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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Abstract

Beech forests are important for biodiversity conservation in Europe and studies to identify

sustainable forest management practices are therefore required. The ground beetle *Carabus*

olympiae Sella, 1855, is a large steno-endemic endangered alpine species with very restricted

ranges. Its known range is only delimited in two beech forests in the western Italian Alps where

beech wood is still harvested. Forty individuals were collected and radio-tracked in 2014-2015 in

order to assess the effects of forest management on microclimatic conditions, microhabitat use and

movements.

Regarding microhabitat selection deadwood and tree bases were preferred, and were used as refuges

during the daytime. Bare ground was not used. The length of the path travelled by individual insects

was more variable and the tortuosity was lower in managed than in unmanaged stands, suggesting

that management induced more constrained trajectories and variable distances.

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

However, the preference for tree bases and deadwood suggests that forest management,

concurrently, may also be beneficial, on the condition that: *i)* the coppice, which provides more

suitable microhabitats, prevails over conversion to high forest, and *ii)* deadwood originating from

cutting (branches and treetops) is properly accumulated.

Introduction

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

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

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

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

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

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

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

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

Study area

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

Methods

Experimental forest management

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

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

ii) strip cuts in the high forest (one stand in 2015, 10 hectares) (management type 1, M1). This had an average stem density of 915 trees ha^{-1}, mean canopy cover of 89%, and mean living aboveground tree volume of 273 m^3 ha^{-1} before treatment. After the cut, which was carried out by removing all trees in 10 elongated openings (350-1250 m^2 each) and leaving a total of 30 m^3 of cut timber on the ground in the form of whole trees, ground cover by deadwood had increased on average from 2% to 6%;

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
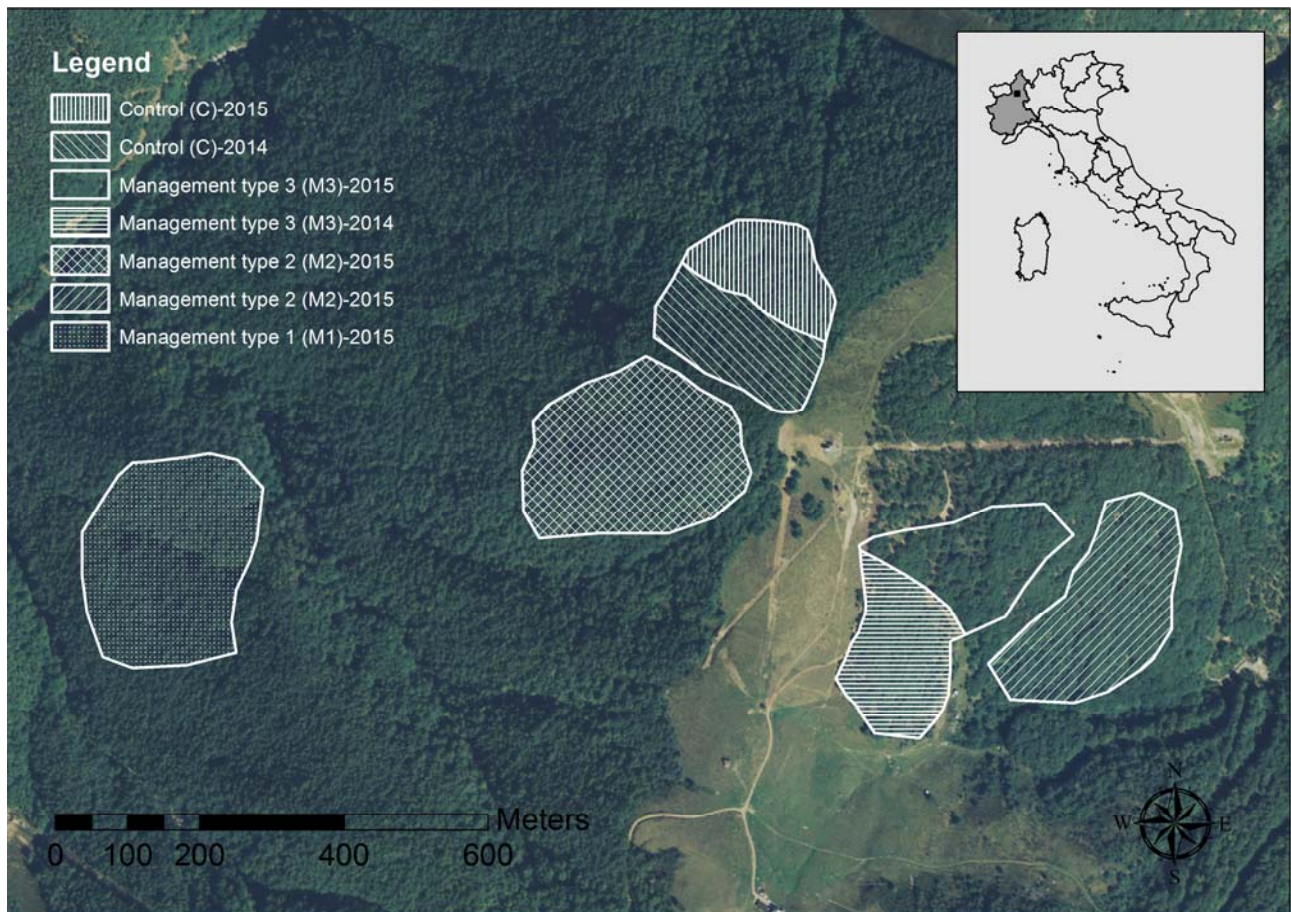
198　　　　　2% to 5 %.

