



Conservation of the Mediterranean coastal pine woodlands: How can management support biodiversity?



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ABSTRACT

Forest management decisions may have unintentional effects on what they were originally not designed for, including effects on woodland species and communities. In protected areas of coastal dune woodlands, some sites are fenced as a part of forestry management. In this study, we hypothesised that different states of disturbance (disturbed vs non-disturbed) created by fencing generate unintentional heterogeneity in species composition (and possibly richness) in plant communities and soil biota. We surveyed vascular plants, oribatid mites and soil properties in fenced and nearby non-fenced places in protected coastal pine woodlands in Italy. The fenced areas were undisturbed for at least 30 years, whereas the non-fenced areas were subjected to thinning and trampling. Effects of fencing on community composition and soil properties were assessed by (distance based) redundancy analysis. Congruence between plant and mite community composition in response to fencing was tested using a series of (partial) Mantel tests. Finally, linear mixed-effects models were used to study species richness. Both plant and mite community composition showed a significant congruent response to fencing. Species richness of plants decreased due to fencing, while that of mites was unaffected. We conclude that the fencing of small areas creates biotic heterogeneity and increases beta diversity in the Mediterranean coastal woodlands. Therefore, we support the use of fencing as a relatively cheap and effective method of conservation management for maintaining the biodiversity of both above- and belowground communities in the coastal pine woodlands of the Mediterranean area.

1. Introduction

Dune woodland, a forest type scattered along the Mediterranean Sea coast, is an important habitat for nature conservation. The dominant species of these dune woodlands are often pines, including *Pinus brutia*, *P. halepensis*, *P. pinaster* and *P. pinea*. They largely belong to the priority habitat 2270 “Wooded dunes with *Pinus pinea* and/or *P. pinaster*” of Annex I of the EU Habitats Directive (92/43/EEC). Most of these woodlands are the result of large reforestation that took place in the second half of the twentieth century in Italy and other Mediterranean countries, mainly for pine nut, resin and timber production and as a shelterbelt for crops (Martínez and Montero, 2004). In spite of this, they are protected since they resemble the structure and plant composition of the original woodlands, hosting specialist species of the coastal

environments (Leone and Lovreglio, 2004; Bonari et al., 2017a; Bonari et al., 2018).

Mediterranean woodlands have been under long-lasting human impact (Scarascia-Mugnozza et al., 2000). Generally, the following characteristics can be considered signs of a well-maintained pine woodland for conservation purposes: natural composition of the canopy (including high cover of *Pinus* individuals in a reproductive age), structural complexity with heterogeneous age structure or well-developed vegetation layers (including a shrub layer), regeneration of canopy trees and shrubs, scarce presence of non-native species, presence of old trees, a variety of dead wood (lying or standing), and low levels of soil compactness and human trampling (Dimopoulos, 2016). Unfortunately, coastal pine woodlands are highly disturbed at many sites due to urban sprawl, coastal erosion and increasing tourism (Malavasi

Abbreviations: EC, electrical conductivity; F, fenced (plots); NF, non-fenced (plots); SOM, soil organic matter

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Fig. 1. Sampled sites (circles) and the distribution of stone pine (*Pinus pinea*) dominated dune woodlands (hatched) in peninsular Italy. Distribution data are extracted from the *CircumMed Pine Forest Database* (Bonari et al., 2019).

et al., 2016). One way of maintaining their biological diversity is creating heterogeneity in the understorey, which might support heterogeneous plant and animal communities. Silvicultural treatments have often been proposed for enhancing the diversity of pine woodlands, e.g. thinning in high-density or homogeneous stands (Ruiz-Benito et al., 2012). An alternative way of creating heterogeneity is the fencing of smaller areas, which creates differences in disturbance between fenced and non-fenced places. These small areas are equivalent to forest parcels which were used by foresters to physically separate areas with different forestry treatments. They are currently adopted in many of the protected areas hosting this forest type, having unknown implications for conservation although originally they were not designed as a conservation measure. Since the effects of fencing of pine dune woodlands on biodiversity are poorly known, scientific studies are needed to understand the impact of this practice (Coll et al., 2018).

It is relatively easy to observe how plants react to fencing management, but there are other components of the ecosystem that may respond to a reduction of disturbance by fencing. For example, the

effects of fencing on soil mesofauna are poorly known. Still, conservation management should be optimized to take into account both flora and fauna (Bonari et al., 2017b; Sierzeza and Eichholz, 2019). Therefore, we studied the effect of fencing on plants and oribatids. Oribatida are an order of mites (also called beetle mites or moss mites) that dwell mostly in soil layers rich in organic material. Being predominantly saprophagous and one of the dominant groups of soil mesofauna, they are among the most important decomposers, contributing significantly to nutrient cycling. They also serve as soil quality indicators (e.g. Salmon, 2018).

In our previous study, we tested the hypothesis of concordance in species composition between vascular plants and oribatid mites under the same disturbance regime (Bonari et al., 2017c). We found that vegetation cover is an important driver of these communities, especially because it influences the insolation of the ground, thus having an effect on soil temperature and moisture, which affect oribatids. In the present study, we focus on protected Mediterranean coastal pine woodlands with mixed management including fenced areas with no silvicultural

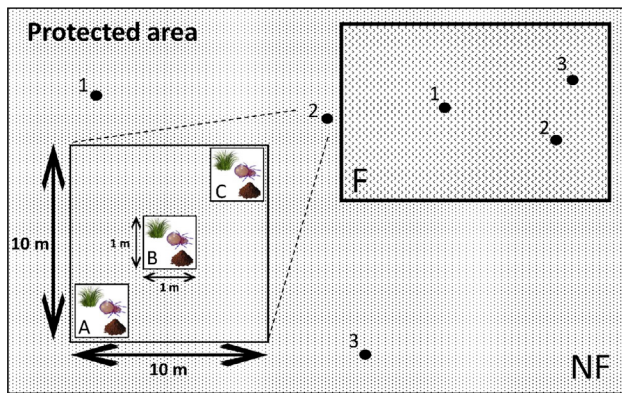


Fig. 2. A scheme of the stratified sampling design adopted in this study showing random points (1, 2, 3) in fenced (F) and non-fenced (NF) parts of a protected area. The magnified inset shows the position of plots (A, B, C) within an area of 100 m². Replicates of the cubic soil blocks used for sampling mites and soil within the plots of 1 m² are not shown. The positions of icons of plants, mites and soil within the plots A, B and C have no meaning.

treatment since decades and non-fenced areas subject to various disturbances. Our hypotheses are that (1) fencing creates heterogeneity in species composition and richness in both plant communities and soil biota, and (2) observed heterogeneity in a plant community can indicate a parallel heterogeneity in a community of soil invertebrates.

