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RESPONSE OF BEETLE COMMUNITIES FIVE YEARS AFTER WILDFIRE IN MEDITERRANEAN FOREST ECOSYSTEMS

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Elia M., Laforteza R., Tarasco E., Sanesi G. – Response of beetle communities five years after wildfire in Mediterranean forest ecosystems.

Wildfires are one of the most important drivers of forest composition and biodiversity in the Mediterranean Basin. Many studies have demonstrated that fires can affect insect diversity by altering the functional traits of species groups. We examined the 5-year response of beetles to wildfires by assessing patterns of community composition across a gradient from forest interior to forest edge to burnt forest area in Southern Italy. Our objective was to characterize the relationship between distance from the forest edge and occurrence of beetle taxonomic assemblages. We analyzed the composition, similarity, and dominance of ground beetle communities in randomly selected plots located along the forest-to-burned-area gradient. We found a negative relationship between community similarity and distance from the forest edge; moreover, the composition of species assemblages (within each family) became increasingly similar with proximity to the forest edge. As the distance from the forest edge into the burned area became greater the dominance of few species increased, and species composition shifted toward habitat generalists. The results partially support the notion that the differences in beetle communities probably are driven by habitat changes caused by fires, especially for those taxa with many specialist species in feeding and oviposition habitats. Understanding the biological effects of wildfires is necessary prior to design management strategies and policies for counteracting the loss of biodiversity at the global, regional and national levels.

KEY WORDS: Wildfire, Beetles community, Similarity index, Mediterranean forests, Biodiversity.

INTRODUCTION

Wildfires are one of the main factors influencing the composition and spatial turnover of insect communities, which represent an important pool of biodiversity in forest ecosystems and landscapes (HOLLIDAY, 1991; VILLA-CASTILLO and WAGNER, 2002; RAINIO and NIMELÄ, 2004). The insect biodiversity can be directly and indirectly affected by wildfire. Direct effects are due to the heat and smoke generated by the fire. Many insects may be able to escape fires through flight, but non-flying insects need to rely on other strategies to survive, including exploiting micro-refugia such as bush clumps, rocks and marshy areas (UYS *et al.*, 2006), burrows (DIPPENAAR-SCHOEMAN, 2002) or moving beneath the soil surface (VILLANI *et al.*, 1999). Indirect effects can be ascribed to the alteration of ecosystem structure, especially vegetation (ELIA *et al.*, 2016), soil and litter composition (WHELAN, 1995; PICKERING, 1997; SWENGEL, 2001; BUDDLE *et al.*, 2006). The consequence is that both species richness and abundance were negatively affected by fire.

A number of studies have focused on the effects of wildfires on different insect taxa, especially ground beetles (LARSEN and WILLIAMS, 1999; GANDHI *et al.*, 2001; SAINT-GERMAIN *et al.*, 2005; JOHANSSON *et al.*, 2011; ELIA *et al.*, 2012). In their study on Mediterranean pine forests, NUNES *et al.* (2006) found that the abundance of ground beetles tends to decrease over a two-year time period after the fire. ELIA *et al.* (2012) confirmed these short-term effects of fires on ground beetle communities. On the other hand, other studies showed a contrasting pattern of increasing abundance during the post-fire period (see: WARREN *et al.*,

1987; REED, 1997; GARCIA-TEJERO *et al.*, 2013). Fire-induced effects on beetles are therefore difficult to generalize because diverse communities may respond differently during the post-fire period in relation to the resilience of the community (i.e., species turnover) and the rate of changing vegetation conditions after fires (TEASDALE *et al.*, 2013; REINHARDT *et al.*, 2006; CAMPBELL *et al.*, 2007; ELIA *et al.*, 2015). Although there is a body of literature focusing on the *cause-and-effect* relationship between wildfires and beetle communities in the short-term period (up two years after fire), only a few studies have investigated this relationship over a longer period of time after a fire.

In this paper, we investigated the response of beetle communities five years after fire by assessing patterns of community composition across a gradient from the forest interior to the burnt forest area. To this end, we analyzed the composition of beetle communities in terms of similarity and dominance in random plots located inside the forest (*control plots*) and over the burnt area at different distances from the nearest forest edge (*burnt forest plots*).

We hypothesized that the community composition of beetles within *control plots* located inside the forest area is representative of the community that would have existed in the absence of wildfire. This community composition should be more similar between control sites than between a control site and a forest location that has been affected by a wildfire (*burnt forest plot*). To test our hypothesis, we assessed the *similarity* of beetle species in a Mediterranean forest ecosystem five years after a fire. We used the distance between burnt forest plots and control plots to explain the spatial distribution of the beetle community across the gradient: *forest interior - forest edge - burnt area* (see Table 1). In a

Table 1 – The gradient used to explain the spatial distribution of beetle community in the study area.

<i>Burnt area</i>		36% of canopy cover and mean tree height of 2.00 m
<i>Forest edge</i>		64% of canopy cover and mean tree height of 4.00 m
<i>Forest interior</i>		91% of canopy cover and mean tree height of 4.50 m

subsequent step, we analyzed the community of beetles within burnt forest plots in terms of *dominance* to test the hypothesis that as the distance from the nearest forest edge increases, only a few species become dominant, whereas a majority of species exist in moderate or low abundance (subordinate or rare species).

We focused our analysis on beetles because they have been extensively studied as indicators of environmental conditions and ecosystem disturbances, including wildfires (LARSEN, 1999; MORETTI *et al.*, 2004; KOIVULA and SPENCE, 2006; PRYKE *et al.*, 2012). In this perspective, the dynamics of the beetle community through space and time could provide insights into the rates of changing vegetation conditions after fire (HOLLIDAY, 1991; VILLA-CASTILLO and WAGNER, 2002; ELIA *et al.*, 2015).

