

# Post-fire selective thinning of *Arbutus unedo* L. coppices keeps animal diversity unchanged: the case of ants

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## Abstract

• **Context** In the Mediterranean area, different post-fire management strategies are used for coppices of resprouting species to promote a more regular forest structure, enhance plant growth, and reduce fire risk. However, the effects of these management treatments on forest-associated fauna are unknown, which in turn could be limiting their beneficial effects.

• **Aims** The aim of this work was to determine whether forest management of a recently burned area dominated by a vigorous resprouting tree species (*Arbutus unedo* L.) affects ant communities.

• **Methods** Ant communities, sampled using pitfall traps, were examined from unmanaged and selective thinning coppices of *A. unedo*. Ants are here used as bioindicators of ecosystem health and surrogates for other animal groups.

• **Results** Very limited effects of these post-fire management strategies on the structure and composition of ant communities were found. The lack of effects could be due to the reported small changes in physical conditions among treatments; or either, the most sensitive ant species to these post-fire management treatments might be the same ones affected by fire and, consequently, the ant species that would potentially be affected most were no longer in the study area.

• **Conclusion** The lack of any significant effects caused by these post-fire management practices on the associated fauna of *A. unedo* coppices points out the suitability of these treatments in these circumstances.

**Keywords** Ants · Biodiversity conservation · Mediterranean · Post-fire management · Selective thinning · Strawberry tree

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**Contribution of the co-authors** Xavier Arnan: designing the experiment, supervising the field and laboratory work, and running the data analysis.

Olga Boet: supervising the laboratory work.

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## 1 Introduction

Logging and wildfires are two forest disturbances that promote major shifts in vegetation cover, which in turn lead to variations in microhabitat conditions and resource availability for associated fauna. Consequently, these changes may affect the composition and structure of animal communities. Accordingly, decreases in the diversity of forest fauna have been documented following different types of forest management activities (e.g., Thiollay 1997; Kavanagh and Webb 1998; Davis et al. 2001). In the case of wildfires, direct effects on forest fauna have also been reported, causing both mortality and fleeing, and indirect effects, such as the modification of forest habitats (e.g., Arnan et al. 2007; Rost et al. 2012). Thus, species that survive a fire or return after a fire may have difficulties living in the new conditions. On the other hand, there are species that are capable of reestablishing in these modified habitats that have different characteristics compared

to before the fire (e.g., McCullogh et al. 1998; Rodrigo et al. 2008; Arnan et al. 2013b).

Wildfires, which are increasingly more frequent and intense (Piñol et al. 1998), are a major source of disturbance that has profound impacts on forests in the Mediterranean basin (Rodrigo et al. 2004). In many burned areas, different forestry management treatments are often applied after a fire, either to harvest burned wood (Rost et al. 2012; Quevedo et al. 2013) or to encourage faster regeneration. For example, selective removal of resprouts may be applied in areas dominated by species with a great capacity for post-fire resprouting (Quevedo et al. 2013). In this context, post-fire forest management practices can be understood as a second disturbance to the ecosystem, which could, in turn, affect the fauna that has established in the burned area and thus hinder their recovery (Apigian et al. 2006).

Ants are among the most diverse, abundant, and ecologically significant organisms on Earth (Hölldobler and Wilson 1990). They can modify the abiotic and biotic properties of their environment by performing a variety of ecological functions (e.g., Hölldobler and Wilson 1990; Zelikova et al. 2011). Consequently, ants are considered to be crucial components of most terrestrial ecosystems (Hölldobler and Wilson 1990; Lach et al. 2010). Ant communities are often used as bioindicators of ecological change, arising from natural or anthropological disturbances (e.g., Vasconcelos et al. 2000; Andersen and Majer 2004; Hoffmann and Andersen 2003; Nakamura et al. 2007). In the case of fire, ant community recovery post-fire depends on both the direct mortality caused by the high temperature as well as by the changes in the vegetation cover caused by fire (Andersen et al. 2006; Arnan et al. 2006, 2007; Rodrigo and Retana 2006; Parr and Andersen 2008). Therefore, fires can generate changes in the composition of ant communities (Andersen et al. 2009; Arnan et al. 2006), including species richness and diversity, which in some cases increase (Mackay et al. 1991; Andersen et al. 2009), in others decrease (York 2000; Rodrigo and Retana 2006), or, as in some Mediterranean ecosystems, are not affected by fire (Arnan et al. 2006).

