

# Complementarity of rarity, specialisation and functional diversity metrics to assess community responses to environmental changes, using an example of spider communities in salt marshes

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## ► To cite this version:

Boris Leroy, Isabelle Le Viol, Julien Pétilion. Complementarity of rarity, specialisation and functional diversity metrics to assess community responses to environmental changes, using an example of spider communities in salt marshes. *Ecological Indicators*, Elsevier, 2014, 46, pp.351-357. 10.1016/j.ecolind.2014.06.037 . hal-01101385

**HAL Id: hal-01101385**

**<https://hal-univ-rennes1.archives-ouvertes.fr/hal-01101385>**

Submitted on 8 Jan 2015

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1           **Complementarity of rarity, specialisation and functional diversity**  
2           **metrics to assess community responses to environmental changes, using**  
3           **an example of spider communities in salt marshes**

4

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17 Number of words (abstract): 229

18 Number of words (full word file): 5180

19

20 **Abstract**

21 The study of community responses to environmental changes can be enhanced by the  
22 recent development of new metrics useful in applied conservation: relative rarity,  
23 ecological specialisation and functional diversity. These different metrics have been  
24 critically assessed independently, but are rarely combined in applied conservation  
25 studies, especially for less-studied taxa such as arthropods. Here we report how these  
26 different metrics can complement each other by using the response of spider  
27 communities to environmental changes in salt marshes as an example. Sampling took  
28 place using pitfall traps in salt marshes of the Mont St Michel Bay (France) during  
29 2004 and 2007. The sampling design was spatially replicated (3 plots per treatment  
30 and 4 traps per plot) and encompassed four habitat treatments: control, sheep  
31 grazing, cutting (annual, in summer) and invasion by the plant *Elymus athericus*. We  
32 observed contrasting responses of spider communities to the different treatments:  
33 grazing had a negative impact on both rarity and functional diversity but a positive  
34 impact on specialisation; cutting had a negative impact on the three metrics; and  
35 invasion only had a negative impact on rarity and specialisation. These contrasting  
36 responses emphasise the necessity of using different complementary community  
37 metrics in such conservation studies. Consequently, rarity-, specialization-, and  
38 functional-based indices should be applied simultaneously more frequently, as they  
39 potentially provide additional complementary information about communities. Such  
40 complementary information is the key to better-informed conservation choices.

41 **Key-words:** Index of Relative Rarity; Community Specialisation Index; Functional  
42 Divergence; Grazing; Cutting; Spiders.

44 **Abbreviations:**

45 CSI: Community Specialisation Index (Julliard et al., 2006)

46 FDiv: Functional Divergence Index (Villéger et al., 2008)

47 *I*<sub>RR</sub>: Index of Relative Rarity (Leroy et al. 2013)

48 SSI: Species Specialisation Index (Julliard et al., 2006)

49

## 50 1. Introduction

51 Addressing responses of biodiversity to environmental changes at the  
52 community rather than species level present the advantage of integrating the  
53 responses of multiple species and their interactions. Responses of communities can  
54 be assessed with different facets (taxonomic, phylogenetic or functional) for which  
55 several new indices have been developed (*e.g.*, Devictor et al., 2010; Meynard et al.,  
56 2011; Strecker et al., 2011). However, these different facets are still rarely used in  
57 applied conservation studies, particularly for less-studied taxa such as arthropods.  
58 The early methods used to assess communities in conservation studies were simple  
59 taxonomic diversity metrics such as species richness or abundance (*e.g.*, Prieto-  
60 Benítez and Méndez, 2011). However these simple metrics only reflect a fraction of  
61 the biodiversity and do not take into account the identity of species and their  
62 characteristics within and between communities, even though these aspects are  
63 crucial to assess biodiversity distribution ( $\alpha$ ,  $\beta$  components), conservation concerns  
64 (rarity), ecosystem functioning (functional traits) and importantly the processes  
65 implied in the impact of environmental changes on this biodiversity (biotic and  
66 functional homogenisation). Hence, species and communities were attributed values  
67 with respect to the conservation goal, for example according to their rarity, or more  
68 recently to their functional characteristics. In this study, we focus on recent  
69 methodological advances regarding three aspects: rarity, ecological specialisation and  
70 functional diversity.

