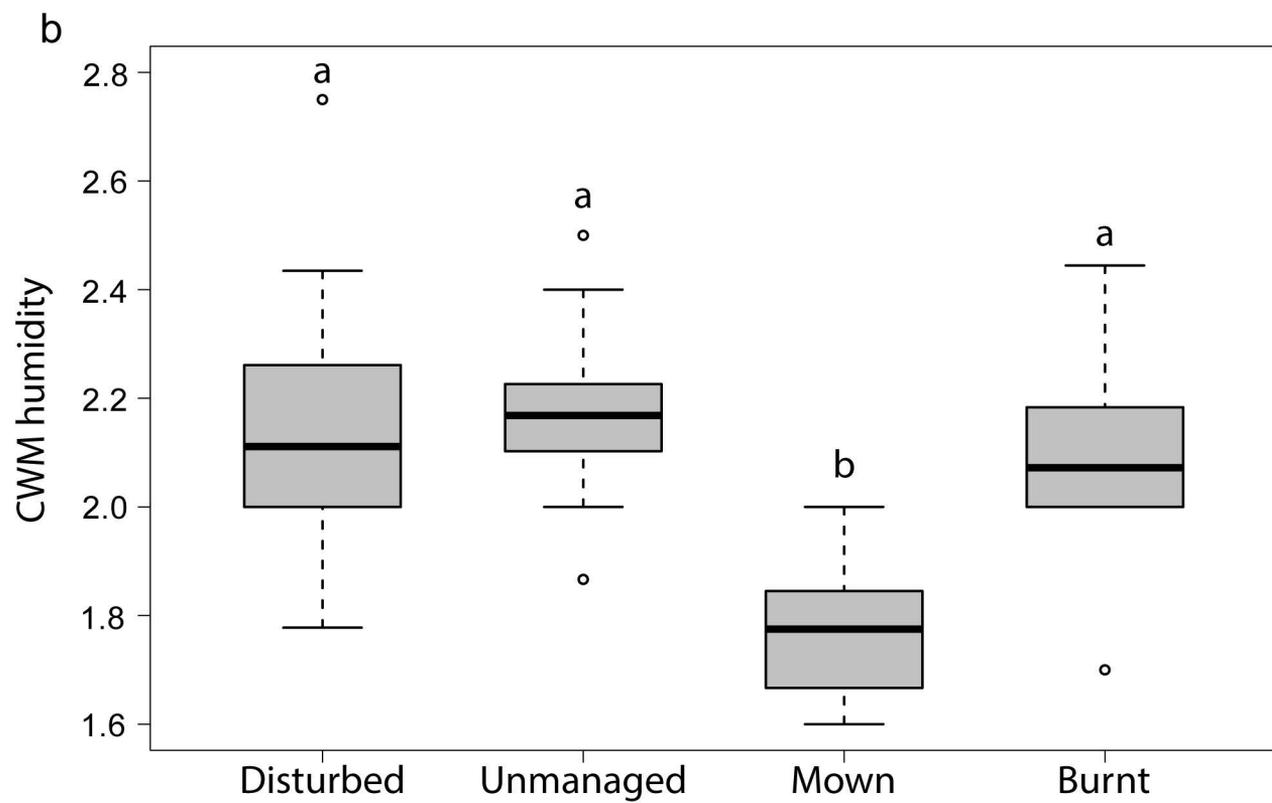
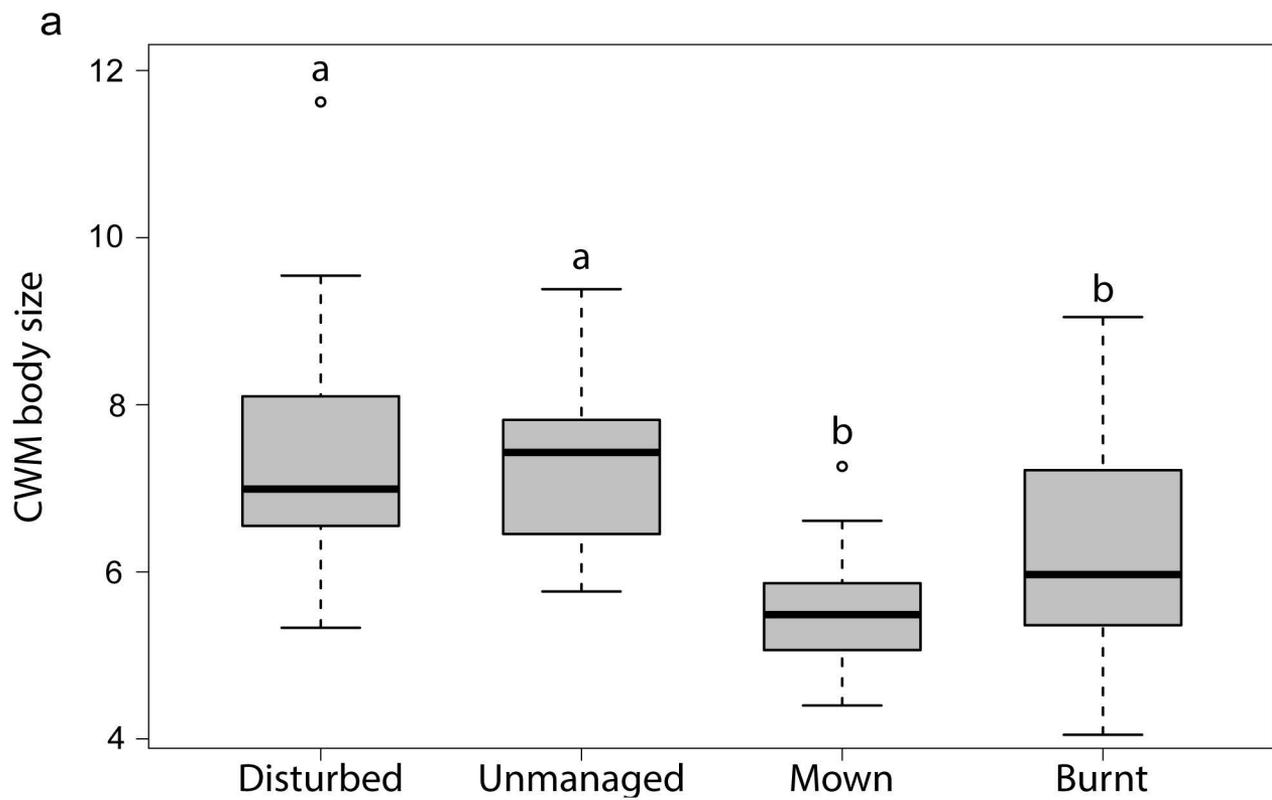


