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SMALL WOODLOTS OUTSIDE FOREST: MULTI-SCALE AND MULTI-TAXA ASSESSMENT FOR ENVIRONMENTAL PLANNING

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ABSTRACT

Trees outside forest (i.e., scattered trees, small woodlots - SWOFs, trees lines) represent important multifunctional resources, providing different ecosystem services at both global and local scales. Nevertheless, more studies are needed to better understand their part in biodiversity conservation. This is particularly true for areas located in important biodiversity hotspots, such as those in the Mediterranean basin.

The main aim of this thesis was to investigate the role of Mediterranean SWOFs for biodiversity conservation, along a gradient of land-use intensification by using a multi-scale and multi-taxa approach.

The adoption of multiple approaches, combining data from cartography to field-research and statistical models, contributed to understand the effect of human-disturbance and environmental conditions on taxa diversity patterns, with several theoretical and applied implications for SWOF management.

An exhaustive census of SWOF in three land-use matrices highlighted that these patches represent a non-negligible component of natural and human-impacted land-uses, supporting a good level of biodiversity in human-altered matrices also.

An intensive field sampling was conducted for almost two years to collect data on vascular plants and six invertebrates: pseudoscorpions, spiders, but also darkling, rove and ground beetles, and ants. This research provides a remarkable contribution to the knowledge of floristic and faunistic species distribution at regional, but also at national level.

For almost all taxonomic groups, the dominant matrices surrounding SWOFs exerted a stronger effect on species composition rather than on species richness; it was observed a general homogenization of taxonomical diversity patterns of multiple taxa across SWOFs surrounded by urban and agricultural matrices, but also on one key functional trait of two congeneric plant species (seed mass).

The high number of key native plants, richness and abundance of almost all invertebrates found in each land-use matrix underlines the need to preserve these remaining patches to avoid that homogenization drives a generalized biodiversity loss.

This study also showed how the highly complex and variable interplay of environmental drivers (spatial-topographic, bioclimatic and landscape-level variables) and biological interactions influenced the cross-taxonomic congruence.

Results support the hypothesis that SWOFs represent an opportunity for natural conversion actions and consequent rewilding ecosystems, with potential benefits for biodiversity conservation. The knowledge provided is a baseline to comprehend the role of small woodlots outside forest and give useful indications for planning and management conservation efforts.

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GENERAL INTRODUCTION

Impacts of human activities, habitat loss and fragmentation on biodiversity

In the last two centuries, the impact of human activities on land has grown enormously (Tylianakis et al. 2008). Across the world, many natural ecosystems have been altered by human activities leading to a complex patchwork of human-dominated landscapes, which has been overlaid on natural heterogeneity of environmental conditions (Tscharntke et al. 2012).

More than half of world ecosystems and biomes have been converted to human use and another large percentage of remnant forestland is expected to be converted to other uses by 2050, due to agriculture and urban sprawl (Millennium Ecosystem Assessment 2005).

The expansion of agriculture activities and urban infrastructures have occurred over most of the Mediterranean region, especially in the Mediterranean basin (Falcucci et al. 2007; Marignani et al. 2017b). Due to the intense human activity occurred in this region for several millennia, the natural spatial patterns have been strongly shaped by this anthropogenic disturbance (Falcucci et al. 2007) resulted in a constantly increasing rate of land consumption (Congedo et al. 2017). According to the last estimates at the national level, the replacement of natural and agricultural areas with artificial land cover in the Italian territory amounts to a total of 23,063 square kilometers and grew up to 24,881 hectares from 2012 to 2017 (Strollo et al. 2020). The growing human pressure along the gradient of land-use intensification (from mountains to urbanized coastal areas) is exerting huge pressures on primary vegetation (Strollo et al. 2020) and natural coastal habitats, strongly affected by tourism (Carranza et al. 2020), leading to a drastic reduction and loss of these important habitats (Malavasi et al. 2013; Marignani et al. 2017a).

These growing human-induced pressures on natural habitats are predicted to become more important in the context of the global environmental change where habitat conversion and degradation, habitat loss and fragmentation, biotic invasions and climate change have been recognized as its main drivers (Tscharntke et al. 2012). Because of this, land-use intensification (Newbold et al. 2015), habitat loss and fragmentation (Fischer and Lindenmayer 2007) have been described as the most significant threat to biodiversity and key topics in landscape ecology (Wilson et al. 2016).

The fragmentation process leads to the reduction of continuous tracts of habitat to smaller and more spatially isolated remnant patches separated and surrounded by a matrix of human-transformed land cover (Haddad et al. 2015). This transformation, destruction, reduction and the consequent isolation of remaining patches directly affect ecological processes at all organization levels by altering gene flows, populations demographic structure, species dispersion dynamics, communities, and the ecosystems' dynamics (Saunders et al. 1991; Fahrig 2003; Cushman et al. 2010).

The role of large patches for biodiversity

In this context, several theories from metapopulation ecology and landscape ecology pointed out the important role of large, intact and well-connected patches for the maintenance of ecological processes and biodiversity conservation (Lindenmayer 2019). Island biogeography theory also applied to “ecological islands” of terrestrial contexts, predicts a species richness variation with island area and isolation, suggesting an increase of richness with the area and the degree of connectivity (MacArthur and Wilson 2001). According to this theory, species–area relationship has shown that only patches with a large area can allow the full assemblage of organisms (Fischer and Lindenmayer 2002; Matthews et al. 2019). To explain this relationship, three biological hypotheses have been proposed: (i) the habitat diversity hypothesis, (ii) the area per se hypothesis, (iii) the passive sampling hypothesis, (iv) resource concentration hypothesis (see Connor and McCoy 2017). The first is based on the assumption that large areas have a greater heterogeneity of habitats than the smaller ones, and they can support habitat specialist species (i.e., species that require specific habitats) and generalist habitat species (i.e., species that require multiple habitats) (Williams 1964), resulting in a high potential number of species that may exist in a large area, as also predicted by niche theory (Vandermeer 1972). The second hypothesis suggests a positive relationship between the increase of species abundance and area, specifying that the probability of species going to extinct is related to the decrease of abundance and area (Simberloff and Abele 1976). The third hypothesis is based on the assumption that larger areas have more chances of receiving more colonists than the smaller ones, which are likely to represent a wider range of species than the pool of colonists arriving on small areas (Connor and McCoy 1979). The fourth one predicts that large habitat patches with large amounts of resources can support a wide number of species (e.g., densities of insects) because resource concentration is more likely to contribute to their longer persistence in those patches (Root 1973).

These mechanisms are not exclusive and may act individually or in combination with others, including those related to the edge effect (Connor and McCoy 2017). Studies on edge effects show that small patches have a larger amount of “edge habitat" than larger patches and only the latter can provide “interior habitat" adapted to sustain a great number of vulnerable species (Laurance 2000), decreasing the probabilities of local extinction (Connor and McCoy 2017).

The role of small patches for biodiversity

Each of the above mechanisms points to the importance of large and well-connected patches for ensuring the persistence of species and populations’ robustness to stochastic perturbations, highlighting the lower ecological value of small, isolated patches with extensive edge environments in the landscape (Wintle et al. 2019; Lindenmayer 2019).

Given the large number of ecological theories, the importance of large, intact and well-connected patches has been recognized by sectoral policies, programs and projects across the world, which commonly integrate large, mostly intact habitats on the conservation policies, often avoiding areas with many small fragments (Fahrig 2017). Most of these key sectoral policies and programs downplay the value of areas containing many small fragments and not consider small isolated remnant patches as valuable complements to the conservation of large patches (Fischer and Lindenmayer 2002; Wintle et al. 2019).

However, the high conservation value and importance of small remnant patches was pointed out with emphasis by several recent reviews and meta-analyses (Le Roux et al. 2015; Tulloch et al. 2016; Fahrig 2017; Wintle et al. 2019; Lindenmayer 2019). Such reviews and meta-analyses demonstrated a consistent number of positive responses to habitat fragmentation, highlighting that there is no justification for considering small patches as habitats with low conservation value (Fahrig 2017; Wintle et al. 2019; Lindenmayer 2019).

Several small patches are capable of supporting a higher level of species richness than a single large patch, regardless of the considered taxonomic group, except birds that showed slightly negative responses to the considered landscape metrics, such as patch density and size metrics (e.g., patch number and patch size), or edge metrics (e.g., edge density) (Fahrig 2017; Morelli et al. 2018).

To explain the observed positive responses to habitat fragmentation, different reasons have been proposed and associated to the (i) amount of these patches in the human-impacted landscapes, (ii) functional connectivity, (iii) landscape heterogeneity and complementation, (iv) edge effect, (v) predator-prey and host-parasitoid mechanisms, (vi) competition at intraspecific and interspecific level (Fahrig 2017; Lindenmayer 2019).

The first common explanation considering the wide-spread distribution of these small patches in human-impacted landscapes (Tulloch et al. 2016) suggests that they often are all that remain where the intense human activities have occurred for several millennia (Lindenmayer 2019). In these human-modified landscapes, small patches can provide sufficient habitat for small-patch specialists or open-country species, playing a crucial role in their conservation (Fischer and Lindenmayer 2002).

The second common explanation is related to the capacity of these patches to represent stepping stones, or nodal points among the remaining large habitats, efficiently acting as connectivity providers due to their size and spatial arrangement (Saura et al. 2014). For example, numerous small patches separated by small distances can increase the probability that dispersing organisms can move successfully across, potentially increasing the functional connectivity (Manning et al. 2006) and promoting the natural regeneration of large habitat areas (Chazdon et al. 2009; Fischer et al. 2009). The loss of these stepping stones and nodal points could hardly be compensated by increasing the habitat amount or the resources and the population sizes in the source patches (Saura et al. 2014).

The most common explanation is related to the landscape heterogeneity in areas characterized by a large number of habitat patches (Fahrig 2017). For example, a large number of patches can capture a much wider environmental heterogeneity than a single large patch, providing more resources for the persistence of species and populations (Tscharntke et al. 2012; Hu et al. 2012). Heterogeneous landscapes with multiple edges and notches can support species that require multiple resources located in the proximity (Slancarova et al. 2014), increasing the "landscape complementation" *sensu* Dunning et al. (1992).

Although some authors suggested that a limited presence of an extensive edge in large habitat can provide a positive effect for interior-patch species (e.g., Laurance 2000). Edge of small patches can represent high-quality habitat for many species (Fahrig 2017). The positive responses to edge effects have been linked to structural diversity and productivity of these marginal habitats, which can provide higher refuges and resources than interior habitat (Henden et al. 2011; Moore et al. 2011).

Another possible explanation is attributed to the increasing of predator–prey and host-parasitoid mechanisms, related to the presence of refuges for prey and host-parasitoid (Fahrig 2017).

Last, but not least, the variation in resources of small patches in human-fragmented landscapes can be particularly helpful when other resources are limited, reducing competition between species (Fargione et al. 2007). The spatial arrangement of patches in human-modified landscapes can delay the competitive exclusion, favoring the repeated immigration of competitors and increasing the coexistence of ecologically equivalent species (Dufour et al. 2006).

Trees outside forest as small resources patches - definitions

We can define as small patches: small woodland and shrubland, linear arboreal elements, green belts, clumps of trees, or large scattered trees embedded in a matrix of human-modified land-use (Wintle et al. 2019; Lindenmayer 2019).

In a more holistic view, these resources are included into the definition of trees outside the forest (hereby TOF), trees or shrub that not reach the thresholds *in situ* (in terms of size, width and height) to be assigned to categories of "Forest" and "Other Wooded Land" (Bellefontaine et al. 2001, 2002). The definition of TOF was provided by Food & Agricultural Organization of United Nations (hereby FAO), based on the definition of forest (FOR) and other wooded land (OWL) and is widely accepted and applied at the international level (FAO 2010).

Forest (FOR)

FAO defines forest as "land spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10%, or trees able to reach these thresholds *in situ*. It does not include land that is predominantly under agricultural or urban land use" (FAO 2010).