2. Materials & methods

2.1. Study area and habitat

The study area comprises the Tyrrhenian and Adriatic coastlines of the Italian Peninsula where stone pine (*Pinus pinea*) woodlands occur. These woodlands cover approximately 46,290 ha in Italy (Tabacchi et al., 2007). *P. pinea*, the dominant pine species forming these woodlands, currently occurs in most of the countries bordering the Mediterranean Sea. However, its natural range is unknown because it has been planted throughout the Mediterranean Basin since antiquity (Martínez and Montero, 2004). Typically, the structure of these woodlands comprises an open-canopy upper tree layer formed by *P. pinea* and a sparse lower tree layer with scattered individuals of *Quercus* spp. or other thermophilous tree species. The character of the shrub layer is heavily dependent on disturbance (and management), but it includes species typical of holm oak (*Quercus ilex*) forests and of the Mediterranean macchia. The herb layer is heterogeneous and generally species-poor, although richer patches with light-demanding species are present. As in the shrub layer, disturbance is the key driver of the occurrence of individual herb species and their abundances. From the successional point of view, the most advanced vegetation is represented by pure holm oak forests with scarce shrubs and few understorey species, including shade-tolerant plants of the evergreen oak forest, typical of the natural vegetation of inner dunes and dune slacks (Acosta et al., 2003; Bonari et al., 2017a). An intermediate successional stage is characterized by sclerophyllous shrub species of the Mediterranean macchia (e.g. *Cistus* spp., *Erica* spp., *Phillyrea* spp., *Rhamnus* spp.). Many stands of these woodlands are considerably disturbed due to thinning, clearcutting or trampling: they often lack any shrub species and are characterized by generalist herbs. These woodlands grow in the inland zone of the coastal dune systems, i.e. on stabilized dunes, mainly on moderately acidic soils, but pH can vary even within small areas (Bonari et al., 2017c). Climate varies with latitude and encompasses both the Mediterranean and temperate macro-bioclimates (Pesaresi et al., 2014). From the syntaxonomic point of view, these woodlands are currently classified in the *Pinetalia halepensis* order of the *Quercetea ilicis* class (Mucina et al., 2016).

2.2. Sampling design

We collected original data during springs of 2014 and 2015 to analyse the effects of fencing on vascular plants and oribatid mites, which were surveyed simultaneously at the same sites. Fig. 1 shows the distribution of our sites and of coastal stone pine woodlands in peninsular Italy based on the data extracted from the *CircumMed Pine Forest Database* (GIVD: EU-00-26; Bonari et al., 2019). We selected seven homogeneous sites (Appendix A, Table S1) with stone pine woodlands along the coastline of the Italian Peninsula fulfilling the following criteria: (i) being a part of a protected area (National Park, Nature Reserve, Regional Park or Site of Community Importance); (ii) being attributable to the priority habitat 2270 “Wooded dunes with *Pinus pinea* and/or *Pinus pinaster*” of Annex I of the EU 92/43/EEC Directive (Gigante et al., 2016); (iii) growing on sandy soils; (iv) containing fenced areas (F), in which forestry practices, e.g. thinning, were not performed for at least 30–50 years and where high fences prevent the access of people and ungulates such as deer and wild boar, and non-fenced areas (NF) subjected to various disturbances such as thinning and trampling by humans and large animals (see examples in Appendix A, Fig. S1).

Our stratified random sampling design is shown in Fig. 2. At each site, we randomly chose six points (three in F and three in NF areas), and delimited a squared area of 100 m² around these points. Within this area, we placed three plots (1 m × 1 m; A, B, C) in the opposite corners and in its centre (corresponding to the above-mentioned random point). Firstly, all the vascular plants (nomenclature after Conti et al., 2005) in each 1 m² plot were recorded with an estimation of their percentage cover, i.e. the vertical ground projection of each species. Secondly, mites were collected as follows: after removal of the undecomposed organic matter including branches, twigs and pine needles, we sampled, in the opposite corners of each plot and in its centre, three replicates as cubic soil blocks to a depth of 10 cm by means of a corer (1000 cm³; not shown in Fig. 2). Thirdly, the same procedure as used for mites was repeated for soil samples (1000 cm³; not shown in Fig. 2). They represent the upper-most part of the soil profile (topsoil) including both the organic layer, when present, and the mineral soil. In summary, our sampling design included 3 plots (1, 2, 3) × 3 replicates (A, B, C) × 2 treatments (F, NF) × 7 sites, resulting in a total of 126 plots.

Soil fauna was extracted in the lab by means of Berlese-Tullgren funnels. Extractions were conducted at ambient temperature for 15 days, and the obtained soil animals were preserved in 75% ethanol for subsequent identification. Upon separation from other taxonomic groups, oribatid mites were identified to species using several keys (Pérez-Iñigo, 1993, 1997; Subías and Arillo, 2001; Subías, 2004; Weigmann, 2006). Numbers of individuals for each species in each sample were counted. Juveniles were not included in this study.

2.3. Soil analysis

In the laboratory, soil samples were dried at +30 °C, manually sieved with a metal-free 2 mm sieve and then homogenized by quartering and mechanical pulverization. Soil pH_(H2O) and electrical conductivity (EC) were measured on the < 2 mm fraction of soil samples in a 1:2.5 (weight/volume) soil:water mixture applying the methods III.1 and IV.1 defined in the Italian Ministerial Decree “Official Methods for Soil Chemical Analysis” (September 13th, 1999). The carbonate content, expressed as calcium carbonate percentage (% CaCO₃), was determined on the pulverized soil samples by calcimetry using De Astis calcimeter. Loss on ignition was used to determine the content of organic matter in soil samples (SOM). An aliquot of about 0.5 g of pulverized soil was oven-dried at +105 °C for 16 h, cooled in a desiccator and weighed; then it was combusted at +375 °C for 16 h in a muffle furnace. After combustion, the sample was cooled in a desiccator and weighed again. Soil organic matter content (% SOM) was calculated as [(oven-dry soil weight – soil weight after combustion)/oven-dry soil

weight] $\times 100$. To determine the particle size distribution, about 200 g of soil sample was placed in a sieve stack consisting of sieves with mesh sizes of 2, 1, 0.4, 0.25, 0.1 and 0.063 mm. After stirring for 10 min, the fraction remaining on each sieve was collected and weighed.