It is also known that forest management practices can affect ant communities (Vasconcelos et al. 2000; Dunn 2004; Gómez and Abril 2011), since in the short term it often implies a decrease in vegetation cover and may also alter food availability. However, as far as we know, there is no information available regarding whether post-fire salvage logging, which involves a modification of forest condition, affects the recovery of ant communities. This is of particular interest for species that inhabit closed canopy forests, as these species are the most affected by fire (York 2000; Parr and Andersen 2008; Andersen et al. 2006, 2009; Arnan et al. 2006; Rodrigo and Retana 2006). Therefore, it is important for the conservation of areas affected by fires, as is the case of the Mediterranean basin, to determine whether the post-fire forest

management treatments often applied to large areas (Quevedo et al. 2013) can be considered a threat to the recovery of ant communities after a fire event.

The aim of this work is to determine whether forest management of a recently burned area dominated by a vigorous resprouting tree species affects the local fauna. In particular, this study focuses on the effects of the application of selective thinning of resprouts and the selective thinning of resprouts plus mechanical clearing of the surrounding vegetation on ant communities established after fire in a strawberry tree (*Arbutus unedo* L.) coppice. We test the following hypotheses: (a) post-fire forest management treatments on the strawberry tree coppices will change the structure and composition of ant communities, since environmental conditions will also change, and (b) the more intense the forest management treatment, the greater the changes to ant communities. This is the first study that analyzes the effects of selective thinning of resprout species on fauna in a post-fire scenario in a Mediterranean area, where wildfires have a great impact on biota.

## 2 Materials and methods

### 2.1 Study area

This study was undertaken in the north of the Baix Llobregat region (41° 35' N, 1° 52' E, Catalonia, NE Iberian Peninsula), in a 240-ha area affected by three canopy wildfires in recent years (1985, 1986, and 1994). Before these fires occurred, this area was covered by Aleppo pine (*Pinus halepensis* Mill.) forests and, after the fire events, developed forested areas of strawberry trees (Arnan et al. 2013a). The study area is located between 390 and 500 m above sea level and in a region with a dry Mediterranean climate (according to Thornthwaite's humidity index), with a mean annual temperature of 13.5 °C and a mean annual precipitation of 650–700 mm.

The strawberry tree (*A. unedo* L.) is a sclerophyllous tree species that belongs to the family of Ericaceae. In our study area, the strawberry tree forests had a density of 1,265±625 individuals/ha and a number of 15,494±7,154 resprouts/ha, before treatments (Quevedo et al. 2013). In general, the strawberry tree has very vigorous resprouts after a disturbance (such as a cut or a fire) and can develop up to 60 resprouts/stump (Quevedo et al. 2013). This vigorous resprouting response involves a slow growth of stems due to the high competition for resources at the individual level. In these conditions, the lack of forest management promotes a high horizontal continuity and, as consequence, a high fire risk (Quevedo et al. 2013).

## 2.2 Experimental design and application of post-fire experimental treatments

In 2006, we established four sampling zones in the study area (6.5 ha each one, hereafter, blocks), distance of 250–400 m between them, with a high density of *A. unedo* vigorously resprouted after the last fire (1994). In each block, we randomly chose three plots of 50 m × 50 m of similar slope (10 to 30 %) and aspect (southwest) (distance between plots 40–125 m). All the selected plots showed similar forest structural characteristics. For a more detailed description of spatial position and characteristics of plots, see Quevedo et al. (2013).

In the winter of 2006–2007, the three plots within each block were assigned one of three management strategies at random: (1) control strategy (henceforth C), in which nothing was done to the plot; (2) a selective thinning treatment (henceforth T), in which resprouts were selectively removed with electric scissors (Electrocoup F3005) from all of the strawberry trees in the plot—one out of five resprouts was retained per individual (preserving the largest diameter resprouts); and (3) a selective thinning plus understory clearing treatment (henceforth U), in which the same selective thinning procedure was applied in tandem with the mechanical clearing of all the understory vegetation of the plot (for more details, see Quevedo et al. 2013).

## 2.3 Field sampling

### 2.3.1 Characterization of the strawberry tree forest structure

Vegetation structure is an important predictor of the structure and composition of ant communities (Retana and Cerdá 2000; Lassau and Hochuli 2004; Arnan et al. 2007). In order to characterize the type of habitat, we established a 40-m × 40-m physical grid in each plot, with each cell measuring 1 × 1 m. Each of the 1,600 cells was visually assigned one of the following three types of habitats: (1) *Arbutus* cover, when the majority of the cell was covered by strawberry tree; otherwise, (2) shrub cover, if plant coverage had a height between 1.20 and 3 m; and (3) herbaceous cover, if that coverage was lower than 1.20 m or if no vegetation cover was present (bare and rocky soil). Then, we established the habitat composition of each plot as the percentage of each type of habitat.

