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The effect of forest management on endangered insects assessed by radio-tracking: The case of the ground beetle *Carabus olympiae* in European beech *Fagus sylvatica* stands

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39 **Abstract**

40

41 Beech forests are important for biodiversity conservation in Europe and studies to identify
42 sustainable forest management practices are therefore required. The ground beetle *Carabus*
43 *olympiae* Sella, 1855, is a large steno-endemic endangered alpine species with very restricted
44 ranges. Its known range is only delimited in two beech forests in the western Italian Alps where
45 beech wood is still harvested. Forty individuals were collected and radio-tracked in 2014-2015 in
46 order to assess the effects of forest management on microclimatic conditions, microhabitat use and
47 movements.

48 Regarding microhabitat selection deadwood and tree bases were preferred, and were used as refuges
49 during the daytime. Bare ground was not used. The length of the path travelled by individual insects
50 was more variable and the tortuosity was lower in managed than in unmanaged stands, suggesting
51 that management induced more constrained trajectories and variable distances.

52 We concluded that logging may exert short-term negative effects on *C. olympiae* ground beetles.

53 However, the preference for tree bases and deadwood suggests that forest management,
54 concurrently, may also be beneficial, on the condition that: *i*) the coppice, which provides more
55 suitable microhabitats, prevails over conversion to high forest, and *ii*) deadwood originating from
56 cutting (branches and treetops) is properly accumulated.

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58

59 **Introduction**

60

61 European beech (*Fagus sylvatica* L.) forests represent a unique ecosystem, stretching from the
62 Mediterranean regions and southeastern Europe to the British Isles and Scandinavia, comprising
63 vast areas in the Alps, Balkans and the Carpathians as well as in the northern European lowlands.
64 The special history and ongoing ecological processes related to beech forests has been
65 acknowledged by the establishment of the transnational UNESCO World Heritage site "Primeval
66 Beech Forests of the Carpathians and the Ancient Beech Forests of Germany", instigated in 2007
67 and 2011. Apart from old-growth remnants, even secondary or managed beech forests harbour
68 relevant ecosystem processes, animal and plant species of conservation interest, and provide an
69 immense array of ecosystem services to man. For these reasons, they feature prominently in the
70 Habitats Directive of the European Commission, Annex I (EU Council 1992), which lists five
71 different beech forest types that deserve conservation at the EU level. Beech (*Fagus sylvatica* L.)
72 forests are important for many autotrophic and heterotrophic organisms like soil macrofungi,
73 ground dwelling arthropods, land snails, saproxylic fungi, hole nesting birds, saproxylic insects,
74 epiphytic lichens and bryophytes and epixylic bryophytes (Brunet et al. 2010, Larrieu et al. 2014a;
75 Larrieu et al. 2014b). European conservation policy also takes into account beech forests. The
76 Annex 1 of the "Habitats Directive" (92/43/EEC) lists eight beech habitat types as worthy of
77 conservation. Current threats to these ecosystems comprise climate change (Gessler et al., 2007; Di
78 Filippo et al., 2012), increased possibility of fire and drought damage (Piovesan et al., 2008; Ascoli
79 et al., 2013), habitat loss and fragmentation (Kunstler et al., 2007), species invasion (Krumm and
80 Vítková 2016) and grazing by domestic or wild ungulates (Vandenberghhe et al., 2007; Olesen and
81 Madsen, 2008). Forestry practices can also affect biodiversity both positively and negatively, and
82 studies focused on the impacts of beech forest management on animal diversity (e.g. Laiolo et al.
83 2004; Kappes 2005 and 2006; Müller et al. 2007; Moning and Müller, 2009; Floren et al. 2014;
84 Chumak et al., 2015) are necessary to identify the practices which are compatible with animal

85 conservation. A case in point are the beech forests of many mountain areas in Italy (Nocentini,
86 2009). Most of them were traditionally managed as coppice, i.e., by repeatedly cutting back sprouts
87 to ground level to stimulate vegetative growth and provide firewood on a short rotation basis (20 to
88 40 years). High forests, i.e., stands where trees regenerate by seed, are rare. However, many
89 coppices are now transitioning to a high-forest structure, due to either abandonment of regular
90 management, or silvicultural conversion by thinning (Nocentini, 2009), yet the impacts of such
91 management changes on animal biodiversity are not fully understood.

92 Ground beetles (Coleoptera: Carabidae) are typical inhabitants of European beech forests. Carabids
93 show a wide range of life history traits and microhabitat requirements, and therefore they have been
94 widely used as biological model of forest management (Rainio and Niemelä, 2003). They are
95 relatively easy and cost-efficient to sample with standardized methods (i.e., pitfall trapping), and are
96 sensitive to environmental factors such as temperature, humidity and vegetation structure (Stork,
97 1990; Butterfield, 1996; Lövei and Sunderland, 1996). Most studies on carabids in different forest
98 habitats have focused on habitat fragmentation (Davies and Margules, 1998; Niemelä, 2001;
99 Koivula and Vermeulen, 2005) and edge effects (Heliölä et al., 2001; Koivula et al., 2004; Negro et
100 al., 2009). The consequences of forestry practices on ground beetle diversity have also been
101 extensively studied in forests (Werner and Raffa, 2000; du Bus de Warnaffe and Lebrun, 2004;
102 Pearce and Venier, 2006; Taboada et al., 2006, Baker et al., 2009; Lange et al., 2014; Skłodowski
103 2014; Macko, 2016). Habitat modification and destruction may be responsible for species decline,
104 which affects particularly large-sized and brachypterous (short or reduced wings) ground beetles,
105 because of their limited dispersal capacity (Kotze and O'Hara, 2003). A common trend in carabids
106 is that populations of large, poorly dispersing species decrease with increasing disturbance and
107 forest management intensity (Rainio and Niemela 2003, Kotze and O'Hara, 2003, Paillet et al.
108 2010). Several ground beetle species are threatened (Assmann and Janssen 1999; Kotze and
109 O'Hara, 2003, Brockerhoff et al., 2005; Matern et al. 2007; Pokluda et al., 2012), despite that, for

110 most species, the status of populations is not known, and therefore field studies are needed in order
111 to develop the most appropriate conservation measures.

112 The species *Carabus olympiae* Sella, 1855, is an endangered alpine species with a very restricted
113 range, limited to two neighboring beech forests (a few hectares each) in the western Italian Alps.
114 After the alarming population decline suffered in the years 1930-42 (Malausa et al., 1983), *C.*
115 *olympiae* has been declared a *priority species* (i.e. a species for the conservation of which the
116 European Community has a particular responsibility) and included in Annexes II and IV of the EU
117 Habitats Directive. It is listed in international agreements such as the Bern Convention, and is
118 considered Vulnerable according to the IUCN red list of Threatened species
119 (<http://www.iucnredlist.org/>). Despite this protection framework, *C. olympiae* still faces threats due
120 to the construction of downhill skiing facilities (Negro et al. 2009, 2010, 2013) and to forestry
121 practices. In addition to removing trees, logging usually alters the shrub, herbaceous, and litter
122 layers. The period immediately after logging may therefore be dangerous for forest ground beetles,
123 which are deprived of protection by the canopy and understory layers. Previous research focused on
124 local ground beetle diversity, and tentatively indicated that the best option to protect this species
125 was low-intensity or no management (Negro et al. 2007, 2013), particularly avoiding thinning that
126 results in a more even tree size distribution (Negro et al., 2014).

127 We hypothesize that bare soil should be avoided by *C. olympiae*, and that the amount of deadwood
128 and dense field vegetation could determine its movements within the study area.

129 We radio tracked *C. olympiae* individuals in multiple stands that differed in terms of forest
130 management and certain structural elements of forests to evaluate their effects on microhabitat use
131 and movements of these beetles to identify those interventions that are compatible with and,
132 possibly favourable to, the conservation of *C. olympiae*.

133 We assessed the short-term effects of different types of logging practices on: *i*) local microclimatic
134 conditions, *ii*) habitat use and *iii*) movements of radio-tagged individuals.