2.4. Data analysis

We considered individual sites (with F and NF plots) as independent observations in data analysis. We computed a matrix of Bray-Curtis dissimilarities in community composition between all pairs of sites. First, we computed principal coordinate analysis (PCoA) based on this matrix to display the changes induced by fencing within the context of the major community composition gradients. Then, we used partial redundancy analysis (RDA) to test the effects of fencing on soil properties and partial distance-based redundancy analyses (db-RDA) to test the effect of fencing on plant and mite community composition. Site identity was used as a covariate in these analyses. The significance of the effect of fencing was assessed by permutation of F and NF plots within sites. Congruence between plant and mite community composition in response to fencing was tested by a series of (partial) Mantel tests. Bray-Curtis dissimilarity matrices of plant and mite communities and a Euclidean distance matrix based on the binary fencing variable (fencing matrix) were used as an input for the Mantel tests. We quantified Pearson correlations between fencing and community composition as well as partial correlations with the community dissimilarity matrix of the other organism group used as a covariate. The resulting correlations were used to partition the effect of fencing into three components: congruent response of the plant and mite community compositions and an independent response of each of these two communities. In addition, we tested overall congruence between plant and mite communities by comparing the two community dissimilarity matrices (both with and without the fencing matrix as a covariate) using the Mantel test. The significance of all effects in multivariate analyses was assessed by permutation tests, with blocks defined by site identity and plot permutation within blocks. All permutation tests used 127 permutations, which was the maximal number of possible combinations with the data available; therefore the minimal achievable type I error probability was $P = 0.008$. The effects of fencing on plant and oribatid mite diversity (measured as species richness and Shannon index) and total oribatid mite abundance were tested by linear mixed-effect models with the respective diversity parameter as a response, site identity as random effect and fencing as a fixed-effect predictor. Vascular-plant cover and soil data were averaged, and mite data were pooled within each plot. The pooled data on mite abundances were square-root transformed to prevent the overwhelming effect of a few dominant species on the analysis. Data for species richness and total oribatid abundance were log-transformed. The analyses were conducted in the R packages vegan (Oksanen et al., 2018) and nlme (Pinheiro et al., 2017).

3. Results

Soils sampled in the F and NF plots were non-saline sandy soils ($EC = 107\text{--}717 \mu S/cm$; Table 1) with a prevalence of the medium and fine sand fractions. Their pH values ranged from 4.5 to 8.2, though most soil samples were near-neutral to moderately alkaline (pH 6.6–8.2). Non-calcareous to slightly calcareous soils with low carbonate content (1–5% $CaCO_3$) had pH values < 7 , while moderately to very calcareous soils (6–24% $CaCO_3$) showed higher pH values up to 8.2. Soil organic matter (SOM) content varied considerably from 3 to 87%.

No statistically significant difference was found in soil properties between F and NF sites (Table 1).

In the seven surveyed sites, 975 unique species-site records were obtained for 191 vascular plant species (Appendix A, Table S2). The average number of species per site was 50 ± 11 (mean \pm SD), the minimum was 38, and the maximum was 71. More species-site records were from NF plots (567 occurrences) than from F plots (408

Table 1

Descriptive statistics of properties of soil samples including mean and standard deviation (SD), minimum (min) and maximum (max) values for fenced and non-fenced sites.

Soil properties	F (Fenced)				NF (Non-fenced)			
	Mean	SD	Min	Max	Mean	SD	Min	Max
pH	6.7	1.2	4.5	8.2	7.0	1.2	4.7	8.1
Electrical conductivity ($\mu S/cm$)	337	143	107	685	358	156	127	717
Carbonate content (% $CaCO_3$)	5.2	3.0	1.0	10.9	6.9	6.3	1.0	23.8
Soil organic matter (%)	15.9	21.8	3.0	86.5	9.0	8.6	3.5	44.4
Particle size distribution (%)								
2–1 mm (very coarse sand)	1.1	1.2	0.1	4.9	1.1	0.9	0.1	3.5
1–0.4 mm (coarse sand)	18.6	18.2	1.4	53.4	18.0	16.6	2.3	60.5
0.4–0.25 mm (medium sand)	39.9	15.7	12.4	66.7	43.4	15.4	17.0	75.3
0.25–0.1 mm (fine sand)	35.8	14.3	16.2	66.8	33.7	18.8	7.9	69.7
0.1–0.063 mm (very fine sand)	3.0	2.7	0.6	9.3	2.2	1.1	0.5	4.3
< 0.063 mm (silt + clay)	1.6	0.7	0.4	2.9	1.7	0.9	0.4	3.7

occurrences). The dominant *Pinus pinea* was present at all sites and in all treatments. Otherwise, the most frequent species were *Smilax aspera* and *Quercus ilex* (all sites, but not in all treatments), followed by *Asparagus acutifolius*, *Carex flacca*, *Dactylis glomerata*, *Hedera helix*, *Myrtus communis*, and *Rubia peregrina*.

We recorded 16,502 individuals of 153 oribatid species across the seven surveyed sites (Appendix A, Table S3). The average number of individuals per site \times treatment ($N = 14$) was 1179 ± 899 (mean \pm SD), the minimum was 157, and the maximum was 3056. The corresponding densities per square metre would be $13,100 \pm 9989 \text{ ind./m}^2$, 1744 ind./m^2 and $33,955 \text{ ind./m}^2$, respectively. The average number of species per site was 41 ± 12 (mean \pm SD), the minimum was 23, and the maximum was 59. Overall, the total abundance was significantly higher in F plots than in NF plots ($F_{1,6} = 14.6$, $P = 0.009$). The most abundant species were the cosmopolitan species *Microppia minus*, *Oppeilla nova* and *Sphaerorchthonius splendidus* (2054, 1989 and 1255 individuals, respectively).

Principal coordinate analyses of vascular plant and oribatid species compositions showed a difference between communities in F and NF plots (Fig. 3; permutation tests of significance of the constrained axis: $F_{1,6} = 2.80$, $P = 0.016$ for plants; $F_{1,6} = 3.28$, $P = 0.016$ for oribatids; 127 permutations), which was also reflected by significant effects of fencing on community composition. In plant communities, woody species (e.g. *Erica arborea*, *Phillyrea angustifolia*, *Quercus ilex*) typical of mid- or late-successional stages and forest species (e.g. *Brachypodium sylvaticum*, *Hedera helix*) were associated with F plots. In contrast, more generalist light-demanding species (e.g. *Piptatherum miliaceum*, *Rosmarinus officinalis*, *Rubus ulmifolius*, *Sonchus bulbosus*, and *Stipa bromoides*) typical of more disturbed areas or of the early-successional stages of dune sclerophyllous scrub were associated with NF plots. The ordination of oribatids, obtained using the partial db-RDA, showed species common in forest soils or those of Mediterranean thermophilous woodlands to be confined to F sites (e.g. *Ceratozetes laticuspidatus*, *Chamobates pusillus*, *Liochthonius brevis*, *Machuella draconis*, *Odontoccephus eleongatus*, and *Suctobelbella subtrigona*). NF sites were characterized by species with a broad ecological range, often present at open or highly disturbed sites, such as *Schelorbates initialis* and *Tectocephus sarekensis*.