To characterize the abiotic environment of each habitat type, we measured different environmental variables during each season of 2008:

- (a) Soil moisture. Three soil samples were taken from each plot, one for each of the three habitat types (total of 36 samples). Each sample, in turn, was formed by three subsamples of the same habitat, taken within the same plot. Soil samples were taken with a manual metal probe

(1,500 cm<sup>3</sup>), placed in a sealed plastic bag, weighed in the laboratory, set to dry at 105 °C for 24 h, and subsequently reweighed. The moisture content of the soil was obtained as the difference between the wet and dry weights.

- (b) Percentage of shade provided by vegetation. This was determined by measuring the photosynthetically active radiation (PAR) with a Decagon Sunfleck SF 40 Ceptometer (Delta Devices, Cambridge, UK) at the time of maximum sunlight, above and below the vegetation canopy. Measures were obtained in each plot and habitat type directly (five repeats per habitat) (total of 36 samples). Then, the percentage of shade in each point was calculated as the percentage of the difference between the two PAR values (above and below) in respect to the above PAR value.
- (c) Temperature. Temperature at ground level was obtained with continuous temperature registers (Stow Away Tidbit Temp Logger; Onset Computer Corp., Pocasset, MA) placed at ground level in each plot and habitat type. The sensors recorded temperature data every 30 min for a week. Since the sensors measured a maximum limit of 38 °C, the percentages of temperature data values below 20 °C and above 35 °C were used, representing the values out of the range of temperatures in which Mediterranean ants are active (Retana and Cerdá 2000).

### 2.3.2 Ant sampling

We used pitfall traps to measure the ground ant composition and abundance of each plot. Sampling was conducted in 2008, 2009, and 2010 and in May and July, when ant activity is highest in Mediterranean ecosystems (Cros et al. 1997). Pitfall traps were 7.5-cm-diameter 9.5-cm-deep plastic vials partially filled with water, soap, and salt. A 4 × 4 grid of traps with 5-m spacing was established in each plot. The traps were placed in the center of the plot to minimize edge effects and were operated for 1 week for each sampling period. The contents of the 16 traps of each plot were lumped together to obtain a single sample per plot. The samples from the two sampling periods (May and July) from each plot and year were pooled for analysis, so that we had one sample for the whole activity period of ants (i.e., total number of ants per 16 traps across 14 days by each year). The ants were then sorted in the laboratory and identified to the species level according to Bernard (1968) and Espadaler (1990), supplemented with the identification of some specimens by Dr. Xavier Espadaler.

## 2.4 Data analysis

The effects of forest management (C, T, and U) on habitat composition were analyzed using blocked univariate

ANOVAs, where treatment was the main factor and block, the random factor. We conducted a separate analysis for each habitat type (*Arbutus* cover, shrub cover, and herbaceous cover). The effects of forest management and habitat type on the different abiotic variables (soil moisture, PAR, and temperature values below 20 °C and above 35 °C) were also analyzed using blocked univariate ANOVAs, where treatment, habitat, and season were the main factors and block, the random factor. We conducted a separate analysis for each abiotic variable and season (winter, spring, summer, and fall). We conducted post hoc comparisons using the Tukey test when the differences for the main factors were significant. The percentages of each habitat type and the PAR were arcsine-square root transformed in order to achieve data homoscedasticity. The statistical package SPSS (SPSS Inc. 2006) was used for these analyses.

To investigate the composition and structure of communities of ants in each plot, we computed the following indexes: (a) overall abundance of ants in pitfall traps; (b) species richness (*S*); (c) Shannon diversity index ( $H' = -\sum_{p=1}^S p_i \ln p_i$ , where  $p_i$  is the proportion of workers of the *i*th species in traps, and *S* is the number of species); and (d) the index of numerical dominance of the most abundant ant species ( $ID = 100(y_1/y)$ , where  $y_1$  is the abundance of the most abundant species and *y* is the overall abundance of all ant species in the plot). Each ant species was assigned to a particular trophic guild according to its diet, as determined from Aman et al. (2013b). We established the following guilds: liquid food (nectar and/or honeydew), liquid food and insects, only insects, insects and seeds, and only seeds. We analyzed the changes in the structure of the ant community and the proportion of individuals belonging to the different dietary guilds among treatments and years by using mixed linear models, where treatment, year, and their interaction were the main factors and block and plot the random factors. Overall abundance was log transformed, while the numerical dominance index and the relative abundance of each trophic guild were arcsine-square root transformed in order to achieve homoscedasticity. The statistical package R (version

2.13.0; R Development Core Team 2008) was used in these analyses (function “lme” in package “nlme”).