71 Rarity primarily provides an insight into the facet of species biodiversity that is  
72 most at risk of extinction (Gaston, 1994), also with respect to the maintenance of  
73 vulnerable ecosystem functions (Mouillot et al., 2013). Different axes of rarity are  
74 usually considered: restricted abundance, restricted geographic distribution and

75 narrow niche breadth. The study of rarity in arthropod communities has recently  
76 been improved by the proposal of new indices based on species occurrence (Leroy et  
77 al., 2012), which provide the possibility of integrating multiple spatial scales (Leroy et  
78 al., 2013). Robust metrics have also been developed to assess the average niche width,  
79 *i.e.* the specialisation of species communities (Devictor et al., 2010a). Using basic  
80 predictions from the ecological niche theory, specialist species should indeed be  
81 favoured in rather stable environments whereas generalists should be more able to  
82 thrive in disturbed habitats (Levins, 1968). In the same way, species functional traits  
83 are increasingly taken into account to provide a better assessment of the functional  
84 responses of communities to environmental changes. The study of this facet of  
85 biodiversity was improved thanks to the proposal of novel metrics which have been  
86 developed and analysed critically (Devictor et al., 2010a; Villéger et al., 2008). The  
87 loss of functional diversity was a criterion that had been overlooked initially, but is of  
88 increasing concern in biological conservation (*e.g.*, Devictor et al., 2010b).

89 All these distinct approaches were successfully applied on arthropod  
90 communities (*e.g.*, Leroy et al., 2013; Penone et al., 2013) to assess their responses to  
91 environmental changes. They have each been critically assessed alone, but how they  
92 complement each other in the case of applied conservation remained to be tested. In  
93 this paper, we report a case study combining different recently developed metrics  
94 (rarity, specialisation, functional diversity) to assess how they complement each other  
95 to assess the response of arthropod communities to environmental changes, by using  
96 the example of spider communities of salt marshes. The environmental changes are  
97 here the replacement of natural vegetation of salt marshes by monospecific stands of  
98 the species *Elymus athericus* (Bockelmann and Neuhaus, 1999), and two  
99 management practices likely to limit the spread of this species: annual cutting and

100 sheep grazing. The impact of *E. athericus* will here be termed as an invasion in  
101 accordance with previous work on this species (Pétillon et al., 2005).

102         Salt marshes are of important conservation value because they host stenotopic  
103 species due to the constraining environmental conditions (Pétillon et al., 2011), and  
104 geographically rare species because of the restricted distribution of salt marshes in  
105 the western Palearctic (Leroy et al., 2013). In addition, salt marshes are subject to  
106 environmental changes (invasion by *E. athericus* and management) which often  
107 result in the replacement of the single dominant plant species by another (Veeneklaas  
108 et al., 2012). These changes in vegetation in turn modify the structure and  
109 composition of arthropod communities (Ford et al., 2012). The impacts of these  
110 environmental changes on salt marsh arthropods are still poorly understood, and  
111 results from scarce literature are often contradictory (*e.g.*, Rickert et al., 2012; van  
112 Klink et al., 2013). Consequently, we expected the application of distinct community  
113 metrics to provide new and complementary information, thus leading to a better  
114 understanding of how environmental changes impact communities. For that purpose,  
115 we compared the impacts of four treatments (control, invasion, cutting and grazing)  
116 on spider communities of salt marshes using community-level indices. Spiders were  
117 selected as a model group as they constitute one of the most abundant and diverse  
118 groups of arthropods in salt marshes (Pétillon et al., 2008) and for their well-known  
119 sensitivity to changes in habitat structure (*e.g.*, Marc et al., 1999).

120

121

## 122 2. Methods

### 123 2.1. Sampling design

124 The impacts of treatments on spider communities were investigated at two sites in  
125 the Mont-Saint-Michel Bay (NW France, 48°37'N, 1°34'W), 1 kilometre apart. Four  
126 treatments were investigated: control, invasion by *E. athericus*, vegetation cutting  
127 (once a year, in July) and grazing by sheep (on average 50 sheep/ha) (Pétillon et al.,  
128 2007). Treatments are representative of the main salt-marsh habitats of the Mont St-  
129 Michel Bay (Pétillon et al., 2007), and covered all together 89% of the 4054 hectares  
130 of salt marshes (*E. athericus*-invaded areas: 35%, cutting: 19%, sheep-grazing: 25%,  
131 natural vegetation: 10%, data from 2007, Valéry and Radureau, personal  
132 communication).

133 Spider communities were sampled between May and June in 2004 and 2007: the  
134 former with control, invasion and cutting treatments, and the latter with control,  
135 invasion and grazing treatments. The comparison between treatments was made in  
136 similar salt-marsh zones within each site and the only apparent varying factors (at the  
137 local and landscape scales) between plots were the presence/absence of management  
138 practices (cutting and grazing) or invasion by *E. athericus*.