The identification of the beetle communities that are most sensitive to wildfires plays a critical role in terms of post-fire management and can be used to provide interpretive results. These outcomes may be applied to post-fire management strategies aimed at restoring forest ecosystems and species diversity (LAFORTEZZA *et al.*, 2015).

MATERIALS AND METHODS

STUDY SITE

Field data on beetle communities were collected in a forest area covering ca. 600 hectares in the region of Apulia, Southern Italy (40°55'43.63" N; 16°36'52.05" E) (Fig. I). During early July 2008, this area experienced a crown-fire ignited by farmers. The fire affected an area of ca. 260 hectares (40% of the total forestland caused a severe reduction of the canopy tree cover (around 95% of tree mortality). The woodland of this area is mainly represented by *Quercus pubescens* Willd. and *Q. coccifera* L., with large open spaces covered by herbs and seedlings such as:

Ruscus aculeatus (L.), *Myrtus communis* (L.), *Calicotome villosa* (Poir.), *Cistus incanus* (L.), *C. monspeliensis* (L.), *C. salvifolius* (L.), *Spartium junceum* (L.), and *Euophorbia dendroides* (L.). The understory vegetation includes *Pistacia lentiscus* (L.), *P. terebinthus* (L.), *Rosa canina* (L.), *Crataegus monogyna* (Jacq.), *Phillyrea* spp, *Rhamnus alaternus* (L.), *Erica arborea* (L.), *Rubus ulmifolius* (Schott.), *Smilax aspera* (L.) and *Arbutus unedo* (L.).

The variations in the structure and composition of this forest are mainly determined by human-induced impacts. These variations include coppicing and firewood collection, which have taken place over the past 20 years, overgrazing, and fires (LAFORTEZZA *et al.*, 2013). Table 1 shows the gradient from *forest interior* to *burnt area* used to explain the spatial distribution of the beetle community.

BEETLE TRAPPING

We sampled the beetle communities in our area by randomly placing three pit-fall traps inside the forest (*control plots*) and six traps in the burnt area at various distances from the nearest forest edge (*burnt forest plots*) (Fig. I). These standard pit-fall traps consisted of an outer and inner cup (9-cm diameter). They were buried in the ground with the rim at surface level and baited with vinegar to attract and preserve falling insects and subsequently covered with a sloped stone to protect them from rain and animal disturbance. The traps were kept active in the period lasting from May to July 2013 and emptied at ca. 10-day intervals. All specimens collected were placed in a plastic container filled with 70% ethyl alcohol and examined in a laboratory for species identification.

DATA ANALYSIS

To test our first hypothesis, we assessed the similarity of community composition (i.e., spatial turnover) across a

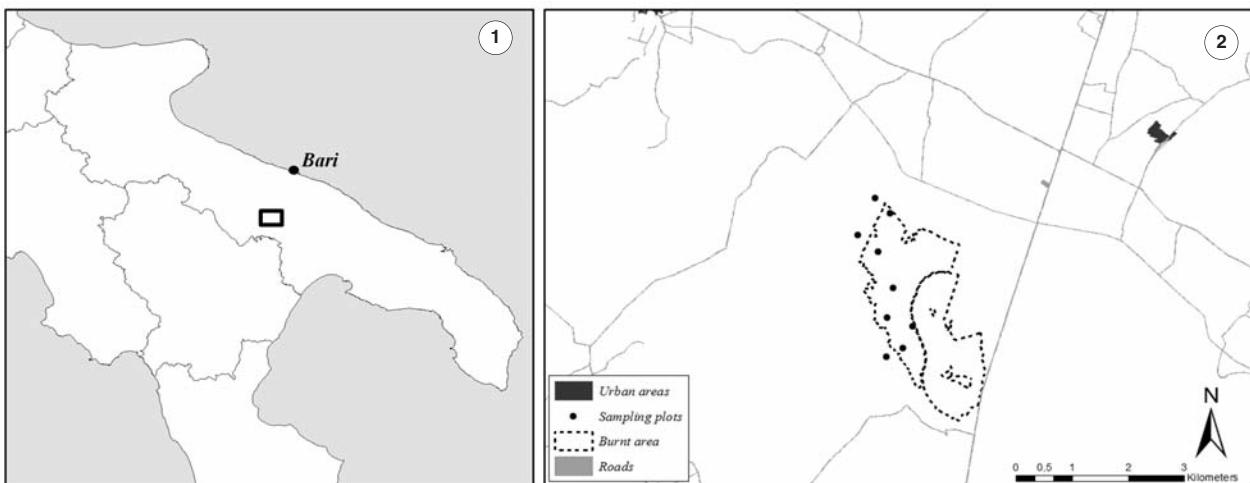


Fig. I – Location of the study area: 1) Apulia Region in Southern Italy; 2) boundary and location of the forest area affected by a fire in August 2008 (dashed line).

gradient from the forest interior to the burnt forest area in our study site.

For the purpose, we used the Bray-Curtis index (BC) – one of the most common measures of community similarity (MAGURRAN, 2004):

$$BC = 1 - \frac{\sum |Y_{ij} - Y_{ik}|}{\sum Y_{ij} + Y_{ik}} \quad (\text{eq. 1})$$

where Y_{ij} is the abundance of species i in site j , Y_{ik} is the abundance of species i in site k , and the summation is all species found at the two sites. The BC values range from 0 (no species in common) to 1 (identical abundance of all species) (LAFORTEZZA *et al.*, 2010). We calculated BC for each plot located in the burnt forest area (*burnt forest plots*) and the nearest control plot of similar altitude. We corrected the similarity value for any confounding effects of spatial autocorrelation and then used regression models to relate community similarity values to measures of distance from the nearest forest edge (D). We calculated similarity values by: (1) considering the entire community of beetles collected during the survey period; and (2) grouping species according to family.