Significant differences in the ant community composition between treatments were determined with multivariate analysis using the program Primer 5.1.2 (Clarke and Gorley 2006). Ant abundance per plot was log transformed to obtain homoscedasticity. First, plots were ordered using nonmetric multidimensional scaling (NMDS), based on the Bray-Curtis similarity matrix. Then, analysis of similarities (ANOSIM) was used to test for significant differences in ant community composition related to forest management treatment (C, T, and U). Ant species that occurred in less than three plots and that had a relative abundance of <1 % were excluded from the analysis. Finally, similarity percentages (SIMPER) were used to identify the ant species that contributed most to differences in community composition. All these analyses were conducted for each year (2008, 2009, and 2010).

### 3 Results

#### 3.1 Habitat composition

The U plots showed significant differences in habitat composition with respect to the C and T plots. These had a lower percentage of shrub cover ( $F=30.5$ ;  $p<0.0001$ ;  $6.2\pm 1$ ,  $19.2\pm 2.0$ , and  $27.5\pm 9.6$  % for the U, C, and T plots, respectively) and a greater percentage of herbaceous cover ( $F=12.8$ ;  $p=0.007$ ;  $64.5\pm 8.9$ ,  $39.6\pm 10.0$ , and  $8.4\pm 38.4$  % for the U, C, and T plots, respectively), with respect to the other two types of plots. There were no significant differences in the habitat composition between plots C and T ( $p>0.05$ ). There were also no significant differences for the *Arbutus* cover category between plots of different treatments ( $p>0.05$ ,  $41.2\pm 11.1$ ,  $34\pm 5.8$ , and  $29.3\pm 8.1$  % for the C, T, and U plots, respectively). According to the different composition of habitats in the plots and to differences in the physical characteristics of habitats (Table 1), plots showed only small differences between treatments in soil moisture in

**Table 1** *F* values of the blocked univariate ANOVA tests for each season and for each variable that we used to characterize habitats (soil moisture (%), percentage of shade exerted by vegetation (PAR), percentage values

of temperature ground level below 20 °C (T20) and percentage above 35 °C (T35)) in the treatment (T), habitat (H), and block (B)

Source	Winter			Spring			Summer			Fall		
	T	H	B	T	H	B	T	H	B	T	H	B
Soil moisture (%)	2.4	0.8	4.1*	3.9*	1.2	2.1	0.1	0.6	1.1	5.6*	1.3	3.1*
Shade (%)	1.9	55.5*	5.7*	0.1	86.5*	5.4*	4.3*	32.9*	12.3*	0.1	24.6*	9.9*
T20	0.6	3.4*	3.4*	0.7	4.7*	43.2*	1.2	2.0	5.9*	0.7	2.3	58.0*
T35	— <sup>a</sup>	— <sup>a</sup>	— <sup>a</sup>	2.0	1.0	1.3	1.7	1.9	3.9*	0.7	2.4	2.7