135

136 **Study area**

137

138 The study was carried out within the upper Sessera Valley, in the North-western Italian Alps
139 (45°40' N; 8°16'E). The area is classified as Site of Community Importance (SCI, IT113002, Val
140 Sessera) with the purpose of protecting *C. olympiae*. It included the upper part of the River Sessera
141 basin, a mountainous catchment, from the valley bottom up to an elevation of 2556 m a.s.l. (average
142 elevation: 1350 m). Annual rainfall is 1700 mm with two equinoctial maxima, and mean annual
143 temperature is 7°C. Snow cover lasts about 5 months (November to March). The most common
144 land cover classes are pasture, shrubland (Alpine rose *Rhododendron ferrugineum* L. and bilberry
145 *Vaccinium myrtillus* L.), secondary forest on former pastures, coniferous plantation, and beech
146 forest (belonging to the association *Luzulo-Fagetum*). In the study area, Alpine rose shrubland is
147 characterized by sparse isolated patches as the result of historical forest cutting and pastoral
148 practices. Beech is traditionally managed as coppice-with-standards (i.e., trees retained to provide a
149 seed source for stump replacement over time) to produce firewood and charcoal. Over recent
150 decades, forest management has been progressively reduced. The last harvest in privately owned
151 coppice stands was carried out in 1960; the density of standards is on average 100 trees per hectare,
152 and they are now about 80 years old. On the other hand, most coppices on public properties have
153 been actively converted to high forest in the 1980s. Conversion has been carried out by progressive
154 thinning (3-4- entries) of sprouts (Giannini and Piussi, 1976), before reaching the final step of
155 “temporary high forest”, i.e., a forest that has the structure of a mono-layered high forest, but
156 which originated from sprouting, to which the seeding cut will be eventually applied. Most of the
157 coppices in conversion are currently between the second and the third thinning, and the trees are
158 70–75 years old, with some trees > 100 years old.

159

160

161 **Methods**

162

163 **Experimental forest management**

164 Experimental forest management was carried out in 2014 and 2015 in seven different stands, with
165 the aim of reproducing “business as usual” as well as alternative management options for beech
166 coppices and high forests (Fig. 1). Forest structure and ground cover were measured before and
167 after logging across 30 sampling plots (details in Negro et al., 2014). In all logging practices
168 adopted (interventions *ii*, *iii* and *iv*) the retention of coarse woody debris was prescribed and the
169 following four treatments were applied:

170

171 *i)* no logging (control, C), only in over-mature coppices (two stands, one in 2014 and one in
172 2015, 3 hectares each). These had an average stem density of 2029 trees ha⁻¹, mean canopy
173 cover of 89%, and mean living aboveground tree volume of 207 m³ ha⁻¹. Ground cover by
174 deadwood (downed logs and branches, including those occurring naturally and 1-2 m wide
175 piles that had been stacked at the latest silvicultural entry) was in the range of 0% to 10%
176 (Negro et al., 2014);

177

178 *ii)* strip cuts in the high forest (one stand in 2015, 10 hectares) (management type 1, M1). This
179 had an average stem density of 915 trees ha⁻¹, mean canopy cover of 89%, and mean living
180 aboveground tree volume of 273 m³ ha⁻¹ before treatment. After the cut, which was carried
181 out by removing all trees in 10 elongated openings (350-1250 m² each) and leaving a total of
182 30 m³ of cut timber on the ground in the form of whole trees, ground cover by deadwood
183 had increased on average from 2% to 6%;

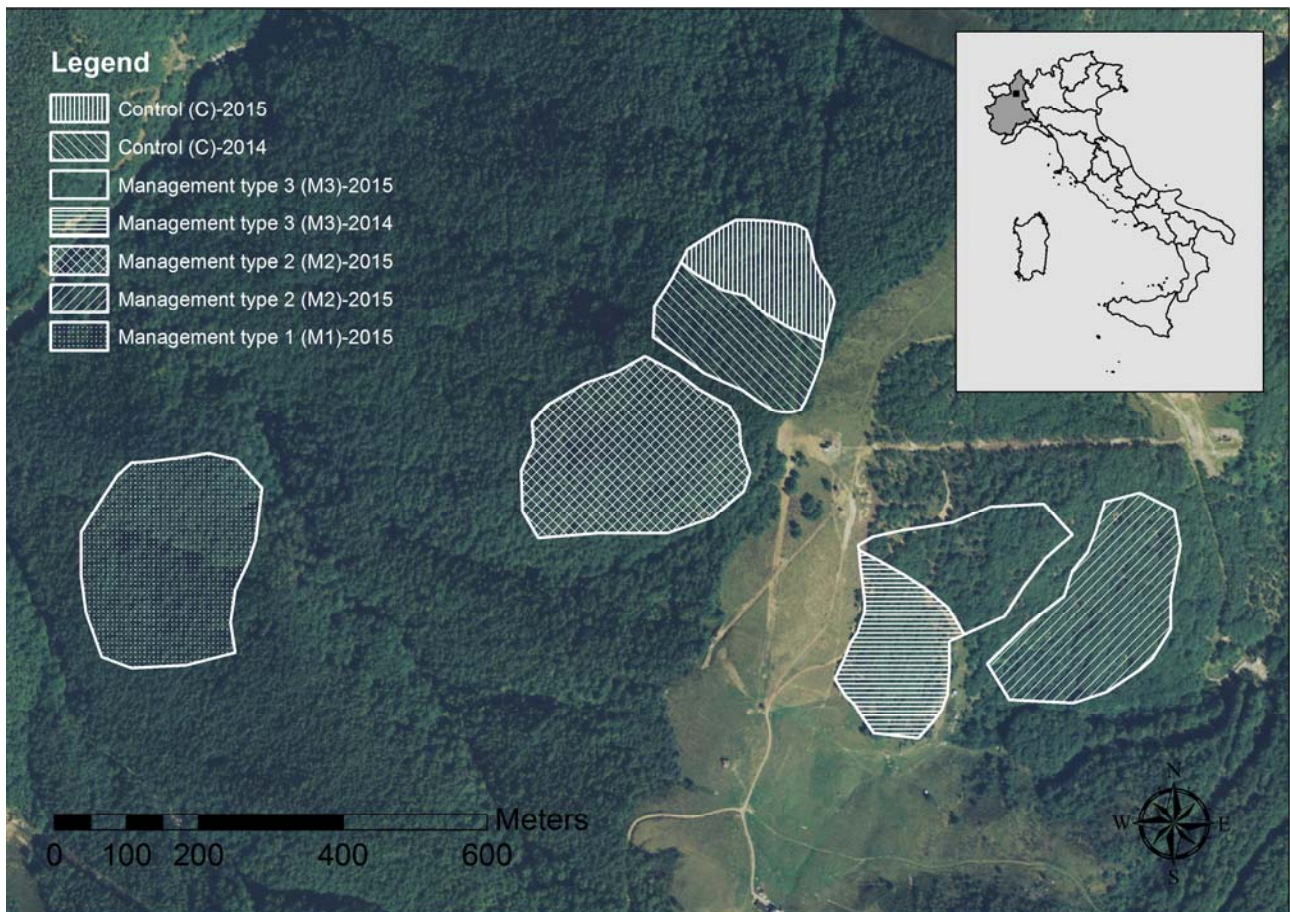
184

185 *iii)* single-tree selection in over-mature coppices (two stands in 2015, 14 and 6 hectares
186 respectively) (management type 2, M2). This was carried out by harvesting individual
187 sprouts, trees, or small groups (tree diameter <25 cm) with the aim to promote the growth of
188 better-quality beech stems, while retaining all other tree species and leaving all branches on
189 the ground. Stem density and living tree volume were reduced on average by 26% and 30%
190 respectively, while ground cover by deadwood increased on average from 2% to 8%;

191

192 *iv)* uniform thinning to convert over-mature coppices to high forest (two stands, one in 2014
193 and one in 2015, 3 hectares each) (management type 3, M3). This represented the “business
194 as usual” scenario for beech coppices in the region, and was carried out by harvesting all but
195 the best-quality sprouts from each stool, and leaving branches in small piles uniformly
196 scattered on the ground. Stem density and living tree volume were reduced on average by
197 85% and 60%, respectively, while ground cover by deadwood increased on average from
198 2% to 5 %.

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Figure 1. Experimental design. Seven different stands underwent four different forest management interventions: i) absence of management in over-mature coppices (control, C); ii) strip cuts in the temporary high forest (management type 1, M1); iii) single-tree selection in over-mature coppices (management type 2, M2); iv) uniform thinning to convert over-mature coppices to high forest (management type 3, M3).

208 **Microclimate monitoring**

209 Five *Thermo/Hygro Button* loggers (Maxim Integrated Products, Inc., Sunnyvale, CA, U.S.A.) were
210 used to record temperature and relative humidity every 1 h in each stand in 2015. The data loggers,
211 attached to wooden poles (2 cm above the soil surface) and sheltered from rain by means of a
212 plastic roof, were located in areas that were representative of the treatment applied in that stand.
213 Recorded data were used to compute daily mean, minimum, and maximum temperature and relative
214 humidity.

215

216 **Pitfall trapping**

217 We captured ground beetles by pitfall trapping in un-managed areas. In each sampling area, five
218 baited pitfall traps were arranged according to a Latin square design, i.e., at the four vertices and at
219 the center of a 20 m-wide square. Each trap (7.5 cm diameter and 9 cm deep), filled with 150 ml of
220 vinegar as an attractant (van den Berghe, 1992), was assembled with a double bottom in order to
221 keep animals alive. A flat stone was placed above each trap to prevent flooding. The exact location
222 of traps was determined by means of a Global Positioning System (GPS) Garmin eTrexR
223 Navigator. A total of 23 sampling areas in 2014 (early July-mid August) and 24 in 2015 (mid June-
224 early August) were set and traps emptied at three-day intervals.