As a subsequent step, we focused only on the data collected in burnt forest plots and used a k-dominance analysis to assess the dominance distribution of individuals among species of beetles. The number of individuals of each species was sorted in descending order and the proportion of the total number of individuals for each species was plotted on a percentage scale against the species rank. The shape of the k-dominance plot provides an indication of the dominance (or evenness) of a given species assemblage. For example, steep lines indicate assemblages with a high dominance, whereas shallower slopes characterize assemblages with low species dominance (i.e., higher level of evenness) (SILVA *et al.*, 2014). A k-dominance analysis displays the cumulative proportion of abundance against the log species rank.

RESULTS

A total of 5326 specimens belonging to 15 families and 64 species were collected during the survey period (Table 2). Twenty-two species of Carabidae, 8 of Nitidulidae, 7 of Staphylinidae, 5 of Tenebrionidae, 4 of Elateridae and Cerambycidae, 3 of Curculionidae, and 2 of Cetoniidae were trapped. We found only one species of Scydmaenidae, Lagridae, Dytiscidae, Lucanidae, Cantharidae, and Oedemeridae. The most abundant family was Nitidulidae (85.2%), followed by Carabidae (up to 6%), Staphylinidae (5.1%) and so forth. Within these families the most abundant species were *Epuraea aestiva*, *Carabus coriaceus* and *Stenus* sp., respectively.

As previously mentioned, we calculated an index of community similarity (BC) to assess differences in the community composition of beetles across the gradient from the forest edge to the burnt forest area (Fig. II). When considering the entire Coleoptera community, BC ranged between 0.78 and 0.50; in other words, five years after the fire, the community composition of beetles in the plots located near the forest edge was 78% similar to the one in the absence of wildfire. Conversely, the community composition of beetles in the burnt forest locations was only 50% similar to the one observed in the control plots. The same pattern occurred when different beetle species were grouped according to taxa (Fig. II).

To further explore these patterns of variation, a number of regression models were developed to explain community similarity (BC) as a function of distance from the nearest forest edge (D). The results are listed in Table 3. All regression models showed a negative relationship between community similarity and distance from the forest edge, with an explanatory power (r^2) ranging between 0.57 and 0.81. These results suggest that, in general, Coleoptera were negatively still affected by wildfire 5 years after the event. However, our regression models show a different slope, which can be seen as an indicator of spatial turnover of the beetle community. For example, the slope of the model describing community similarity in Nitidulidae appears significantly different (less steep) from that for Cerambycidae or Tenebrionidae. This suggests that the spatial turnover in Nitidulidae is less affected by wildfire than it is in other taxa of specialist species (SANTOS *et al.*, 2014).

To assess the dominance of species assemblage in our study area, we grouped species abundance data according to three classes of distance from the nearest forest edge: 0-120 m; 120-240 m; and >240 m. The dominance distribution of individuals among species across the landscape is illustrated in Fig. III. The k-dominance plot shows that as the distance

Table 2 – Number of beetle specimens collected during the survey period.

Coleoptera	Species	Abundance		
Cantharidae		1	Dytiscidae	
	<i>Cantharis livida</i> L.	1	<i>Colymbetes fuscus</i> L.	2
Carabidae		316	Elateridae	51
	<i>Abax ater</i> Vill.	33	<i>Agryotes aterrimus</i> L.	44
	<i>Amara fulva</i> Müller	1	<i>Elater ferrugineus</i> L.	2
	<i>A. plebeja</i> Gyll.	2	<i>Lacon punctatus</i> Herbst	4
	<i>Brachinus crepitans</i> L.	4	<i>Melanotus villosus</i>	1
	<i>Calathus cinctus</i> Mots.	55	Fourcroy	
	<i>C. fuscipes</i> G.	4		2
	<i>Calosoma sycophanta</i> L.	8	<i>Dorcus parallelepipedus</i> L.	2
	<i>Carabus coriaceus</i> L.	116	Nitidulidae	4539
	<i>C. glabratus</i> Payk.	4	<i>Carpophilus dimidiatus</i>	615
	<i>C. intricatus</i> L.	7	<i>Fabr.</i>	
	<i>C. violaceus</i> L.	16	<i>C. marginellus</i> Mots.	603
	<i>Cyprus caraboides</i> L.	7	<i>Epuraea aestiva</i> L.	701
	<i>C. italicus</i> Bon.	2	<i>E. pallescens</i> Stephens	650
	<i>C. rostratus</i> L.	2	<i>E. unicolor</i> Olivier	648
	<i>Harpalus rufipes</i> De Geer	1	<i>Pocadius ferrugineus</i> Fabr.	602
	<i>H. luteicornis</i> Duft.	7	<i>Soronia grisea</i> L.	168
	<i>Leistus ferrugineus</i> L.	1	<i>Urophorus humeralis</i> Fabr.	552
	<i>L. spinibarbis</i> Fabr.	27	Oedemeridae	1
	<i>Nebria brevicollis</i> Fabr.	1	<i>Oedemra flavipes</i> Fabr.	1
	<i>Pterostichus melas</i> Creut.	7	Scydmaenidae	9
	<i>P. nigrita</i> Payk.	9	<i>Mastigus ruficornis</i> Mots.	9
	<i>Zabrus tenebrioides</i> G.	2	Silphidae	2
Cerambycidae		20	<i>Phosphuga atrata</i> L.	1
	<i>Cerambyx cerdo</i> L.	15	<i>Silpha carinata</i> Herbst	1
	<i>Clytus arietis</i> L.	2	Staphylinidae	271
	<i>Purpuricenus kaehleri</i> L.	2	<i>Astrapaeus ulmi</i> Rossi	4
	<i>Ropalopus clavipes</i> Fabr.	1	<i>Staphylinus caesareus</i>	1
Cetoniidae		15	Ceder.	
	<i>Cetonia aurata</i> L.	7	<i>S. olens</i> Müller	64
	<i>Potosia cuprea</i> Fabr.	8	<i>Stenus</i> sp.	140
Curculionidae		4	<i>Tachyporus hypnorum</i> Fabr.	7
	<i>Pachytychius hordei</i> Brullé	1	<i>Tachinus laticollis</i> Gravenhorst	37
	<i>Pissodes notatus</i> Fabr.	3	Tenebrionidae	26
Dermestidae		67	<i>Blaps gibba</i> Lap.	5
	<i>Attagenus unicolor</i> Brahm	62	<i>B. mucronata</i> Latr.	4
	<i>Dermestes laniarius</i> Illiger	2	<i>Pedinus helopoides</i> Ahrens	5
	<i>D. maculatus</i> De Geer	3	<i>Tentyria laevigata</i> Steven	3
			<i>Tenebrio molitor</i> L.	5
			<i>Lagria hirta</i> L.	4