\* $p\leq 0.05$ ; indicates significant values

<sup>a</sup> Analysis not available due to lack of data

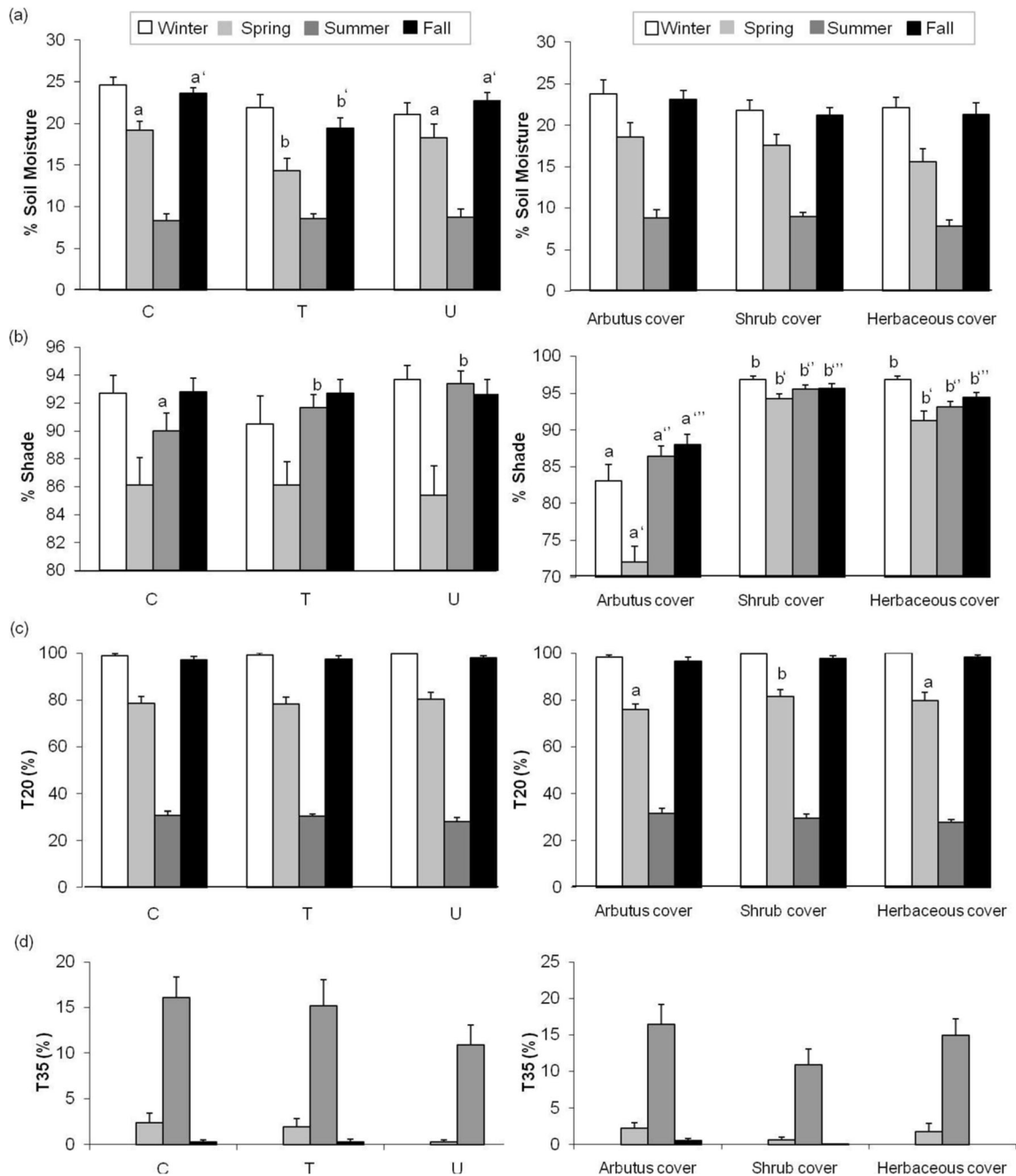
spring and fall and in shadow in summer, but no differences for temperature (Fig. 1).

### 3.2 Ant community structure

In total, 38,159 worker ants from 31 species were collected (12,741, 9,600, and 15,818 from years 2008,

2009, and 2010, respectively) (Table 2). Species richness varied between 19 and 22 in the C plots, between 22 and 25 in the T plots, and between 20 and 22 in the U plots.

The mixed linear models showed no effects of treatments, years, or their interaction on species richness, the Shannon diversity index ( $H'$ ), and the index of



**Fig. 1** Environmental variables measured for each forest management treatment and habitat for each season: winter (white bars), spring (light gray bars), summer (dark gray bars), and fall (black bars): **a** percentage of soil moisture, **b** percentage of shade, **c** percentage values of temperature below 20 °C ( $T_{20}$ ), and **d** percentage values of temperature above

35 °C ( $T_{35}$ ). The vertical bars indicate standard error (SE) of the mean. Different letters indicate significant differences between categories, according to the Tukey post hoc tests. C control, T selective thinning, U selective thinning plus understory clearing



**Table 2** List of ant species and number of worker ants collected in the plots of the three treatment intensities during the 3 years of study (2008, 2009, and 2010). The intensities of treatment are control (C), selective thinning (T), and selective thinning plus understory clearing (U)