199

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).

Microclimate monitoring

Five *Thermo/Hygro Button* loggers (Maxim Integrated Products, Inc., Sunnyvale, CA, U.S.A.) were

used to record temperature and relative humidity every 1 h in each stand in 2015. The data loggers,

attached to wooden poles (2 cm above the soil surface) and sheltered from rain by means of a

plastic roof, were located in areas that were representative of the treatment applied in that stand.

Recorded data were used to compute daily mean, minimum, and maximum temperature and relative

humidity.

Pitfall trapping

10

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 terrariurm 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

Microhabitat assessment

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

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

Data analysis

Microhabitat use

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

In our study design, microhabitat use was measured for each individual, whilst microhabitat availability was measured at the population scale (i.e. inside each managed stand), assuming that microhabitat types were equally available to all monitored animals (Thomas & Taylor 1990). To 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 *compana*

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) $S = \sum_{i=1}^{P} \sum_{j=1}^{K} \frac{(u_{ij} - p_i u_{+j})^2}{p_i u_{+j}}$

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 = standard deviation/mean × 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**

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

Table 1. Number of fixes, daily mean ± SE and total distances covered by radio-tagged ground beetles in 2014-2015.

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

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328

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

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

radio-tagged individuals had more litter, shrubs and tree bases, while managed areas had more

deadwood and bare soil (Fig. 2, on top).
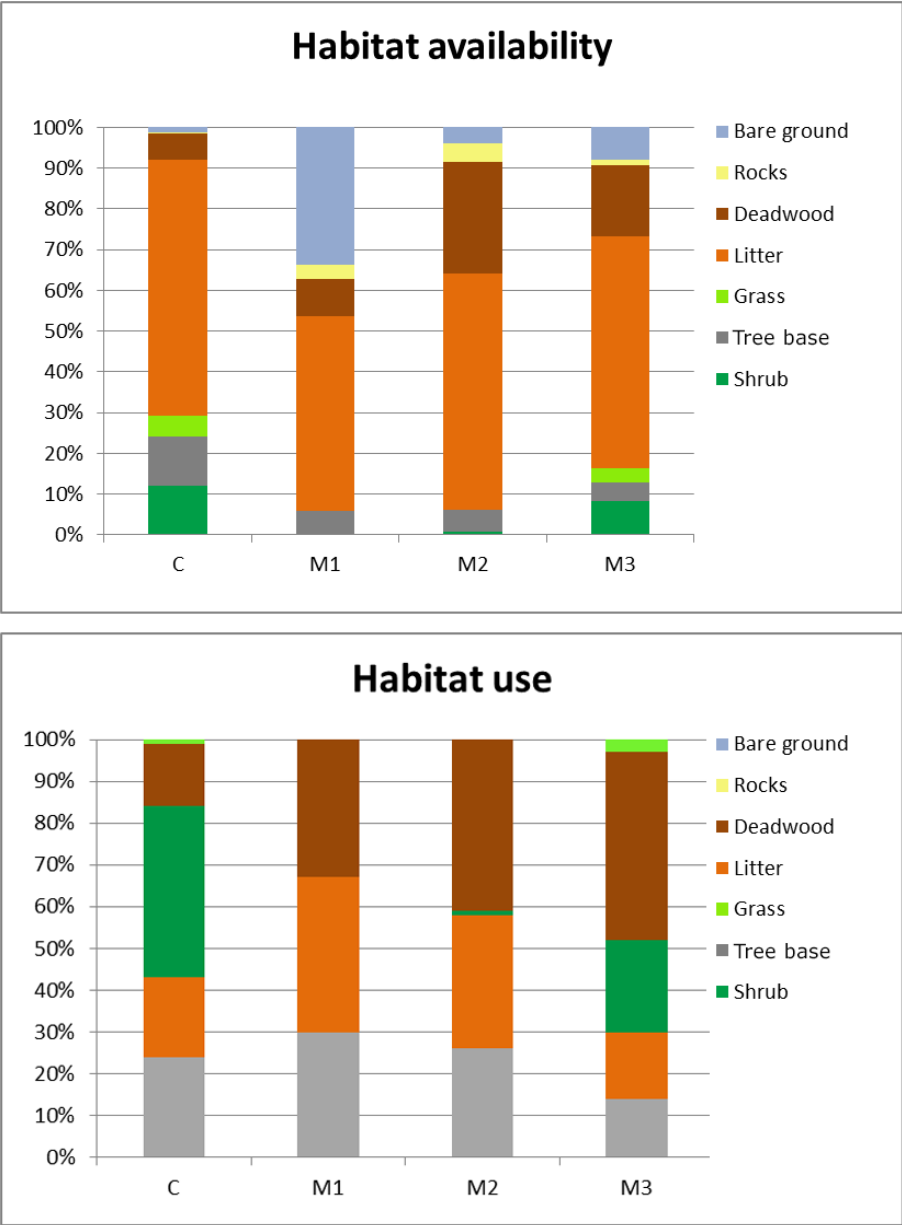
Fig 2. Available (top) vs used (bottom) microhabitat types in managed and control areas used by
radio-tracked individuals.