Variation partitioning showed a similar effect of fencing on plant and mite communities (Fig. 4). The partial effect of fencing accounted for 2.0% ($P = 0.023$) in plant communities and 1.7% ($P = 0.016$) for

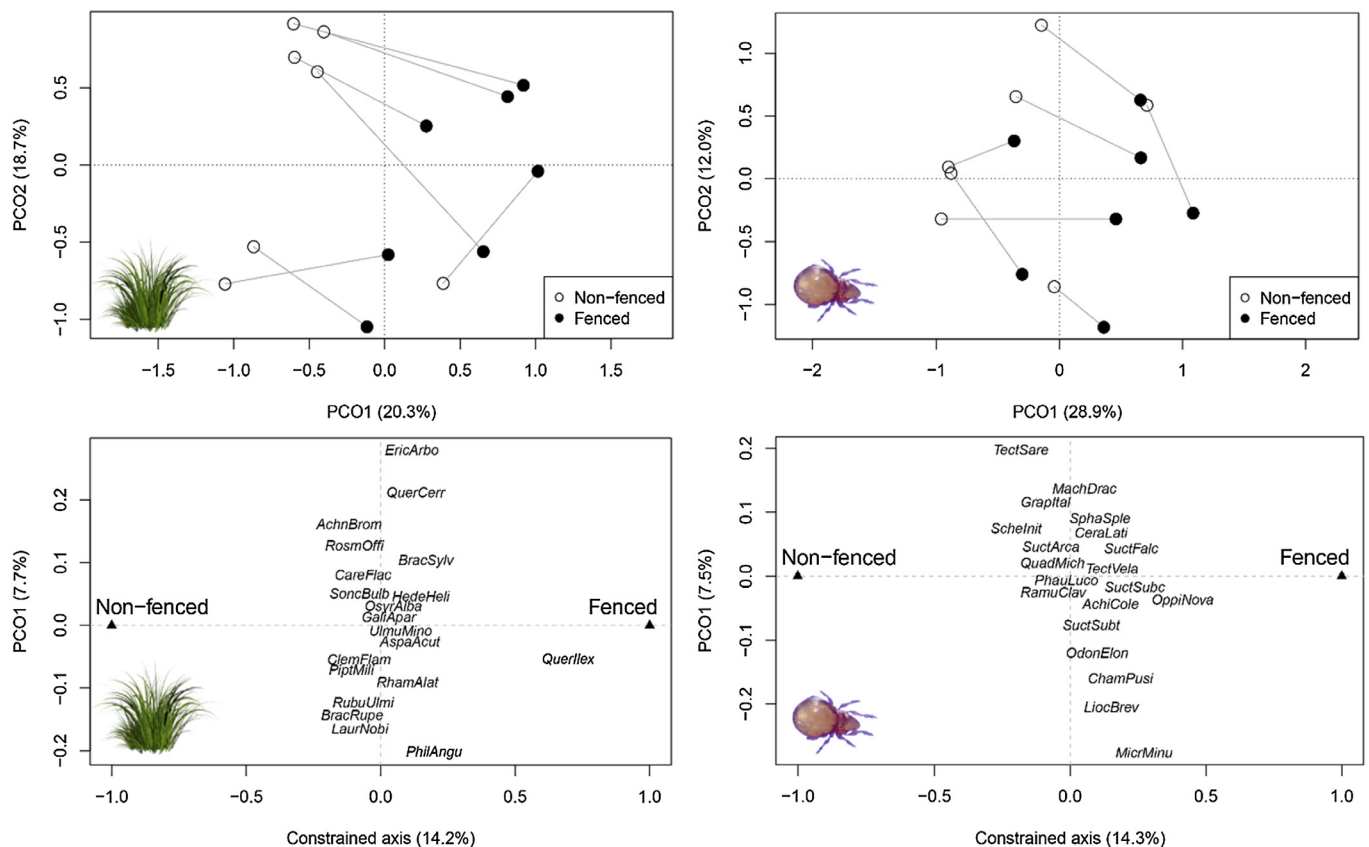


Fig. 3. PCoA and partial db-RDA diagrams of plant and mite communities. Plots within individual sites are connected by lines on the PCoA plots. Species scores are displayed on db-RDA plots. The full species names of plants and mites can be found in Appendix A (Tables S2 and S3, respectively).

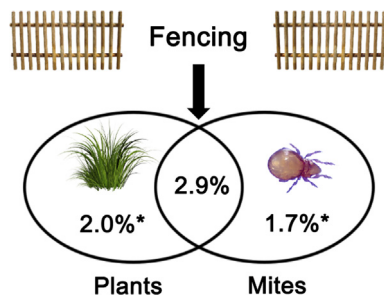


Fig. 4. Variation partitioning of the effect of fencing on species composition of plant and mite communities. Partial effects on the two community types as well as the shared effect between them are displayed. Percentages are derived from R^2 based on partial Mantel tests (127 permutations). Note that the effect of fencing on the shared component of community composition was not tested.

oribatid communities. The shared effect accounted for 2.9% (not tested). There was a significant congruence between plant and oribatid community composition (Mantel test: $R^2 = 0.21$; $P = 0.008$; Mantel test when fencing was set as a covariable: $R^2 = 0.021$; $P = 0.008$; 127 permutations).

Fencing had a significant negative effect on plant species richness (mixed-effect model, $F_{1,6} = 16.9$, $P = 0.007$) and no effect on oribatid species richness ($F_{1,6} = 2.52$, $P = 0.16$). We did not find any significant effect of fencing on the Shannon diversity index of plant ($F_{1,6} = 0.14$, $P = 0.72$) and oribatid communities ($F_{1,6} = 0.06$, $P = 0.81$).

4. Discussion

Our study demonstrates that fencing affects biota in the Mediterranean coastal pine woodlands. Both vascular plant and

oribatid mite species composition responded to fencing, while no difference between fenced and non-fenced plots was found for the measured soil properties. Species richness of plants was lower in fenced plots, while species richness of oribatids did not differ between fenced and non-fenced plots. We will separately discuss the effect of fencing on plant and oribatid species composition and soil properties, and species richness. Then, we will try to synthesise our findings and provide conservation management recommendations.

4.1. Responses of species composition and soil properties to fencing

Direct comparisons of different taxa sampled in the same sets of plots are often used to obtain deeper insight into the effects of various factors and processes on communities (e.g. Lososová et al., 2011; Chytrý et al., 2012; Sitzia et al., 2017; Burrascano et al., 2018; Barbato et al., 2019). In this study, we found a common response to fencing at the community level for vascular plants and oribatids in Mediterranean coastal pine woodlands. This supports the results of our previous local-scale study in which community patterns of plants and oribatids in this habitat showed a high degree of congruence (Bonari et al., 2017c). Moreover, we confirmed the results of a former study that mite community composition changes in response to the management of plant communities (Gormsen et al., 2006). We also found that soil properties were not affected by fencing, implying that the differences in community composition between fenced and non-fenced treatments were not caused by the measured soil factors (see also Angiolini et al., 2018 for the response of dune species to soil factors). Nevertheless, we observed in the field that litter cover was much higher in fenced plots, possibly providing more shelter and food resources to oribatids (Wehner et al., 2016). Soil organic matter did not significantly differ between fenced and non-fenced plots. However, the patchy litter distribution within the non-fenced plots might have contributed to this result.