Ant species	Diet	C plots			T plots			U plots		
		2008	2009	2010	2008	2009	2010	2008	2009	2010
<b>Myrmicinae</b>										
<i>Aphaenogaster gibbosa</i>	i, s	23 (2)	13 (1)	25 (2)	–	–	5 (1)	4 (3)	4 (2)	4 (2)
<i>Aphaenogaster subterranea</i>	i	6 (2)	–	3 (1)	3 (2)	3 (2)	4 (1)	10 (1)	4 (1)	12 (1)
<i>Crematogaster scutellaris</i>	l	–	–	–	3 (2)	5 (2)	11 (2)	27 (1)	40 (1)	110 (1)
<i>Crematogaster sordidula</i>	l	62 (3)	56 (2)	70 (2)	–	–	–	13 (1)	7 (1)	33 (2)
<i>Goniomma blanci</i>	s	1 (1)	–	–	–	–	–	–	–	–
<i>Leptothorax acervorum</i>	i, l	–	–	–	2 (1)	–	1 (1)	–	4 (1)	–
<i>Messor barbarus</i>	s	–	–	257 (1)	4 (2)	2 (1)	–	–	–	–
<i>Messor bouvieri</i>	s	76 (1)	–	–	751 (1)	2 (1)	2,127 (1)	–	–	–
<i>Messor capitatus</i>	s	487 (2)	849 (2)	1,317 (2)	29 (2)	365 (2)	127 (2)	130 (2)	421 (3)	704 (2)
<i>Myrmecina graminicola</i>	i, l	–	1 (1)	–	–	–	–	1 (1)	–	–
<i>Myrmica spinosior</i>	i	2 (1)	1 (1)	1 (1)	19 (3)	11 (2)	20 (2)	26 (1)	9 (2)	16 (1)
<i>Pheidole pallidula</i>	i, s	844 (4)	425 (4)	728 (4)	504 (4)	227 (4)	723 (4)	193 (4)	183 (4)	502 (4)
<i>Solenopsis</i> sp.	i, s	1 (1)	–	2 (2)	1 (1)	–	3 (2)	2 (2)	1 (1)	1 (1)
<i>Temnothorax niger</i>	i, l	57 (4)	25 (3)	36 (4)	19 (3)	12 (3)	34 (3)	17 (4)	13 (4)	19 (4)
<i>Temnothorax racovitzai</i>	i, l	23 (4)	6 (4)	29 (4)	9 (3)	6 (4)	19 (4)	27 (3)	25 (4)	30 (4)
<i>Temnothorax recedens</i>	i, l	–	1 (1)	1 (1)	–	1 (1)	7 (2)	–	1 (1)	–
<i>Tetramorium semilaeve</i>	i, s	2 (1)	–	2 (1)	–	–	1 (1)	–	–	–
<b>Formicinae</b>										
<i>Camponotus aethiops</i>	i, l	–	–	1 (1)	–	–	–	–	–	–
<i>Camponotus cruentatus</i>	l	1 (1)	–	–	405 (1)	94 (1)	98 (1)	–	–	–
<i>Camponotus lateralis</i>	i, l	–	–	–	1 (1)	5 (2)	9 (2)	–	1 (1)	–
<i>Camponotus piceus</i>	l	50 (4)	56 (4)	39 (4)	89 (4)	51 (4)	36 (4)	23 (3)	24 (3)	47 (3)
<i>Camponotus pilicornis</i>	l	55 (4)	71 (4)	73 (4)	45 (4)	55 (4)	55 (4)	28 (4)	58 (4)	84 (4)
<i>Camponotus sylvaticus</i>	l	90 (3)	102 (2)	88 (2)	237 (4)	226 (4)	190 (3)	185 (3)	479 (3)	250 (4)
<i>Formica gagates</i>	i, l	133 (2)	72 (2)	235 (2)	572 (3)	323 (2)	189 (2)	1,228 (1)	1,229 (2)	755 (2)
<i>Formica gerardi</i>	i, l	6 (2)	11 (3)	6 (2)	80 (4)	20 (2)	47 (3)	2 (2)	1 (1)	1 (1)
<i>Formica subrufa</i>	i, l	459 (1)	465 (2)	441 (2)	432 (3)	480 (3)	507 (3)	1,088 (2)	2,050 (2)	3,547 (2)
<i>Lasius cinereus</i>	l	1,192 (3)	15 (1)	17 (3)	1 (1)	14 (1)	4 (2)	1,076 (2)	69 (3)	476 (2)
<i>Lasius myops</i>	l	–	4 (1)	–	–	1 (1)	2 (2)	–	–	3 (1)
<i>Plagiolepis pygmaea</i>	l	467 (4)	191 (4)	380 (4)	511 (4)	201 (4)	346 (4)	343 (4)	289 (4)	556 (4)
<i>Polyergus rufescens</i>	i, s	–	–	–	1 (1)	–	–	–	–	1 (1)
<b>Dolichoderinae</b>										
<i>Tapinoma nigerrimum</i>	i, l	223 (4)	65 (4)	101 (4)	158 (4)	73 (4)	118 (3)	182 (3)	82 (3)	132 (3)
Total number of individuals captured		4,260	2,429	3,852	3,876	2,177	4,683	4,605	4,994	7,283
Species richness		22	19	22	23	22	25	20	22	21