The FAO definition of "Forest" (FAO 2010) implies that:

- Forest is determined both by the presence of trees and the absence of other predominant land-uses;

- The trees should be able to reach a minimum height of 5 meters *in situ*;
- Includes areas with young trees that have not yet reached, but which are expected to reach, the thresholds of canopy cover and height;
- Includes temporarily unprotected areas with forest management practices or affected by natural disasters, which are expected to be regenerated within 5 years;
- Includes forest roads, firebreaks and other small open areas;
- Forest in national parks, nature reserves and other protected areas (e.g., those of environmental, scientific, historical, cultural or spiritual interest) are included;
- Includes windbreaks, shelterbelts and corridors of trees with an area of more than 0.5 hectares and width of more than 20 meters;
- Excludes tree stands in agricultural production systems (fruit tree plantations, oil palm plantations etc.) and agroforestry systems when crops are grown under tree cover.

Other Wooded Land (OWL)

Following the FAO definition, lands that are not classified as Forest, i.e. spanning more than 0.5 hectares, trees higher than 5 meters and a canopy cover of 5-10%, or trees able to reach these thresholds *in situ*, or with a combined cover of shrubs, bushes and trees above 10%, fall within the classification of "Other Wooded Land" (OWL) (FAO 2010). This category does not include land that is predominantly under agricultural or urban land use (FAO 2010).

The FAO definition of Other Wooded Land (FAO 2010) implies that:

- The canopy cover of trees is between 5 and 10%;
- The threshold value of combined cover of shrubs, bushes and trees is more than 10%;
- Includes areas with shrubs and bushes where no trees are present.

Other Land (OL)

All areas that are not classified as “Forest” or “Other Wooded Land” are considered as "Other Land": agricultural land, meadows and pastures, built-up areas, arid land, land under permanent ice, etc.

The FAO definition of “Other Land” (FAO 2006) implies that:

- The difference between Forest and Other Land with tree cover is defined by the type of land use;
- Clusters of trees and scattered trees in agricultural areas, park, garden and urban landscapes are included, provided that the criteria for extension, coverage and height are met;
- Agricultural production systems (e.g., fruit tree plantations) or agroforestry systems are included;
- Scattered trees with coverage less than 10%, groups of trees with an extension of less than 0.5 ha and rows of trees less than 20 m in width are excluded.

As a sub-category of "Other Land", we find "Other Land with Tree Cover" (OLwTC) (FAO 2010). This sub-category was integrated to take into account the formations meeting the “Forest” criteria but grow in agriculture or urban areas as a prevalent land-use (FAO 2006). Therefore "Other Land with Tree Cover" (OLwTC) sub-category represents those areas, classified as "Other Land" (OL), with an extension greater than 0.5 hectares and trees with coverage greater than 10% able to reach a height of 5 m at maturity (FAO 2006).

Trees Outside Forest (TOF)

Referring to the definitions of “Forest”, "Other Wooded Lands" and "Other Lands" (FAO 2006, 2010), TOF include:

- **Groups of trees** capable of reaching a height of 5 m (**height ≥ 5 m**) maturity *in situ* and with an **area of less than 0.5 hectares**, whatever the degree of coverage (see definition of Forest);
- **Trees** capable of reaching a height of 5 m (**height ≥ 5 m**) at maturity *in situ*, but with a coverage of less than 5% (**cc <5**) (see definition of Other Wooded Lands);
- **Trees or shrubs** that do not reach a height of 5 m (**height <5 m**) at maturity *in situ*, but with a combined coverage less than 10% (**cc <10%**) (see definition of Other Wooded Lands);
- **Linear and riparian formations** with a width less than 20 m (**width <20 m**), consisting of trees capable of reaching a height of 5 m (**height ≥ 5 m**) at maturity *in situ* (see definition of Forest).

The above definitions (FAO 2006, 2010) therefore imply that are considered as TOF:

- Not only trees but also shrubs that do not reach the threshold of 5 m in height as long as a combined coverage of less than 10% is achieved;

- Any tree that grows in “other lands” (OL);
- Any tree or shrub that grows in an agricultural or urban land use, excluded from the category of "Forests" or "Other Wooded Lands";
- Those that meet the thresholds even if not included in agricultural or urban land use.

Based on the reported thresholds, TOF can assume different spatial patterns and be found along streams, canals, roads and highways, in almost all land uses (urbanized, industrial and commercial areas, more or less heterogeneous mining or agricultural areas, etc.) (de Foresta et al. 2013).

Although a clear and shared classification of these highly diverse resources is still needed, attempts of classification have been elaborated in agroforestry, there is not yet any classification system that includes all TOF, although a formal classification of TOF appears to be necessary to allow map representations and monitor these resources (Kleinn 2000).

Following Kleinn (2000), in classifying TOF two or more general criteria can be considered useful:

1. the land-use in which they grow (agricultural land, urban areas, etc.);
2. their spatial configuration (linear, clustered, isolated and scattered);
3. the functions they perform (e.g., fencing, windbreak, shade, landscape, etc.);
4. their origin (planted or residues).

Based on the land-use in which TOF grow (de Foresta et al. 2013), they can be classified into three main categories:

- **Trees on land predominantly under agricultural use** (**TOF-AGRI**), all trees and/or shrubs (including plantations, orchards and agroforestry systems excluding those used for forestry), whatever their spatial model (linear, grouped or scattered), found in this type of land-use;
- **Trees on land predominantly under urban use** (**TOF-URB**), all trees and / or shrubs (including trees in private gardens, parks, along roads, in parking lots, etc.), regardless of the spatial model (linear, clustered or scattered), found in this type of land-use;
- **Trees Outside Forests, on land not predominantly under agricultural or urban use** (**TOF-NON A/U**), all trees and/or shrubs growing in areas not predominantly occupied by the above land-use types.

According to geometry or the spatial configuration (Kleinn 2000; FAO 2010; de Foresta et al. 2013), TOF have also been categorized as:

1. **Groups of trees and shrubs** (Small Woodlots Outside Forests, hereafter **SWOF**) planted or grown naturally, regardless of height and coverage, with an extension between five hundred and five thousand square meters (**0.5> area ≥ 0.05**);
2. **Trees in lines** (**width <20 m, length ≥ 20 m**) natural or planted, whatever the level of their coverage;
3. **Scattered trees**, **single trees** (**height ≥ 5 m**) **or groups of trees** with a no-linear arrangement and an extension less than five hundred square meters (**area <0.05 ha**).

The importance of TOF

Apart from the studies that focused on the importance of small patches, the role of TOF as an important multifunctional resource has been widely acknowledged both from the socio-economic and environmental point of view(Paletto et al. 2006; Marchetti et al. 2018a, b). They provide fundamental ecosystem services (Paletto et al. 2006; Marchetti et al. 2018a, b) and contribute to the human wellbeing at both the global and local scale (Plieninger et al. 2012; de Foresta et al. 2013; Schnell et al. 2015).

From the socio-economic point of view, the trees outside the forest are used for various primary and secondary productions, such as timber, firewood, bark, branches for pets, honey, and other edible products (e.g., small fruits, honey) (Mezzalira 1997). TOF also offers a wide range of cultural services, providing resources for ecotourism, recreation, and education, especially in residential areas (Schnell et al. 2015; Sallustio et al. 2018).

From an environmental point of view, the ecosystem functioning provided by trees outside forest is believed to be disproportionate relative to the small area occupied, as a result, they are defined as keystone structures of the landscapes (Manning et al. 2006; Fischer et al. 2010; Prevedello et al. 2018).

At the broad scale, TOF play an important role in climate change adaptation, mitigation and carbon sequestration (Schnell et al. 2015). Woody plants in TOF contribute to improving the air quality, filtering and absorbing air pollutants, and mitigating the impacts of livestock and industrial odours (Rahman et al. 2018).

At the local scale, woody plants in TOF contribute to microclimate moderation through reducing wind and light penetration and intercepting solar radiation and precipitation (Rossi et al. 2016). TOF have a positive impact on soil fertility and contribute to reducing the risk of soil erosion and desertification, increasing the concentration of water in the soil, especially in dry environments (Eldridge and Freudenberger 2005). Woody plants in TOF contribute to maintaining water quality and quantity by reducing the impact of flood and storm-surge events (slowing rainfall-runoff), reducing the sediment and nutrient loads, and protecting stream banks from erosion (Manning et al. 2006).

Thanks to their wide distribution, TOF promote the conservation of biodiversity by representing ecological corridors and offering the living fences for numerous animal and plant species within fragmented landscapes (Bellefontaine et al. 2001, 2002; Manning et al. 2006).

The important role for biodiversity conservation of TOF has been recognized in commercial forests (Mazurek and Zielinski 2004; Matveinen-Huju et al. 2006), agricultural areas (Dunn 2000; Harvey et al. 2006; Manning et al. 2006; DeMars et al. 2010; Fischer et al. 2010; Frizzo and Vasconcelos 2013) and urban landscapes (Yasuda and Koike 2009; Stagoll et al. 2012). Many individual studies focused on studying areas supporting scattered trees considering one or some taxonomic groups, including for example plants (Schlawin and Zahawi 2008; Poltz and Zotz 2011; Rivest et al. 2013; Rossetti et al. 2015), birds (Harvey et al. 2006; DeMars et al. 2010; Pizo and Santos 2011; Stagoll et al. 2012; Barth et al. 2015), bats (Lumsden and Bennett 2005; Harvey et al. 2006; Fischer et al. 2010), collembolans (Rossetti et al. 2015), ants, beetles or other invertebrates (Dunn 2000; Reyes-López et al. 2003; Oliver et al. 2006; Matveinen-Huju et al. 2006; Ohsawa 2007; Yasuda and Koike 2009), and mites (Brooker et al. 2008).

However, only a few studies analyzed and compared abundance and richness of multiple taxonomic groups across different areas and landscape contexts supporting scattered trees (Frizzo and Vasconcelos 2013; Azihou et al. 2013; Le Roux et al. 2015; Prevedello et al. 2018). Surprisingly enough, no studies focused on abundance and richness of multiple taxonomic groups in the Mediterranean areas are conducted considering, for example, SWOFs located in different land-use matrices and their different attributes at multiple scales.

Aims of the thesis

Considering that policymakers, land planners and conservation organizations had generally focused its efforts on large, intact and well-connected areas by underestimating the importance of small fragments (Wintle et al. 2019), more empirical studies on taxa inhabiting small patches is urgently needed to support and promote their conservation (Lindenmayer 2019).

Given this urgent need, this thesis aims at investigating the spatial distribution and importance of Mediterranean SWOFs for biodiversity conservation, along a gradient of land-use intensification from natural areas to urbanized zones (natural and semi-natural, agriculture and artificial areas), by achieving four general goals:

1. focus on SWOF, including orchards with fruit-producing trees located outside forests, with the specific aims:
    a) map the spatial distribution of SWOFs along a gradient of land-use intensification;
    b) assess differences in native and alien species richness and composition among SWOFs located in different land-use matrices;
    c) identify key native and alien taxa related to each land-use matrix;
    d) assess the structural attribute differences among SWOFs surrounded by different land-use matrices;
    e) explore the effect of environmental factors and structural attribute on native and alien communities.
2. define the floristic and faunistic composition of the sampled taxonomic groups;
3. assess diversity patterns of several taxonomical groups - vascular plants and six groups of ground-dwelling arthropods (pseudoscorpions, spiders, darkling, rove and ground beetles, ants) – with the specific aims:
    a) evaluate diversity pattern in terms of species richness and composition among land-use matrices,
    b) explore the effect of environmental variables and biological interactions on taxa concordance in SWOFs located along the gradient of land-use intensification.
4. study the functional traits of vascular plants in SWOFs surrounded by different land-use types. Data of plant height and the seed of herb layer communities have been collected and

the relative databases built up. As an example, data of seed mass of two congeneric species were analyzed with the aims to:

a) assess the influence of land-use matrices on seed mass and its variability at intra- and interspecific levels;

b) explore the relationship between the intraspecific variation of seed mass and different environmental filters.