from the forest edge increases, the dominance of some species increases. The k-dominance curves associated with the distance classes “1” (0-120 m) and “2” (120-240 m) appear rather similar in terms of shape and steepness; indeed, species abundance in these two distance classes is almost the same: 35 and 36 species, respectively (see Table 4). The k-dominance curve became steeper in distance class “3” (>240 m). In this case the number of species decreased to 31. The number of specimens increased across the gradient from the forest edge to the burnt forest area (from 1035 to 1170), while the number of families appears rather stable (ca. 11 to 12) (Table 4).

Our results suggest that species assemblage over the three distance locations is largely dominated by Nitidulidae, likely owing to the fact that these species are strongly generalist (ARBOGAST, 2009). On the other hand, the number of species and relative abundance of Carabidae tend to decrease as the distance from the forest edge increases. We found 12 species in distance classes “1” (0-120 m) and “2” (120-240

m) and 8 species in class “3” (>240 m). For example, in our study area *Carabus coriaceus* was the most abundant species of the Carabidae taxon; such abundance decreased significantly as the distance from the forest edge decreased. In fact, we found 41 specimens in distance class “1”, 29 in distance class “2”, and 14 in distance class “3” (Table 4).

This decreasing pattern of species abundance can also be found for other Carabidae species, as well as for some species of Elateridae. For instance, the abundance of species such as *Calathus cinctus* (Carabidae) and *Agryotes aterrimus* (Elateridae) dropped from 43 to 4 and from 20 to 5, respectively, over the distance classes. On the other hand, given the fact that some taxa are more generalist than others, certain species showed a contrasting pattern with gains or losses in terms of specimen number along the same spatial gradient. For example, the abundance of *Staphylinus olens* (Staphylinidae) increased from the “1” to “2” distance class (from 8 to 16 specimens) and then decreased in class “3” (4 specimens).