Parentheses indicate the number of plots in which each species appears for each year and intensity of treatment. Diet data are based on Arnan et al. (2013b)

*i* insects, *s* seeds; *l* liquid food (nectar and/or honeydew)

numerical dominance of the most abundant ant species (Table 3). Significant differences were found between the years 2008 and 2009, as well as between 2009 and 2010, in terms of total ant abundance ( $467 \pm 61$ ,  $356 \pm 88$ , and  $705 \pm 158$  ants per plot for the years 2008, 2009, and 2010, respectively) (Table 3).

### 3.3 Ant community composition

The ANOSIM analyses showed only a slight effect of treatments on ant community composition in the long term (year 2010; global  $R = -0.238$ ,  $p = 0.014$ , Fig. 2;  $p > 0.05$  for the years 2008 and 2009). The only significant differences were

**Table 3** Effect of treatment (forest management C, T, and U) and year (2008, 2009, and 2010) on variables describing structure of ant communities. *F* values, significance (*p*), and degrees of freedom (df) come from the ANOVA tables of the mixed linear models

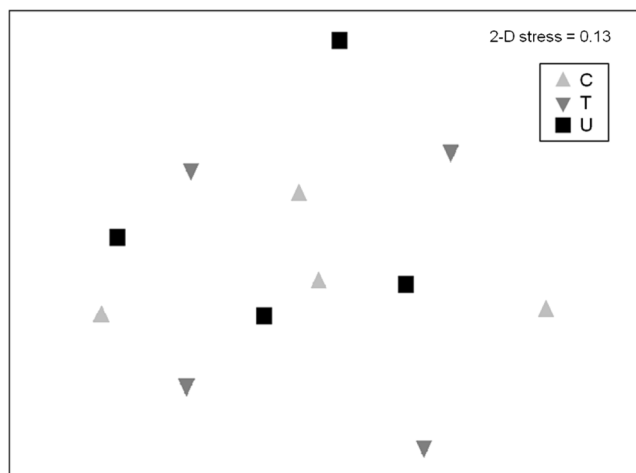
Source of variation	df	Total ant abundance		Richness		Shannon diversity index ( <i>H'</i> )		Numerical dominance	
		<i>F</i>	<i>p</i>	<i>F</i>	<i>p</i>	<i>F</i>	<i>p</i>	<i>F</i>	<i>p</i>
Treatment	2	1.2	0.374	1.7	0.256	1.8	0.247	2.3	0.185
Year	2	9.3	0.002*	1.6	0.236	0.3	0.777	0.2	0.828
Treatment×year	4	0.4	0.819	1.4	0.278	0.1	0.981	0.1	0.961

\* $p < 0.05$ ; indicates significant value

between the C and U plots (post hoc analyses:  $p = 0.029$  between the C and U plots,  $p = 0.114$  between the C and T plots, and  $p = 0.057$  between the T and U plots). Nevertheless, Fig. 2 shows no clear separation of treatments. In fact, the negative value of the global R statistic indicates a very large variability within plots of the same treatment, but very low variability between treatments. So, despite the fact that it seems that there were significant differences between ant community from C and U plots, they were very slight and difficult to assess given the high variability within treatments. The SIMPER analysis revealed that the species that contributed to the slight dissimilarity between C and U plots were *Messor capitatus* (12.07 %), *Formica subrufa* (10.52 %), *Formica gagates* (10.17 %), and *Pheidole pallidula* (8.78 %).

### 3.4 Ant dietary guilds

The relative abundance of each ant dietary guild was not significantly different between treatments or years. Their interaction was not significant either (mixed linear models,  $p > 0.05$  in all cases).