**a****b**





**a****b****c****d**





**Table 1.** Species richness, abundance of Red List species, conservation value, functional diversity and composition (mean values + SD) of ground-dwelling and vegetation-dwelling spiders in relation to individual management methods. Effects of management methods were tested using GLMMs and LMMs. Significant results are marked with boldface.

Measured indicators	Disturbance (N=12)	No-management (N=12)	Mowing (N=12)	Burning (N=12)	p
<b>Ground-dwelling 2017</b>					
Species richness	26.67 ± 7.46	26.00 ± 7.32	26.08 ± 5.32	27.0 ± 7.94	0.976
Red List species	22.67 ± 12.67	22.67 ± 14.99	22.17 ± 10.58	23.17 ± 13.37	0.965
Conservation value	10.75 ± 3.72	10.25 ± 2.99	10.42 ± 3.00	12.25 ± 4.07	0.477
FD (RaoQ)	0.11 ± 0.03	0.11 ± 0.02	0.12 ± 0.03	0.12 ± 0.04	0.406
CWM body size	7.24 ± 0.45	7.47 ± 0.36	7.51 ± 0.56	7.33 ± 0.27	0.166
CWM humidity	2.14 ± 0.26	1.96 ± 0.22	2.00 ± 0.19	2.05 ± 0.24	<b>0.036</b>
CWM hunting	1.04 ± 0.03	1.03 ± 0.04	1.03 ± 0.03	1.05 ± 0.05	0.580
<b>Ground-dwelling 2018</b>					
Species richness	19.42 ± 4.80	20.17 ± 4.39	21.17 ± 3.74	22.42 ± 4.64	0.406
Red List species	11.75 ± 9.16	16.92 ± 7.61	17.50 ± 9.46	23.33 ± 14.97	<b>&lt; 0.001</b>
Conservation value	7.33 ± 3.75	9.50 ± 3.15	10.50 ± 2.97	11.33 ± 3.63	<b>0.009</b>
FD (RaoQ)	0.10 ± 0.02	0.08 ± 0.02	0.09 ± 0.02	0.09 ± 0.01	<b>0.017</b>
CWM body size	7.94 ± 0.54	7.87 ± 0.43	7.87 ± 0.29	8.03 ± 0.46	0.522
CWM humidity	1.71 ± 0.25	1.51 ± 0.26	1.52 ± 0.13	1.55 ± 0.21	<b>0.002</b>
CWM hunting	1.02 ± 0.02	1.02 ± 0.02	1.02 ± 0.02	1.02 ± 0.01	0.974
<b>Vegetation-dwelling 2017</b>					
Species richness	2.08 ± 1.62	6.33 ± 3.17	3.08 ± 1.16	3.75 ± 1.60	<b>&lt; 0.001</b>
Red List species	0.42 ± 0.79	3.50 ± 2.94	1.17 ± 0.94	2.17 ± 2.04	<b>0.007</b>
Conservation value	0.58 ± 1.08	2.08 ± 1.31	1.25 ± 0.97	1.67 ± 1.44	<b>0.024</b>
FD (RaoQ)	0.08 ± 0.10	0.20 ± 0.08	0.13 ± 0.07	0.14 ± 0.05	<b>0.005</b>
CWM body size	5.49 ± 1.12	6.42 ± 1.39	6.10 ± 2.49	5.55 ± 1.32	0.377
CWM humidity	1.98 ± 0.39	2.14 ± 0.35	1.79 ± 0.40	1.91 ± 0.22	0.073
CWM hunting	1.87 ± 0.21	1.72 ± 0.16	1.65 ± 0.28	1.69 ± 0.27	0.087
<b>Vegetation-dwelling 2018</b>					
Species richness	3.75 ± 1.48	5.33 ± 1.16	3.58 ± 1.50	4.83 ± 1.33	0.118
Red List species	1.50 ± 1.68	3.08 ± 2.81	2.00 ± 2.70	2.83 ± 2.98	0.333
Conservation value	1.08 ± 1.16	2.42 ± 1.16	1.83 ± 1.40	1.92 ± 1.44	0.095
FD (RaoQ)	0.10 ± 0.04	0.16 ± 0.07	0.12 ± 0.09	0.13 ± 0.72	0.198
CWM body size	7.52 ± 1.70	7.31 ± 1.00	5.57 ± 0.78	6.29 ± 1.38	<b>&lt; 0.001</b>
CWM humidity	2.14 ± 0.26	2.17 ± 0.17	1.78 ± 0.14	2.09 ± 0.19	<b>&lt; 0.001</b>
CWM hunting	1.96 ± 0.05	1.88 ± 0.12	1.87 ± 0.11	1.87 ± 0.10	0.062