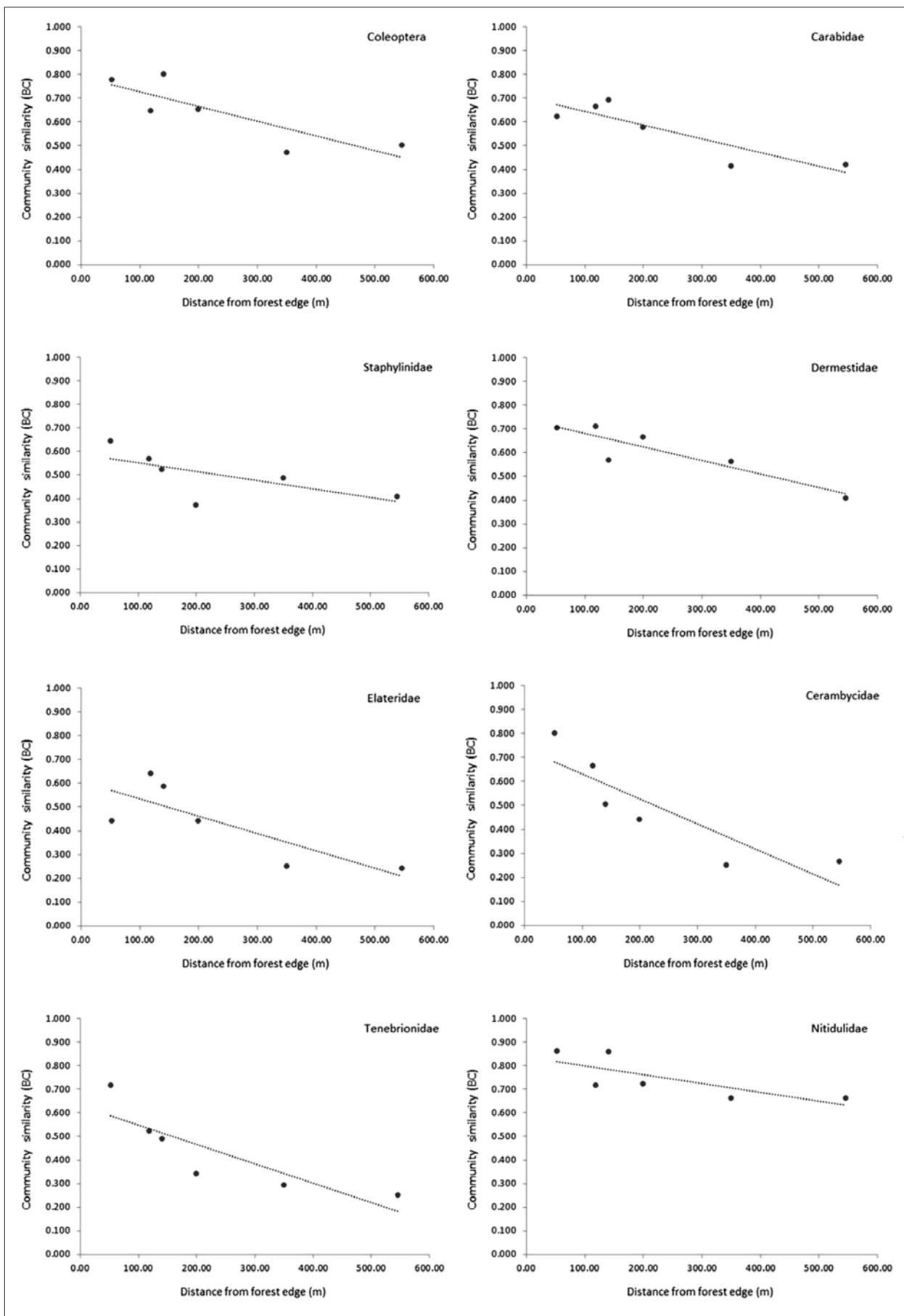


Fig. II – Linear regression models describing the relationship between beetle community similarity (Bray-Curtis index) and distance from the forest edge (D).

Table 3 – Linear regression models explaining variation of community similarity as a function of distance from the nearest forest edge (D).

Coleoptera	Regression model	r^2
Beetle community	0.790 - 0.0006 * D	0.701
Carabidae	0.704 - 0.0006 * D	0.767
Staphylinidae	0.589 - 0.0004 * D	0.453
Dermestidae	0.738 - 0.0006 * D	0.815
Elateridae	0.606 - 0.0007 * D	0.641
Cerambycidae	0.734 - 0.001 * D	0.766
Tenebrionidae	0.630 - 0.0008 * D	0.734
Nitidulidae	0.838 - 0.0004 * D	0.578

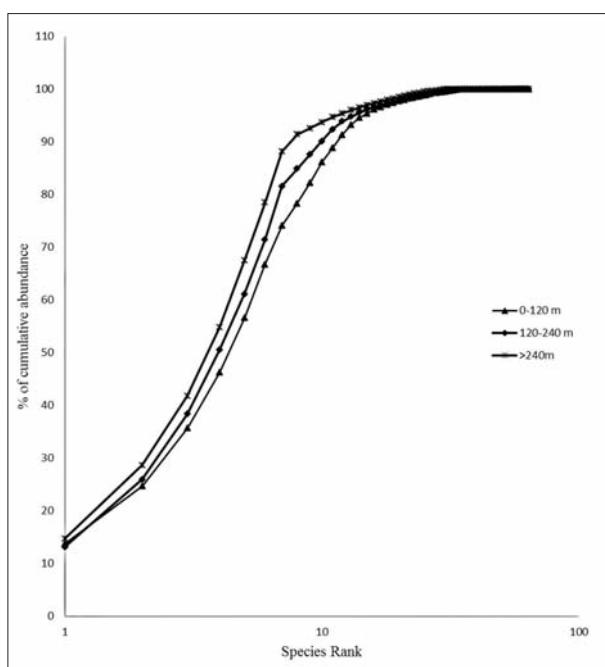


Fig. III – K-dominance curves for beetle species assemblage across three different distance classes from forest edge in a burnt area.

DISCUSSION

This study provides an integration of previous research conducted in Mediterranean forest ecosystems with the aim to evaluate the spatial and temporal response of insect community to fire (see ELIA *et al.*, 2012). Evidence has been provided on the relationship between wildfire occurrence and the mid-term response of beetle communities. Additional recent studies have provided similar evidence for other biogeographical regions (e.g., boreal and tropical) and support our decision to use epigaeic or surface-dwelling fauna (e.g., ground beetles) as indicators of wildfire disturbance (YANOVSKY and KISELEV, 1996; MCGEOCH, 1998; ORGEAS and ANDERSEN, 2001; SIMILÄ *et al.*, 2002; JOHANSSON *et al.*, 2010). Families such as Carabidae (ground beetles), for example, are ideal *taxa* because they are sensitive to environmental change and can be easily caught in pitfall traps (NIELD, 1990; BUSE and GOOD, 1993; NIEMELÄ *et al.*, 1996; SPENCE *et al.*, 1996). Pit-fall traps are frequently employed in many areas of

ecological research. This is perhaps due to the fact that other methods of trapping are more difficult and expensive to implement, particularly for surface-active invertebrates. Sometimes pitfall trapping is the only feasible method, as when population densities are low or when minimal impact methods are required for sensitive sites (SPENCE and NIEMELÄ, 1994; MELBOURNE, 1999).

Our findings indicate the existence of a negative relationship between beetle species composition and distance from the nearest forest edge in a burnt area. As the distance from the forest edge increases, the community similarity between burnt forest plots and control plots decreases, while the dominance of a few species increases. This result could be explained by the fact that the distance from the forest edge can be assumed as proxy of fire spread across the forest area and the subsequent changes in habitats. Fires affect the amount of sunlight reaching the forest floor and the diversity and spatial distribution of plant and animal communities (REED, 1997; BUDDLE *et al.*, 2000; MORETTI *et al.*, 2004). In addition, fire affects forest heterogeneity (BENGSSON, 2002) by creating a mosaic of open spaces and edge areas; the mosaic is required for some insect species to engage in breeding, oviposition, foraging and nectaring (GRUNDEL *et al.*, 1998).