139 The sampling protocol was designed to be comparable among treatments: within  
140 each site, each treatment was applied to three plots during the same sampling period.  
141 Plots had a surface area of 100m<sup>2</sup> and were spaced 100m apart. Within each plot,  
142 ground-dwelling spiders were sampled with four pitfall traps, set up regularly in a  
143 square grid and placed 10m apart, as this is the minimum distance to avoid  
144 interference between traps (Topping and Sunderland, 1992). Traps consisted of  
145 polypropylene cups (10 cm diameter, 17 cm deep) containing ethylene-glycol as a  
146 preservative. Traps were covered with a raised wooden roof to exclude the rain and



147 were visited weekly, tides permitting (i.e. three times per month during May and  
148 June). To summarise, there were 36 traps per site (3 treatments \* 3 plots \* 4 traps)  
149 and thus a total of 72 traps for the whole sampling protocol. To verify the impacts of  
150 treatments on vegetation, percentage covers of all plant species were estimated once  
151 within a radius of 1m around all traps.

## 152       2.2. Spider community-level indices

153 We calculated the average rarity, specialisation, and functional diversity of each  
154 community (pitfall trap) using species characteristics obtained from spider datasets  
155 (rarity, specialisation), and the literature (hunting strategy).

156 Data came from i) the western France spider database, and ii) the Catalogue of Spider  
157 Species from Europe and the Mediterranean Basin (both datasets were detailed in  
158 Leroy et al., 2013).

### 159           2.2.1. Multiscale Index of Rarity

160 For each spider species, we calculated rarity weights ( $w_{Mi}$ ) according to the method  
161 described in Leroy et al. (2013) (details in Appendix A). These rarity weights integrate  
162 information on the occurrence of species at two spatial scales: the western France  
163 scale and the western Palearctic scale. Weight values range from 0 to 2. The rarer the  
164 species, the higher the weights, with species which are rare at both scales receiving  
165 higher rarity weights than species which are rare at a single scale.

166 The Index of Relative Rarity ( $I_{RR}$ ) of each pitfall community was then calculated as  
167 the average weight of rarity of individuals of all the species of the considered  
168 community, and was subsequently normalized to values between 0 (no rare species in  
169 the community) and 1 (all individuals of the community belong to species with the

170 maximum rarity weight):  $I_{RR} = ([\sum(a_i \times w_{Mi})/N] - w_{\min})/(w_{\max} - w_{\min})$  where  $a_i$  and  
171  $w_{Mi}$  respectively are the abundance and rarity weight of the  $i^{th}$  species of the  
172 community;  $N$  is the total number of individuals in the community; and  $w_{\min}$  and  
173  $w_{\max}$  are the minimum and maximum possible weights, respectively.

#### 174 *2.2.2. Community Specialisation Index*

175 Each species was characterised for habitat specialisation through using the  
176 calculation of a Species Specialisation Index (SSI) according to Julliard et al. (2006).  
177 To define habitat specialisation, the western France spider database was used. For  
178 each species, the frequencies of occurrence in each habitat class were calculated (see  
179 details in Appendix B). The coefficient of variation of these frequencies of occurrence  
180 (standard deviation/average) as a measure of each Species Specialisation Index (SSI)  
181 (Julliard et al., 2006) was then calculated. The SSI varied between 0.76 (most  
182 generalist species) and 4.69 (most specialised species) (Table 1 and Appendix B). The  
183 Community Specialisation Index (CSI) of each pitfall community was calculated as  
184 the average SSI of the species detected, weighted by local species abundance, as  
185 follows:  $CSI = [\sum(a_i SSI_i)/N]$ , where  $a_i$  and  $SSI_i$  respectively are the abundance and  
186 species specialisation indices of species  $i$ ; and  $N$  is the total number of individuals in  
187 the community (Devictor et al., 2008). The CSI was then standardised between 0 and  
188 1 in a similar manner to the Index of Relative Rarity (Leroy et al., 2012); a CSI of 1  
189 means that a community is composed of individuals of the most specialised species of  
190 the database, whereas a CSI of 0 means that a community is composed of individuals  
191 of the most generalist species.

#### 192 *2.2.3. Functional diversity*

193 The functional diversity of spider communities was calculated on the basis of  
194 functional traits related to hunting modes, because of the important impacts of  
195 predator hunting modes in ecosystem functioning (Schmitz, 2009). We used  
196 functional traits at the family level on the basis of the trait matrix of spider families of  
197 Cardoso et al. (2011). The Functional Divergence (FDiv) index (Villéger et al., 2008)  
198 was calculated in order to take into account both the occupation of functional space  
199 by the different families and the relative abundance of families in communities.