Site	Management	Plot	Vegetation cover (%)
1	Burning	1	72
1	Burning	2	80
1	Burning	3	68
1	Burning	4	78
1	Disturbance	1	30
1	Disturbance	2	60
1	Disturbance	3	70
1	Disturbance	4	50
1	Mowing	1	85
1	Mowing	2	80
1	Mowing	3	90
1	Mowing	4	90
1	No-management	1	92
1	No-management	2	92
1	No-management	3	92
1	No-management	4	75
2	Burning	1	63
2	Burning	2	74
2	Burning	3	70
2	Burning	4	70
2	Disturbance	1	62
2	Disturbance	2	21
2	Disturbance	3	53
2	Disturbance	4	30
2	Mowing	1	88
2	Mowing	2	85
2	Mowing	3	75
2	Mowing	4	60
2	No-management	1	90
2	No-management	2	90
2	No-management	3	73
2	No-management	4	89
3	Burning	1	65
3	Burning	2	75
3	Burning	3	70
3	Burning	4	70
3	Disturbance	1	30
3	Disturbance	2	28
3	Disturbance	3	30
3	Disturbance	4	40
3	Mowing	1	80
3	Mowing	2	70
3	Mowing	3	75
3	Mowing	4	60
3	No-management	1	88
3	No-management	2	75

3	No-management	3	80
3	No-management	4	92

<b>Vegetation height (cm)</b>	<b>Proportion of bare soil (%)</b>
35	28
33	20
32	32
40,6	20
31	67
33	37
34,4	30
30,6	47
15,8	15
13,8	20
10,6	16
11,4	10
51,4	5
49,6	5
46	5
34	25
34,4	37
40,8	25
37,6	27
45	30
51	37
42	79
35,6	46
30	70
11,4	12
14,2	15
11,4	18
9	40
57,6	15
66	8
41	7
44,2	8
32,6	55
31,4	30
42,2	30
27,4	31
30	62
35,8	82
30,2	55
38,6	30
11,8	18
10	29
11,4	25
9,6	15
41,6	12
41,6	15