GENERAL FRAMEWORK

The research period was divided into three principal phases: pre-fieldwork phase, fieldwork phase and post-fieldwork phase (see figure 1).

Pre-fieldwork phase

The pre-fieldwork phase was carried out by photointerpretation and classification of digital aerial orthophotos to provide a Small Woodlots Outside Forests (SWOF) census in the study area (Chapter 1, aim 1). The mapped Small Woodlots Outside Forests (SWOF) were classified according to the first hierarchical level of regional land-use map (scale 1:25.000; RAS 2008) into three main categories: SWOFs in natural and semi-natural areas (NAT), SWOFs in agricultural areas (AGR), SWOFs in urban and artificial surfaces (URB), SWOFs surrounded by mixed land-use areas. 30 SWOF were selected with stratified random sampling, proportionally to the number of Small Woodlots Outside Forests (SWOF) present in each land use stratum.

Environmental variables were measured using several tools in geographical information system and categorized in three main descriptor sets: (i) spatial-topographic factors, (ii) landscape measures, (iii) bioclimatic variables (Chapter 1).

Fieldwork phase

The fieldwork phase was performed by applying a systematic sampling design for each site with the use of (i) five plots of 1x1 m and (ii) five pitfall traps systematically placed along a transect, and (iii) a Cross-vanes Window Flight Trap (CWFT) for each SWOFs centroid.

By using 1 sqm plots and pitfall traps, data of vascular plants and six groups of ground-dwelling invertebrates - pseudoscorpions (Arachnida, Pseudoscorpiones), spiders (Arachnida, Araneae), darkling beetles (Insecta, Coleoptera, Tenebrionidae), rove beetles (Insecta, Coleoptera, Staphylinidae), ground beetles (Insecta, Coleoptera, Carabidae), and ants (Insecta, Hymenoptera, Formicidae) – were collected and determined (Chapter 2 and 3, aims 2-3).

By using the cross-vanes window flight traps, data of pseudoscorpions (Arachnida, Pseudoscorpiones), spiders (Arachnida, Araneae), cicadas *sensu lato* (Insecta, Rhynchota, Cicadidae and Cicadellidae), saproxylic and ground beetles (Insecta, Coleoptera) were also recorded. The determination of these data is still in progress, but among one of the determined species resulted new for the island fauna and the new records were published (Chapter 2, aim 2).

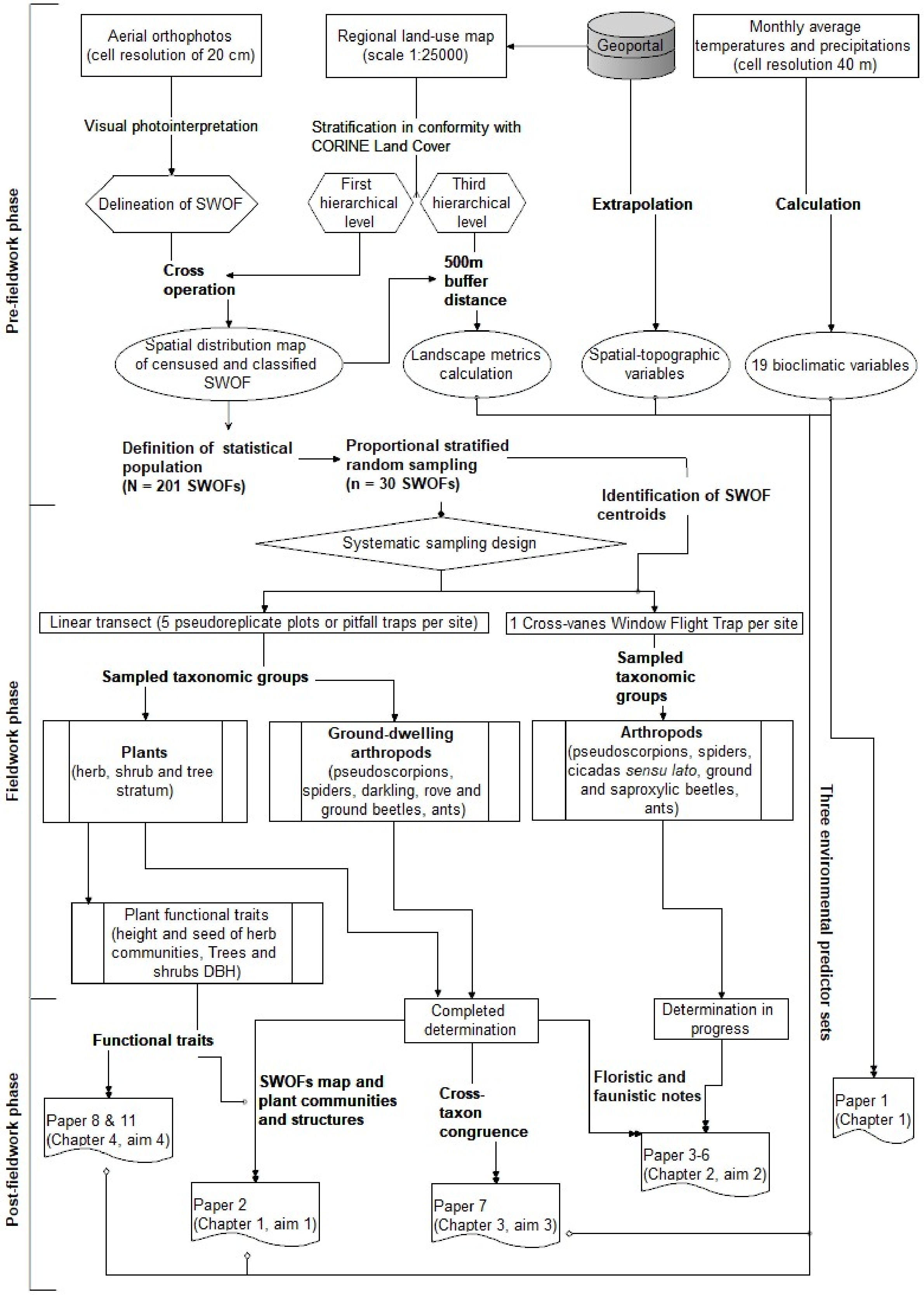
In addition, a selection of plant traits was collected at plot level for trees, shrub and herb layers (i) trees and shrubs diameter at breast height (DBH) (Chapter 1, aim 1) ii) plant height of the herbaceous layer (Chapter 4, aim 4), (iii) seed mass of the herbaceous layer (Chapter 4, aim 4).

Post fieldwork phase

During the post fieldwork phase, GIS and field data were combined to perform all statistical analyses and writing the papers.

Statistical analyses were performed following an increasing level of complexity: (i) focusing on the relative importance of single environmental drivers (e.g., the dominant land-use) (Chapter 1 and 4), (ii) analysing a combination of several drivers, such as climate (Chapter 1), abiotic, and biotic filters varying at given spatial scale, allowing to explore multiple combination of effects on diversity patterns (Chapter 3).

Figure 1. Logical framework of the research with phases and outputs.



PAPERS PUBLISHED OR PREPARED DURING THE PHD

Papers related to environmental data of the thesis

1. **Bazzato E.,** Rosati L., Canu S., Fiori M., Farris E., Marignani M., 2021. High spatial resolution bioclimatic variables to support ecological modelling in a Mediterranean biodiversity hotspot. Ecological Modelling, 441: 109354. https://doi.org/10.1016/j.ecolmodel.2020.109354

Paper related to the mapping of small woodlots outside forests

2. **Bazzato E.**, Lallai E., Serra E., Melis M.T., Marignani M. (submitted). Key role of small woodlots outside forest in a Mediterranean fragmented landscape. Submitted to Forest Ecology and Management

Papers related to taxonomic groups sampled during the thesis

3. Rosati L., Fascetti S., Romano V.A., Potenza G., Lapenna M.R., Capano A., Nicoletti, P., Farris E., de Lange P.J., Del Vico E., Facioni L., Fanfarillo E., Lattanzi E., Cano-Ortiz A., Marignani M., Fogu M.C., **Bazzato E.**, Lallai E., Laface V.L.A., Musarella C.M., Spampinato G., Mei G., Misano G., Salerno G., Esposito A., Stinca A., 2020. New Chorological Data for the Italian Vascular Flora. Diversity, 12(1): 22. https://doi.org/10.3390/d12010022
4. **Bazzato E.**, Marignani M., Ancona C., Caria M., Cillo D., Serra E. (submitted). First record of *Ozognathus cornutus* (Leconte, 1859) (Coleoptera: Ptinidae) from Sardinia, Italy. Submitted to Redia.
5. Caria M., Pantini P., Alamanni F., Ancona C., Cillo D., **Bazzato E.** (accepted). New records and distributional data for the Sardinian spider fauna (Arachnida, Araneae). Accepted by Fragmenta entomologica
6. Schifani E., Nalini E., Gentile G., Aleffi M., Alamanni F., Ancona C., Caria M., Cillo D., **Bazzato E.**, 2021. Ants of Sardinia: an updated checklist based on new faunistic, morphological and biogeographical notes. Redia 104, 21–35. https://doi.org/10.19263/REDIA-104.21.03.

Paper related to the cross-taxon congruence

7. **Bazzato E.**, Caria M., Lallai E., Schifani E., Cillo D., Maccherini S., Bacaro G., Marignani M., (prepared). Cross-taxon congruence in small woodlots outside forest along a land-use intensification gradient.

Papers related to the functional results of the thesis

8. **Bazzato E.**, Serra E., Maccherini S., Marignani M. (submitted). Reduction of intraspecific seed mass variability along a land-use intensification gradient. Submitted to Ecological indicator

Papers related to the environmental planning

9. **Bazzato E.,** Cillo D., Bacaro G., Marignani M. (prepared). Multiple approach leads to multi-solutions – an index to plan functional urban areas.
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CHAPTER 1 – ENVIRONMENTAL SPATIAL DATA AND SWOF MAPPING

The main aim of this chapter was focused on SWOFs: a map of their spatial distribution was provided and their structural attribute together with native and alien plant species richness and composition were assessed. Hence, the effect of different structural attribute and environmental factors (spatial-topographic, bioclimatic and landscape-level variables) on species composition were explored (Subchapter 1.2, paper 2). Among the considered environmental factors, an original dataset of 19 high-spatial resolution bioclimatic variables was produced (Subchapter 1.1, paper 1).

Subchapter 1.1: high-spatial resolution bioclimatic variables

Paper 1

Bazzato E., Rosati L., Canu S., Fiori M., Farris E., Marignani M., 2021. High spatial resolution bioclimatic variables to support ecological modelling in a Mediterranean biodiversity hotspot. Ecological Modelling, 441: 109354. https://doi.org/10.1016/j.ecolmodel.2020.109354

Subchapter 1.2: SWOF map, plant community and structure

Paper 2

Bazzato E., Lallai E., Serra E., Melis M.T., Marignani M. (submitted). Key role of small woodlots outside forest in a Mediterranean fragmented landscape. Submitted to Forest Ecology and Management

Subchapter 1.1: high-spatial resolution bioclimatic variables

High spatial resolution bioclimatic variables to support ecological modelling in a Mediterranean biodiversity hotspot

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Abstract

Understanding the effects of climate on biodiversity and its different levels of response to climatic variation is important for addressing conservation-based questions: the use of bioclimatic variables and species modelling tools is common in environmental, agricultural and biological sciences. Unfortunately, most of the ecological local studies are limited to the use of global data with coarse spatial resolutions, while fine-grain climate data are necessary to capture environmental variability and perform reliable modelling. We propose a high-resolution dataset (40 m grid) of the suite of original coarse-grain bioclimatic variables proposed by WorldClim 2 for the island of Sardinia (Italy); variations amongst our dataset and WorldClim 2 were calculated and mapped to show the spatial distribution of differences between all pairs of variables.