### 200 *2.3. Analyses*

201 Because rarity weights and specialisation indices of species describe two axes of  
202 rarity, we first analysed their correlation for the sampled species with Pearson's  
203 correlation coefficient. Species richness and the three indices ( $I_{RR}$ , CSI and FDiv)  
204 were then calculated for each community of spiders of the 72 traps. The FDiv index  
205 was calculated for only 58 out of 72 communities, because functional divergence  
206 cannot be estimated when communities contain less than three functionally singular  
207 species. The effects of treatments on the calculated metrics were then analysed using  
208 linear mixed-effects models because of the hierarchical nature of the sampling design.  
209 We fitted linear mixed-effect models with the treatment as a fixed effect, and sites  
210 nested in year as random effects. The significance of fixed effects was tested by  
211 comparing models with vs. without fixed effects using likelihood ratio tests. We then  
212 performed multiple comparisons between treatments with Tukey's post-hoc tests  
213 with a correction on probability values by controlling the false discovery rate  
214 (Benjamini and Hochberg, 1995). Welsh two sample t-tests were used to compare the  
215 average abundance between treatments, of several species. Indices were calculated  
216 with Rarity version 1.2-1 (Leroy, 2013) and FD version 1.0-11 (Laliberté and Shipley,  
217 2011). Linear mixed effect models were performed in R 3.0.2 (R Core Team, 2013)

218 with the package lme4 version 1.0-5 (Bates et al., 2013), and post-hoc tests with the  
219 package multcomp version 1.3-1 (Hothorn et al., 2008).

220

## 221 **3. Results**

### 222 *3.1. Samplings*

223 A total of 3826 spiders, representing 31 species, 10 families and 6 functional groups,  
224 was collected (Table 1). The dominance of single plant species per treatment was  
225 confirmed: *Atriplex portulacoides* in the control plots (mean±sd=66.8±11.5%,  
226 range=[55%-88%], n=12), *Elymus athericus* in the invaded plots  
227 (mean±sd=89.4±7.3%, range=[70%-93%], n=12), *Festuca rubra* in the cut plots  
228 (mean±sd=62.9±5.4%, range=[55%-70%], n=12) and *Puccinellia maritima* in the  
229 grazed plots (mean±sd=77.5±21.4%, range=[30%-90%], n=12).

### 230 *3.2. Correlation between rarity weights and specialisation indices of species*

231 We observed a positive correlation between rarity weights and specialisation indices  
232 of species (Pearson's  $R = 0.75$ ;  $P < 0.001$ ;  $N = 31$ ) (Fig. 1). Nevertheless, some species  
233 had very high specialisation values but low rarity weights (*e.g.*, *Erigone longipalpis*),  
234 or had very high rarity weights but intermediate specialisation values (*e.g.*, *Pardosa*  
235 *purbeckensis*).

### 236 *3.3. Effects of treatments on communities*

237 We detected a significant treatment effect on all the measured metrics (see Figure 1  
238 and details in Appendix B). The species richness was significantly lower in traps  
239 located in invaded plots than in traps associated with other treatments ( $\chi^2=13.5$ ,  
240  $df=3$ ,  $p=0.004$ ) (Fig. 2A). The relative rarity of communities was significantly higher  
241 in control plot traps than in those located in managed or invaded plots ( $\chi^2=24.1$ ,  
242  $df=3$ ,  $p<0.001$ ) (Fig. 2B). This increase in rarity was associated with a higher mean  
243 abundance of the rare species *P. purbeckensis* in the control treatment than in other  
244 treatments ( $t=5.78$ ,  $df=31.50$ ,  $p<0.001$ ) (Table 1). Conversely, although the CSI was

245 significantly higher in the control treatment than under invasion or cutting  
246 treatments, the highest CSI was observed in the grazed plots, and was significantly  
247 higher than in control and in invaded or cut plots ( $\chi^2=24.40$ ,  $df=3$ ,  $p<0.001$ ) (Fig.  
248 2C). The higher specialisation under the control compared to invasion and cutting  
249 treatments was associated with higher average abundance of specialised species such  
250 as *P. purbeckensis* ( $t=6.60$ ,  $df=30.22$ ,  $p<0.001$ ) and *Silometopus ambiguus* ( $t=2.90$ ,  
251  $df=33.08$ ,  $p=0.007$ ) (Table 1). On the other hand, the very high specialisation value  
252 observed under the grazing treatment was associated to the high abundance of the  
253 specialised species *E. longipalpis*, with on average  $13.83\pm 9.76$  individuals per trap  
254 versus  $1.13\pm 2.05$  individuals per trap (mean $\pm$ sd) in other treatments ( $t=4.49$ ,  
255  $df=11.20$ ,  $p<0.001$ ). However, grazing, and to a lesser extent cutting, appeared to  
256 have a negative impact on the functional divergence of communities, with a  
257 significantly lower FDiv in traps of grazed plots than in those of invaded or control  
258 traps ( $\chi^2=10.4$ ,  $df=3$ ,  $p=0.016$ ) (Fig. 2D). We compared this lower functional  
259 diversity to the presence of functional groups under the grazing treatment, and we  
260 observed only two dominant functional groups (“Ground hunters” and “Other  
261 hunters”), while other treatments were more balanced, with an additional functional  
262 group (“Orb web weavers”) (Table 1).