41,8  
37,6

10  
6

Species	Site 1	Site 2	Site 3	Conservation status
<i>Aculepeira ceropegia</i>	45	65	26	ES
<i>Agroeca brunnea</i>	6	2	6	ES
<i>Agroeca cuprea</i>	20	18	27	LC
<i>Agroeca proxima</i>	9	3	1	ES
<i>Agyneta rurestris</i>	6	3	8	ES
<i>Agyneta saxatilis</i>	2	3	2	ES
<i>Allagelena gracilens</i>	1	0	2	ES
<i>Alopecosa cuneata</i>	1374	663	566	ES
<i>Alopecosa farinosa</i>	183	219	135	ES
<i>Alopecosa pulverulenta</i>	150	65	272	ES
<i>Alopecosa striatipes</i>	191	122	74	CR
<i>Alopecosa trabalis</i>	59	15	186	ES
<i>Araneus sturmi</i>	0	1	0	LC
<i>Argiope bruennichi</i>	9	2	2	ES
<i>Asagena phalerata</i>	10	3	8	ES
<i>Aulonia albimana</i>	16	1	52	ES
<i>Callobius claustrarius</i>	0	2	7	ES
<i>Ceratinella brevis</i>	1	2	0	ES
<i>Ceratinella scabrosa</i>	3	1	3	ES
<i>Cercidia prominens</i>	1	0	0	ES
<i>Cicurina cicur</i>	18	8	8	ES
<i>Civizelotes pygmaeus</i>	1	3	6	VU
<i>Clubiona neglecta</i>	1	5	1	ES
<i>Clubiona terrestris</i>	3	1	10	ES
<i>Coelotes terrestris</i>	3	1	8	ES
<i>Crustulina guttata</i>	0	0	1	ES
<i>Cybaeus angustiarum</i>	0	0	6	ES
<i>Dictyna arundinacea</i>	8	1	4	ES
<i>Diplocephalus cristatus</i>	2	1	5	ES
<i>Diplocephalus picinus</i>	1	1	0	ES
<i>Diplostyla concolor</i>	2	0	0	ES
<i>Drassodes lapidosus</i>	1	2	14	ES
<i>Drassodes pubescens</i>	62	45	31	ES
<i>Drassyllus lutetianus</i>	1	0	0	ES
<i>Drassyllus praeficus</i>	7	15	14	ES
<i>Drassyllus pumilus</i>	3	1	0	EN
<i>Drassyllus pusillus</i>	55	43	30	ES
<i>Dysdera moravica</i>	0	1	1	LC
<i>Enoplognatha latimana</i>	3	1	1	ES
<i>Enoplognatha thoracica</i>	2	3	2	ES
<i>Entelecara acuminata</i>	2	0	0	ES
<i>Erigone atra</i>	1	2	0	ES
<i>Erigone dentipalpis</i>	0	2	1	ES
<i>Euophrys frontalis</i>	0	1	0	ES

<i>Euryopis flavomaculata</i>	1	0	0	ES
<i>Evarcha arcuata</i>	3	1	7	ES
<i>Evarcha laetabunda</i>	68	42	39	VU
<i>Hahnia nava</i>	0	3	5	ES
<i>Haplodrassus dalmatensis</i>	1	2	1	VU
<i>Haplodrassus signifer</i>	106	106	111	ES
<i>Haplodrassus silvestris</i>	1	2	2	ES
<i>Haplodrassus soerenseni</i>	0	0	1	LC
<i>Harpactea hombergi</i>	0	0	1	ES
<i>Harpactea lepida</i>	2	0	2	ES
<i>Harpactea rubicunda</i>	1	0	2	ES
<i>Heliophanus cupreus</i>	1	1	2	ES
<i>Heliophanus flavipes</i>	12	6	17	ES
<i>Histopona torpida</i>	0	1	2	ES
<i>Hypsosinga albovittata</i>	18	44	26	LC
<i>Hypsosinga pygmaea</i>	1	0	0	LC
<i>Hypsosinga sanguinea</i>	10	11	1	ES
<i>Cheiracanthium campestre</i>	0	1	0	LC
<i>Cheiracanthium oncognathum</i>	1	0	0	VU
<i>Inermocoelotes inermis</i>	4	14	73	ES
<i>Lasaeola tristis</i>	1	0	0	LC
<i>Linyphia hortensis</i>	1	1	0	ES
<i>Linyphia triangularis</i>	1	0	2	ES
<i>Mangora acalypha</i>	164	130	245	ES
<i>Megalephyphantes pseudocollinus</i>	0	0	1	LC
<i>Metellina segmentata</i>	0	0	1	ES
<i>Micaria formicaria</i>	21	7	14	VU
<i>Micaria fulgens</i>	9	4	34	LC
<i>Micaria pulicaria</i>	0	1	6	ES
<i>Micrargus herbigradus</i>	1	1	1	ES
<i>Microlinyphia pusilla</i>	1	0	0	ES
<i>Micrommata virescens</i>	0	0	1	ES
<i>Misumena vatia</i>	1	0	2	ES
<i>Neottiura bimaculata</i>	3	0	0	ES
<i>Neottiura suaveolens</i>	2	0	0	VU
<i>Neriere emphana</i>	0	1	0	ES
<i>Neriere radiata</i>	0	1	0	ES
<i>Nigma flavescens</i>	1	1	0	ES
<i>Oxyopes ramosus</i>	0	0	1	LC
<i>Ozyptila atomaria</i>	2	2	3	ES
<i>Ozyptila claveata</i>	32	18	24	LC
<i>Ozyptila praticola</i>	0	0	1	ES
<i>Pachygnatha degeerii</i>	0	1	12	ES
<i>Pachygnatha listeri</i>	0	0	1	ES
<i>Palliduphantes pallidus</i>	5	3	1	ES
<i>Pardosa lugubris</i>	14	1	188	ES
<i>Pardosa palustris</i>	394	1186	304	ES