Plant composition changed due to fencing in coastal pine woodlands. This result suggests that vegetation succession is ongoing within fenced areas on old inland dunes unaffected by silvicultural treatments or trampling by humans. On stabilized dunes, the climax is represented by holm oak forests (Carranza et al., 2008). Nevertheless, when pines are present, the light-demanding *Quercus ilex* rarely becomes dominant, usually occurring in the lower tree layer only. It can become dominant when pines die or are not present at all, but such situations are relatively rare along the coasts of the Italian Peninsula. The most widespread stage of the succession in fenced plots is characterized by the presence of tall shrubs. This physiognomy suggests that coastal pine woodlands dominated by *Pinus pinea* are interconnected with Mediterranean macchia, representing a transitional successional stage. This also implies that the fenced plots, with generally more closed canopy than the woodlands in the non-fenced areas, tend to develop towards the climax, with a decrease of annual species and rapid increase of perennial species, including woody ones (Bonet, 2004).

Non-fenced areas were subjected to silvicultural treatments which interfere with vegetation dynamics by preventing the establishment of macchia and then of the holm oak forest. However, these treatments also favour the formation of grassland patches by creating empty space within pine woodland understorey. Moreover, non-fenced plots host typical species of coastal habitats, although subjected to more frequent and more intensive disturbance, e.g. cutting of understorey shrubs, trampling, harvesting of pine cones and camping (Leone and Lovreglio, 2004). Such disturbances also influence the plant community composition by supporting ruderal species. However, a large part of the disturbance is to be attributed to the massive touristic use, with off-trail trampling by vacationers being relatively common (see also Ciccarelli, 2014 for the effects on foredune communities). It is remarkable that disturbance in this habitat type, when too intense, decreases the presence (and sometimes the survival) of plant communities, mostly those of shrubs and herbs. Under such conditions, an impoverished aspect of the pine woodland with no shrubs and abundance of generalist herbs prevails.

Diverging species composition of oribatids is expected under diverging environmental conditions (Maaß et al., 2014). Also in this study, oribatid species composition differed substantially between the fenced and non-fenced plots. Typical forest species characterized the fenced plots, whereas opportunistic and disturbance-tolerant species dominated in the non-fenced plots. This can be explained by two factors, both related to the heavy disturbance levels in the non-fenced plots due to forestry management (e.g. thinning), trampling by human visitors, and grazing and trampling by game. One factor would be the more open canopy in the non-fenced plots (observed), leading to higher insolation levels of the ground and more pronounced temperature and moisture dynamics in the topsoil (not measured; Iovieno et al., 2010; Bonari et al., 2017c). The other factor would be a thinner and more patchy organic layer in the non-fenced plots, which was observed when visiting the sites, though larger litter accumulation in the fenced plots only showed as a trend in our analysis of soil organic matter content (see, e.g. Roig et al., 2005). Soil organic matter is generally measured in mineral soil, excluding the undecomposed litter, which was also the case in our sampling. It has to be noted that this removal of litter might have led to some bias in the results regarding oribatids, some of which were probably also present in this layer, though it tended to be very dry most of the time. The thickness of the organic layer, providing food and higher moisture levels, also indicated the higher mite abundances in the fenced plots (N'Dri et al., 2017). Due to a high mass of litter, coniferous forests support a higher abundance of mites than beech forests (Erdmann et al., 2012) and also than fallows, arable fields, grazed pastures and riverine forests (Maraun and Scheu, 2000).

As reported, for instance, by Salmon (2018), forest dynamics have a strong effect on soil invertebrates. In our study, forest dynamics in the non-fenced plots were substantially altered by forest management. This was mainly represented by thinning operations, opening up the canopy

and disturbing the topsoil. Effects of logging on oribatids have been detected even after 15 years in montane spruce forests of the Šumava Mountains in the Czech Republic (Kokořová and Starý, 2017), but the disturbance caused by thinning at our sites was much less severe. In general, oribatid communities have the ability to regenerate after disturbance (Gergőcs and Hufnagel, 2017), which could have partially masked the magnitude of the effect of the thinning operations. Despite this, the differences between oribatids of fenced and non-fenced plots were distinct. Regrettably, a detailed chronological record of the forestry-related operations is lacking for our plots. The fenced plots had been fenced for at least 30 years (many for much longer), whereas all the types of disturbance mentioned above continued in the non-fenced plots. Those caused by human visitors, and probably also game, increased after parts of the forests became inaccessible due to fencing, and our data do not allow us to discriminate between the effects caused by the individual types of disturbance and the effect of the resulting greater canopy openness.

4.2. Species richness in fenced and non-fenced areas

Our study confirms that fencing negatively influences species richness of plants, indicating that richness decreases with successional age and under lower disturbance level, in agreement with the Intermediate Disturbance Hypothesis (Huston, 1979). Our result is also consistent with what was found in other types of Mediterranean forests (Amici et al., 2013) or other habitat types (Chytrý et al., 2001), where disturbance resulted in increased species richness (but see also other studies where it decreased or did not change; Battles et al., 2001; Grifffis et al., 2001, respectively). Our finding was expected because late-successional stages of dune woodlands lead to the holm oak forest, which is a dark sclerophyllous forest with low species richness in the herb layer. In contrast, the more heterogeneous environment of non-fenced plots leads to higher species richness (see also Section 4.1). Nevertheless, we demonstrate that species richness of coastal pine woodlands is closely connected to disturbance and post-disturbance vegetation succession. Yet, a shallow water table along with low values of salinity could have contributed to the overall species richness found in our plots (Antonellini and Mollema, 2010). Our findings indicate that the effect of trampling previously described for foredune plant diversity is also valid in stabilized dunes (Santoro et al., 2012).

Unlike for plants, we did not find a difference in species richness of oribatids between fenced and non-fenced plots. A fine-scale heterogeneity in soils is known to increase species richness of intermediate-sized soil fauna (Nielsen et al., 2010). In our study, the sampling design tried to minimize the soil heterogeneity by selecting sites as homogeneous as possible, thus focusing on the net effect of management on oribatid diversity. Therefore, we suggest that species richness of oribatids in sandy soil is driven by other factors than forest management (Farská et al., 2014; Kokořová and Starý, 2017), which can act at different spatial resolution from those we explored in this study (Lindo and Winchester, 2009).

5. Conclusions and implications for conservation

We studied whether fencing affects biological components of coastal pine woodlands including vascular plants and oribatid mites. Our findings suggest that the current mosaic-like management with fencing of smaller areas within larger non-fenced areas is a suitable choice for nature conservation because it increases beta diversity. We provide evidence that the current diversity of dune woodlands seems to be partly increased by the differentiation of forest management, which maintains patches with a low degree of disturbance, thus creating heterogeneity that promotes biodiversity.

To summarise, conservation authorities should keep fenced and non-fenced areas within protected areas of coastal pine woodlands – or create such a mosaic where it does not exist – as an effective and

relatively cheap nature conservation measure. Managers should consider applying patchy management of dune woodlands with a periodic cutting of the understorey to prevent further succession, and simultaneous abandonment of other parts, which will develop towards the climax. Such a choice would make a positive contribution to the maintenance of biodiversity in the stone pine woodlands in coastal areas.