263

#### 264 **4. Discussion**

265 In this paper, we combined different metrics (rarity, specialisation and functional  
266 diversity) to analyse how spider communities of salt marshes respond to changes in  
267 their environment, whether induced by a biological invasion (*Elymus athericus*) or by  
268 management practices (grazing, cutting). As expected, environmental changes  
269 generally had negative impacts on spider communities, which led to decreases in  
270 rarity, specialisation and functional diversity of communities. While the general trend  
271 was similar among metrics, we observed divergent responses to the relative impacts  
272 of treatments between the three measured metrics.

##### 273 *Influence of community composition on metrics*

274 In particular, the most interesting divergence was observed for the effect of grazing  
275 treatment. Specialisation was higher in grazed plots than in control plots, whereas the  
276 opposite was observed for rarity and functional diversity. This is surprising given the  
277 observed positive relationship between rarity weights and specialisation indices at the  
278 species level. However, this relationship has exceptions such as the widespread but  
279 specialist species *E. longipalpis*. Incidentally, *E. longipalpis* was highly abundant in  
280 the grazed plot, leading to a divergence between specialisation and rarity indices. This  
281 pattern shows that metrics expected to vary conjointly can produce divergent  
282 outcomes because of their sensitivity to the composition of communities. This  
283 sensitivity is very important to highlight unexpected impacts of environmental  
284 changes on communities.

285 Another important property of the metrics used here is that they are not biased by the  
286 species richness or abundance: because the metrics are based on the average trait  
287 (rarity, specialisation) of the species in the community, then a species-poor  
288 community *can* have a higher value than a species-rich community (Julliard et al.,

289 2006; Leroy et al., 2012; Villéger et al., 2008). If we take the example of the CSI, a  
290 community with only a few individuals from very specialised species will have a  
291 higher CSI than a community with many individuals of generalist species. Because of  
292 this averaging property, a decrease in CSI may result from a decrease of specialists  
293 and/or from an increase of generalist species. Typically disturbances are expected to  
294 negatively impact specialist species while fostering generalist species, as both  
295 predicted by theory and empirically observed (see Devictor and Robert, 2009). As a  
296 consequence it is expected that changes in community composition resulting from  
297 environmental disturbances will generally result in lower CSI values. However, in  
298 extreme cases, the reverse might be observed: for example a disturbance  
299 corresponding to the extreme values of the abiotic gradient may have positive  
300 impacts on only a few specialist species with particular adaptation to this disturbance  
301 (*e.g.*, Doxa et al., 2010), which would thus result in a higher CSI value. Though this is  
302 a particular case, it highlights the importance of verifying, as a second step, how  
303 community composition changed. This impact is similar on the functional divergence  
304 index: the loss of species occupying intermediate functions will increase the  
305 functional value. However, this is not relevant in our case study since spider  
306 functional groups are strongly divergent; thus, a decrease in abundance of any  
307 functional group will decrease the functional value.

308 To summarise, the three applied metrics accurately synthesise how changes in  
309 community composition affect different facets of biodiversity; such changes cannot  
310 be detected by species richness alone. However, understanding these changes imply  
311 to look at changes in community composition as a second step.

### 312 *Implications for salt marsh conservation*

313 The indices showed that grazing induced a decrease in the average rarity and  
314 functional diversity of communities, but an increase in their specialisation. This



315 pattern is mainly due to the increase in widespread but specialist species such as *E.*  
316 *longipalpis*. This positive impact of grazing on specialist species of salt marshes such  
317 as *E. longipalpis* was reported by Ford et al. (2012), although they did not investigate  
318 species rarity. The negative effect of grazing, and to a lesser extent cutting, on  
319 functional diversity is clearly due to a reduction in the number of vegetation strata,  
320 which negatively impacts spiders living in higher vegetation such as orb web weavers  
321 (Uetz et al., 1999). On the other hand, no significant impact of grazing was detected  
322 on species richness alone, unlike Rickert et al. (2012) and van Klink et al. (2013). In  
323 fact, a negative impact on species richness was only observed for the invaded  
324 treatment. We also found higher specialisation and rarity in control plots than in  
325 invaded and cut plots, in accordance with a previous study that revealed more  
326 continental, non-specialised, spider species in cut or invaded salt marshes (Pétillon et  
327 al., 2005).

328 To summarise, our results suggest that undisturbed salt marshes generally support a  
329 higher specialisation, rarity and functional diversity than disturbed salt marshes, with  
330 the notable exception of the positive impact of grazing on specialised species. To  
331 preserve all the facets of spider diversity, we would recommend maintaining the  
332 grazing management in areas invaded by *E. athericus* rather than the cutting  
333 treatment.