<i>Pardosa pullata</i>	22	22	6	ES
<i>Pardosa riparia</i>	77	42	144	ES
<i>Pelecopsis elongata</i>	4	1	0	LC
<i>Pelecopsis radiccicola</i>	2	1	1	ES
<i>Pellenes tripunctatus</i>	0	1	0	LC
<i>Philodromus albidus</i>	0	0	1	ES
<i>Philodromus cespitum</i>	0	2	0	ES
<i>Philodromus dispar</i>	1	0	1	ES
<i>Phlegra fasciata</i>	34	21	35	ES
<i>Phrurolithus festivus</i>	0	0	1	ES
<i>Phrurolithus minimus</i>	1	0	0	VU
<i>Phylloneta impressa</i>	37	47	5	ES
<i>Pisaura mirabilis</i>	8	5	19	ES
<i>Porrhomma errans</i>	3	0	0	ES
<i>Robertus arundineti</i>	2	0	1	ES
<i>Robertus lividus</i>	5	0	0	ES
<i>Robertus neglectus</i>	1	0	0	ES
<i>Scotophaeus quadripunctatus</i>	1	0	0	ES
<i>Segestria senoculata</i>	0	0	2	ES
<i>Sibianor aurocinctus</i>	1	0	0	ES
<i>Sittipub pubescens</i>	1	0	0	ES
<i>Stemonyphantes lineatus</i>	3	0	1	ES
<i>Synema globosum</i>	2	1	5	LC
<i>Talavera aequipes</i>	3	3	2	ES
<i>Talavera aperta</i>	1	0	0	LC
<i>Talavera petrensis</i>	5	4	3	VU
<i>Tapinocyba insecta</i>	1	0	0	ES
<i>Tegenaria campestris</i>	2	0	6	ES
<i>Tegenaria silvestris</i>	1	0	0	ES
<i>Tenuiphantes cristatus</i>	0	0	1	ES
<i>Tenuiphantes flavipes</i>	5	6	3	ES
<i>Tenuiphantes mengei</i>	1	0	0	ES
<i>Tenuiphantes tenuis</i>	2	2	0	ES
<i>Thanatus arenarius</i>	318	362	24	VU
<i>Thanatus formicinus</i>	101	39	34	LC
<i>Thanatus striatus</i>	1	0	0	LC
<i>Tibellus oblongus</i>	1	0	0	ES
<i>Tiso vagans</i>	1	2	2	ES
<i>Titanoeca quadriguttata</i>	0	0	1	ES
<i>Trachyzelotes pedestris</i>	4	2	16	ES
<i>Trichoncus affinis</i>	0	0	1	VU
<i>Trichopterna cito</i>	1	0	0	ES
<i>Trochosa ruricola</i>	20	5	55	ES
<i>Trochosa terricola</i>	42	9	138	ES
<i>Walckenaeria antica</i>	1	0	0	ES
<i>Walckenaeria atrotibialis</i>	1	1	0	ES
<i>Walckenaeria dysderoides</i>	1	0	0	ES

<i>Xerolycosa miniata</i>	13	41	26	ES
<i>Xerolycosa nemoralis</i>	5	5	3	ES
<i>Xysticus bifasciatus</i>	32	18	28	ES
<i>Xysticus cristatus</i>	75	62	36	ES
<i>Xysticus erraticus</i>	9	0	2	ES
<i>Xysticus kochi</i>	38	76	47	ES
<i>Xysticus lanio</i>	0	0	2	ES
<i>Zelotes aeneus</i>	9	6	2	LC
<i>Zelotes aurantiacus</i>	1	1	24	LC
<i>Zelotes electus</i>	84	86	9	LC
<i>Zelotes latreillei</i>	8	3	4	ES
<i>Zelotes longipes</i>	6	100	2	LC
<i>Zelotes petrensis</i>	71	75	93	ES
<i>Zora nemoralis</i>	1	1	0	ES
<i>Zora silvestris</i>	3	0	6	ES
<i>Zora spinimana</i>	4	2	5	ES

Degree of rareness	Hunting strategy	Humidity preference	Body size
VA	2	3	14,00
VA	1	3	7,70
S	1	1	4,45
S	1	3	6,35
VA	2	1	2,40
VA	2	3	2,15
A	2	3	10,00
VA	1	1	7,75
A	1	1	9,00
VA	1	3	9,25
VR	1	2	13,00
S	1	2	11,75
VA	2	3	5,25
A	2	3	15,95
A	2	2	5,10
A	1	1	4,25
A	2	4	10,50
VA	1	3	1,90
S	1	4	1,90
S	2	3	6,25
VA	1	4	6,00
R	1	1	2,00
A	1	2	7,00
VA	1	3	7,00
VA	2	3	11,50
A	2	2	2,25
A	2	4	8,00
VA	2	1	3,50
VA	1	3	2,10
VA	1	3	2,00
VA	2	4	2,75
VA	1	1	12,00
VA	1	1	7,45
A	1	4	6,25
A	1	1	6,50
R	1	1	3,90
A	1	2	4,75
R	1	1	7,50
S	2	2	5,00
A	2	2	4,45
A	1	3	2,20
VA	1	4	2,30
VA	1	4	2,30
A	1	2	3,80