By means of community similarity analyses (BC index) across the gradient from the forest interior to the forest edge and the burnt forest area, we found clear evidence that beetle communities are affected by wildfires still at 5 years after the fire event. Our results support the first hypothesis that the community composition and thus spatial turnover will vary across the gradient as a result of fire impacts. The similarity between forest edge and the interior forest (i.e., control plots), higher than that found between forest edge and area burned, suggests that the main habitat components did not differ significantly across this gradient, but provided suitable microclimatic conditions for forest species to move across the forest edge. For instance, BRIGIĆ *et al.* (2014) found that the carabid beetle assemblages near the forest edge in Croatian forests were more similar to those in the forest interior than those in the adjacent open area. Certainly, these patterns of similarity are likely to be strongly dependent on the specific taxa that we used as indicators. The life histories and dispersal mechanisms of beetles make them sensitive to changes in environmental conditions at relatively small scales (10s of meters) (LACSELLA *et al.*, 2015). Indeed, if we were to consider other *taxa* they would respond differently depending on the habitats needed for their life cycle. For example, wild bees are often positively correlated with forest edge habitat, as they require canopy closure for nesting and open areas for foraging (DIAZ-FORERO *et al.*, 2013; BOGUSCH *et al.*, 2014). On the other hand, butterflies are primarily associated with open areas instead of the forest habitat (RICKETTS, 2001).

The k-dominance analysis suggests that the species assemblage in burnt forest locations is dominated by a few (generalist) species. There is a consensus that post-fire simplification in vegetation structure may cause the establishment of a few dominant (generalist) species (HERRANDO *et al.*, 2003; SANTOS *et al.*, 2014), which in turn can affect a diverse arrangement of beetle communities. Generalist species are favored in disturbed environments, while specialists prefer slowly changing environmental conditions (GILCHRIST, 1995; SULTAN, 2001; VAN BUSKIRK, 2002; RICHMOND *et al.*, 2005). Carabidae, which are more specialist, decrease in abundance and species richness as the distance from the forest edge increases, whereas abrupt habitat changes generated by disturbances could promote

Table 4 – K-dominance and abundance values (in parentheses) of beetle species within the burnt area in the three different distance classes.