Radio-tagged individuals used five microhabitat types, i.e. litter, tree bases, deadwood, shrubs and

grass (Fig. 2, on bottom). Bare ground and rocks were not used. Microhabitat use modelling did not

show any significant differences in the use of litter, deadwood or tree bases due to year of sampling,

sex, mean temperature or humidity. The use of these microhabitat types significantly changed

according to management type only. In particular, litter was used more frequently in M1 than in

control stands, and deadwood was used more frequently in all managed than in control stands

(Table 2).

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

The ranking of microhabitats obtained through Compositional Analysis showed that deadwood and

tree bases were the most preferred microhabitats both in managed and control stands, whilst litter

and grass were the least preferred ones. Shrubs was a high-ranking microhabitat in control stands,

but a low ranking one in M2 and M3 stands (Table 3 and Appendix I).

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

	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	0
Lambda	0.068	0.088	0.021	0.134
Significance	**	*	**	**

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).

393
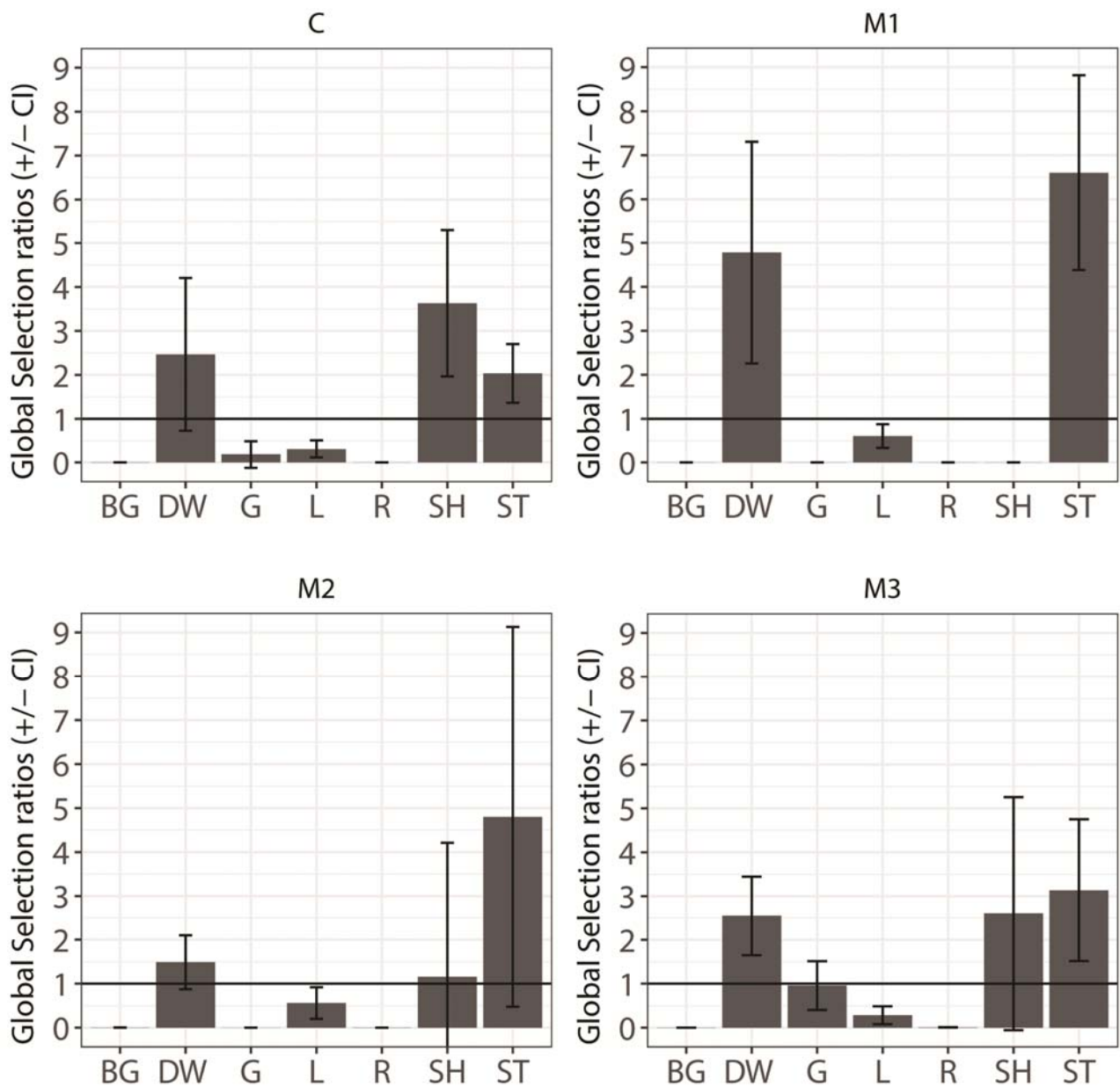
Figure 3. Microhabitat selection in control (C), and managed stands (M1, M2 and M3) as revealed
by Manly selection ratio Wi (± SE). Microhabitat codes as follows: shrubs (SH), tree bases (ST),
deadwood (DW), grass (G), bare ground (BG), rocks (R) and litter (L).