A	2	3	3,75
VA	1	4	7,00
S	1	2	4,70
S	2	1	1,75
R	1	1	5,85
VA	1	1	8,45
A	1	3	8,40
S	1	3	6,30
A	1	3	4,50
VA	1	3	6,00
VA	1	2	10,00
A	1	1	5,20
A	1	1	4,50
VA	2	3	6,50
S	2	1	4,80
S	2	5	4,60
S	2	1	4,70
VR	1	2	7,90
R	1	1	11,50
VA	2	3	11,25
S	2	1	3,75
A	2	3	5,05
VA	2	3	6,00
VA	2	2	5,75
VR	2	1	5,00
VA	2	3	7,75
R	1	1	6,25
A	1	1	5,50
VA	1	3	3,60
VA	1	3	2,15
VA	2	3	4,25
VA	1	3	14,30
VA	1	3	9,25
VA	2	2	2,60
R	2	1	2,25
A	2	3	5,25
A	2	3	5,00
A	2	3	3,65
S	1	2	8,00
S	1	1	5,00
S	1	1	3,50
S	1	4	4,15
VA	2	3	3,85
VA	2	4	4,75
VA	2	2	1,95
VA	1	2	6,40
VA	1	3	6,50

VA	1	3	5,00
A	1	3	5,50
S	1	2	2,05
A	1	3	1,60
S	1	1	7,80
A	1	3	4,80
VA	1	3	5,30
S	1	3	5,05
A	1	1	6,45
VA	1	2	2,80
R	1	1	2,85
VA	2	2	4,50
VA	2	2	13,50
VR	2	1	2,75
A	2	3	2,38
VA	2	3	3,50
S	2	3	2,25
S	1	3	12,65
VA	2	3	8,50
A	1	1	3,40
VA	1	1	5,15
A	2	1	6,75
R	1	3	7,40
A	1	1	2,45
R	1	2	2,65
S	1	2	3,10
A	1	3	1,80
S	2	3	7,50
A	2	4	7,50
VA	2	4	2,75
VA	2	3	2,15
VA	2	3	2,05
A	2	3	3,15
R	1	1	7,00
A	1	1	9,45
A	1	4	5,35
S	1	1	9,00
A	1	3	2,10
A	2	1	5,00
S	1	2	8,00
R	1	3	2,45
S	1	1	1,65
VA	1	3	11,00
VA	1	3	10,50
VA	1	3	2,35
VA	1	3	2,60
VA	1	3	2,05

S	1	1	6,30
VA	1	2	6,25
VA	1	4	7,90
VA	1	3	6,35
A	1	1	7,00
VA	1	2	8,40
S	1	3	7,05
R	1	1	6,30
R	1	2	4,10
S	1	1	4,40
VA	1	3	7,40
R	1	1	6,40
A	1	2	6,00
A	1	3	4,70
A	1	2	3,75
VA	1	5	6,35

<b>Ground-dwelling spiders</b>				
	<b>Species</b>	<b>Indicator</b>	<b>p value</b>	<b>Year</b>
<b>Disturbance</b>	<i>Agroeca cuprea</i>	0,351	0,033	2017
	<i>Xerolycosa miniata</i>	0,367	0,031	2018
<b>Mowing</b>	<i>Haplodrassus signifer</i>	0,400	0,006	2017
<b>No-management</b>	<i>Agroeca cuprea</i>	0,370	0,020	2018
<b>Herb-dwelling spiders</b>				
	<b>Species</b>	<b>Indicator</b>	<b>p value</b>	<b>Year</b>
<b>No-management</b>	<i>Mangora acalypha</i>	0,468	0,009	2017
	<i>Aculepeira ceropegia</i>	0,355	0,009	2017
	<i>Aculepeira ceropegia</i>	0,420	0,011	2018
	<i>Argiope bruennichi</i>	0,333	0,014	2017