225

226 **Radio-tracking**

227 Individuals were radio tagged with Micro-Pip radio-tag transmitters (about 0.3 g, 15 × 5 × 4 mm)
228 developed by Biotrack Ltd, (Wareham, England, www.biotrack.co.uk). Transmitters were attached
229 with cyanoacrylate on top of the elytrae, with the short antenna (2.5 cm) directed backwards, in
230 keeping with Negro et al. (2008). After radio-tagging, individuals were kept for some hours in a
231 dark terrarium before releasing them back to the wild. In the field, radio-signals could be detected
232 from about 300 m, with a battery life span of about three weeks. Tagged carabids were relocated
233 once a day, in the morning, using a receiver and a hand-held Yagi directional antenna. The exact
234 radio-telemetry location (fix) was determined in the field by a GPS, where a coloured peg was also
235 driven into the soil. At the end of the experiment period, radio-tagged individuals were caught to
236 retrieve transmitters before being released.

237

238 *C. olympiae* are more active at night than by day. During the day, they do not move or move only
239 up to few meters, whilst movements of several tens of meters were almost exclusively at night
240 (Negro et al. 2008). The fixes collected in the present study reflect therefore the microhabitat used
241 by individuals as a refuge or shelter during the day time.

242

243 **Microhabitat assessment**

244 Microhabitat availability for ground beetles was assessed in October 2015 by transect sampling. We
245 set six linear transects (60 m long) per stand in the same areas used by radio-tracked individuals (a
246 total of 42 transects, i.e. 6 transects x 7 stands). Quadrats (60 x 60 cm) along transects were set at
247 two metres one from another (i.e. 30 quadrats for each transect) for a total of 1260 quadrats (30
248 quadrats x 42 transects). We took a picture of ground cover in each quadrat and the dominant
249 microhabitat was later classified into one of the following categories: shrubs (SH), tree bases (ST),
250 deadwood (DW, i.e. coarse woody debris such as pieces of branches or trunks), grass (G), bare
251 ground (BG), rocks (R) and litter (L).

252 Individual microhabitat use was assessed by recording the microhabitat used by radio-tagged
253 individuals as shelter during the day (i.e. the dominant microhabitat category at the fix).

254

255 **Data analysis**

256

257 **Microhabitat use**

258 We used a binomial Generalized Linear Mixed Model to model microhabitat use (i.e. 1 used, 0
259 unused) as a function of management (i.e. C, M1, M2 and M3), year of study (2014 and 2015), sex,
260 and microclimate (average humidity and temperature), specifying the identity of radio-tagged
261 individuals as a random effect. We modelled the use of three microhabitat types, i.e. litter,
262 deadwood and tree bases, for which the sample size was statistically adequate not to have
263 convergence problems of the model.

264 In our study design, microhabitat use was measured for each individual, whilst microhabitat
265 availability was measured at the population scale (i.e. inside each managed stand), assuming that
266 microhabitat types were equally available to all monitored animals (Thomas & Taylor 1990). To
267 compare use and availability in this kind of study design (i.e. to study microhabitat selection), we

268 used Compositional Analysis, calculating the significance of Wilk's Λ and t statistics by
 269 randomization tests (Aebischer et al. 1993). Analyses were performed using the function *compans*
 270 in the package *adehabitatHS* for R (Calenge, 2011; R Core team, 2016). We also calculated the
 271 selection ratio: $w_j = u_j/a_j$ where u_j is the proportion of use of the habitat class j and a_j is the
 272 proportion of availability (relative frequency) of habitat class j (Manly et al., 2002). If individuals
 273 use all habitats in relation to their availability, the ratio is 1, if they use a certain habitat
 274 proportionally less than its availability, the ratio is a value between 1 and 0, while if they use the
 275 habitat proportionally more than its availability, the ratio is > 1 (i.e., the higher the value, the higher
 276 the preference for that habitat).

277 Since the selection ratio showed that microhabitat selection was different from one animal to
 278 another, we investigated these differences by means of eigenanalysis of selection ratios (Calenge,
 279 2006). If W is the table containing the selection ratios for each animal (rows) and each microhabitat
 280 type (columns), the eigenanalysis consists of a non-centred and non-scaled principal component
 281 analysis of the table $W - 1$, using the proportion of availability of each microhabitat type as column
 282 weights and the number of relocations of each animal as row weights. This analysis partitions the
 283 statistics:

284 (1)
$$\lambda = \frac{\sum_{i=1}^p \sum_{j=1}^q (u_{ij} - p_i u_{+j})^2}{p_i u_{+j}}$$

285

286 where u_{ij} is the number of relocations of animal j in microhabitat i, p_i is the proportion of available
 287 resource units in microhabitat i, and u_{+j} is the total number of relocations of animal j (White and
 288 Garrott, 1990).

289 **Movements**

290 Movements were studied by measuring the linear distances (with a telemeter) and the turning angles
 291 between consecutive fixes (Cain, 1989). Vagility indices were computed for each individual,

292 consistent with Negro et al., 2008. Independence among successive angles was tested by the non-
293 parametric Runs Test above and below the median, which is used to test the randomness of a
294 sequence of a series of observations (turning angles, in our case) (Sokal & Rohlf, 1995). To
295 approach normality (checked by using normal probability plots), distance data were square-root
296 transformed (Sokal & Rohlf, 1995).

297 To test the agreement among turning angles, we computed an index of angular concordance, or
298 angular concentration (Zar, 1999; Fortin & Dale, 2005), converting angles in a circular system
299 using Batschelet's (1981) protocol. The mean vector length takes a value of 1 when all the angles
300 are the same and a value of 0 when the vectors cancel each other out (Upton & Fingleton, 1989;
301 Fortin & Dale, 2005).

302 To check the compactness of the paths travelled by each radio-tagged individual, we calculated the
303 Index of Tortuosity (T) that considers both distances and angles. This index is based on the convex
304 hull, which is the smallest convex polygon that fully contains the path. T is equal to the L/M ratio,
305 where L is the total distance covered and M is the major diameter of the convex hull (Claussen et
306 al., 1997). The coefficient of variation ($CV = \text{standard deviation}/\text{mean} \times 100$) of distance was used
307 to test the variability of daily distance between fixes among different forest management types.

308 Movement data were merged with those of 21 individuals radio-tracked in Alpine rose shrubberies
309 and over-mature beech coppices in years 2005 and 2006 (Negro et al., 2008). This merge
310 significantly increased the size of the data set and gave us the opportunity to study movements in a
311 gradient of increasing human impact (from undisturbed forest to managed forest to shrubs resulting
312 from past forest harvesting).

313 All vagility parameters (i.e. tortuosity index, angular concordance and CV of distance) were
314 modelled by means of GLMs, in relation to the sex of the individual and forest management.

315

316 **Results**

317 A total of 40 *C. olympiae* individuals were collected and radio-tracked within stands: 15 in 2014 (in
 318 two stands i.e. C and M3) and 25 in 2015 (in four stands i.e. C, M1, M2 and M3) (Table 1). Forest
 319 management increased deadwood (M1: from 2 to 6%; M2: from 2 to 8%; M3: from 2 to 5%) and
 320 bare ground cover (M1: from 4 to 9%, M2: from 3 to 9%; M3: from 3 to 29%) in all managed
 321 stands. Cover by litter decreased greatly (M1: from 80 to 60%, M2: from 75 to 63%; M3: from 75
 322 to 32%), while grass cover exhibited only little change (M1: from 0 to 3%, M2: from 5 to 1%; M3:
 323 from 5 to 4%). The harvest did not affect ground cover by tree bases, shrubs and rocks.