We observed relevant differences for the bioclimatic variables related to rainfall (mean RMSE = 39.79; mean nRMSE = 0.21) compared to the temperature ones (mean RMSE = 4.81; mean nRMSE = 0.11). Moreover, discrepancies are not evenly distributed in the territory: the greater differences correspond to the areas characterized by complex orographic systems.

Results recommend caution in making ecological assessments based on bioclimatic variables derived from global data with coarse spatial resolutions in physiographically complex landscapes, especially in the Mediterranean regions, characterized by seasonal climatic variations and high levels of biodiversity and biogeographical complexity.

These new data will support a new generation of research studies in a broad array of ecological applications at a much finer scale than previously possible

Keywords: Sardinia WorldClim, Species distribution model, Seasonal climatic variations, Data reliability

Bazzato E., Rosati L., Canu S., Fiori M., Farris E., Marignani M., 2021. High spatial resolution bioclimatic variables to support ecological modelling in a Mediterranean biodiversity hotspot. Ecological Modelling, 441: 109354. https://doi.org/10.1016/j.ecolmodel.2020.109354

Introduction

Climate varies across space and species can shift their distribution in order to find appropriate climatic conditions where they can live suitably (Bellard et al., 2012). In the same way, climate fluctuations drive the ecological changes in species, populations, ecological networks and ecosystems functions and processes (Parmesan, 2006). Climate variation over time, including year-to-year variability, has been linked to a shift in phenology and physiology of plants and animals (Bellard et al., 2012; Parmesan, 2006); moreover, also latitudinal and altitudinal range shifts are well documented for a wide number of species (Lenoir and Svenning, 2015), especially for those with high dispersal capacities like marine invertebrates, birds, and insects (Parmesan, 2006).

Bioclimatic variables, unlike climate data, are developed focusing on relevant combination of variables, considering biotic thresholds; hence they better describe, and predict, the response of living organisms (Jennings and Harris, 2017; Rivas-Martínez et al., 2011).

In the middle 1980s, the earliest computer-based methods were developed for estimating mean climate conditions of a given site on Earth's surface, by using point location data sets (Sutherst and Maywald, 1985) or spatially local gridded climate data (e.g., Booth et al., 1987). Following the development of more sophisticated and complex spatial interpolation methods (Hutchinson and Gessler, 1994), modellers have rapidly built spatially gridded climatologies, appropriately scaled on land elevation (Hutchinson, 1995). Subsequently, spatially interpolated gridded climate data have become available for researchers, improving environmental information in sites where there was a lack of local data (Hijmans et al., 2005).

Open data on gridded bioclimate datasets, which differ in their quality over time, space and resolution (from 30 s ~1 km2 to 10 min ~340 km2 at the equator), are for example WorldClim (Fick and Hijmans, 2017), MerraClim (Vega et al., 2017), CHELSA (Karger et al., 2017), CliMond (Kriticos et al., 2012), EuMedClim (Fr´ejaville and Benito Garzo´n, 2018) and ENVIREM (Title and Bemmels, 2018). Most of these global datasets consist of monthly average temperature (minimum, maximum and medium), monthly precipitation and solar radiation assessed across a large temporal range, as well as bioclimatic variables. Bioclimatic variables, originally devised by Nix (1986) and deriving from the monthly temperature and rainfall values, describe annual trends (e.g., mean annual temperature and precipitation), seasonal trends (e.g., annual range in temperature and precipitation) and extreme or limiting environmental factors (e.g., temperature of the coldest and warmest month, and precipitation of the wet and dry quarters).

Considering their peculiar characteristics, bioclimatic variables were considered suitable for studying species distributions, under current or possible future conditions, using species distribution modelling (SDM) tools (Kriticos et al., 2012). The use of bioclimatic variables and species modelling tools have thus found a widespread use in environmental, agricultural and biological sciences (Booth et al., 2014; Di Febbraro et al., 2018; Guisan and Thuiller, 2005; Pecchi et al., 2019): assessing the environmental niche of species or their invasion and proliferation; quantifying the impact of climate and other environmental changes on species distributions; modelling species assemblages from individual species predictions; testing biogeographical, ecological and evolutionary hypotheses; identifying sites of high potential of occurrence for rare species; developing strategies and action plans to ensure a long-term conservation of species.

For many applications, fine spatial grain climate data is considered necessary to capture environmental variability, especially in physiographically complex landscapes (Hijmans et al., 2005); for example, they are preferable to study distribution of species with low-dispersal ability (Chust et al., 2004; Franklin et al., 2013; Guisan et al., 2007), species corridors and effects of barriers, or for others detailed ecological or conservation studies (Elith and Leathwick, 2009; Hess et al., 2006; Nezer et al., 2017). Fine-grain climate grids are able to detect potential microrefugia (Hannah et al., 2014; Meineri and Hylander, 2017), i.e. sites with peculiar microclimates that support populations of species outside their main distribution area. Microrefugia are thus particularly relevant to understand the spatial distribution of species in response to climate change (Dobrowski, 2011) and the demographic and genetic performance of populations at the periphery of their range (Papuga et al., 2018; Pironon et al., 2017).

Unfortunately, fine-grain climate grids are only available for limited parts of the world (Hijmans et al., 2005) and most of the ecological local studies are limited to and by the use of global data with coarse spatial resolutions.

The development of high spatial resolution bioclimatic data is particularly important in the Mediterranean basin, one of the 35 terrestrial biodiversity hot spots of the world (Medail, 2017), where climate-driven habitat loss was recognized as a major threat to biodiversity (Barredo et al., 2016). Nevertheless, as for Sardinia, the second-largest island of the Mediterranean basin, several studies to assess the distribution of plants (e.g., Casazza et al., 2014; Fois et al., 2018a, 2018b; Ongaro et al., 2018) or animals (e.g., Iannella et al., 2019; Russo et al., 2014; Sýkora et al., 2017) relied on coarse-grain bioclimatic open data such as Worldclim. To fill this gap in Sardinia, we propose a novel high-resolution dataset (40 m grid, equal to ~ 1.69 arcsec) of the suite of bioclimatic variables proposed by WorldClim 2 (Fick and Hijmans, 2017), one of the most used dataset in ecological modelling (Marchi et al., 2019).

We calculated the suite of 19 bioclimatic variables using a high- resolution monthly climatologies of temperature and precipitation of Sardinia, based on long-term climate time series and local topography. To assess the differences amongst our fine-grain dataset and the original coarse-grain bioclimatic variables of WorldClim 2, we performed a quantitative comparison and spatial distribution of errors between all pairs of variables of these datasets.

The high-resolution data produced can be particularly suited for studying species distributions under current conditions, improving ecological studies at finer spatial scales.

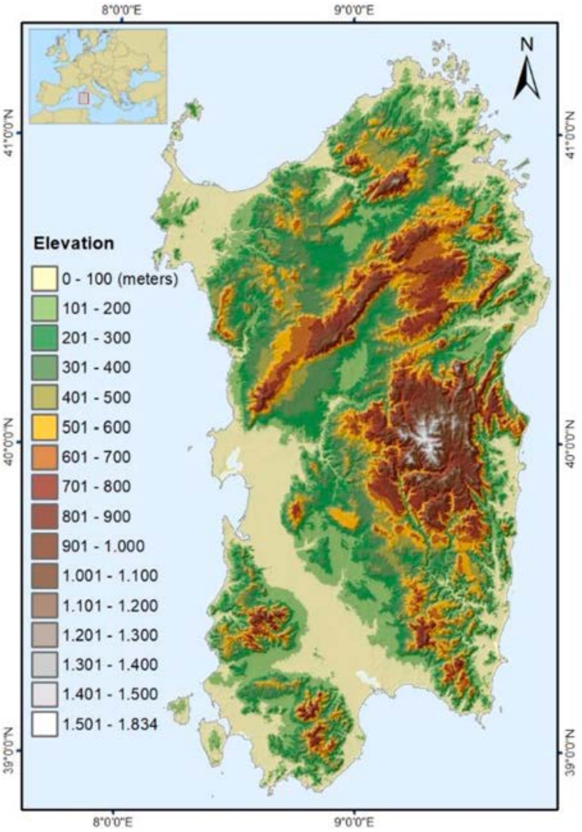
Study area

The island of Sardinia, one of the two largest Mediterranean islands, is located in the middle of the western Mediterranean Basin and covers a surface area of around 24,000 km2 with a coastline of about 1,900 km, marked by a variety of landforms (cliffs, sandy dunes, long or pocket beaches). Due to its large extension, the territory is characterized by a complex orographic pattern with hilly lands, plateaus, mountain and plains (Fig. 1), placed on heterogeneous geological substrata for age and typology.

More than 600 formations and more second-rank lithostratigraphic units have been recognized (Carmignani et al., 2016): Palaeozoic magmatic intrusive units and metamorphic complexes related to Hercynian Orogenesis; sedimentary successions linked to Mesozoic and Tertiary marine transgression; volcano-sedimentary successions related to the opening of the Tyrrhenian Sea; Quaternary deposits of various origin (alluvial, aeolian, lacustrine, littoral and slope movement-related) covering the previous geological formations.

The climate is typically Mediterranean, with mild and poorly rainy winters, warm and dry summers. Recent detailed bioclimate mapping, using the bioclimatic classification of Rivas-Martínez et al. (2011), identified that the island is characterized by two macrobioclimates (Mediterranean pluviseasonal oceanic and Temperate oceanic), four classes of continentality (from weak semihyperoceanic to weak sub-continental), eight thermotypic horizons (from lower thermomediterranean to upper supratemperate) and seven ombrothermic horizons (from lower dry to lower hyperhumid), whose combination resulted in 43 different isobioclimates (Canu et al., 2015).

Fig. 1. Sardinia is the second main island in the Mediterranean and it is characterized by a complex orographic pattern.



The heterogeneous climate, morphology and geological substrata of the island determine a high rate of endemism (Fois et al., 2017) and a wide variety of Potential Natural Vegetation *sensu* Farris et al. (2010), described in detail by Bacchetta et al. (2009).

Methods

Monthly average temperatures (minimum, maximum and mean) and precipitations were originally interpolated to produce the bioclimatic map of Sardinia (Canu et al., 2015). The data at 40 m resolution were created using high quality meteorological data from 203 rain gauges and 68 temperature gauges of the regional climatic database of the Weather and Climate Department (ARPA Sardegna) for the time period 1971–2000. Monthly average temperature and precipitation were interpolated by Regression Kriging, combining a Multiple Linear Regression with an Ordinary Kriging of the regression residuals. Factors such as latitude, longitude, altitude, sea distance and local topography were considered as independent geographic variables to account for topographic effects (Canu et al., 2015).

Starting from this baseline data, we calculated bioclimatic variables at 40 m resolution using the C++ code included in the System for Automated Geoscientific Analysis (SAGA) version 7.5.0 (Conrad et al., 2015). The free and open-source Geographical Information System SAGA under the GNU public license was specifically developed for regional climate and environmental modelling applications (Conrad et al., 2015).

In order to evaluate the extreme or limiting environmental factors, we defined the quarterly parameters by following the definitions provided by WorldClim (Hijmans et al., 2005) and ANUCLIM (Xu and Hutchinson, 2013).

Three types of bioclimatic variables were evaluated (Table 1): variables related to temperature (BIO01-BIO07 and BIO10-BIO11); variables related to rainfall (BIO12-BIO17); variables related to both temperature and rainfall (BIO08-BIO09 and BIO18-BIO19).

The calculation of bioclimatic variables related to temperature was performed using average monthly maximum, minimum and mean temperatures. Cell-by-cell calculations of bioclimatic variables related to rainfall were conducted using monthly average precipitation.