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Author contributions

G.B., with contributions of M.C., J.T., C.A. and M.M., conceived the study. G.B. did the field sampling and identified plants. M.M. identified mites. F.N. did the soil analyses. J.T. performed statistical analyses. G.B. led the writing of the paper, with contributions of M.C., C.A., M.M., J.T., G.P., F.N. and J.S. All the authors commented on the manuscript before submission.

Appendix A. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.foreco.2019.04.005>.

References

- Acosta, A., Stanisci, A., Ercole, S., Blasi, C., 2003. Sandy coastal landscape of the Lazio region (Central Italy). *Phytocoenologia* 33 (4), 715–726. <https://doi.org/10.1127/0340-269X/2003/0033-0715>.
- Amici, V., Santi, E., Filibeck, G., Diekmann, M., Geri, F., Landi, S., Scoppola, A., Chiarucci, A., 2013. Influence of secondary forest succession on plant diversity patterns in a Mediterranean landscape. *J. Biogeogr.* 40 (12), 2335–2347. <https://doi.org/10.1111/jbi.12182>.
- Angiolini, C., Bonari, G., Landi, M., 2018. Focal plant species and soil factors in Mediterranean coastal dunes: an undisclosed liaison? *Estuar. Coast. Shelf Sci.* 211, 248–258. <https://doi.org/10.1016/j.ecss.2017.06.001>.
- Antonellini, M., Mollema, P.N., 2010. Impact of groundwater salinity on vegetation species richness in the coastal pine forests and wetlands of Ravenna, Italy. *Ecol. Eng.* 36 (9), 1201–1211. <https://doi.org/10.1016/j.ecoleng.2009.12.007>.
- Battles, J.J., Shlisky, A.J., Barrett, R.H., Heald, R.C., Allen-Diaz, B.H., 2001. The effects of forest management on plant species diversity in a Sierran conifer forest. *For. Ecol. Manage.* 146 (1), 211–222. [https://doi.org/10.1016/S0378-1127\(00\)00463-1](https://doi.org/10.1016/S0378-1127(00)00463-1).
- Barbato, D., Perini, C., Mocali, S., Bacaro, G., Tordoni, E., Maccherini, S., Marchi, M., Cantiani, P., De Meo, I., Bianchetto, E., Landi, S., Bruschini, S., Bettini, G., Gardin, L., Salerni, E., 2019. Teamwork makes the dream work: disentangling cross-taxon congruence across soil biota in black pine plantations. *Sci. Total Environ.* 656, 659–669. <https://doi.org/10.1016/j.scitotenv.2018.11.320>.
- Bonari, G., Acosta, A.T.R., Angiolini, C., 2017a. Mediterranean coastal pine forest stands: understorey distinctiveness or not? *For. Ecol. Manage.* 391, 19–28. <https://doi.org/10.1016/j.foreco.2017.02.002>.
- Bonari, G., Acosta, A.T.R., Angiolini, C., 2018. EU priority habitats: rethinking Mediterranean coastal pine forests. *Rendiconti Lincei. Scienze Fisiche e Naturali* 29 (2), 295–307. <https://doi.org/10.1007/s12210-018-0684-9>.
- Bonari, G., Fajmon, K., Malenovsky, I., Zelený, D., Holuša, J., Jongepierová, I., Kočárek, P., Konvička, O., Uříčář, J., Chytrý, M., 2017b. Management of semi-natural grasslands benefiting both plant and insect diversity: the importance of heterogeneity and tradition. *Agric. Ecosyst. Environ.* 246, 243–252. <https://doi.org/10.1016/j.agee.2017.06.010>.
- Bonari, G., Knollová, I., Vlčková, P., Xystrakis, F., Čoban, S., Sağlam, C., Didukh, Y.P., Hennekens, S.M., Acosta, A.T.R., Angiolini, C., Bergmeier, E., Bertacchi, A., Costa, J.C., Fanfarillo, E., Gigante, D., Guarino, R., Landi, M., Neto, C.S., Pesaresi, S., Rosati, L., Selvi, F., Sotiriou, A., Stinca, A., Turcato, C., Tzonev, R., Viciani, D., Chytrý, M., 2019. CircumMed Pine Forest Database: an electronic archive for Mediterranean and Submediterranean pine forest vegetation data. *Phytocoenologia*. <https://doi.org/10.1127/phyto/2018/0311>.
- Bonari, G., Migliorini, M., Landi, M., Protano, G., Fanciulli, P.P., Angiolini, C., 2017c. Concordance between plant species, oribatid mites and soil in a Mediterranean stone pine forest. *Arthropod-Plant Interact.* 11 (1), 61–69. <https://doi.org/10.1007/s11829-016-9466-4>.
- Bonet, A., 2004. Secondary succession of semi-arid Mediterranean old-fields in south-eastern Spain: insights for conservation and restoration of degraded lands. *J. Arid Environ.* 56 (2), 213–233. [https://doi.org/10.1016/S0140-1963\(03\)00048-X](https://doi.org/10.1016/S0140-1963(03)00048-X).
- Burrascano, S., de Andrade, R.B., Paillet, Y., Ódor, P., Antonini, G., Bouget, C., Blasi, C., 2018. Congruence across taxa and spatial scales: are we asking too much of species data? *Glob. Ecol. Biogeogr.* 27 (8), 980–990. <https://doi.org/10.1111/geb.12766>.
- Carranza, M.L., Acosta, A.T.R., Stanisci, A., Pirone, G., Ciaschetti, G., 2008. Ecosystem classification for EU habitat distribution assessment in sandy coastal environments: an application in central Italy. *Environ. Monit. Assess.* 140 (1–3), 99–107. <https://doi.org/10.1007/s10661-007-9851-7>.