The eigenanalysis of selection ratios (Calenge and Dufour, 2006) graphically showed that all

individuals avoided bare ground, grass and rocks and showed variable preference in the use of tree

bases, shrubs and deadwood (Appendix II). The radio-tracked individuals selected shrubs,

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).

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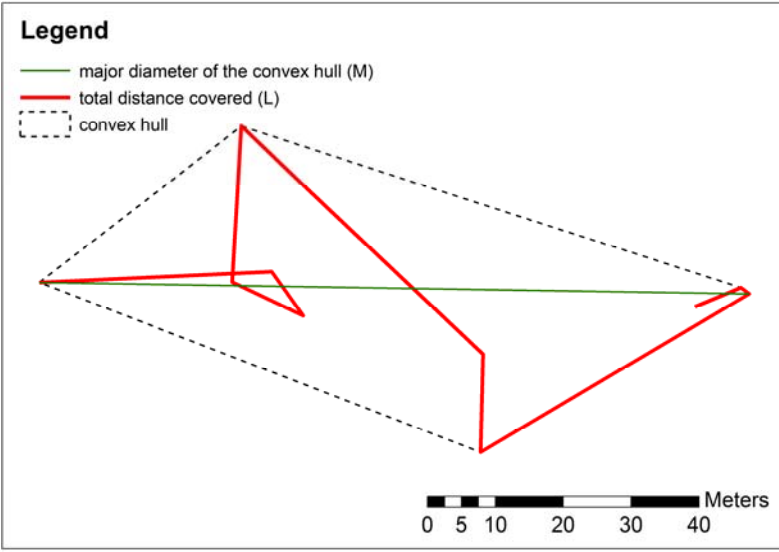
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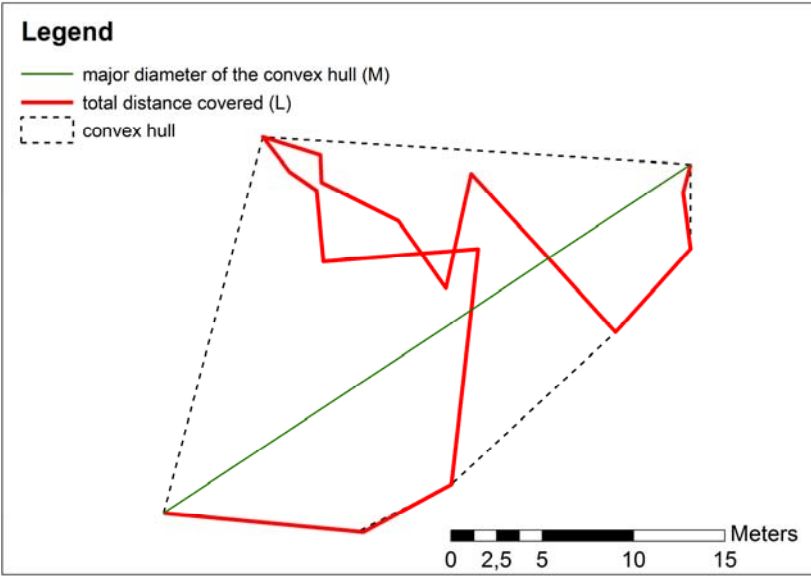
412

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.