Distance class						
0-120 m		120-240 m		>240 m		
<i>Epurea aestiva</i>	13.8 (143)	<i>Epurea pallescens</i>	13.2 (141)	<i>Epurea unicolor</i>	14.7 (172)	
<i>Carpophilus dimidatus</i>	24.7 (113)	<i>Carpophilus dimidatus</i>	25.9 (137)	<i>Epurea aestiva</i>	28.6 (163)	
<i>Epurea pallescens</i>	35.7 (113)	<i>Epurea aestiva</i>	38.3 (133)	<i>Epurea pallescens</i>	41.8 (154)	
<i>Carpophilus marginellus</i>	46.3 (110)	<i>Epurea unicolor</i>	50.5 (130)	<i>Pocadius ferrugineus</i>	54.8 (152)	
<i>Urophorus humeralis</i>	56.6 (107)	<i>Urophorus humeralis</i>	61.1 (114)	<i>Carpophilus marginellus</i>	67.4 (148)	
<i>Epurea unicolor</i>	66.7 (104)	<i>Pocadius ferrugineus</i>	71.5 (111)	<i>Carpophilus dimidatus</i>	78.5 (129)	
<i>Pocadius ferrugineus</i>	74.1 (77)	<i>Carpophilus marginellus</i>	81.5 (108)	<i>Urophorus humeralis</i>	88.1 (113)	
<i>Calathus cinctus</i>	78.3 (43)	<i>Soronia grisea</i>	84.9 (36)	<i>Stenus sp.</i>	91.4 (38)	
<i>Carabus coriaceus</i>	82.2 (41)	<i>Carabus coriaceus</i>	87.6 (29)	<i>Carabus coriaceus</i>	92.6 (14)	
<i>Stenus sp.</i>	86.2 (41)	<i>Stenus sp.</i>	90.1 (27)	<i>Soronia grisea</i>	93.7 (13)	
<i>Abax ater</i>	88.8 (27)	<i>Attagenus unicolor</i>	92.4 (24)	<i>Tachinus laticollis</i>	94.6 (11)	
<i>Soronia grisea</i>	91.3 (26)	<i>Staphylinus olens</i>	93.8 (16)	<i>Attagenus unicolor</i>	95.4 (9)	
<i>Agriotes aterrimus</i>	93.2 (20)	<i>Leistus spinibarbis</i>	94.8 (10)	<i>Cyphrus caraboides</i>	96.0 (7)	
<i>Attagenus unicolor</i>	94.6 (14)	<i>Carabus violaceus</i>	95.5 (8)	<i>Pterostichus nigrita</i>	96.5 (6)	
<i>Staphylinus olens</i>	95.4 (8)	<i>Agriotes aterrimus</i>	96.1 (6)	<i>Agriotes aterrimus</i>	96.9 (5)	
<i>Tachyporus hypnorum</i>	96.1 (8)	<i>Tachinus laticollis</i>	96.5 (5)	<i>Calathus cinctus</i>	97.3 (4)	
<i>Harpalus luteicomis</i>	96.6 (5)	<i>Pterostichus melas</i>	96.9 (4)	<i>Potosia cuprea</i>	97.6 (4)	
<i>Carabus intricatus</i>	97.0 (4)	<i>Mastigus ruficornis</i>	97.2 (3)	<i>Staphylinus olens</i>	97.9 (4)	
<i>Carabus violaceus</i>	97.4 (4)	<i>Pedinus helopioides</i>	97.5 (3)	<i>Brachinus crepitans</i>	98.2 (3)	
<i>Cetonia aurata</i>	97.7 (3)	<i>Tentyria laevigata</i>	97.8 (3)	<i>Carabus glabratus</i>	98.5 (3)	
<i>Leistus spinibarbis</i>	98.0 (3)	<i>Abax ater</i>	97.9 (2)	<i>Cetonia aurata</i>	98.7 (3)	
<i>Pterostichus melas</i>	98.3 (3)	<i>Astrapaeus ulmi</i>	98.1 (2)	<i>Mastigus ruficornis</i>	99.0 (3)	
<i>Amara plebeja</i>	98.5 (2)	<i>Calathus cinctus</i>	98.3 (2)	<i>Colymbetes fuscus</i>	99.1 (2)	
<i>Cerambyx cerdo</i>	98.6 (2)	<i>Calathus fuscipes</i>	98.5 (2)	<i>Dorcus parallelepipedus</i>	99.3 (2)	
<i>Dermestes laniarius</i>	98.8 (2)	<i>Carabus intricatus</i>	98.7 (2)	<i>Elater ferrugineus</i>	99.5 (2)	
<i>Pissodes notatus</i>	99.0 (2)	<i>Harpalus luteicomis</i>	98.9 (2)	<i>Blaps mucronata</i>	99.6 (1)	
<i>Potosia cuprea</i>	99.2 (2)	<i>Pterostichus nigrita</i>	99.1 (2)	<i>Carabus violaceus</i>	99.7 (1)	
<i>Brachinus crepitans</i>	99.3 (1)	<i>Zabrus tenebrioides</i>	99.3 (2)	<i>Clytus arietis</i>	99.7 (1)	
<i>Calathus fuscipes</i>	99.4 (1)	<i>Cerambyx cerdo</i>	99.3 (1)	<i>Nebria brevicollis</i>	99.8 (1)	
<i>Cantharis livida</i>	99.5 (1)	<i>Harpalus rufipes</i>	99.4 (1)	<i>Pissodes notatus</i>	99.9 (1)	
<i>Clytus arietis</i>	99.6 (1)	<i>Melanotus villosus</i>	99.5 (1)	<i>Purpuricenus kaehleri</i>	100 (1)	
<i>Cyphrus rostratus</i>	99.7 (1)	<i>Pachytychius hordei</i>	99.6 (1)	<i>Abax ater</i>	100 (0)	
<i>Lagria hirta</i>	99.8 (1)	<i>Phosphuga atrata</i>	99.7 (1)	<i>Amara fulva</i>	100 (0)	
<i>Pterostichus nigrita</i>	99.9 (1)	<i>Potosia cuprea</i>	99.8 (1)	<i>Amara plebeja</i>	100 (0)	
<i>Silpha carinata</i>	100 (1)	<i>Purpuricenus kaehleri</i>	99.9 (1)	<i>Astrapaeus ulmi</i>	100 (0)	
<i>Amara fulva</i>	100 (0)	<i>Staphylinus caesareus</i>	100 (1)	<i>Blaps gibba</i>	100 (0)	
<i>Astrapaeus ulmi</i>	100 (0)	<i>Amara fulva</i>	100 (0)	<i>Calathus fuscipes</i>	100 (0)	
<i>Blaps gibba</i>	100 (0)	<i>Amara plebeja</i>	100 (0)	<i>Calosoma sycophanta</i>	100 (0)	
<i>Blaps mucronata</i>	100 (0)	<i>Blaps gibba</i>	100 (0)	<i>Cantharis livida</i>	100 (0)	
<i>Calosoma sycophanta</i>	100 (0)	<i>Blaps mucronata</i>	100 (0)	<i>Carabus intricatus</i>	100 (0)	
<i>Carabus glabratus</i>	100 (0)	<i>Brachinus crepitans</i>	100 (0)	<i>Cerambyx cerdo</i>	100 (0)	
<i>Colymbetes fuscus</i>	100 (0)	<i>Calosoma sycophanta</i>	100 (0)	<i>Cyphrus italicus</i>	100 (0)	
<i>Cyphrus caraboides</i>	100 (0)	<i>Cantharis livida</i>	100 (0)	<i>Cyphrus rostratus</i>	100 (0)	
<i>Cyphrus italicus</i>	100 (0)	<i>Carabus glabratus</i>	100 (0)	<i>Dermestes laniarius</i>	100 (0)	
<i>Dermestes maculatus</i>	100 (0)	<i>Cetonia aurata</i>	100 (0)	<i>Dermestes maculatus</i>	100 (0)	
<i>Dorcas parallelepipedus</i>	100 (0)	<i>Clytus arietis</i>	100 (0)	<i>Harpalus rufipes</i>	100 (0)	
<i>Elater ferrugineus</i>	100 (0)	<i>Colymbetes fuscus</i>	100 (0)	<i>Harpalus luteicomis</i>	100 (0)	
<i>Harpalus rufipes</i>	100 (0)	<i>Cyphrus caraboides</i>	100 (0)	<i>Lacon punctatus</i>	100 (0)	
<i>Lacon punctatus</i>	100 (0)	<i>Cyphrus italicus</i>	100 (0)	<i>Lagria hirta</i>	100 (0)	
<i>Leistus ferrugineus</i>	100 (0)	<i>Cyphrus rostratus</i>	100 (0)	<i>Leistus ferrugineus</i>	100 (0)	

(continued)