**Fig. 2** Nonmetric multidimensional scaling (NMDS) ordination in two dimensions of plots based on ant abundance data in 2010. C control, T selective thinning, U selective thinning with understory clearing

## 4 Discussion

This work has demonstrated that there is virtually no effect of post-fire selective thinning on the structure of ant communities inhabiting the *A. unedo* coppices of our study area. The applied forest management treatments did not cause any impact, either positive or negative, on the total abundance, species richness, diversity, dominance, or the composition of ant community, neither in the short or the medium term (Table 3, Fig. 2). This result contradicts numerous works, much from the tropics of Amazon, where it has been found that the application of selective thinning leads to a negative impact on the species richness or the composition of ant species (e.g., Roth et al. 1994; Vasconcelos et al. 2000; Dunn 2004; Nakamura et al. 2007; Palladini et al. 2007), and other research on faunal groups also used as bioindicators, as for example, beetles (Davis et al. 2001), birds (Thiollay 1997), small mammals, reptiles, and amphibians (Kavanagh and Webb 1998). However, there are other studies that show similar results to those provided in this work that indicate that some forest harvesting does not negatively affect the overall animal biodiversity (Azevedo-Ramos et al. 2006) and have a minimal impact on ant communities (Vasconcelos et al. 2000; Andersen et al. 2009). Several of these works agree that one of the key factors that determine if forest harvesting will have an impact on the richness and/or composition of biodiversity is the intensity, and also the frequency, of treatments (Dunn 2004; Andersen et al. 2009; Gómez and Abril 2011). The more intense the treatment, the more change is generated on the vegetation cover, causing a larger impact on the whole ecosystem (Maeto and Sato 2004; Palladini et al. 2007). Nonetheless, we were unable to detect this larger effect in even our most intense treatment, which involved both the selective thinning and the clearing of understory vegetation.

Soil moisture determines ant foraging activity, the abundance of food resources, the suitability of places to make their nests, as well as predation by other species of ants (Wang et al. 2001). Also, ground temperature has a direct effect on nest temperature levels and an indirect effect on food supply via vegetation growth (York 2000). In fact, when we applied selective thinning along with the clearing of the understory,

we were able to modify proportions of habitats, resulting in a greater availability of more open habitats. However, there are only small differences in ground temperature between habitats but similar soil moisture, probably due to an early resprouting of the understory vegetation. As a consequence, although this change in open habitat proportion, differences between treatments as a whole in physical conditions were very small (Fig. 1). So, these small changes were not great enough to affect the structure or the composition of the ant community.

Our results also show that these treatments could not modify the relative abundance of ant dietary guilds in the study area. Although it might make more sense that clearing undergrowth would cause a decrease in food resources, such as seeds generated by the shrub layer and insects that inhabit this environment (termites, springtails, other ants, etc.) (Hölldobler and Wilson 1990; Arnan et al. 2006), this did not yet have any effect or was not important enough to negatively impact ants that feed on these food resources. On the other hand, the shrubs present in our study area have a high resprouting capacity after cutting (*Pistacia lentiscus*, *Quercus coccifera*, *Thymus vulgaris*, *Viburnum tinus*, etc.), so that they recover quickly and with ease.

One possible explanation for the lack of any effect of post-fire forest management treatments on ant community might be that the ant species most sensitive to these forest practices are the same ones that are also most affected by changes in vegetation promoted by fire, i.e., tree-dwelling ant species (Arnan et al. 2013b). Consequently, the ant species that would be most affected by post-fire management treatments were no longer in the study area. As such, post-fire forest management might affect the process of post-fire colonization in the long term although we did not detect any evidence in the short and midterm.

At any rate, the act of reducing forest cover by selective thinning has minimal effect on the ant community of this area, nor does it if the understory is also removed. This implies that the application of such treatments that have clear advantages in the recovery of the structure of these burned forests and in the decrease of fire risk (Quevedo et al. 2013) may be advisable on a regional basis without any negative effect in the ant fauna. In fact, as these treatments favor the height and diameter growth of the stands of the strawberry tree (Quevedo et al. 2013), accelerating the recovery of these forests towards more structured and closed might even encourage, in the medium and long term, the appearance of ant species normally only found in more closed areas (Arnan et al. 2006; Rodrigo and Retana 2006). However, despite the fact that ants can be used as bioindicators of the response of fauna in general (Hoffmann and Andersen 2003; Andersen and Majer 2004), it would be appropriate to check the effect of these treatments on other faunal groups. In this context, analyzing the effect of similar treatments on the faunal community in the medium or long term is important in order to be able to understand the overall

effect of these post-fire forest management treatments in the biodiversity of burned areas.

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