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

GLMs showed that the coefficient of variation of the distances and the tortuosity index varied according to the management; the length of the paths of individuals in M2 and M3 stands was more variable than in control stands and the tortuosity index was significantly lower in shrubbery and in 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	**

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

431
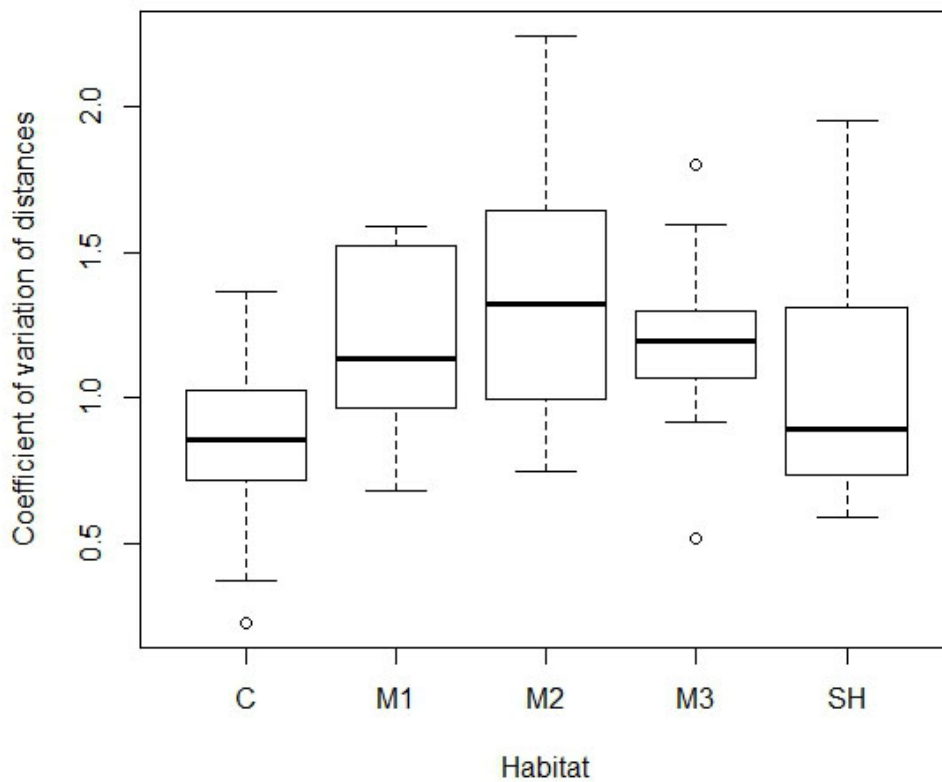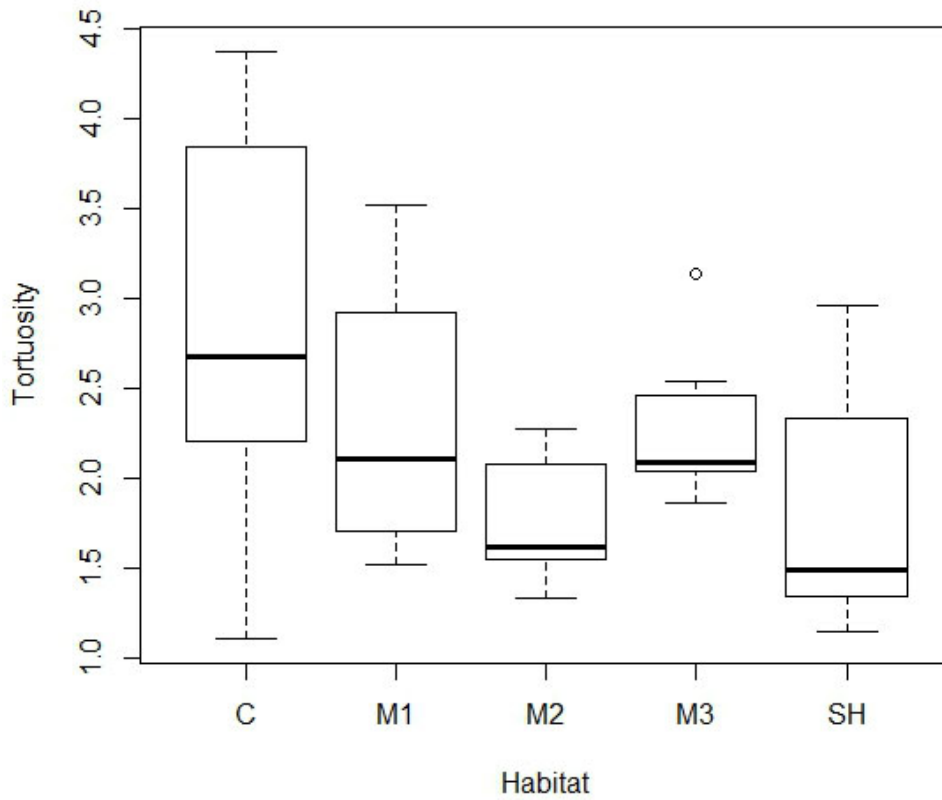
Fig.5 Box plots of the tortuosity index (top) and of the coefficient of variation of the distances
(bottom) between management types (Control C, Management M1, Management M2, Management
M3 and Shrubs SH)