### 334 *Conclusion*

335 We therefore showed contrasting responses of spider communities to the different  
336 environmental changes (invasion and management measures). These contrasts  
337 emphasise the necessity of using different and complementary community metrics in  
338 such conservation studies. As a consequence, rarity-, specialization- and functional-  
339 based indices should be more frequently applied together, as they can potentially

340 contribute a wide range of complementary information about species communities.

341 Such complementary information is the key to better-informed conservation choices.

### 342 **Acknowledgements**

343 We thank Simon Potier, Alain Radureau and Vincent Schricke for fruitful discussion,

344 Damien Nougès and Mathieu de Flores for their contribution in field work and

345 spider identification. Aldyth Nys provided useful editing of the English of an earlier

346 draft. Loïc Valéry provided data on the spatial distribution of treatments. We thank

347 two anonymous reviewers for useful additions in an earlier draft.

348

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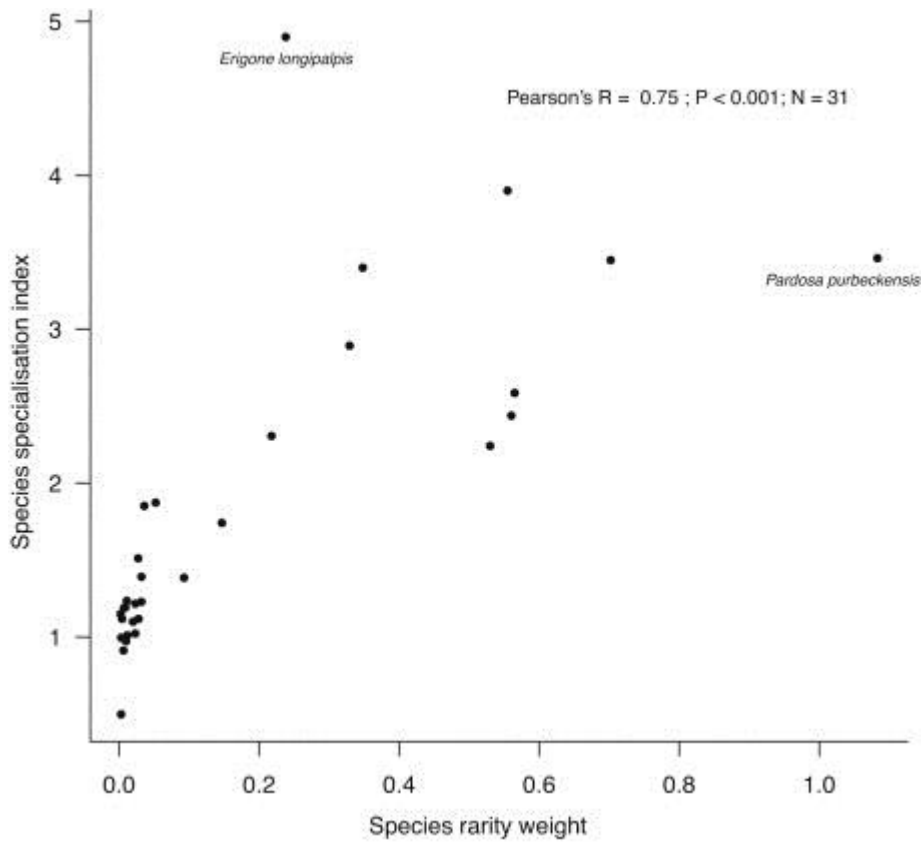
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**Table 1.** Multiscale rarity weights, specialisation indices and functional groups of the sampled spider species, and their average  $\pm$  sd abundance in pitfall traps. The functional groups are based on Cardoso *et al.* (2011).