324 Table 1. Number of fixes, daily mean \pm SE and total distances covered by radio-tagged ground beetles in
 325 2014-2015.
 326

No.	Management	Sex	Year	Fix No.	Total distances (m)	Mean (m) \pm SE
1	Control	F	2014	31	151.10	5.04 \pm 0.76
2	Control	M	2014	6	12.91	2.58 \pm 1.25
3	Control	M	2014	14	130.05	10.00 \pm 2.65
4	Control	M	2014	4	97.10	32.37 \pm 6.93
5	Control	F	2014	20	88.90	5.56 \pm 0.84
6	Control	F	2014	9	134.80	19.26 \pm 4.44
7	Control	M	2014	3	8.65	4.33 \pm 2.98
8	M3	F	2014	6	76.91	15.38 \pm 11.78
9	M3	M	2014	6	57.99	11.60 \pm 6.74
10	M3	M	2014	6	130.47	26.09 \pm 11.05
11	M3	M	2014	31	310.72	11.10 \pm 3.23
12	M3	F	2014	11	125.00	15.63 \pm 6.03
13	M3	M	2014	11	195.30	19.53 \pm 5.65
14	M3	F	2014	4	3.40	1.13 \pm 0.34
15	M3	F	2014	22	319.40	15.21 \pm 3.54
16	M3	M	2014	16	73.40	4.89 \pm 1.36
17	Control	M	2015	37	806.50	22.40 \pm 3.09
18	Control	M	2015	38	388.60	10.50 \pm 1.40
19	Control	M	2015	6	7.00	1.40 \pm 0.44
20	Control	F	2015	13	67.65	5.64 \pm 1.91
21	Control	M	2015	29	654.15	23.36 \pm 3.79
22	Control	F	2015	4	71.80	23.93 \pm 3.14
23	Control	F	2015	15	95.80	6.84 \pm 2.50
24	M2	F	2015	24	259.00	11.26 \pm 2.73
25	M2	F	2015	24	27.28	1.19 \pm 0.55
26	M2	M	2015	19	135.50	7.53 \pm 2.65
27	M2	M	2015	24	63.30	2.75 \pm 1.27
28	M1	M	2015	14	91.20	7.02 \pm 2.16
29	M1	M	2015	11	66.80	6.68 \pm 3.35

30	M1	M	2015	6	110.00	22 ± 11.49
31	M1	M	2015	6	155.10	15.51 ± 4.74
32	M1	F	2015	26	306.85	12.27 ± 2.38
33	M1	F	2015	15	102.73	7.34 ± 2.98
34	M2	M	2015	11	201.85	20.19 ± 4.77
35	M2	F	2015	17	289.35	18.08 ± 4.75
36	M2	F	2015	15	109.13	7.80 ± 2.48
37	M2	M	2015	14	54.75	4.21 ± 1.74
38	M3	F	2015	20	343.30	18.07 ± 5.10
39	M3	F	2015	13	162.20	13.52 ± 4.66
40	M3	M	2015	7	88.50	14.75 ± 6.68
41	M3	M	2015	26	453.55	17.44 ± 6.28

327

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329 *Microclimatic conditions*

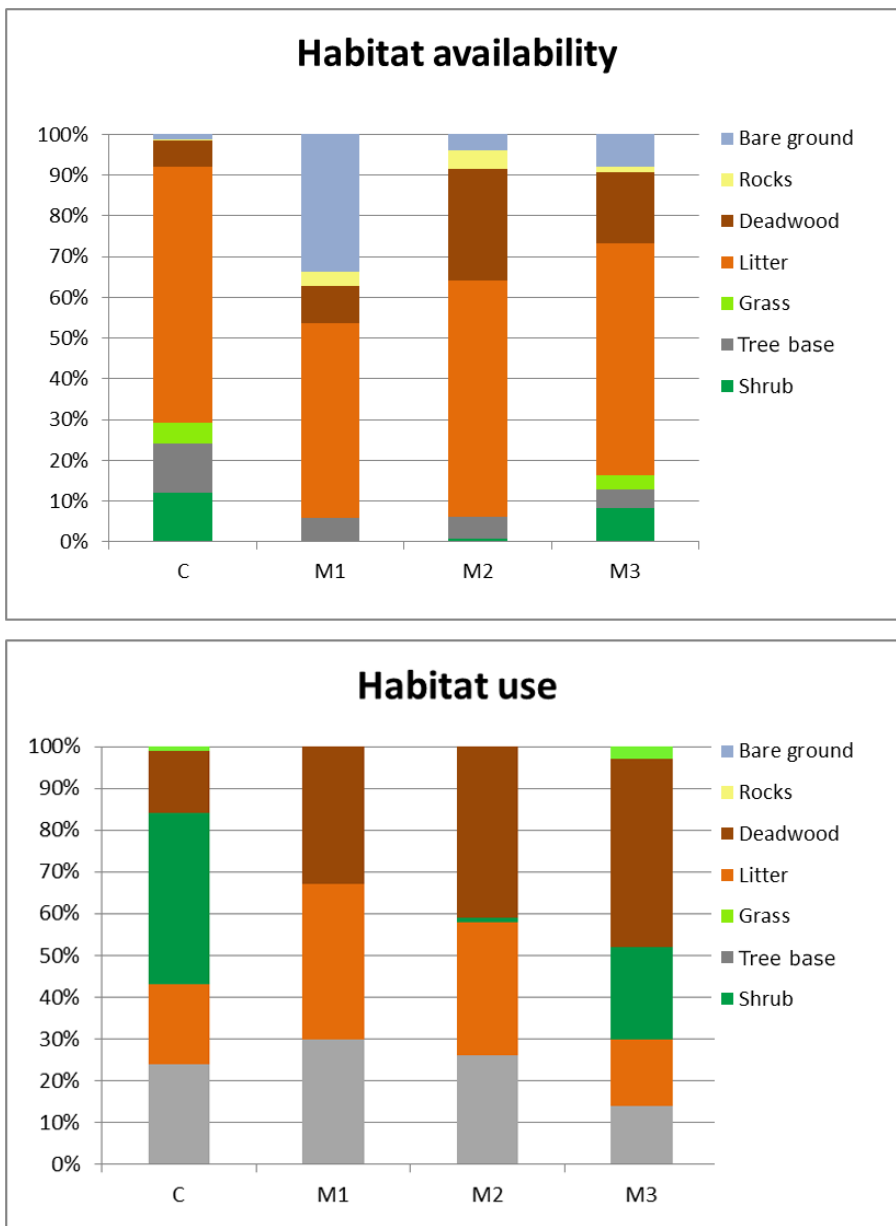
330 A complete series of climate data was only available for C and M3 stands in 2015, where data
331 loggers ran without interruption from 18 June to 5 August. Microclimatic conditions in these
332 control and managed stands were significantly different. No significance differences were found
333 between control and M1 and M2 stands, likely because the incompleteness of the data collected; the
334 maximum temperature was on average significantly higher in M3 than in C (Gaussian GLM: beta
335 5.0297, SE 0.4861, t value 10.35, $p < 2e-16$), whilst the minimum and average humidity were lower
336 (minimum humidity: Gaussian GLM: beta -3.27, SE 0.86, t value -3.77, $p < 0.000185$; average
337 humidity: Gaussian GLM: beta -18.74, SE 1.53, t value -12.2, $p < 0.000185$).

338

339 *Microhabitat use*

340 Five microhabitats were available for radio-tagged ground beetles in all managed and control areas.
341 The most available microhabitat was litter (min= 48% of quadrats in M1, max= 63% in C),
342 followed by deadwood (min = 6% in C, max=27% in M2), bare soil (min = 1% in C, max=34% in
343 M1), tree bases (min = 4% in M3, max=12% in C) and rocks (min= 1% in M3, max=5% in M2).
344 Shrubs were absent from M1 and grass cover from M1 and M2. Overall, control areas used by

345 radio-tagged individuals had more litter, shrubs and tree bases, while managed areas had more
346 deadwood and bare soil (Fig. 2, on top).



347
348 Fig 2. Available (top) vs used (bottom) microhabitat types in managed and control areas used by
349 radio-tracked individuals.

350
351 Radio-tagged individuals used five microhabitat types, i.e. litter, tree bases, deadwood, shrubs and
352 grass (Fig. 2, on bottom). Bare ground and rocks were not used. Microhabitat use modelling did not
353 show any significant differences in the use of litter, deadwood or tree bases due to year of sampling,

354 sex, mean temperature or humidity. The use of these microhabitat types significantly changed
 355 according to management type only. In particular, litter was used more frequently in M1 than in
 356 control stands, and deadwood was used more frequently in all managed than in control stands
 357 (Table 2).

358 Table 2. Binomial Generalized Linear Mixed Models of the frequency of the use of Litter, Tree
 359 bases and Deadwood diurnal shelter in the different forest management stands. Control was set as
 360 reference category. Significance codes: ‘***’ $p < 0.001$; ‘**’ $p < 0.01$; ‘*’ $p < 0.05$; ‘.’ $p < 0.1$. NS Not
 361 Significant

Litter				
	Estimate	St. Error	z values	Significance
(Intercept)	-1.61	0.28	-5.75	***
M1	1.02	0.48	2.13	*
M2	0.69	0.43	1.61	NS
M3	-0.06	0.4	-0.14	NS
Deadwood				
	Estimate	St. Error	z values	
(Intercept)	-1.83	0.27	-6.87	***
M1	1.1	0.45	2.45	*
M2	1.49	0.39	3.78	***
M3	1.67	0.35	4.76	***
Tree bases				
	Estimate	St. Error	z values	
(Intercept)	-1.15	0.22	-5.14	***
M1	0.26	0.42	0.63	NS
M2	0.03	0.37	0.09	NS
M3	-0.68	0.35	-1.95	.

362

363

364

365 The ranking of microhabitats obtained through Compositional Analysis showed that deadwood and
 366 tree bases were the most preferred microhabitats both in managed and control stands, whilst litter
 367 and grass were the least preferred ones. Shrubs was a high-ranking microhabitat in control stands,
 368 but a low ranking one in M2 and M3 stands (Table 3 and Appendix I).