Some descriptive statistics such as mean, minimum and maximum were used to describe the results. To assess the rate of dispersion of data, for each bioclimatic variable we calculated the coefficient of variation (in percentage).

Comparisons with the WorldClim 2 bioclimatic variables

Comparison amongst WorldClim 2 and the new high-resolution bioclimatic variables of Sardinia was possible because they were based on the same temporal range: 1970–2000 for WorldClim 2 (Fick and Hijmans, 2017) and 1971–2000 for Sardinia (Canu et al., 2015).

Table 1. Bioclimatic variables, types, descriptions and units of the new high-resolution bioclimatic variables of Sardinia (Italy).

Filename	Type of variable	Description	Unit
BIO01.tif	Temperature-related variable	Annual Mean Temperature	°C
BIO02.tif	Temperature-related variable	Mean Diurnal Range (Mean of monthly (max temp - min temp))	°C
BIO03.tif	Temperature-related variable	Isothermality (BIO02/BIO07) (x 100)	Index
BIO04.tif	Temperature-related variable	Temperature Seasonality (standard deviation x 100)	Index
BIO05.tif	Temperature-related variable	Maximum Temperature of Warmest Month	°C
BIO06.tif	Temperature-related variable	Minimum Temperature of Coldest Month	°C
BIO07.tif	Temperature-related variable	Temperature Annual Range (BIO05-BIO06)	°C
BIO08.tif	Temperature-related and rainfall-related variable	Mean Temperature of Wettest Quarter	°C
BIO09.tif	Temperature-related and rainfall-related variable	Mean Temperature of Driest Quarter	°C
BIO10.tif	Temperature-related variable	Mean Temperature of Warmest Quarter	°C
BIO11.tif	Temperature-related variable	Mean Temperature of Coldest Quarter	°C
BIO12.tif	Rainfall-related variable	Annual Precipitation	mm
BIO13.tif	Rainfall-related variable	Precipitation of Wettest Month	mm
BIO14.tif	Rainfall-related variable	Precipitation of Driest Month	mm
BIO15.tif	Rainfall-related variable	Precipitation Seasonality (Coefficient of Variation)	Index

BIO16.tif	Rainfall-related variable	Precipitation of Wettest Quarter	mm
BIO17.tif	Rainfall-related variable	Precipitation of Driest Quarter	mm
BIO18.tif	Temperature-related and rainfall-related variable	Precipitation of Warmest Quarter	mm
BIO19.tif	Temperature-related and rainfall-related variable	Precipitation of Coldest Quarter	mm

Variables were compared following three main steps: (i) at first, we resampled WorldClim 2 data to the resolution of our variables (40 m) using the nearest neighbour method; (ii) then we checked errors in raster alignment and adjusted alignment using the nearest neighbour method and one of our raster as snap raster, to ensure all cells were properly aligned; (iii) finally we performed the quantitative comparison analyses. To assess if and where the two datasets are different, we calculated Spearman's correlation coefficient (ρ), the root mean square error (RMSE) and the normalized root mean square error by the mean (nRMSE) between all pairs of variables.

For each variable we mapped the spatial distribution of errors, by means of the difference between the two datasets, namely the new high spatial resolution dataset minus WorldClim 2.

All data manipulation and geographic analyses were performed with R (R Core Team, 2020), using raster (Hijmans, 2020) and gdalUtils (Greenberg and Mattiuzzi, 2020) packages. Metadata of rasters were added using ArcGIS software by Esri.

Results

We generated a high-resolution suite of 19 bioclimatic variables of Sardinia: all rasters are provided at roughly 1.69 arcsec (40 m cell size) resolution and in the WGS84 geographic coordinate system (EPSG 4326). GeoTIFF rasters of all 19 bioclimatic variables of Sardinia (Italy) are included in Annex I. Metadata files include file name, thumbnail, tags and description for all rasters.

Table 2. Mean, minimum, maximum and coefficient of variation (CV) values for each of the new high spatial resolution bioclimatic variables of Sardinia (Italy).

Variable	**Name of variable**	**Mean**	**Minimum**	**Maximum**	**CV (%)**
BIO01	Annual Mean Temperature	15.44	8.61	18.12	8.51
BIO02	Mean Diurnal Range (Mean of monthly (max temp - min temp))	9.69	4.06	13.46	13.64
BIO03	Isothermality (BIO02/BIO07) (x 100)	37.08	22.71	43.69	7.99
BIO04	Temperature Seasonality (standard deviation x 100)	571.91	461.06	655.64	4.98
BIO05	Maximum Temperature of Warmest Month	30.59	24.16	33.96	4.40
BIO06	Minimum Temperature of Coldest Month	4.57	-2.13	9.79	33.78
BIO07	Temperature Annual Range (BIO05-BIO06)	26.02	17.45	31.16	7.25
BIO08	Mean Temperature of Wettest Quarter	11.85	3.22	17.36	22.54
BIO09	Mean Temperature of Driest Quarter	23.54	17.43	26.16	5.27
BIO10	Mean Temperature of Warmest Quarter	24.30	18.90	26.16	4.02
BIO11	Mean Temperature of Coldest Quarter	8.60	0.98	12.29	17.98
BIO12	Annual Precipitation	690.58	418.54	1376.38	19.22
BIO13	Precipitation of Wettest Month	100.73	57.82	209.09	19.23
BIO14	Precipitation of Driest Month	8.21	0.46	22.32	44.88
BIO15	Precipitation Seasonality (Coefficient of Variation)	50.23	39.75	60.48	7.28

BIO16	Precipitation of Wettest Quarter	288.68	164.44	582.96	19.26
BIO17	Precipitation of Driest Quarter	36.71	11.24	82.44	32.89
BIO18	Precipitation of Warmest Quarter	39.97	11.24	90.93	33.77
BIO19	Precipitation of Coldest Quarter	222.29	113.18	495.01	23.73

The coefficient of variation (in%) of bioclimatic variables was lower in the temperature-related variables (mean CV = 11.39), higher for the precipitation-related ones (mean CV = 23.79) and intermediate in the variables related to both temperature and precipitation (mean CV = 21.33). In particular, the maximum coefficient of variation of the temperature ones amounts to 33.78 for the Minimum Temperature of Coldest Month (BIO06), with BIO10 having the minimum value (CV = 4.02, Table 2).

Within rainfall-related bioclimatic variables, the Precipitation of Driest Month (BIO14) had a higher variation (CV = 44.88) than other variables, representing the highest level of variability in all the dataset. Regarding variables related to both temperature and precipitation the Precipitation of Warmest Quarter (BIO18) showed the highest level of variation (CV = 33.77), while the Mean Temperature of Driest Quarter (BIO09) showed the lowest one (CV = 5.27).

The spatial distribution of the calculated bioclimatic variables is shown according to the three groups temperature (Fig. 2), rainfall (Fig. 3), temperature and rainfall related variables (Fig. 4).

Fig. 2. Temperature-related bioclimatic variables (BIO01-BIO07 and BIO10-BIO11) of Sardinia (Italy).

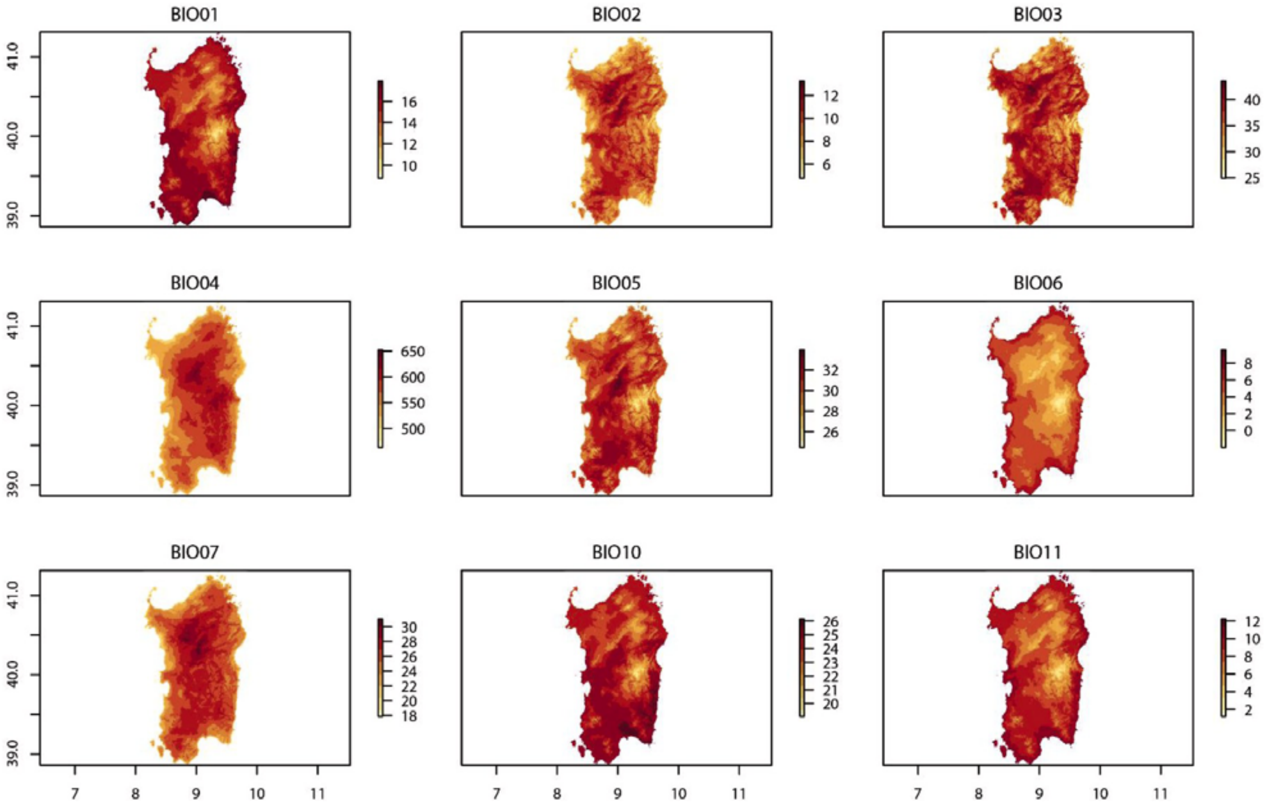


Fig. 3. Rainfall-related bioclimatic variables (BIO12-BIO17) of Sardinia (Italy).