Species	Multiscale rarity weights	Species specialisation indices	Functional groups	Abundance $\pm$ sd			
				Control	Invasion	Cutting	Grazing
Family Araneidae							
<i>Argiope bruennichi</i>	0.009	1.272	Orb web weavers	0.00 $\pm$ 0.00	0.08 $\pm$ 0.28	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Larinioides cornutus</i>	0.004	1.225	Orb web weavers	0.04 $\pm$ 0.20	0.08 $\pm$ 0.28	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Neoscona adianta</i>	0.002	1.271	Orb web weavers	0.00 $\pm$ 0.00	0.04 $\pm$ 0.20	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
Family Clubionidae							
<i>Clubiona stagnatilis</i>	0.144	1.877	Other hunters	0.00 $\pm$ 0.00	0.04 $\pm$ 0.20	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
Family Dictynidae							
<i>Argenna patula</i>	0.344	3.247	Ground hunters	0.71 $\pm$ 1.16	3.29 $\pm$ 4.22	0.83 $\pm$ 0.83	0.92 $\pm$ 1.00
Family Gnaphosidae							
<i>Drassyllus pusillus</i>	0.031	1.429	Ground hunters	0.04 $\pm$ 0.20	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Silometopus ambiguus</i>	0.697	3.164	Other hunters	3.29 $\pm$ 3.38	1.58 $\pm$ 2.21	0.08 $\pm$ 0.29	2.33 $\pm$ 1.97
<i>Zelotes latreillei</i>	0.026	1.749	Ground hunters	0.00 $\pm$ 0.00	0.08 $\pm$ 0.41	0.08 $\pm$ 0.29	0.00 $\pm$ 0.00
Family Linyphiidae							
<i>Agyneta conigera</i>	0.551	4.286	Sheet web weavers	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00	0.08 $\pm$ 0.29	0.00 $\pm$ 0.00
<i>Bathyphantes gracilis</i>	0.012	1.368	Sheet web weavers	0.21 $\pm$ 0.51	0.08 $\pm$ 0.28	0.17 $\pm$ 0.39	0.08 $\pm$ 0.29
<i>Collinsia inerrans</i>	0.525	2.222	Other hunters	0.04 $\pm$ 0.20	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Erigone atra</i>	0.010	1.346	Other hunters	0.46 $\pm$ 0.78	0.04 $\pm$ 0.20	1.50 $\pm$ 1.62	0.67 $\pm$ 0.89
<i>Erigone dentipalpis</i>	0.003	1.195	Other hunters	0.17 $\pm$ 0.38	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00
<i>Erigone longipalpis</i>	0.234	4.690	Other hunters	2.50 $\pm$ 2.69	0.25 $\pm$ 0.53	0.17 $\pm$ 0.39	13.83 $\pm$ 9.76
<i>Oedothorax fuscus</i>	0.022	1.224	Other hunters	4.83 $\pm$ 6.68	0.04 $\pm$ 0.20	4.25 $\pm$ 5.14	2.50 $\pm$ 1.24
<i>Oedothorax retusus</i>	0.028	1.353	Other hunters	0.58 $\pm$ 0.78	0.04 $\pm$ 0.20	5.50 $\pm$ 5.89	0.25 $\pm$ 0.45
<i>Pelecopsis parallela</i>	0.051	1.830	Other hunters	0.00 $\pm$ 0.00	0.33 $\pm$ 0.82	0.42 $\pm$ 0.67	0.00 $\pm$ 0.00
<i>Savignia frontata</i>	0.325	3.720	Other hunters	0.00 $\pm$ 0.00	0.00 $\pm$ 0.00	0.08 $\pm$ 0.29	0.00 $\pm$ 0.00