369

370 Table 3. Microhabitat selection as revealed by Compositional Analysis. Microhabitats are ranked in
 371 ascending order of preference. Significance codes: ‘***’ p < 0.001; ‘**’ p<0.01; ‘*’ p<0.05; ‘.’
 372 p<0.1. NS Not Significant

373

374

	C	M1	M2	M3
Shrub	5	-	2	2
Tree base	6	4	4	5
Grass	2	-	-	4
Litter	3	2	3	3
Deadwood	4	3	5	6
Rocks	0	1	0	1
Bare ground	1	0	1	10
Lambda	0.068	0.088	0.021	0.124
Significance	**	*	**	**

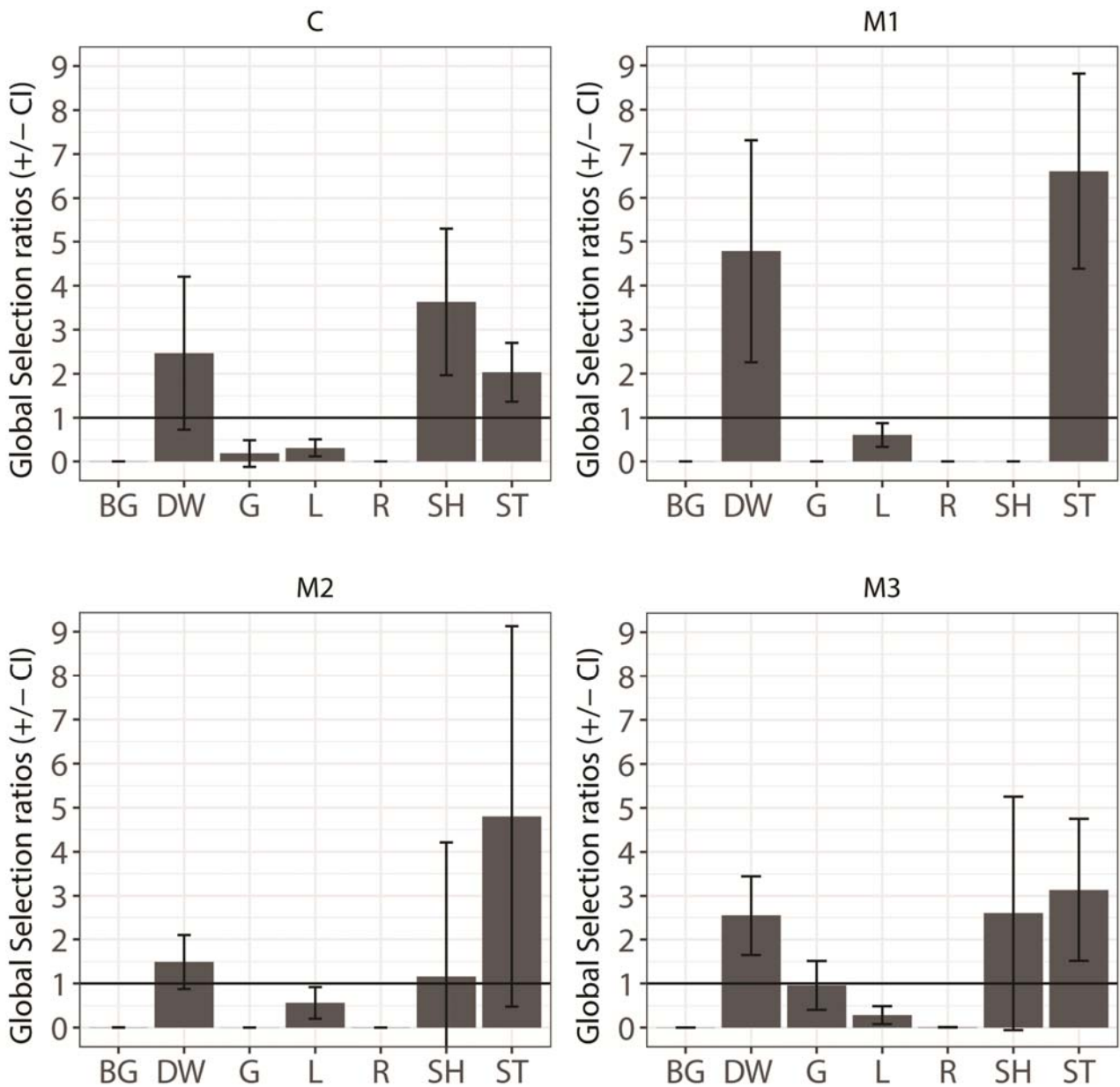
384

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386 Manly’s selection ratio W_i for deadwood, tree bases and shrubs was higher than 1 (i.e. positive
 387 selection) and ranged between 0 and 1 for litter and grass (negative selection) both in the managed
 388 and control stands. The selection ratio for tree bases was higher in M1 and M2 stands (4.74 and
 389 4.40, respectively), suggesting a high preference for that microhabitat, and close to 1 for shrubs in
 390 M2 stand (1.06), suggesting a weak positive selection or a random use for that microhabitat (Fig. 3).

391

392



393

394 Figure 3. Microhabitat selection in control (C), and managed stands (M1, M2 and M3) as revealed
 395 by Manly selection ratio $W_i (\pm SE)$. Microhabitat codes as follows: shrubs (SH), tree bases (ST),
 396 deadwood (DW), grass (G), bare ground (BG), rocks (R) and litter (L).

397

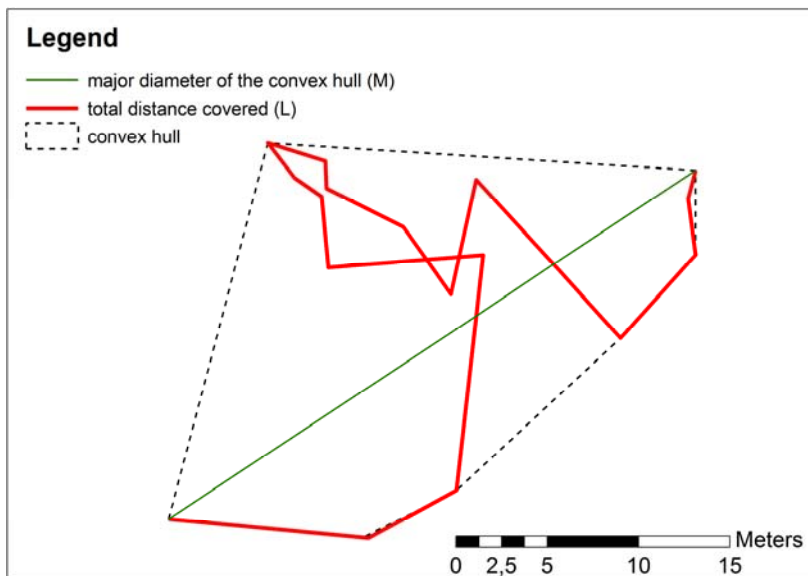
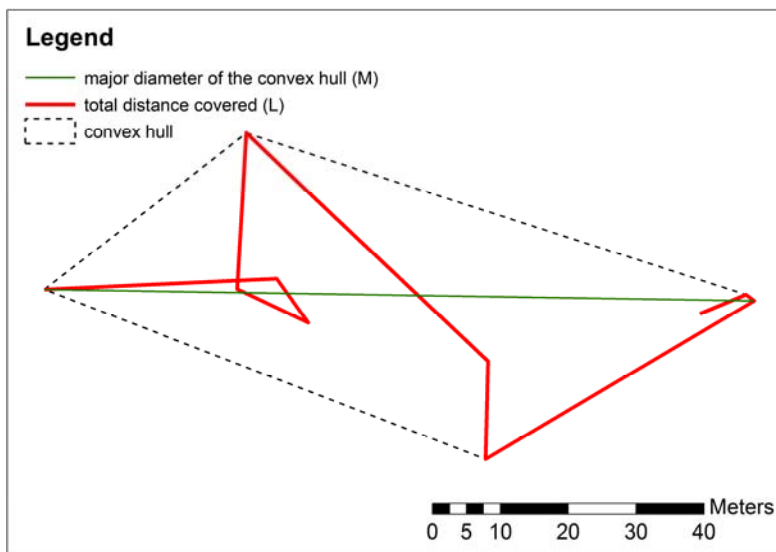
398 The eigenanalysis of selection ratios (Calenge and Dufour, 2006) graphically showed that all
 399 individuals avoided bare ground, grass and rocks and showed variable preference in the use of tree
 400 bases, shrubs and deadwood (Appendix II). The radio-tracked individuals selected shrubs,
 401 deadwood and tree bases in C, in M1 they selected deadwood and tree bases, in M2 some

402 individuals selected shrubs and deadwood and some selected tree bases, while in M3 most
403 individuals selected tree bases and deadwood.