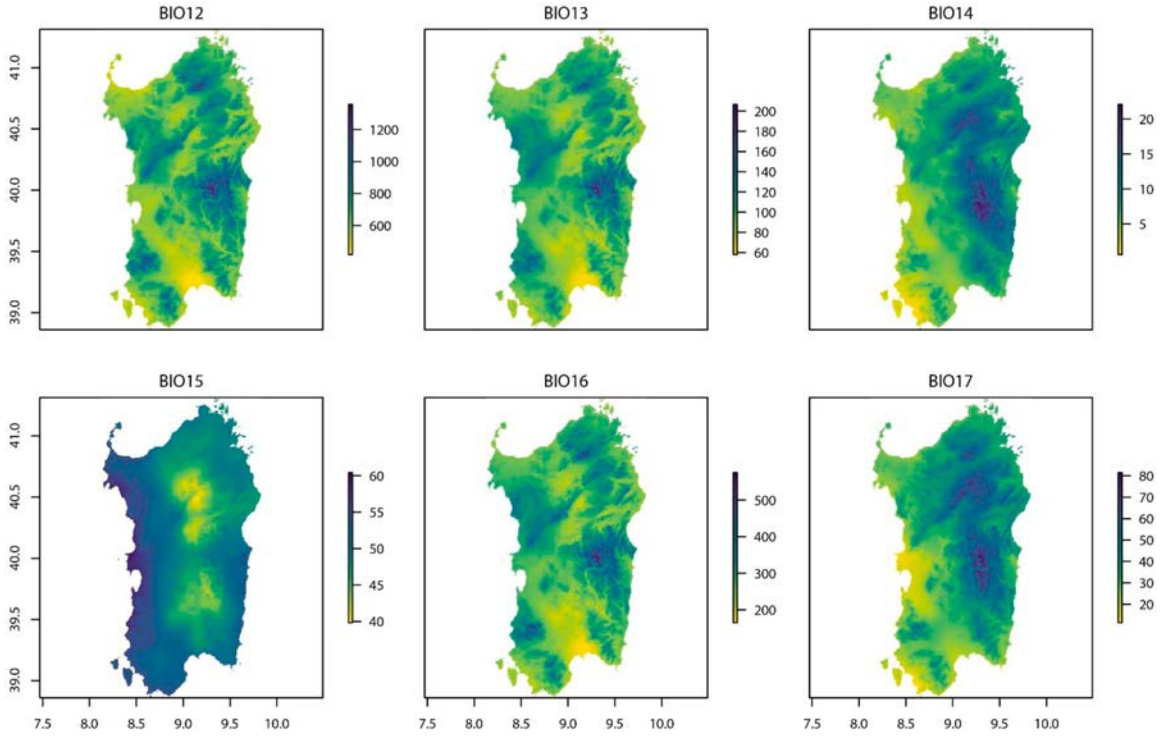
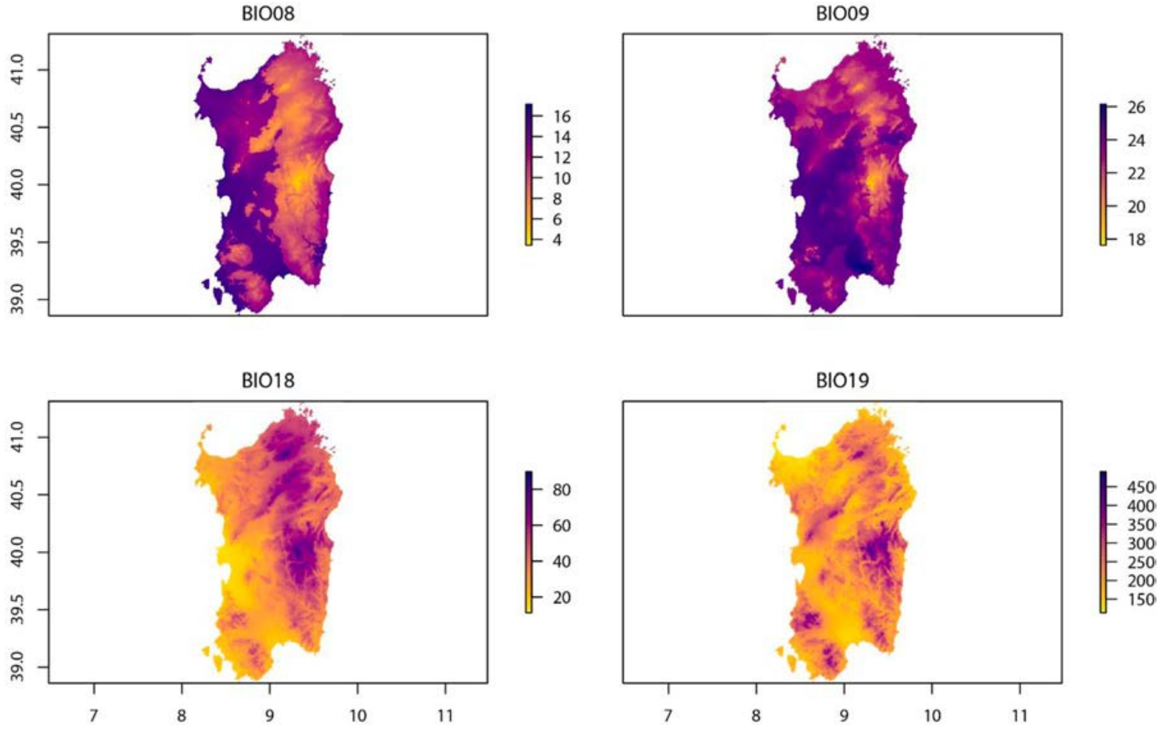


Fig. 4. Bioclimatic variables of Sardinia (Italy) related to both temperature and precipitation (BIO08-BIO09 and BIO18-BIO19).



Comparisons with the WorldClim 2 bioclimatic variables

The comparison of our high-resolution dataset vs. WorldClim 2 in terms of Spearman's correlation coefficient (ρ) showed significant linear correlations (all p-values < 0.001) for all the 19 bioclimatic variables (Table 3) with the highest correlation amongst BIO11 values and the lowest for BIO15.

The normalized root mean square error (nRMSE) revealed relevant differences (Table 3), in particular for the bioclimatic variables related to rainfall showing a higher discrepancy (mean RMSE = 39.79; mean nRMSE = 0.21) compared to the temperature ones (mean RMSE = 4.81; mean nRMSE = 0.11).

All bioclimatic variables related to temperature showed a high correlation (ρ > 0.70) with WorldClim 2 (Table 3).

Rainfall-related bioclimatic variables were less strongly correlated with WorldClim 2 than temperature-related ones; Seasonality trend of precipitation (BIO15) was poorly correlated (ρ = 0.42).

Table 3: Pairwise comparison of the new high-resolution bioclimatic variables and WorldClim 2 bioclimatic variables in terms of Spearman's correlation coefficient (ρ) and normalized root mean square error (nRMSE).

Variable	Name of variable	Spearman's rho	RMSE	nRMSE
BIO01	Annual Mean Temperature	0.94	0.49	0.03
BIO02	Mean Diurnal Range (Mean of monthly (max temp - min temp)	0.76	1.00	0.10
BIO03	Isothermality (BIO02/BIO07) (x 100)	0.72	2.22	0.06
BIO04	Temperature Seasonality (standard deviation x 100)	0.76	31.10	0.05
BIO05	Maximum Temperature of Warmest Month	0.77	3.00	0.10
BIO06	Minimum Temperature of Coldest Month	0.93	2.04	0.45
BIO07	Temperature Annual Range (BIO05-BIO06)	0.79	1.47	0.06
BIO08	Mean Temperature of Wettest Quarter	0.77	1.90	0.16
BIO09	Mean Temperature of Driest Quarter	0.68	1.20	0.05
BIO10	Mean Temperature of Warmest Quarter	0.87	1.53	0.06
BIO11	Mean Temperature of Coldest Quarter	0.96	0.41	0.05
BIO12	Annual Precipitation	0.60	137.00	0.20
BIO13	Precipitation of Wettest Month	0.61	19.57	0.19
BIO14	Precipitation of Driest Month	0.75	2.41	0.29
BIO15	Precipitation Seasonality (Coefficient of Variation)	0.42	4.90	0.10
BIO16	Precipitation of Wettest Quarter	0.56	64.76	0.22
BIO17	Precipitation of Driest Quarter	0.81	10.09	0.27
BIO18	Precipitation of Warmest Quarter	0.55	21.75	0.54
BIO19	Precipitation of Coldest Quarter	0.54	52.59	0.24

With regard to bioclimatic variables related to both temperature and precipitation, mean Temperature of Wettest and Driest quarters showed good correlations with corresponded WorldClim 2 bioclimatic variables (ρ > 0.60). On the contrary, the precipitation of the driest and warmest quarters highlighted a low correlation (ρ < 0.60).

The spatial distribution of errors, calculated as the difference between the two datasets i.e. new high-resolution dataset minus WorldClim2, showed the heterogeneous distribution of the spatial discrepancies of the variables (Figs. 5–7) and a specific pattern according to the different bioclimatic variable analysed. In these figures red colours indicate areas where a given variable was overestimated by WorldClim 2, blue colours the underestimated ones.

For example, the spatial distributions of the differences for the Annual Mean Temperature (BIO01) highlighted lower values modelled by WorldClim 2 compared to our dataset in the mountain areas, with differences of more than 2 ◦C (Fig. 5). Maximum/minimum temperatures (BIO05/06/10) are generally underestimated by WorldClim 2, with peaks up to 7 ◦C for the maximum temperature of the warmest month (BIO05). Accordingly, the annual range of extreme temperature conditions (BIO07) was underestimated by WorldClim 2 in the internal and mountain areas of the island, being overestimated in coastal areas in the north, south and eastern coast (Fig. 5).

Regarding rainfall-related bioclimatic variables, a markedly different spatial distribution was observed for the Annual precipitation (BIO12), showing a gradient moving from north-west to south-east: in the NW areas WorldClim 2 overestimated, while the SE areas are underestimated, with strong differences (higher than 500 mm) (Fig. 6). Precipitation Seasonality (Coefficient of Variation, BIO15) is strongly overestimated by WorldClim 2 in the internal and mountain areas and slightly underestimated along the S-E coast. The Precipitation of Driest Quarter (BIO17) is generally overestimated, in particular in the NW zones. An asymmetry similar to the one observed for BIO12 was detected for the Mean Temperature of Wettest Quarter (BIO08), with underestimated values in the western part of the island and overestimated ones in the eastern ones (Fig. 7). Furthermore, a general overestimation was observed for the Precipitation of Warmest Quarter (BIO18) in all western areas of the island, from north to south, up to 40 mm (Fig. 7).

Fig. 5. Spatial distribution of the differences between all pairs of the new high spatial resolution dataset and WorldClim 2 temperature-related bioclimatic variables (BIO01-BIO07 and BIO10-BIO11). Red colours indicate overestimated areas by WorldClim 2, blue colours the underestimated ones.

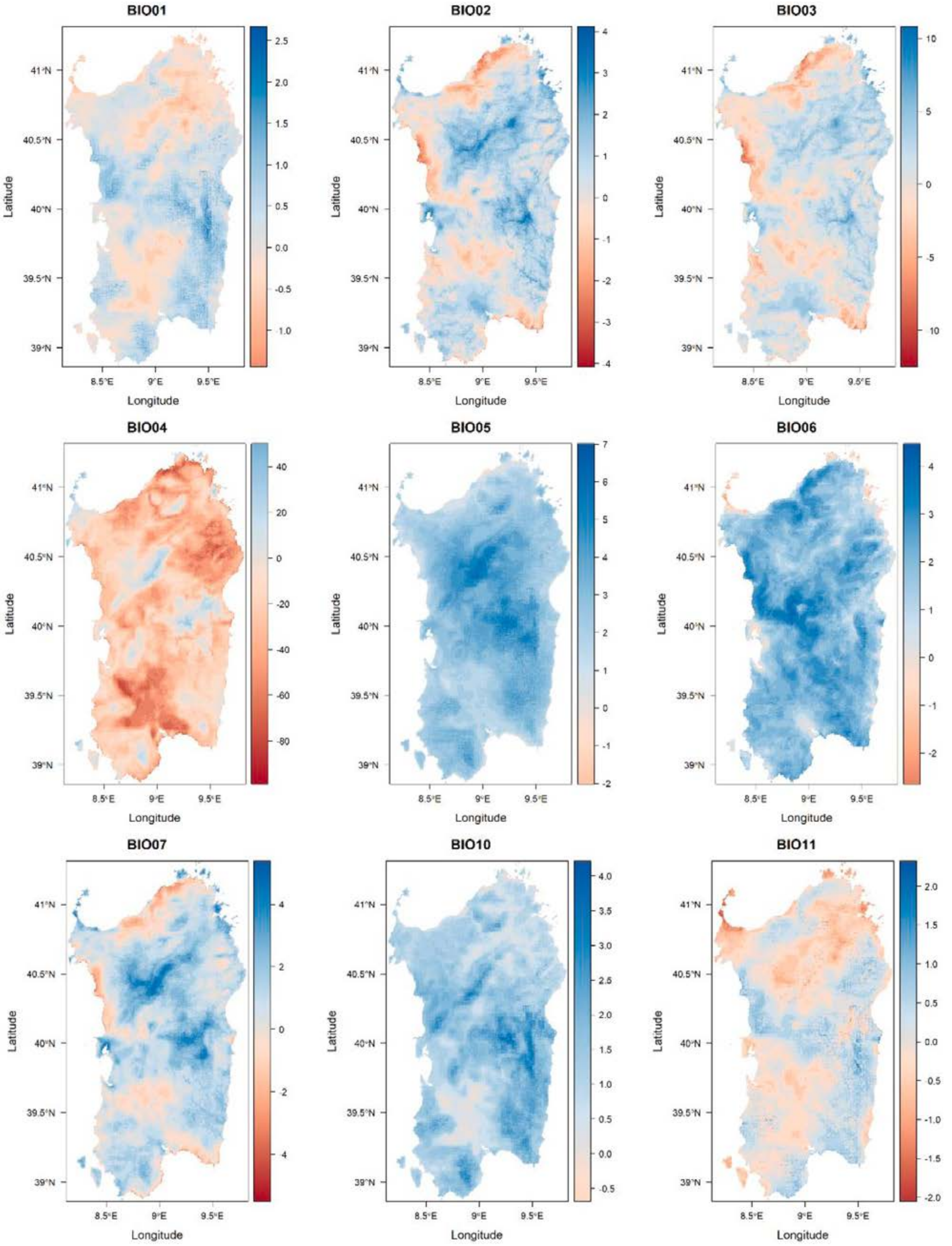


Fig. 6. Spatial distribution of the differences between all pairs of the new high spatial resolution dataset and WorldClim 2 rainfall-related bioclimatic variables (BIO12-BIO17). Red colours indicate overestimated areas by WorldClim 2, blue colours the underestimated ones.