435

Discussion

Microclimatic conditions

Our results confirmed that ground beetles strongly depend on particular humidity and temperature requirements (Skłodowski 2014). Data loggers showed that temperature was higher and humidity lower in M3 than in control stands, suggesting that uniform logging may induce significant short term climatic changes. Warmer and drier microclimates may negatively affect *C. olympiae* both directly (this species likes fresh and humid beech forests, Negro et al. 2014) and indirectly, reducing the local availability of snails and slugs which are the preferred prey of most large carabids (Kotze et al. 2011) (*Arianta arbustorum*, *Arion* and *Limas* spp. in the study area (Negro pers. obs.)). Canopy and shrub cover usually take several years to recover after a cut, depending on the intensity of the removal. Montagnoli et al. (2012) compared canopy cover and soil temperature between May and October at 5 cm depth in over-mature beech coppices (not cut for 40 years) against those following uniform conversion to high forest (75% of trees removed). Even 14 years after the cut, canopy cover was on average 22% lower and soil temperature 1 °C warmer than in the uncut forest. The low-intensity interventions (M1 and M2) may have resulted in only negligible changes in micro-climate, shadiness and shrub cover by preserving patches or corridors of trees throughout the stands. Small canopy gaps (20 to 30 m diameter) have been found to increase maximum soil temperature by +0.2 - +1.9 °C relative to an uncut control, but the effect was already cancelled out at the edge of the gap (Prévost and Raymond, 2012). Gap cutting may be even more conservative in terms of soil moisture, which has been found to be unchanged, or even increased, in small (300-400 m^2) and large (500-700 m^2) gaps compared to below fully closed canopy (Bilek et al., 2014). Compared to other broadleaves, beech has a strong ability to fill canopy gaps by crown extension (Brunet et al. 2010). If the regeneration strategy is successful, a new beech layer will establish in recently opened gaps within a few years, resulting in moderate microclimatic changes with a new canopy (Dovciak and Brown, 2014). Finally, the release of deadwood on the ground may provide 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 m^3 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

The quantity of deadwood in Europe's forests has decreased significantly since the middle of the

19th century due to intense forest exploitation. Since 2000, however, a small overall increase in

deadwood has been observed (Forest Europe 2011). This may be due to management practices that

deliberately increase the amount of woody debris in managed forests (as in our case), either thanks

to legal or policy prescriptions, or as a measure to comply with forest certification requirements

(European Environment Agency 2015). The increases in forest cover and stem density in

many European forests, following extensification of forest use, have also led to a increased natural

disturbance levels (Seidl et al. 2014). Wind storms and bark beetle disturbance events may have

influenced the production of dead wood in some areas (Kulakowski et al. 2017), although this

should not be relevant for the area studied here (no evidence for large disturbances in the managed

beech forest was found). Finally, climate change may also contribute to increased disturbance

frequency and severity, and increased deadwood input (Seidl et al. 2017).

All the above confirms that the retention of deadwood is thus crucial in conservation of forest

organisms, and specific attention should be paid for securing its spatio-temporal continuity in

managed forests devoted to animal diversity conservation.

Very large bases of living and dead trees were clearly preferred by *C. olympiae* individuals that

probably used this resource as shelter for daytime rest or against micro-climatic variation. Coppices

may provide more of this habitat than high forests, since the tree base tends to expand if it must

support a larger number of sprouts. Interventions aimed at maintaining over mature coppices are

therefore preferable to those aimed at converting over-mature coppices to high forest, consistent

with previous analyses (Negro et al., 2014, Seidle et al. 2017).