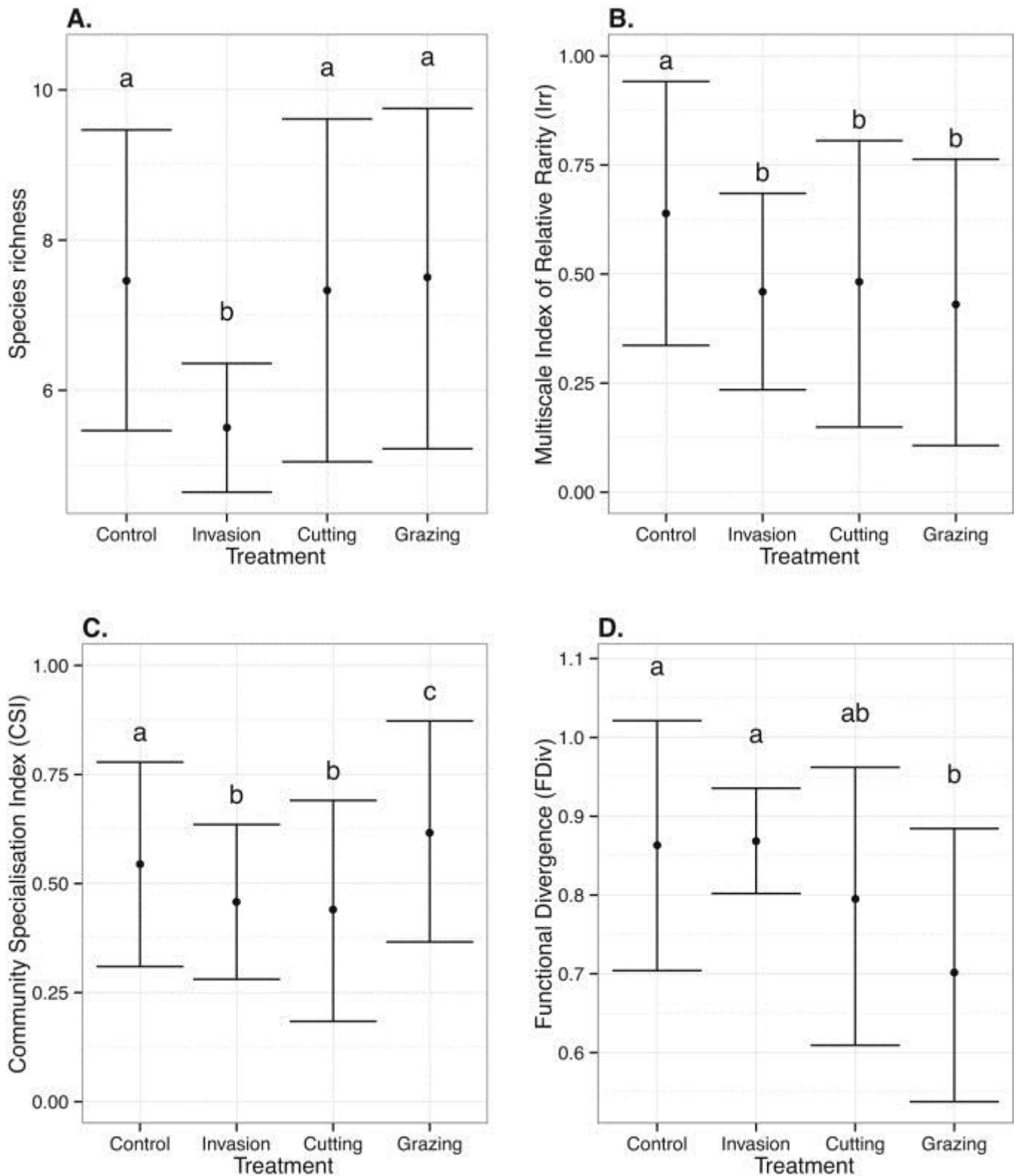
<i>Stemonyphantes lineatus</i>	0.031	1.424	Other hunters	0.04±0.20	0.13±0.34	0.00±0.00	0.00±0.00
<i>Tenuiphantes tenuis</i>	0.003	0.757	Sheet web weavers	0.75±1.03	0.75±1.11	0.50±0.67	1.33±0.78
Family Liocranidae							
<i>Agroeca lusatica</i>	0.555	2.442	Ground hunters	0.33±0.56	0.42±0.83	0.08±0.29	0.00±0.00
Family Lycosidae							
<i>Alopecosa pulverulenta</i>	0.007	1.464	Ground hunters	0.00±0.00	0.08±0.28	0.00±0.00	0.00±0.00
<i>Arctosa fulvolineata</i>	0.560	2.462	Ground hunters	3.13±2.42	2.17±1.97	4.00±3.49	2.00±1.28
<i>Pardosa palustris</i>	0.035	1.785	Ground hunters	0.04±0.20	0.00±0.00	0.08±0.29	0.08±0.29
<i>Pardosa prativaga</i>	0.023	2.046	Ground hunters	0.04±0.20	0.04±0.20	0.00±0.00	0.00±0.00
<i>Pardosa proxima</i>	0.019	1.507	Ground hunters	0.00±0.00	0.08±0.28	0.17±0.39	0.00±0.00
<i>Pardosa purbeckensis</i>	1.077	3.307	Ground hunters	21.13±10.11	6.17±5.26	7.17±4.13	13.17±6.67
Family Tetragnathidae							
<i>Pachygnatha clercki</i>	0.010	1.545	Orb web weavers	0.00±0.00	0.00±0.00	0.33±0.65	0.00±0.00
<i>Pachygnatha degeeri</i>	0.006	0.888	Orb web weavers	2.54±2.28	2.83±4.57	1.83±2.29	0.33±0.65
Family Theridiidae							
<i>Enoplognatha mordax</i>	0.214	2.375	Space web weavers	0.08±0.28	0.04±0.20	0.08±0.29	0.08±0.29
Family Thomisidae							
<i>Ozyptila simplex</i>	0.091	1.675	Ambush hunters	0.04±0.20	0.00±0.00	0.00±0.00	0.08±0.29
Total abundances				41.00±14.63	18.71±11.96	27.42±11.77	37.67±10.53

## Figure captions



**Figure 1.** Correlation between specialisation indices and rarity weights of the sampled species. Two species highly abundant in some of the sampled sites and with high weight values are indicated: *Erigone longipalpis* and *Pardosa purbeckensis*.





**Figure 2.** Average values of indices (A. Species richness, B. Multiscale Index of Relative Rarity, C. Community Specialisation Index, and D. Functional Divergence) and 95% confidence intervals (estimated by linear mixed-effects models (LMMs)) for the four treatments. Different successive letters indicate significant differences in means (revealed by corrected Tukey's post-hoc test on treatment effect following LMMs, with correction for the false discovery rate).