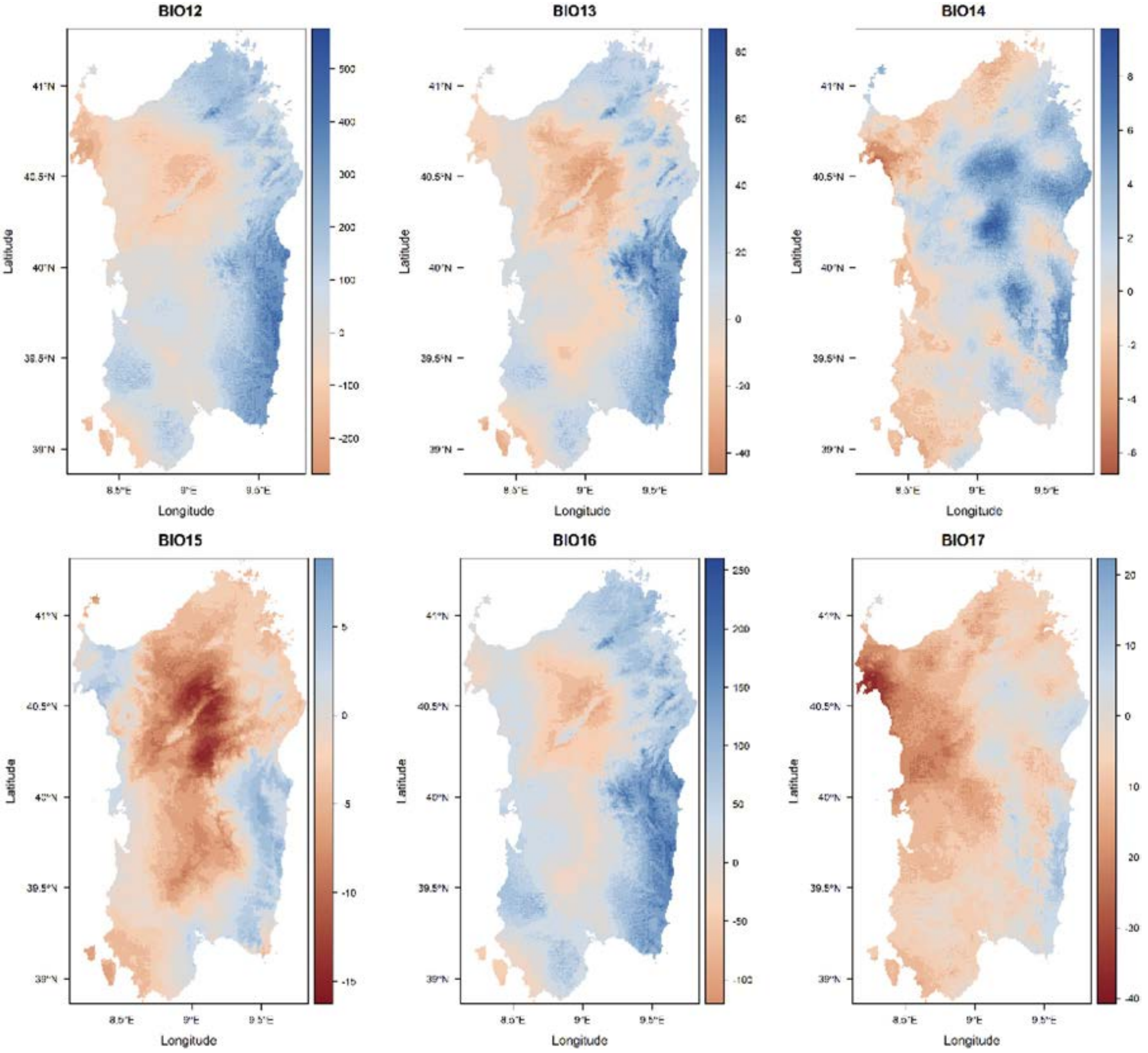
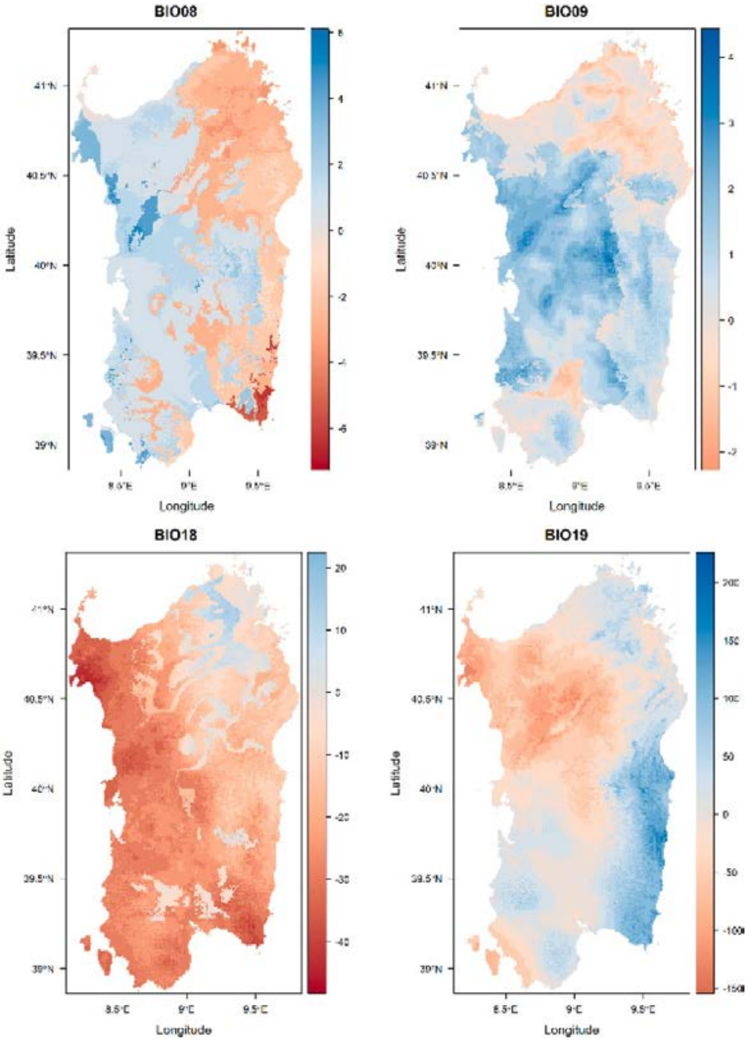


Fig. 7. Spatial distribution of the differences between all pairs of the new high spatial resolution dataset and WorldClim 2 bioclimatic variables related to both temperature and precipitation (BIO08-BIO09 and BIO18-BIO19). Red colours indicate overestimated areas by WorldClim 2, blue colours the underestimated ones.



Discussions

Bioclimatic variables are fundamental for understanding and modelling the ecological processes and the distribution of biodiversity of earth (Jennings and Harris, 2017; Rivas-Martínez et al., 2011). Nevertheless, our study demonstrated that comparing the global dataset to a high spatial resolution one, revealed that we should pay attention on the accuracy of coarse spatial resolutions data, especially in areas of high heterogeneity where weather stations are few and sparsely distributed (Sandoval et al., 2020), like the Mediterranean area.

We observed that the discrepancies existing amongst our high spatial resolution dataset and WorldClim 2 are evident and each bioclimatic variable behaved in a different way: we did not detect a general over/ underestimation pattern (or trend) of the bioclimatic variables, but we rather observed variable-specific patterns mainly linked to the local orographic conditions and to the direction of the dominant winds driving weather perturbations.

The new high spatial resolution dataset compared to WorldClim 2 showed that the larger discrepancies were spotted in the bioclimatic variables related to precipitation. Those inconsistencies are not evenly distributed in the territory: the greater differences between the two datasets correspond to the areas characterized by complex orographic systems. Moreover, since the dominant air mass perturbations in Sardinia come from west and the rain shadow effect is not considered in the model, the global dataset, probably limited by an uneven distribution of meteorological stations, strongly underestimated the annual precipitation in the eastern zones and overestimated the annual and the summer precipitations (i.e., driest and warmest quarter) in the western zones (up to 40 mm). These discrepancies, if applied, for example, to vascular plant species distribution models, can cause biases in the comprehension of the distribution of thermo-xerophilous species particularly (or exclusively) abundant in the western coast of the island (like *Chamaerops humilis, Polygala rupestris, Viola arborescens*, amongst the others (Biondi et al., 2001), and, contrarily, of mesophilous species of non-Mediterranean origin that can surprisingly colonize low elevation (down to the sea level) in the eastern coast, like *Ostrya carpinifolia* (Bacchetta et al., 2004a) and *Taxus baccata* (Farris et al., 2012).

Similar limitations of the WorldClim spatial dataset accuracy, especially in isolated mountainous areas, were indicated for the first and second versions of this dataset by Hijmans et al. (2005) and Fick and Hijmans (2017), respectively. In particular for Italy, Pesaresi et al. (2014, 2017) highlighted that the lower accuracy in precipitation spatialization of WorldClim, could be explained by the scarcity of meteorological stations density respect to the topographic complexity and heterogeneity of the Italian territories. Bedia et al. (2013) highlighted that the discrepancy in precipitation-related variables between local and WorldClim datasets could determine a lack of robustness of the species distribution models leading, for example, to artifacts in the projections of climate change scenarios at regional or local scales. This can eventually compromise the successful use of models in biodiversity conservation and management actions.

The maximum/minimum temperature of the warmest/coldest month are underestimated, i.e. compared to the high spatial resolution dataset, the global dataset generally models cooler summer maximum (up to 7 ◦C) and colder winter minimum (more than 4 ◦C). Accordingly, the coarse scale dataset models a smaller temperature annual range in the mountains and a wider one on the coasts, underrating the continentality values in the internal areas and exaggerating it on the coasts. These discrepancies are particularly important in the Mediterranean climate, since the seasonal distribution of rainfall and the extreme temperatures determine the limits for species survival. Since Sardinia shows many plains and depressed areas in the internal parts of the island, often surrounded by hills or mountains, it is of crucial importance to discriminate areas with higher temperature annual range (i.e., more continental) from those characterized by a smaller temperature annual range (i.e., more oceanic). In the more continental areas, no matter the altitude above the sea level, species like *Arbutus unedo* L., *Laurus nobilis* L. and *Myrtus communis* L., amongst the others, are very rare if not completely absent (Bacchetta et al., 2007; Farris et al 2007a), whereas species more tolerant to continentality like *Quercus* gr. *pubescens* are relatively abundant even at lower elevation (Bacchetta et al., 2004b).