The eigenanalysis of selection ratios showed that microhabitat preferences changed from one

individual to another.

512

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 $P < 0.05$.
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Habitat type

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

M1	ST	L	DW	R	BG
ST	0	+++	+	+++	+++
L	---	0	---	+++	+++
DW	-	+++	0	+++	+++
R	---	---	---	0	+++
BG	---	---	---	---	0

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

| M3	SH	ST	G	L	DW	R	BG
SH	0	---	-	-	---	+++	+++
ST	+++	0	+++	+++	-	+++	+++
G	+	---	0	+	---	+++	+++
L	+	---	-	0	---	+++	+++
DW	+++	+	+++	+++	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
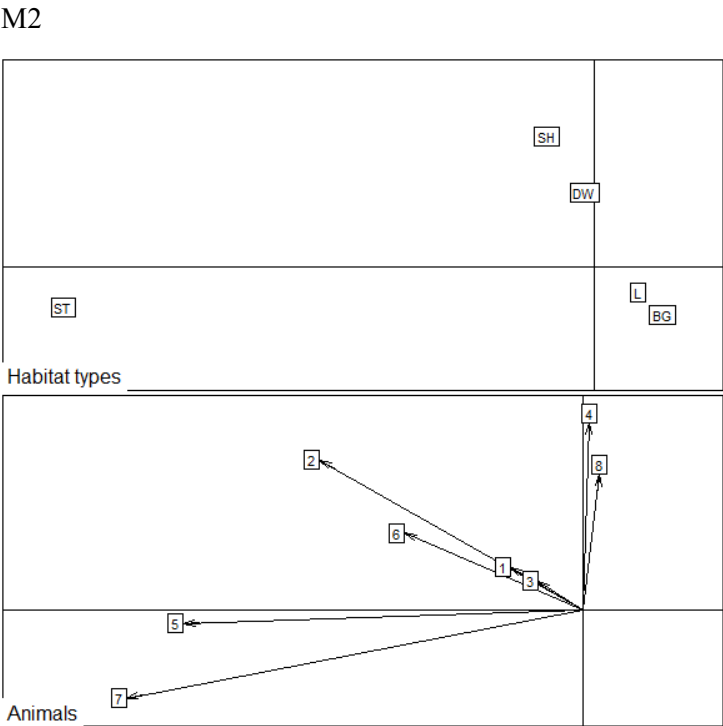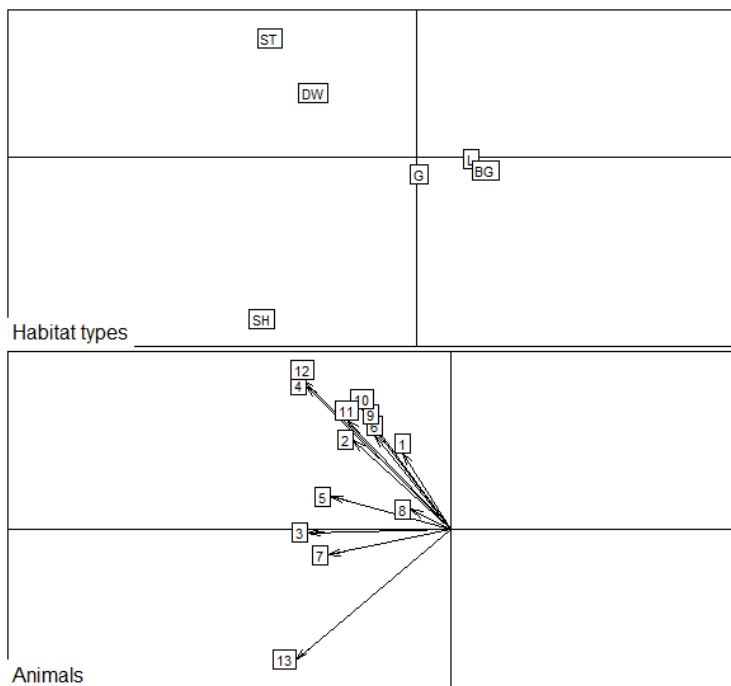907   individual.