404 *Movements*

405 By pooling old (2005 and 2006) and new (2014 and 2015) fixes together, a four-year integrated data
406 set of 1161 fixes (527 + 634) relative to 61 (21 + 40) ground beetles was obtained. All individuals
407 displayed zig-zag trajectories (Fig 4).

408
409



413 Fig 4. Trajectories of two randomly selected *Carabus olympiae* individuals (i.e. male No. 13, top
414 and female No. 5, bottom) radio-tracked in 2014. The individual's path (red line) and the major
415 diagonal of the Minimum Convex Polygon (green line) that includes the entire path are shown.

416 The total distances covered by radio-tracked individuals varied between 3.40 and 806.50 metres,
 417 with mean daily distances ranging from 1.13 to 32.37 metres (Table 1). Angles were evenly
 418 distributed in all stands (shrubberies, control and in managed stands), indicating the lack of a
 419 prevailing walking direction (Appendix III).

420 GLMs showed that the coefficient of variation of the distances and the tortuosity index varied
 421 according to the management; the length of the paths of individuals in M2 and M3 stands was more
 422 variable than in control stands and the tortuosity index was significantly lower in shrubbery and in
 423 M2 stands than in control, and significantly higher in males than in females (Table 4 and Fig. 5).

CV Distance

	Estimate	St..Error	t values	Significance
(Intercept)	0.86	0.07	11.88	***
M1	0.31	0.16	1.93	NS
M2	0.5	0.14	3.49	***
M3	0.33	0.12	2.7	***
Shrubbery	0.18	0.13	1.41	NS

Tortuosity

	Estimate	St..Error	t values	Significance
(Intercept)	0.83	0.08	9.76	***
M1	-0.18	0.16	-1.11	NS
M2	-0.39	0.15	-2.6	*
M3	-0.16	0.13	-1.3	NS
Shrubbery	-0,47	0.11	-4.22	***
Sex	0.29	0.09	3.34	**

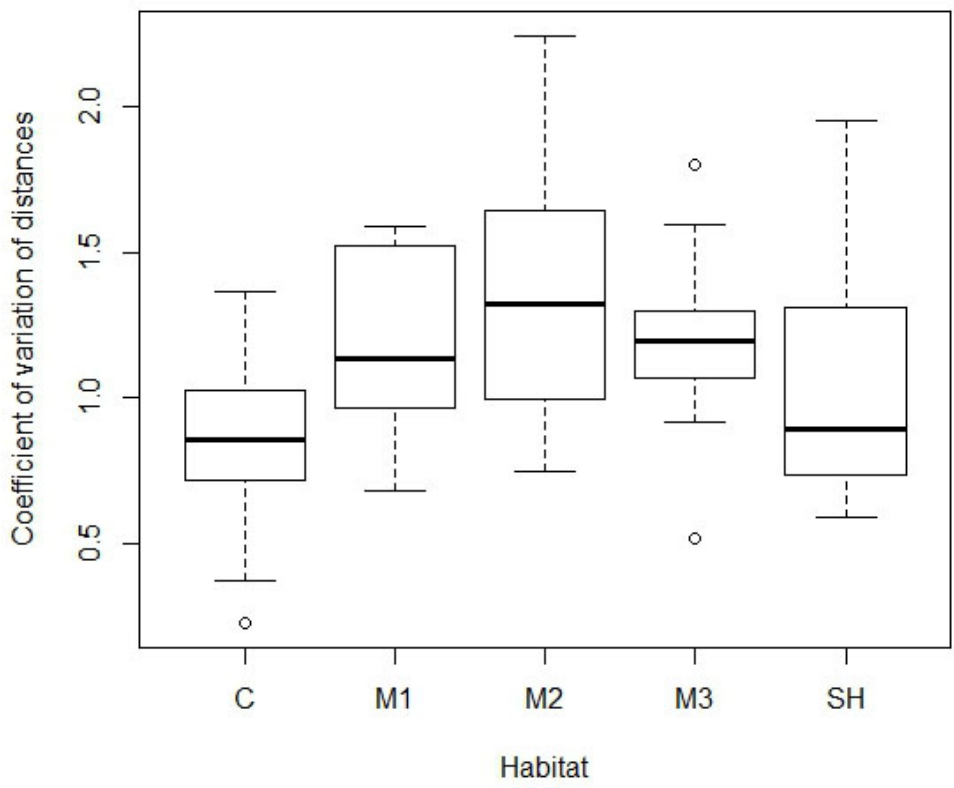
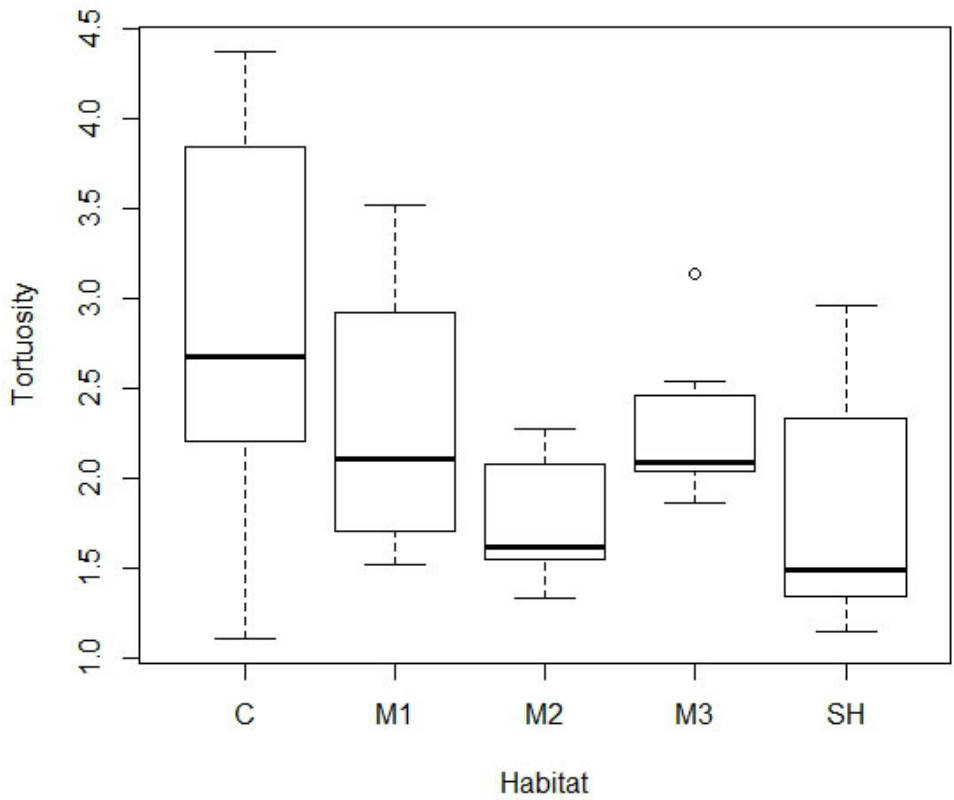
424

425

426 Table 4. Generalized Linear Mixed Models of CV of distances and tortuosity in the different forest
 427 management and shrubland stands. Control was set as reference category. Significance codes: '****'
 428 $p < 0.001$; '***' $p < 0.01$; '**' $p < 0.05$; '.' $p < 0.1$. NS Not Significant

429

430



431

432 Fig.5 Box plots of the tortuosity index (top) and of the coefficient of variation of the distances
 433 (bottom) between management types (Control C, Management M1, Management M2, Management
 434 M3 and Shrubs SH)

435

436 **Discussion**

437 **Microclimatic conditions**

438 Our results confirmed that ground beetles strongly depend on particular humidity and temperature
439 requirements (Skłodowski 2014). Data loggers showed that temperature was higher and humidity
440 lower in M3 than in control stands, suggesting that uniform logging may induce significant short
441 term climatic changes. Warmer and drier microclimates may negatively affect *C. olympiae* both
442 directly (this species likes fresh and humid beech forests, Negro et al. 2014) and indirectly, reducing
443 the local availability of snails and slugs which are the preferred prey of most large carabids (Kotze
444 et al. 2011) (*Arianta arbustorum*, *Arion* and *Limas* spp. in the study area (Negro pers. obs.)).
445 Canopy and shrub cover usually take several years to recover after a cut, depending on the intensity
446 of the removal. Montagnoli et al. (2012) compared canopy cover and soil temperature between May
447 and October at 5 cm depth in over-mature beech coppices (not cut for 40 years) against those
448 following uniform conversion to high forest (75% of trees removed). Even 14 years after the cut,
449 canopy cover was on average 22% lower and soil temperature 1 °C warmer than in the uncut forest.
450 The low-intensity interventions (M1 and M2) may have resulted in only negligible changes in
451 micro-climate, shadiness and shrub cover by preserving patches or corridors of trees throughout the
452 stands. Small canopy gaps (20 to 30 m diameter) have been found to increase maximum soil
453 temperature by +0.2 - +1.9 °C relative to an uncut control, but the effect was already cancelled out
454 at the edge of the gap (Prévost and Raymond, 2012). Gap cutting may be even more conservative in
455 terms of soil moisture, which has been found to be unchanged, or even increased, in small (300-400
456 m²) and large (500-700 m²) gaps compared to below fully closed canopy (Bilek et al., 2014).
457 Compared to other broadleaves, beech has a strong ability to fill canopy gaps by crown extension
458 (Brunet et al. 2010). If the regeneration strategy is successful, a new beech layer will establish in
459 recently opened gaps within a few years, resulting in moderate microclimatic changes with a new
460 canopy (Dovciak and Brown, 2014). Finally, the release of deadwood on the ground may provide
461 shadier microsites that mitigate the absence of a more uniform canopy or shrub cover as a

462 consequence of the cut (Marzano et al., 2013) and contribute to preserving soil moisture (Ritter and
463 Vesterdal, 2006).