	<b>Variables</b>	<b>Estimate</b>	<b>z value</b>	<b>p value</b>
<b>Species richness</b>	management (N/D)	-0.026	-0.283	0.992
	management (M/D)	-0.022	-0.241	0.995
	management (B/D)	0.001	0.104	1.000
	management (M/N)	0.004	0.042	1.000
	management (B/N)	0.036	0.387	0.980
	management (B/M)	0.032	0.345	0.986
<b>Red List species</b>	management (N/D)	-0.087	-0.433	0.973
	management (M/D)	-0.072	-0.362	0.984
	management (B/D)	-0.014	-0.069	1.000
	management (M/N)	0.014	0.071	1.000
	management (B/N)	0.073	0.366	0.983
	management (B/M)	0.059	0.294	0.991
<b>Conservation value</b>	management (N/D)	-0.048	-0.378	0.982
	management (M/D)	-0.032	-0.251	0.994
	management (B/D)	0.131	1.083	0.700
	management (M/N)	0.016	0.127	0.999
	management (B/N)	0.178	1.459	0.463
	management (B/M)	0.162	1.332	0.542
<b>FD (RaoQ)</b>	management (N/D)	-0.001	-0.145	0.999
	management (M/D)	0.003	0.384	0.981
	management (B/D)	0.010	1.325	0.547
	management (M/N)	0.004	0.529	0.952
	management (B/N)	0.012	1.470	0.456
	management (B/M)	0.007	0.941	0.783
<b>CWM body size</b>	management (N/D)	0.232	1.616	0.370
	management (M/D)	0.271	1.892	0.231
	management (B/D)	0.083	0.581	0.938
	management (M/N)	0.040	0.276	0.993
	management (B/N)	-0.148	-1.034	0.729
	management (B/M)	-0.188	-1.311	0.556
<b>CWM humidity</b>	management (N/D)	-0.178	-2.715	<b>0.033</b>
	management (M/D)	-0.137	-2.082	0.159
	management (B/D)	-0.087	-1.331	0.543
	management (M/N)	0.041	0.633	0.921
	management (B/N)	0.090	1.384	0.509
	management (B/M)	0.049	0.751	0.876
<b>CWM hunting</b>	management (N/D)	-0.007	-0.600	0.932
	management (M/D)	-0.006	-0.509	0.957
	management (B/D)	0.006	0.562	0.943
	management (M/N)	0.001	0.093	1.000
	management (B/N)	0.013	1.168	0.647
	management (B/M)	0.012	1.069	0.709

	<b>Variables</b>	<b>Estimate</b>	<b>z value</b>	<b>p value</b>
<b>Species richness</b>	management (N/D)	0.038	0.413	0.976
	management (M/D)	0.086	0.951	0.777
	management (B/D)	0.144	1.605	0.375
	management (M/N)	0.048	0.539	0.950
	management (B/N)	0.106	1.194	0.631
	management (B/M)	0.057	0.656	0.914
<b>Red List species</b>	management (N/D)	0.405	2.545	0.053
	management (M/D)	0.414	2.619	<b>0.044</b>
	management (B/D)	0.708	4.590	<b>&lt; 0.001</b>
	management (M/N)	0.01	0.064	0.999
	management (B/N)	0.304	2.066	0.164
	management (B/M)	0.294	2.009	0.184
<b>Conservation value</b>	management (N/D)	0.259	1.824	0.261
	management (M/D)	0.359	2.584	<b>0.048</b>
	management (B/D)	0.435	3.182	<b>0.008</b>
	management (M/N)	0.1	0.774	0.866
	management (B/N)	0.176	janv-39	0.505
	management (B/M)	0.076	0.618	0.926
<b>FD (RaoQ)</b>	management (N/D)	-0.023	-3.412	<b>0.004</b>
	management (M/D)	-0.013	-1.982	0.195
	management (B/D)	-0.011	-1.669	0.340
	management (M/N)	0.01	1.444	0.472
	management (B/N)	0.119	1.756	0.295
	management (B/M)	0.002	0.299	0.991
<b>CWM body size</b>	management (N/D)	-0.067	-0.499	0.959
	management (M/D)	-0.068	-0.506	0.958
	management (B/D)	0.977	0.723	0.888
	management (M/N)	-0.001	-0.007	1.000
	management (B/N)	0.165	1.222	0.613
	management (B/M)	0.166	1.229	0.608
<b>CWM humidity</b>	management (N/D)	-0.195	-3.600	<b>0.002</b>
	management (M/D)	-0.186	-3.433	<b>0.003</b>
	management (B/D)	-0.162	-2.991	<b>0.015</b>
	management (M/N)	0.009	0.166	0.998
	management (B/N)	0.003	0.607	0.930
	management (B/M)	0.024	0.442	0.971
<b>CWM hunting</b>	management (N/D)	-0.003	-0.416	0.976
	management (M/D)	-0.001	-0.224	0.996
	management (B/D)	-0.001	-0.158	0.999
	management (M/N)	0.001	0.192	0.997
	management (B/N)	0.002	0.258	0.994
	management (B/M)	0.000	0.065	1.000