- Chytrý, M., Lososová, Z., Horská, M., Uher, B., Čejka, T., Danihelka, J., Fajmon, K., Hájek, O., Juříčková, L., Kintrová, K., Lániková, D., Otýpková, Z., Řehořek, V., Tichý, L., 2012. Dispersal limitation is stronger in communities of microorganisms than macroorganisms across Central European cities. *J. Biogeogr.* 39 (6), 1101–1111. <https://doi.org/10.1111/j.1365-2699.2011.02664.x>.
- Chytrý, M., Sedláková, I., Tichý, L., 2001. Species richness and species turnover in a successional heathland. *Appl. Veg. Sci.* 4 (1), 89–96. <https://doi.org/10.1111/j.1654-109X.2001.tb00238.x>.
- Ciccarelli, D., 2014. Mediterranean coastal sand dune vegetation: influence of natural and anthropogenic factors. *Environ. Manage.* 54 (2), 194–204. <https://doi.org/10.1007/s00267-014-0290-2>.
- Coll, L., Ameztegui, A., Collet, C., Löf, M., Mason, B., Pach, M., Verheyen, K., Abrudan, I., Barbati, A., Barreiro, S., Bielak, K., Bravo-Oviedo, A., Ferrari, B., Govedar, Z., Kulhavy, J., Lazdina, D., Metslaid, M., Mohren, F., Pereira, M., Peric, S., Rasztovtits, E., Short, I., Spathelf, P., Sterba, H., Stojanovic, D., Valsta, L., Zlatanov, T., Ponette, Q., 2018. Knowledge gaps about mixed forests: What do European forest managers want to know and what answers can science provide? *For. Ecol. Manage.* 407, 106–115. <https://doi.org/10.1016/j.foreco.2017.10.055>.
- Conti, F., Abbate, G., Alessandrini, A., Blasi, C., 2005. An annotated checklist of the Italian Vascular Flora (Ministero dell'Ambiente e della Tutela del Territorio, Direzione per la Protezione della Natura, Dipartimento di Biologia Vegetale, Università "La Sapienza". Palombi, Rome, Italy).
- Dimopoulos, P., 2016. G3.7 Mediterranean lowland to submontane *Pinus* woodland. European Red List of Habitats. < <https://forum.eionet.europa.eu/european-red-list-habitats/library/terrestrial-habitats/g-forests/g3.7-mediterranean-lowland-submontane-pinus-woodland> > (accessed on December 1st 2018).
- Erdmann, G., Scheu, S., Maraun, M., 2012. Regional factors rather than forest type drive the community structure of soil living oribatid mites (Acari, Oribatida). *Exp. Appl. Acarol.* 57 (2), 157–169. <https://doi.org/10.1007/s10493-012-9546-9>.
- Farská, J., Prejzlková, K., Rusek, J., 2014. Management intensity affects traits of soil microarthropod community in montane spruce forest. *Appl. Soil Ecol.* 75, 71–79. <https://doi.org/10.1016/j.apsoil.2013.11.003>.
- Gergőcs, V., Hufnagel, L., 2017. Comparing the natural variation of oribatid mite communities with their changes associated with anthropogenic disturbance. *Environ. Monit. Assess.* 189 (4). <https://doi.org/10.1007/s10661-017-5897-3>.
- Gigante, D., Attorre, F., Venanzoni, R., Acosta, A.T.R., Agrillo, E., Aleffi, M., Alessi, N., Allegranza, M., Angelini, P., Angiolini, C., Assini, S., Azzella, M.M., Bagella, S., Biondi, E., Bolpagni, R., Bonari, G., Bracco, F., Brullo, S., Buffa, G., Carli, E., Caruso, G., Casavecchia, S., Casella, L., Cerabolini, B.E.L., Ciaschetti, G., Copiz, R., Cutini, M., Del Vecchio, S., Del Vico, E., Di Martino, L., Facioni, L., Fanelli, G., Foggi, B., Frattaroli, A.R., Galdenzi, D., Gangale, C., Gasparri, R., Genovesi, P., Gianguzzi, L., Gironi, F., Giusso Del Galdo, G., Gualmini, M., Guarino, R., Lasen, C., Lastrucci, L., Maneli, F., Pasta, S., Paura, B., Perrino, E.V., Petraglia, A., Pirone, G., Poponessi, S., Prisco, I., Puglisi, M., Ravera, S., Sburlino, G., Sciadrello, S., Selvaggi, A., Spada, F., Spampinato, G., Strumia, S., Tomaselli, M., Tomaselli, V., Uzunov, D., Viciani, D., Villani, M., Wagensommer, R.P., Zitti, S., 2016. A methodological protocol for Annex I Habitats monitoring: the contribution of Vegetation science. *Plant Sociol.* 53 (2), 77–87. <https://doi.org/10.7338/pls2016532/06>.
- Gormsen, D., Hedlund, K., Huifu, W., 2006. Diversity of soil mite communities when managing plant communities on set-aside arable land. *Appl. Soil Ecol.* 31 (1–2), 147–158. <https://doi.org/10.1016/j.apsoil.2005.03.001>.
- Griffis, K.L., Crawford, J.A., Wagner, M.R., Moir, W.H., 2001. Understorey response to management treatments in northern Arizona ponderosa pine forests. *For. Ecol. Manage.* 146 (1), 239–245. [https://doi.org/10.1016/S0378-1127\(00\)00461-8](https://doi.org/10.1016/S0378-1127(00)00461-8).
- Huston, M., 1979. A general hypothesis of species diversity. *Am. Nat.* 113 (1), 81–101. <https://doi.org/10.1086/283366>.
- Iovieno, P., Alfani, A., Băăth, E., 2010. Soil microbial community structure and biomass as affected by *Pinus pinea* plantation in two Mediterranean areas. *Appl. Soil Ecol.* 45 (1), 56–63. <https://doi.org/10.1016/j.apsoil.2010.02.001>.
- Kokořová, P., Starý, J., 2017. Communities of oribatid mites (Acari: Oribatida) of naturally regenerating and salvage-logged montane spruce forests of Šumava Mountains. *Biologia* 72 (4), 445–451. <https://doi.org/10.1515/biolog-2017-0050>.
- Leone, V., Lovreglio, R., 2004. Conservation of Mediterranean pine woodlands: scenarios and legislative tools. *Plant Ecol.* 171 (1/2), 221–235. <https://doi.org/10.1023/>