464

465 **Microhabitat use**

466 In addition to microclimatic conditions, forest management may change ground cover and
467 understory, and induce a different microhabitat use. Even though we were able to consider only 1-2
468 stands per management and further replicates were not possible due to the limited known range of
469 the species, we nonetheless described in details and by means of several sampling points the
470 variance within each treatments. This allowed us to identify that the use of microhabitat types by
471 radio-tagged *C. olympiae* individuals indeed changed significantly according to the management.
472 The increase in bare ground after logging is obviously a negative effect, given that individuals
473 cannot hide in such a microhabitat (Niemelä et al., 1996, Skłodowski 2008). This finding is
474 consistent with previous analyses that showed that *C. olympiae* abundance was negatively related to
475 bare ground cover (Negro et al., 2014). However, the increase in deadwood availability may be
476 beneficial. Microhabitat selection analyses (i.e. Compositional Analysis and Manly ratio) showed
477 that deadwood and tree bases were preferred and litter was avoided in all managed and control
478 stands, suggesting that microhabitat preferences did not significantly change with forest
479 interventions. At the same time, univariate models showed that deadwood was used more frequently
480 in all managed stands than in control stands. This major use, given the consistency of microhabitat
481 preferences, may be easily explained by the greater deadwood availability in managed stands. The
482 abundance of deadwood in managed stands was a consequence of experimental logging that aimed
483 at retaining coarse woody debris in the form of whole cut stems (30 m³ over 10 hectares in
484 treatment ii) or branches left on the ground (all branches from cut stems in treatments iii and iv)
485 (Negro et al. 2014)

486 Deadwood is crucial for hundreds of rare and threatened species in Europe alone (Lassauce et al.
487 2011, Floren et al 2014, Chumak et al. 2015, Gossner et al. 2016)

488

489 The quantity of deadwood in Europe's forests has decreased significantly since the middle of the
490 19th century due to intense forest exploitation. Since 2000, however, a small overall increase in
491 deadwood has been observed (Forest Europe 2011). This may be due to management practices that
492 deliberately increase the amount of woody debris in managed forests (as in our case), either thanks
493 to legal or policy prescriptions, or as a measure to comply with forest certification requirements
494 (European Environment Agency 2015). The increases in forest cover and stem density in
495 many European forests, following extensification of forest use, have also led to a increased natural
496 disturbance levels (Seidl et al. 2014). Wind storms and bark beetle disturbance events may have
497 influenced the production of dead wood in some areas (Kulakowski et al. 2017), although this
498 should not be relevant for the area studied here (no evidence for large disturbances in the managed
499 beech forest was found). Finally, climate change may also contribute to increased disturbance
500 frequency and severity, and increased deadwood input (Seidl et al. 2017).

501 All the above confirms that the retention of deadwood is thus crucial in conservation of forest
502 organisms, and specific attention should be paid for securing its spatio-temporal continuity in
503 managed forests devoted to animal diversity conservation.

504 Very large bases of living and dead trees were clearly preferred by *C. olympiae* individuals that
505 probably used this resource as shelter for daytime rest or against micro-climatic variation. Coppices
506 may provide more of this habitat than high forests, since the tree base tends to expand if it must
507 support a larger number of sprouts. Interventions aimed at maintaining over mature coppices are
508 therefore preferable to those aimed at converting over-mature coppices to high forest, consistent
509 with previous analyses (Negro et al., 2014, Seidle et al. 2017).

510 The eigenanalysis of selection ratios showed that microhabitat preferences changed from one
511 individual to another.

512

513 **Movements**

514 The present research also indicates that forest management may affect *C. olympiae* movements.
515 Both the coefficient of variation of the distances and the tortuosity index varied according to the
516 management. The length of the paths of individuals in managed stands was more variable than in
517 control stands, suggesting that movements of individuals were random in terms of direction and
518 length when relocated in forest stands subjected to logging. Trajectories were rather tortuous, fitting
519 the hunting behaviour of a typical “olfactory-tactile” predator that looks for prey by systematically
520 exploring its territory. The high tortuosity is also in keeping with the use of decaying fruits and
521 vegetables (as shown by multiple choice tests carried out in the lab by Negro and Palestrini,
522 unpublished). We did not track individuals in between two successive telemetric locations. This
523 means that true paths were likely longer and trajectories more tortuous than those measured. The
524 tortuosity index was significantly lower in Alpine rose shrubland and in M2 stands than in control.
525 Negro et al. (2008) found that most individuals located in shrubland were hidden under Alpine rose
526 shrubs, which probably were used as shelter from extreme temperatures and protection from
527 predators. We believe the low availability of shrubs (which were isolated and sparse) may have
528 constrained trajectories (individuals were compelled to move from shrub to shrub) and reduced their
529 tortuosity. Analogously, changes in microhabitat and/or resource availability in some managed
530 forest stands may have significantly reduced tortuosity in comparison with controls.
531 Tortuosity index was significantly higher in males than in females. Males of insects are able to
532 detect pheromones produced and emitted by females (Gullan & Cranston 1994). We therefore
533 hypothesize the paths of males were more tortuous in order to diminish the time allocated for the
534 searching for females (in keeping with Negro et al. 2008).
535 We assumed that microhabitat use and movements of radio-tracked individuals were unaffected by
536 radio-tags. Although their weight and dimensional impediment were low, concerns about the effects
537 of tagging on animal behaviour may arise and a reduction in dispersal ability cannot be excluded. In
538 fact, the weight of the tag (0.3 g) is about 40% of the average weight of individuals, and the tag
539 fixed on top of the elytra may be a hindrance to the movements across narrow passages (leaf-litter,

540 thick grass, etc.). In a previous paper on movements of the common ground beetle species *C.*
541 *coriaceus*, Riecken & Raths (1996) stated that foraging success was not substantially reduced by
542 radiotags. If a reduction in dispersal existed, however, this would have affected both individuals in
543 control and in managed stands to the same degree. Our results, focused on the comparative analyses
544 between managed and un-managed forest patches are therefore reliable.

545

546 **Conclusions**

547 Our results support the evidence that logging may exert short-term negative effects on *C. olympiae*
548 ground beetles (as suggested by the increase in bare ground, and changes in climatic conditions and
549 movements). However, the preference for tree bases and deadwood suggests that forest
550 management may be compatible to beetle's maintenance, on the condition that: *i*) the coppice,
551 which provides more suitable microhabitats, prevails over conversion to high forest, and *ii*)
552 deadwood originating from cutting (branches and treetops) is properly accumulated.

553 In general deadwood continuity and abundance must be secured while harvesting, and very large
554 clearings (which probably make the microclimate too warm and dry for this species) should be
555 replaced by only thinning or gap harvesting.