	<b>Variables</b>	<b>Estimate</b>	<b>z value</b>	<b>p value</b>
<b>Species richness</b>	management (N/D)	1.112	4.822	< <b>0.001</b>
	management (M/D)	0.392	1.514	0.424
	management (B/D)	0.588	2.356	0.084
	management (M/N)	-0.720	-3.591	<b>0.002</b>
	management (B/N)	-0.524	-2.786	<b>0.027</b>
	management (B/M)	0.196	0.882	0.812
<b>Red List species</b>	management (N/D)	2.041	3.623	<b>0.002</b>
	management (M/D)	0.900	1.546	0.400
	management (B/D)	1.517	2.605	<b>0.043</b>
	management (M/N)	-1.141	-2.616	<b>0.042</b>
	management (B/N)	-0.525	-1.591	0.373
	management (B/M)	0.616	1.353	0.519
<b>Conservation value</b>	management (N/D)	1.273	2.975	<b>0.015</b>
	management (M/D)	0.762	1.663	0.337
	management (B/D)	1.050	2.386	0.077
	management (M/N)	-0.511	-1.564	0.393
	management (B/N)	-0.223	-0.744	0.877
	management (B/M)	0.288	0.842	0.831
<b>FD (RaoQ)</b>	management (N/D)	0.116	3.543	<b>0.002</b>
	management (M/D)	0.046	1.399	0.500
	management (B/D)	0.062	1.901	0.228
	management (M/N)	-0.070	-2.144	0.139
	management (B/N)	-0.054	-1.642	0.355
	management (B/M)	0.016	0.502	0.959
<b>CWM body size</b>	management (N/D)	0.965	1.426	0.483
	management (M/D)	0.644	0.951	0.777
	management (B/D)	0.099	0.147	0.999
	management (M/N)	-0.321	-0.486	0.962
	management (B/N)	-0.865	-1.309	0.557
	management (B/M)	-0.544	-0.823	0.844
<b>CWM humidity</b>	management (N/D)	0.182	1.250	0.595
	management (M/D)	-0.180	-1.252	0.593
	management (B/D)	-0.066	-0.454	0.969
	management (M/N)	-0.362	-2.534	0.055
	management (B/N)	-0.248	-1.767	0.289
	management (B/M)	0.113	0.795	0.857
<b>CWM hunting</b>	management (N/D)	-0.160	-1.624	0.365
	management (M/D)	-0.231	-2.349	0.087
	management (B/D)	-0.184	-1.877	0.238
	management (M/N)	-0.071	-0.742	0.880
	management (B/N)	-0.025	-0.258	0.994
	management (B/M)	0.046	0.483	0.963

	<b>Variables</b>	<b>Estimate</b>	<b>z value</b>	<b>p value</b>
<b>Species richness</b>	management (N/D)	0.352	1.810	0.267
	management (M/D)	-0.045	-0.213	0.997
	management (B/D)	0.254	1.277	0.576
	management (M/N)	-0.398	-2.017	0.181
	management (B/N)	-0.098	-0.543	0.948
	management (B/M)	0.299	1.487	0.444
<b>Red List species</b>	management (N/D)	0.691	1.614	0.370
	management (M/D)	0.226	0.504	0.958
	management (B/D)	0.614	1.434	0.477
	management (M/N)	-0.465	-1.130	0.671
	management (B/N)	-0.076	-0.194	0.997
	management (B/M)	0.389	0.939	0.783
<b>Conservation value</b>	management (N/D)	0.802	2.404	0.075
	management (M/D)	0.526	1.504	0.433
	management (B/D)	0.571	1.644	0.351
	management (M/N)	-0.276	-0.977	0.761
	management (B/N)	-0.232	-0.830	0.839
	management (B/M)	0.044	0.149	0.999
<b>FD (RaoQ)</b>	management (N/D)	0.060	2.023	0.179
	management (M/D)	0.020	0.697	0.898
	management (B/D)	0.028	0.949	0.778
	management (M/N)	-0.039	-1.338	0.539
	management (B/N)	-0.032	-1.092	0.694
	management (B/M)	0.008	0.264	0.994
<b>CWM body size</b>	management (N/D)	-0.235	-0.412	0.976
	management (M/D)	-2.295	-4.493	<b>&lt; 0.001</b>
	management (B/D)	-1.558	-2.735	<b>0.031</b>
	management (M/N)	-2.060	-3.618	<b>0.002</b>
	management (B/N)	-1.323	-2.590	<b>0.047</b>
	management (B/M)	0.737	1.295	0.564
<b>CWM humidity</b>	management (N/D)	0.027	0.355	0.985
	management (M/D)	-0.364	-4.767	<b>&lt; 0.001</b>
	management (B/D)	-0.049	-0.647	0.917
	management (M/N)	-0.391	-5.122	<b>&lt; 0.001</b>
	management (B/N)	-0.077	-1.003	0.748
	management (B/M)	0.315	4.120	<b>&lt; 0.001</b>
<b>CWM hunting</b>	management (N/D)	-0.081	-1.955	0.205
	management (M/D)	-0.092	-2.212	0.120
	management (B/D)	-0.093	-2.232	0.115
	management (M/N)	-0.011	-0.257	0.994
	management (B/N)	-0.012	-0.277	0.993
	management (B/M)	-0.001	-0.020	1.000