- B:VEGE.0000029377.59216.e5.
- Lindo, Z., Winchester, N.N., 2009. Spatial and environmental factors contributing to patterns in arboreal and terrestrial oribatid mite diversity across spatial scales. *Oecologia* 160 (4), 817–825. <https://doi.org/10.1007/s00442-009-1348-3>.
- Lososová, Z., Horsák, M., Chytrý, M., Čejka, T., Danihelka, J., Fajmon, K., Hájek, O., Juříčková, L., Kintrová, K., Láníková, D., Otýpková, Z., Řehořek, V., Tichý, L., 2011. Diversity of Central European urban biota: effects of human-made habitat types on plants and land snails: biodiversity of Central European cities. *J. Biogeogr.* 38 (6), 1152–1163. <https://doi.org/10.1111/j.1365-2699.2011.02475.x>.
- Maaß, S., Migliorini, M., Rillig, M.C., Caruso, T., 2014. Disturbance, neutral theory, and patterns of beta diversity in soil communities. *Ecol. Evol.* 4 (24), 4766–4774. <https://doi.org/10.1002/ece3.1313>.
- Malavasi, M., Santoro, R., Cutini, M., Acosta, A.T.R., Carranza, M.L., 2016. The impact of human pressure on landscape patterns and plant species richness in Mediterranean coastal dunes. *Plant Biosyst.* 150 (1), 73–82. <https://doi.org/10.1080/11263504.2014.913730>.
- Maraun, M., Scheu, S., 2000. The structure of oribatid mite communities (Acari, Oribatida): patterns, mechanisms and implications for future research. *Ecography* 23 (3), 374–382. <https://doi.org/10.1111/j.1600-0587.2000.tb00294.x>.
- Martínez, F., Montero, G., 2004. The *Pinus pinea* L. woodlands along the coast of South-western Spain: data for a new geobotanical interpretation. *Plant Ecol.* 175, 1–18.
- Mucina, L., Bültmann, H., Dierßen, K., Theurillat, J.-P., Raus, T., Čarni, A., Šumberová, K., Willner, W., Dengler, J., Gavilán García, R., Chytrý, M., Hájek, M., Di Pietro, R., Iakushenko, D., Pallas, J., Daniëls, F.J.A., Bergmeier, E., Santos Guerra, A., Ermakov, N., Valachovič, M., Schaminée, J.H.J., Lysenko, T., Didukh, Y.P., Pignatti, S., Rodwell, J.S., Capelo, J., Weber, H.E., Solomeshch, A., Dimopoulos, P., Aguiar, C., Hennekens, S.M., Tichý, L., 2016. Vegetation of Europe: hierarchical floristic classification system of vascular plant, bryophyte, lichen, and algal communities. *Appl. Veg. Sci.* 19, 3–264. <https://doi.org/10.1111/avsc.12257>.
- N'Dri, J.K., Seka, F.A., Pokou, P.K., N'Da, R.A.G., Lagerlöf, J., 2017. Abundance and diversity of soil mite (Acari) communities after conversion of tropical secondary forest into rubber plantations in Grand-Lahou, Côte d'Ivoire. *Ecol. Res.* 32 (6), 909–919. <https://doi.org/10.1007/s11284-017-1499-3>.
- Nielsen, U.N., Osler, G.H.R., Campbell, C.D., Neilson, R., Burslem, D.F.R.P., van der Wal, R., 2010. The enigma of soil animal species diversity revisited: the role of small-scale heterogeneity. *PLoS One* 5 (7), e11567. <https://doi.org/10.1371/journal.pone.0011567>.
- Oksanen, J., Guillaume Blanchet, F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Henry, M., Stevens, H., Szoecs, E., Wagner, H., 2018. *Vegan: Community Ecology Package*. R package version 2.5-3. < <https://CRAN.R-project.org/package=vegan> > .
- Pérez-Iñigo, C., 1993. Acari. Oribatei, Poronota. In: Ramos M.A., Tercedor J.A., Ros X.B., Noguera J.G., Sierra A.G., Macpherson Mayol E., Piera F.M., Marino, J.S. González J. T. (Eds.), *Fauna Iberica*, volume 3. Museo Nacional de Ciencias Naturales, CSIC, Madrid.
- Pérez-Iñigo, C., 1997. Acari. Oribatei, Gymnionota I. In: Ramos, M.A., Tercedor, J.A., Ros, X.B., Noguera, J.G., Sierra, A.G., Macpherson Mayol, E., Piera, F.M., Marino, J.S., González, J.T. (Eds.), *Fauna Iberica*, volume 9. Museo Nacional de Ciencias Naturales, CSIC, Madrid.
- Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., R Core Team, 2017. nlme: Linear and nonlinear mixed effects models. R package version 3.1-137. Available at: < <https://CRAN.R-project.org/package=nlme> > (last accessed 10 December 2018).
- Pesaresi, S., Galdenzi, D., Biondi, E., Casavecchia, S., 2014. Bioclimate of Italy: application of the worldwide bioclimatic classification system. *J. Maps* 10 (4), 538–553. <https://doi.org/10.1080/17445647.2014.891472>.
- Roig, S., del Río, M., Cañellas, I., Montero, G., 2005. Litter fall in Mediterranean *Pinus pinaster* Ait. stands under different thinning regimes. *For. Ecol. Manage.* 206 (1–3), 179–190. <https://doi.org/10.1016/j.foreco.2004.10.068>.
- Ruiz-Benito, P., Gómez-Aparicio, L., Zavala, M.A., 2012. Large-scale assessment of regeneration and diversity in Mediterranean planted pine forests along ecological gradients: planted pine forests in the Mediterranean. *Divers. Distrib.* 18 (11), 1092–1106. <https://doi.org/10.1111/j.1472-4642.2012.00901.x>.
- Salmon, S., 2018. Changes in humus forms, soil invertebrate communities and soil functioning with forest dynamics. *Appl. Soil Ecol.* 123, 345–354. <https://doi.org/10.1016/j.apsoil.2017.04.010>.
- Santoro, R., Jucker, T., Prisco, I., Carboni, M., Battisti, C., Acosta, A.T.R., 2012. Effects of trampling limitation on coastal dune plant communities. *Environ. Manage.* 49 (3), 534–542. <https://doi.org/10.1007/s00267-012-9809-6>.
- Scarascia-Mugnozza, G., Oswald, H., Piussi, P., Radoglou, K., 2000. Forests of the Mediterranean region: gaps in knowledge and research needs. *For. Ecol. Manage.* 132 (1), 97–109. [https://doi.org/10.1016/S0378-1127\(00\)00383-2](https://doi.org/10.1016/S0378-1127(00)00383-2).
- Sierzeza, K.P., Eichholz, M.W., 2019. Linking conservation implications of modified disturbance regimes, plant communities, plant associations, and arthropod communities. *Oecologia* 189 (1), 267–277.
- Sitzia, T., Campagnaro, T., Dainese, M., Cassol, M., Cortivo, M., Gatti, E., Padovan, F., Sommacal, M., Nascimbene, J., 2017. Contrasting multi-taxa diversity patterns between abandoned and non-intensively managed forests in the southern Dolomites. *IForest – Biogeosci. Forest.* 10 (5), 845–850. <https://doi.org/10.3832/forest2181-010>.
- Subías, L.S., 2004. Listado sistemático, sinónimo y biogeográfico de los ácaros oribátidos (Acariformes, Oribatida) del mundo. *Graellsia*, 60 (número extraordinario), pp. 3–305.
- Subías, L.S., Arillo, A., 2001. Acari. Oribatei, Gymnionota II. In: Ramos, M.A., Tercedor, J. A., Ros, X.B., Noguera, J.G., Sierra, A.G., Macpherson Mayol, E., Piera, F.M., Marino, J.S., González, J.T. (Eds.), *Fauna Iberica*, volume 15. Museo Nacional de Ciencias Naturales, CSIC, Madrid.
- Tabacchi, G., De Natale, F., Di Cosmo, L., Floris, A., Gagliano, C., Gasparini, P., Genchi, L., Scrinzi, G., Tosi, V., 2007. INFC, 2007 – Le stime di superficie 2005 – Prima parte. *Inventario Nazionale delle Foreste e dei Serbatoi Forestali di Carbonio*. MiPAAF – Ispettorato Generale Corpo Forestale dello Stato, CRA – ISAF, Trento (Italia).
- MiPAAF – Ispettorato Generale Corpo Forestale dello Stato, CRA – ISAF, Trento (Italia) < <https://www.sian.it/inventarioforestale/caricaDocumento?idAlle=496> > (accessed on December 1st 2018).
- Wehner, K., Norton, R.A., Blüthgen, N., Heethoff, M., 2016. Specialization of oribatid mites to forest microhabitats—the enigmatic role of litter. *Ecosphere* 7 (3), e01336. <https://doi.org/10.1002/ecs2.1336>.
- Weigmann, G., 2006. Acari, Actinochaetida. Hornmilben (Oribatida). Goecke & Evers, Keltern.