556

557

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886 Appendix I. Ranking matrices of Compositional Analysis for ground beetles based on comparing
 887 proportional microhabitat use of fixes with proportions of total available microhabitat types in
 888 Control (C) and Managed (M1, M2, M3) stands. Microhabitat codes as follows: shrubs (SH),
 889 stumps (ST), deadwood (DW), grass (G), bare ground (BG), rocks (R) and litter (L). Each mean
 890 element in the matrix was replaced by its sign; a triple sign represents significant deviation from
 891 random at $P < 0.05$.
 892

Habitat type

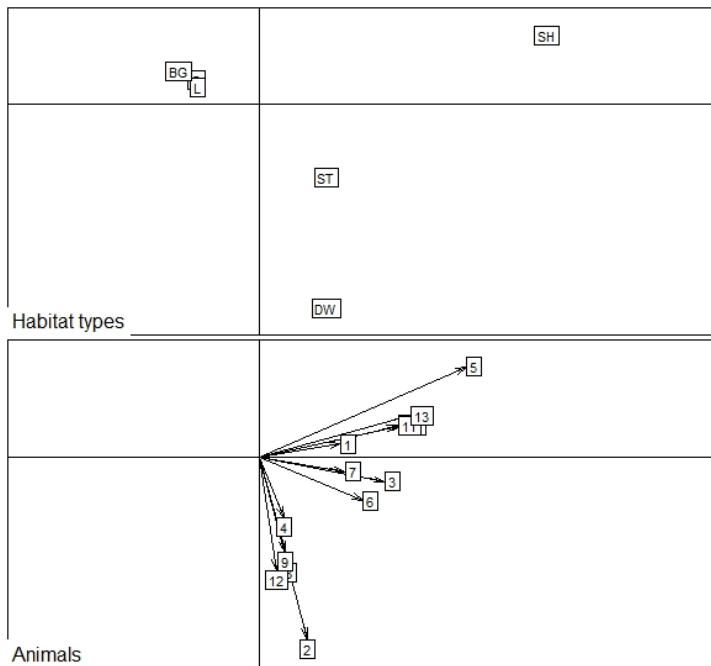
C	SH	ST	G	L	DW	R	BG	M1	ST	L	DW	R	BG
SH	0	-	+++	+++	+	+++	+++	ST	0	+++	+	+++	+++
ST	+	0	+++	+++	+	+++	+++	L	---	0	---	+++	+++
G	---	---	0	---	---	+++	+++	DW	-	+++	0	+++	+++
L	---	---	+++	0	---	+++	+++	R	---	---	---	0	+++
DW	-	-	+++	+++	0	+++	+++	BG	---	---	---	---	0
R	---	---	---	---	---	0	---						
BG	---	---	---	---	---	+++	0						

M2	SH	ST	L	DW	R	BG	M3	SH	ST	G	L	DW	R	BG
SH	0	-	-	---	+++	+++	SH	0	---	-	-	---	+++	+++
ST	+	0	+	-	+++	+++	ST	+++	0	+++	+++	-	+++	+++
L	+	-	0	---	+++	+++	G	+	---	0	+	---	+++	+++
DW	+++	+	+++	0	+++	+++	L	+	---	-	0	---	+++	+++
R	---	---	---	---	0	---	DW	+++	+	+++	+++	0	+++	+++
BG	---	---	---	---	+++	0	R	---	---	---	---	---	0	+++
							BG	---	---	---	---	---	---	0

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898 Appendix II. Results of the eigenanalysis of selection ratio carried out to highlight microhabitat
 899 selection by *Carabus olympiae* individuals radiotracked in Control (C) and Managed (M1, M2 and
 900 M3) stands. Top panel: microhabitat type loadings on the first two factorial axes. Bottom panel:
 901 animal scores on the first factorial plane (see Calenge & Dufour 2006 for further details).
 902 Microhabitat codes as follows: shrubs (SH), stumps (ST), deadwood (DW), grass (G), bare soil
 903 (BG), rocks (R) and litter (L). Numbers indicate animals in each treatment (i.e. 13 individuals in
 904 controls, 6 in M1, 8 in M2 and 13 in M3). The distance of the label from the center of the graph in
 905 the top panel indicates microhabitat preference (those close to the origin not being selected), the
 906 length and direction of the arrows in the bottom panel indicates the preferred selection for each
 907 individual.

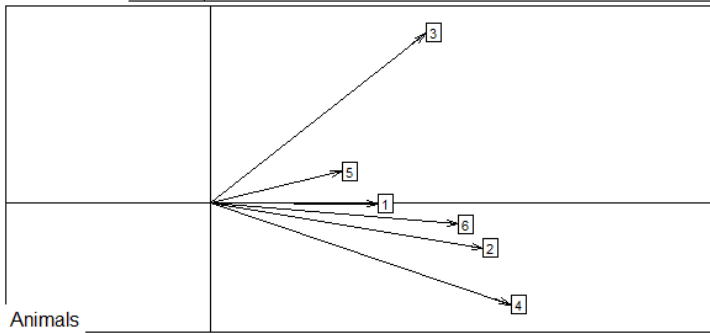
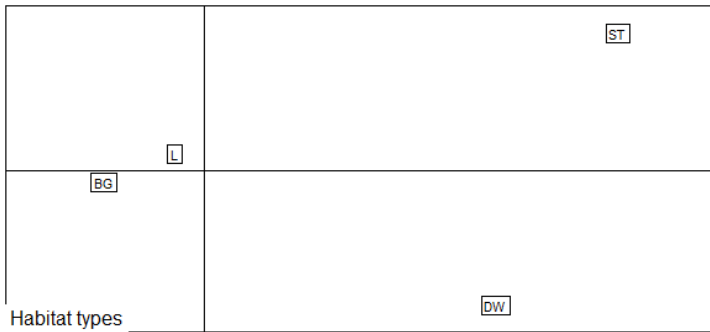
908 Control



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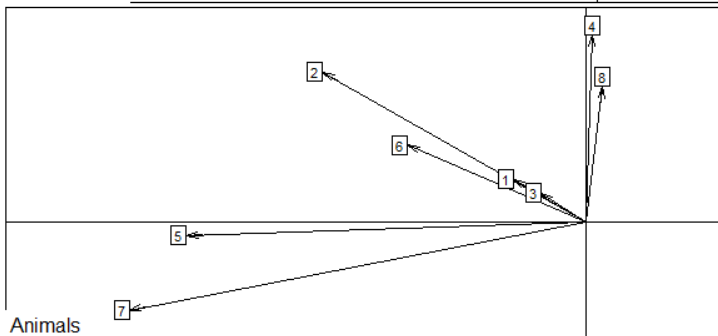
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911 M1



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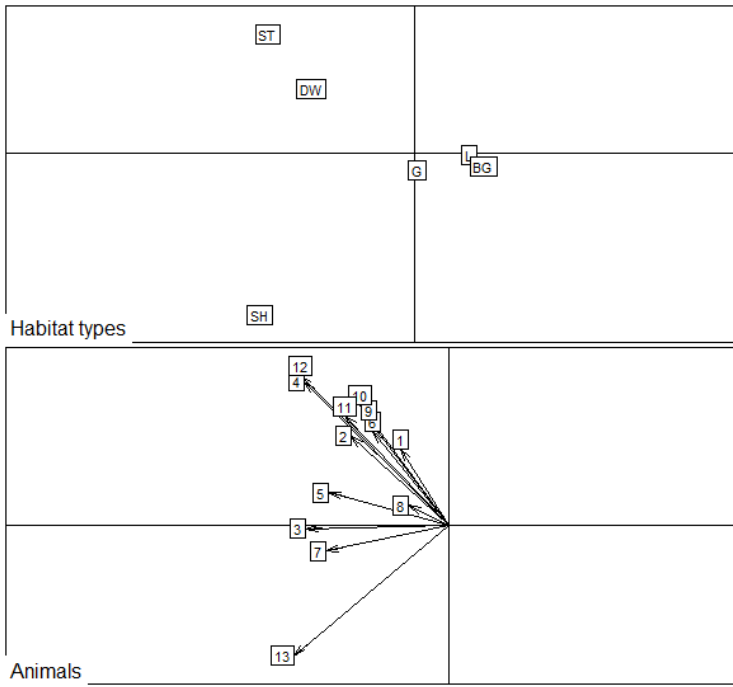
913 M2



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916 M3



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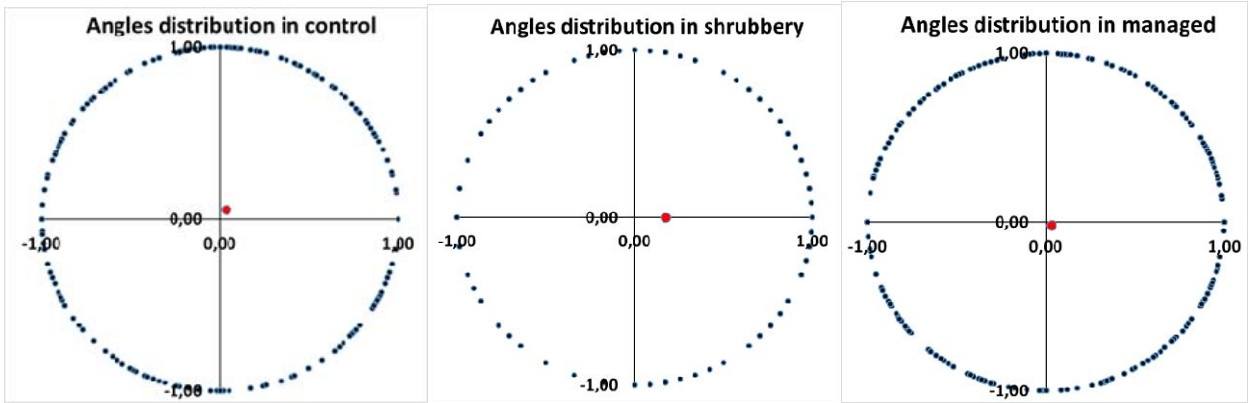
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949 Appendix III. Even distribution of angles in shrubbery, control and managed stands.

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