Continued Table 4

Distance class								
0-120 m			120-240 m			>240 m		
<i>Mastigus ruficornis</i>	100	(0)	<i>Dermestes laniarius</i>	100	(0)	<i>Leistus spinibarbis</i>	100	(0)
<i>Melanotus villosus</i>	100	(0)	<i>Dermestes maculatus</i>	100	(0)	<i>Melanotus villosus</i>	100	(0)
<i>Nebria brevicollis</i>	100	(0)	<i>Dorcus parallelepipedus</i>	100	(0)	<i>Oedemra flavipes</i>	100	(0)
<i>Oedemra flavipes</i>	100	(0)	<i>Elater ferrugineus</i>	100	(0)	<i>Pachytychius hordei</i>	100	(0)
<i>Pachytychius hordei</i>	100	(0)	<i>Lacon punctatus</i>	100	(0)	<i>Pedinus helopiooides</i>	100	(0)
<i>Pedinus helopiooides</i>	100	(0)	<i>Lagria hirta</i>	100	(0)	<i>Phosphuga atrata</i>	100	(0)
<i>Phosphuga atrata</i>	100	(0)	<i>Leistus ferrugineus</i>	100	(0)	<i>Pterostichus melas</i>	100	(0)
<i>Purpuricenus kaehtleri</i>	100	(0)	<i>Nebria brevicollis</i>	100	(0)	<i>Ropalopus clavipes</i>	100	(0)
<i>Ropalopus clavipes</i>	100	(0)	<i>Oedemra flavipes</i>	100	(0)	<i>Silpha carinata</i>	100	(0)
<i>Staphylinus caesareus</i>	100	(0)	<i>Pissodes notatus</i>	100	(0)	<i>Staphylinus caesareus</i>	100	(0)
<i>Tachinus laticollis</i>	100	(0)	<i>Ropalopus clavipes</i>	100	(0)	<i>Tachyporus hypnorum</i>	100	(0)
<i>Tentyria laevigata</i>	100	(0)	<i>Silpha carinata</i>	100	(0)	<i>Tentyria laevigata</i>	100	(0)
<i>Tenebrio molitor</i>	100	(0)	<i>Tachyporus hypnorum</i>	100	(0)	<i>Tenebrio molitor</i>	100	(0)
<i>Zabrus tenebrioides</i>	100	(0)	<i>Tenebrio molitor</i>	100	(0)	<i>Zabrus tenebrioides</i>	100	(0)
TOTAL		1035	TOTAL			1072	TOTAL	
								1170

more generalist species (FUTUYMA and MORENO, 1988). Open spaces, potentially created by fires, support a greater number of generalist species as well as provide a suitable refuge and resources for species associated with such habitats. For example, Nitidulidae may be found in numerous diverse habitats feeding on flowers, fruits, sap, fungi, decaying and fermenting plant tissues or dead animal tissue (ARBOGAST, 2009). This finding is in line with other studies of plants (SPARKS *et al.*, 1996; PETERKEN and FRANCIS, 1999; MULLEN *et al.*, 2003) and other groups of arthropods (OXBROUGH *et al.*, 2006; LARRIVÉE *et al.*, 2008; BOSSART and OPUNI-FRIMPONG, 2009; LACASELLA *et al.*, 2015).

The ability of a beetle community to recover from environmental change caused by wildfire (ISAAC *et al.*, 2009) has been linked to: (1) the diversity of responses at the ecosystem level (ELMQVIST *et al.*, 2003); (2) the diversity and heterogeneity at the community level (BENGTSSON, 2002); and (3) particular functional traits (e.g., generalist species) at the species level (NORDEN *et al.*, 2009).

Taken together, our analyses of the spatial patterns of beetle communities highlight an important point – they provide evidence on how varying *taxa* respond differently to fire as a consequence of their habitat requirements, modes of reproduction and dispersal mechanisms, all of which influence the way a given species, or family, responds to the spatial structure of a landscape. The positive impact of habitat heterogeneity on biodiversity is a well-known and predictable rule in ecology (TEWS *et al.*, 2004). To preserve biodiversity, landscape management might provide mosaics of open and forest areas (PONS *et al.*, 2003) maximizing the expansion of both open- and forest-specialist species and enhancing species turnover after new disturbances in burnt areas (BROTONS *et al.*, 2005).

Finally, some specific limitations related to this study must be taken into consideration. We limited our field data collection to a single forest area (of 600 ha) because over the years this area has proved to be relatively unaffected by intensive grazing and hunting activities. Other forest areas have also been considered across the region, but field

experiments could not be finalized due to significant damage to pit-falls traps. However, we plan to locate new field experiments in other forest areas affected by different wildfires intensity, because not all wildfires are the same and within a wildfire there is much variability in severity that could explain response. Furthermore, future research is needed to understand the effects of additional co-factors on beetle community composition, such as forest loss and fragmentation.

CONCLUSIONS

Adding to a growing list of researches on the importance wildfire impacts in preserving biodiversity in forest landscapes, this study suggested that five years after a fire, there is still an effect of fire on beetle community composition.

The majority of beetle communities surveyed reflect significant differences between burnt and unburnt sites as well as several differences in relation to the distance from the forest edge in a burnt area. The differences were probably driven by vegetation structure changes across the gradient from the burnt area to the forest interior, especially for those *taxa* with many specialist species in feeding and oviposition habitats.

Given the short term of our study, it would be, therefore, unwise to consider its conclusions as definitive, and further studies could investigate how beetle communities respond to forest recover. This will enable the prediction of how changes in habitats conditions within the gradient and as well as their managements, may influence assemblages of beetles.

Estimating the effects of wildfires on beetle communities is a prerequisite to establishing strategies and policies for counteracting biodiversity loss at the global, regional and national levels (CBD SECRETARIAT, 2001). The relationship between beetle assemblage similarity, the distance from forest edge, and the increased dominance at different distance classes in disturbed areas can help to achieve post-fire management objectives. The main goal of forest managers, responsible for wildlife landscapes, is to preserve

natural biodiversity by focusing on the conservation of endemic species. To this end, the maintenance of landscape heterogeneity of fire mosaics could be an appropriate management practice (BRADSTOCK *et al.*, 2005; DRISCOLL *et al.*, 2010; ELIA *et al.*, 2012). In areas historically affected by wildfires, long-term studies should be conducted to: i) evaluate the effectiveness of post-fire management in terms of benefits for biodiversity; ii) recognize ecological mechanisms related to insect-fire interactions; and iii) identify reliable indicators (e.g., species) of post-fire processes.

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