908   Control
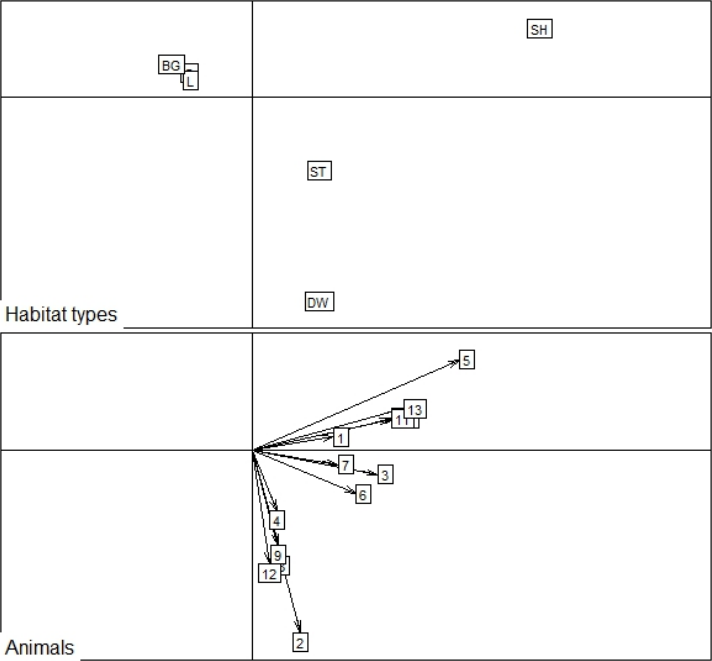
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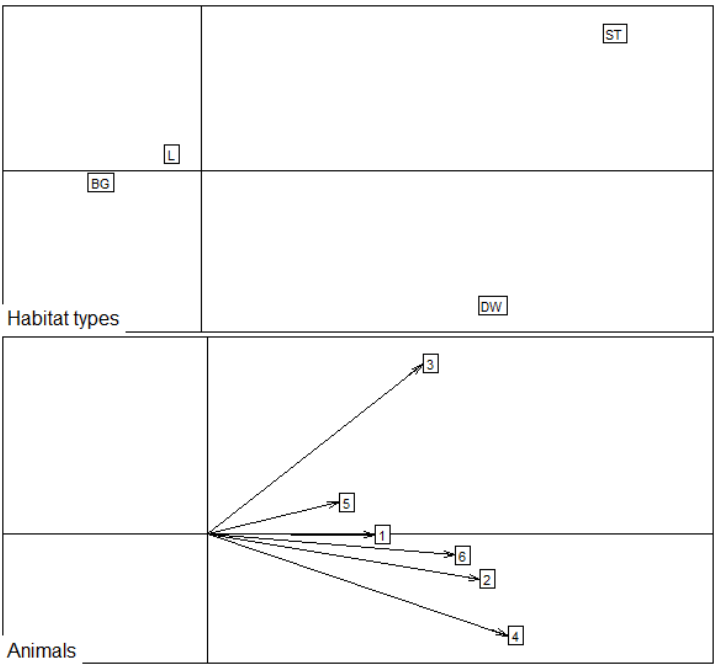
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911   M1

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913 M2

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916 M3

Habitat types

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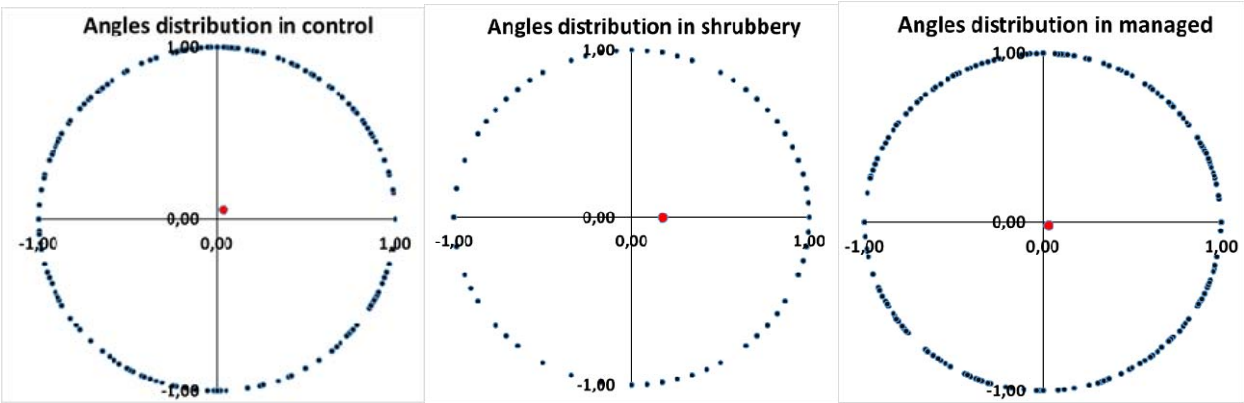
949   Appendix III. Even distribution of angles in shrubbery, control and managed stands.
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