The observed differences related to the precipitation of driest/ warmest periods also influence the delimitation between Mediterranean and Temperate macro-bioclimates (Rivas-Martínez et al., 2011). The definition of this ecological boundary can be particularly important in a Mediterranean island where zones with a Temperate bioclimate are crucial for the conservation of small, isolated populations of plant species of boreal-temperate origin, often living at their rear edge and therefore with important conservation concerns such as *Daphne laureola, Isopyrum thalictroides, Lotus* *alpinus* and *Sanicula europaea* (Farris et al., 2018; Rosati et al., 2020) but also characterized by a high evolutionary potential (Hampe and Petit, 2005). In the same way, those Temperate areas in a Mediterranean context host non-sclerophyllous plant communities like woods with *Quercus* gr. *pubescens, Ostrya carpinifolia, Taxus baccata* and *Ilex aquifolium,* as shrubs with *Sorbus torminalis, Malus pumila, Pyrus communis* and *Juniperus nana* (Bacchetta et al., 2009; Farris et al., 2012) and perennial pasturelands with *Anthoxanthum odoratum* and *Cynosurus cristatus* (Farris et al., 2013), identified as habitats of European concern.

Treating the bioclimatic indices individually helps us to understand which are more reliable, indicating the critical issues to be faced when one is forced to use global datasets such as WorldClim 2 in a Mediterranean territory. According to our results, in Sardinia the most consistent indices regard temperature, with the Annual Mean Temperature being the most reliable one. Yet, the spatial distribution of the variables highlights that mountain areas are difficult to model; in fact, even the annual mean temperature shows some variations.

On the contrary, WorldClim 2 does not seem to be reliable on the precipitation indices, influencing the combined indices too: amongst the least performing variables we can identify Minimum Temperature of Coldest Month (BIO06) and the Precipitation of Warmest Quarter (BIO18).

Given the high discrepancy identified in several sectors of our study area, we recommend being cautious in making ecological assessments based on bioclimatic variables derived from global data with coarse spatial resolutions. The high degree of variability of the new high-resolution bioclimatic variables of the island underlined the need to use fine spatial resolution data to capture the ecological response in physiographically complex landscapes (Hijmans et al., 2005).

Conclusions

In this paper we present and make available the first high spatial resolution dataset for the second largest island in the Mediterranean (Sardinia, Italy), including the 19 bioclimatic variables proposed in WorldClim and widely used for ecological studies (e.g., Iannella et al., 2019; Sýkora et al., 2017).

Increasing the availability of high spatial resolution data to improve ecological understanding of variation at finer scales is extremely important, especially in the Mediterranean regions where past geographical and climatic changes and current environmental heterogeneities have determined high levels of biodiversity and biogeographical complexity (Medail, 2017; Thompson, 2020). Tree species composition and richness in Europe is shaped and strongly influenced by both historical and environmental conditions, in particular climate (Svenning and Skov, 2005): high levels of divergence have been highlighted, particularly on islands, which have been attributed to the combined effects of climatic changes, current ecological conditions, and anthropogenic factors, that have originated a long history of population isolation (Gonza´lez-Martínez et al., 2010).

These new data will support a new generation of research studies in a broad array of ecological applications at a much finer scale than previously possible. This sharpening of analysis is particularly urgent in those areas considered as climate-change hotspots (Giorgi, 2006), like the Mediterranean basin (Giorgi and Lionello, 2008): in southern European mountains boreo-temperate species are suspected to undergo a serious decline in future decades, as a consequence of the climatic change (Erschbamer et al., 2009; Normand et al., 2007; Stanisci et al., 2005).

Coarse-scale data is certainly useful for studying patterns on a global scale, but to model in order to obtain reliable results for planning conservation actions and biodiversity management, we need data with good spatial resolution (Sandoval et al., 2020), showing the variability of our territories.

Author contributions

EB designed the study, developed the new high-resolution biologically meaningful variables, and drafted a first version of the manuscript. MM, LR, MF, EF and SC designed the study and helped draft the manuscript. All authors revised the manuscript and approved the final version of the manuscript for publication.

Subchapter 1.2: SWOF map, plant community and structure

Key role of small woodlots outside forest in a Mediterranean fragmented landscape

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Abstract

Even if the ecological role of trees outside forests (TOF) is acknowledged, few studies provided an exhaustive census and defined their principal characteristics, considering not only the patch size but also the effect of the dominant land-use matrix (natural and semi-natural, agricultural, and artificial areas). Using a multiphase sampling design, we (i) provided a map showing the spatial distribution of small woodlots outside forest (SWOF), a type of TOF, located in different land-use matrices in Sardinia, a Mediterranean hotspot of biodiversity conservation. Moreover, we (ii) assessed the differences in native and alien plant species richness and composition among SWOFs located in different land-use matrices; (iii) identified key native and alien plant species related to each land-use matrix; (iv) assessed the structural attribute differences among SWOFs surrounded by different land-use matrices; (v) explored the effect of environmental factors and structural attribute on native and alien contingent of the plant community.

Overall, our findings indicated that coverage and spatial distribution of Mediterranean SWOFs were not negligible both in natural and human-impacted land-use.

The percentage of cover increased following the gradient of land-use intensification, reaching the highest value in urban zones, while the average patch size decreased along the same gradient, reaching the smallest value in the urban matrix. Our results support the evidence that a good level of naturality is present in human-altered matrix and particularly in agricultural landscapes. The level of naturality of agricultural landscapes, both in terms of vascular plant community composition and stand-structure variation, can be considered as an opportunity for natural conversion to forest land and consequent rewilding ecosystems with important benefits for biodiversity and ecosystem services.

Notably, the contribution of alien species to the overall diversity was low both in natural and human-impacted land-uses. We found dissimilarities in terms of species richness and composition between land-use types; different land-use matrices were characterized by alien and native indicator species.

Alien and native taxa showed different responses to environmental drivers: alien contingents were not affected by landscape variables, while the native ones were influenced by all investigated environmental drivers and by the SWOF's structural complexity.

These findings suggested the chance to rely on SWOFs to build up and extend the green infrastructure network in both natural and human-impacted land-uses. In this framework, monitoring small woodlots outside forest should be a priority for both urban planners and conservationists.

Keywords: spatial distribution map, indicator taxa, native plants, alien species, structural attribute

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Introduction

The intensification and expansion of human-activities over the last decades are increasing the need to develop new strategies for tree resources and woodlands managing to meet the challenge of global change, such as habitat degradation and fragmentation (Franklin et al., 2002; Larsen and Nielsen, 2007; O'Hara and Ramage, 2013; Wagner et al., 2014). Habitat degradation and fragmentation are considered as one of the major threats to biodiversity worldwide (Balmford et al., 2003).

Different approaches and criteria have been proposed to guide the conservation and management of tree resources (e.g., McElhinny et al., 2005; Sabatini et al., 2016). Many of these proposals aim to maintain native species and communities, as well as the heterogeneity and complexity of structures to promote greater ecosystem functioning, stability (Angelini et al., 2015; Caviedes and Ibarra, 2017), green infrastructure and corridors in human-shaped areas (Benedict et al., 2012; Clergeau and Blanc, 2013). In this framework, trees outside forest (hereby TOF) play a crucial role in promoting the conservation of biodiversity and reduce the effects of habitat fragmentation (Fischer et al., 2010; Gibbons et al., 2008), representing ecological corridors, as well as habitats for numerous animal and plant species (Bellefontaine et al., 2001). Examples of TOF are scattered trees, small woodlots (hereby SWOF), linear arboreal elements, and green belts outside the forested matrix, already widely acknowledged as important multifunctional resources (Manning et al., 2006) thanks to their capacity of providing fundamental ecosystem services (Marchetti et al., 2018a, 2018b; Paletto et al., 2006).

Monitoring the spatial distribution, ecological characteristics, and attributes of TOF may represent a key strategy to promote sustainable management and enhance conservation policy of these well-recognized important resources (Angelini et al., 2015; FAO, 2010; Marchetti et al., 2018a).

In addition, being considered as a prominent feature both for agricultural areas and built-up landscape (Bellefontaine et al., 2001; de Foresta et al., 2013; Gutzwiller, 2002; Konijnendijk et al., 2005), notably TOF provide supporting, provisioning and regulating ecosystem services (Lumsden and Bennett, 2005; Manning et al., 2006; Plieninger et al., 2004), as well as important social, aesthetic, and economic benefits (McDonnell et al., 2009; Tyrväinen et al., 2005).

On the other hand, a proportion of these resources is represented by ornamental TOF in urban areas, with species ranging from native ones, to naturalized or recently introduced ones (Rossi et al., 2016), where exotic ornamental species are frequently planted near houses and buildings (Pauleit et al., 2005). In this context, transportation infrastructure such as road and railway could facilitate the spread of alien species (Bacaro et al., 2015; Benedetti and Morelli, 2017) to more natural areas, and TOF themselves could then act as corridors not only for native species, but also for these newly introduced species (Rossi et al., 2016).

During the last two decades, many research activities are developing and implementing a variety of methodological approaches to estimate and infer TOF population parameters at large spatial scale (Corona, 2016; de Foresta et al., 2013; Kleinn, 2000). Most of these forest inventory methodologies, performed over large scales, adopt classical two-phase sampling schemes (pre-fieldwork phase and fieldwork phase), some adopt one-phase inventories and a few others (e.g., the Italian NFI) introduce a third phase to further reduce survey costs and time (Baffetta et al., 2011; Corona, 2016; Fattorini et al., 2006). For example, robust large-scale TOF surveys have been performed in Kenya (Holmgren et al., 1994), India (Tewari et al., 2014), Sweden (Fridman et al., 2014), France (Bellefontaine et al., 2002), Switzerland (Price et al., 2017), and Italy (Sallustio et al., 2018).

Nonetheless, in national forest inventories TOF are generally absent (de Foresta et al., 2013; Kleinn, 2000) or the information of their characteristics are inconsistent, considering for example their spatial or temporal pattern and their coverage on different land-use types (Price et al., 2017; Schnell et al., 2015). A general integration into national tree inventories appears to be necessary to support landscape planning in many countries (Schnell et al., 2015). Similarly, data and studies at local scale appear to be limited (i.e., for Italy at regional level, Corona and Fattorini, 2006; Fattorini et al., 2016; Marchetti et al., 2018a; Paletto et al., 2006), mainly due to the lack of historical data and high-cost sampling efforts (Novotný et al., 2017).

This lack underlines the need to implement the monitoring of these important resources, especially in Mediterranean areas recognized as the most important hotspots for biodiversity conservation (Marignani et al., 2017a, 2017b; Médail, 2017), but strongly affected by human activities (Marignani et al., 2017b), and considered particularly sensitive to the growing threats of climate change (Giorgi and Lionello, 2008), and biological invasions.

Indeed, for these patches, in addition to land-use changes, another threat closely linked to landscape dynamics, could just be represented by biological invasions (Mooney and Hobbs, 2000; With, 2004, 2002), particularly in those surrounded by human-shaped environments (Vilà and Ibáñez, 2011), where ornamental species are frequently planted near houses and buildings (Pauleit et al., 2005).

Given the need for more comprehensive knowledge on spatial distribution and attributes of these important resources, we aimed to fill this gap, focusing in particular on SWOFs, still too little studied in our area. More specifically, our main objectives were to (i) map the spatial distribution of SWOFs along a gradient of land-use intensification; (ii) assess differences in native and alien species richness and composition among SWOFs located in different land-use matrices; (iii) identify key native and alien taxa related to each land-use matrix; (iv) assess the structural attribute differences among SWOFs surrounded by different land-use matrices; (v) explore the effect of environmental factors and structural attribute on native and alien contingent of the plant community.

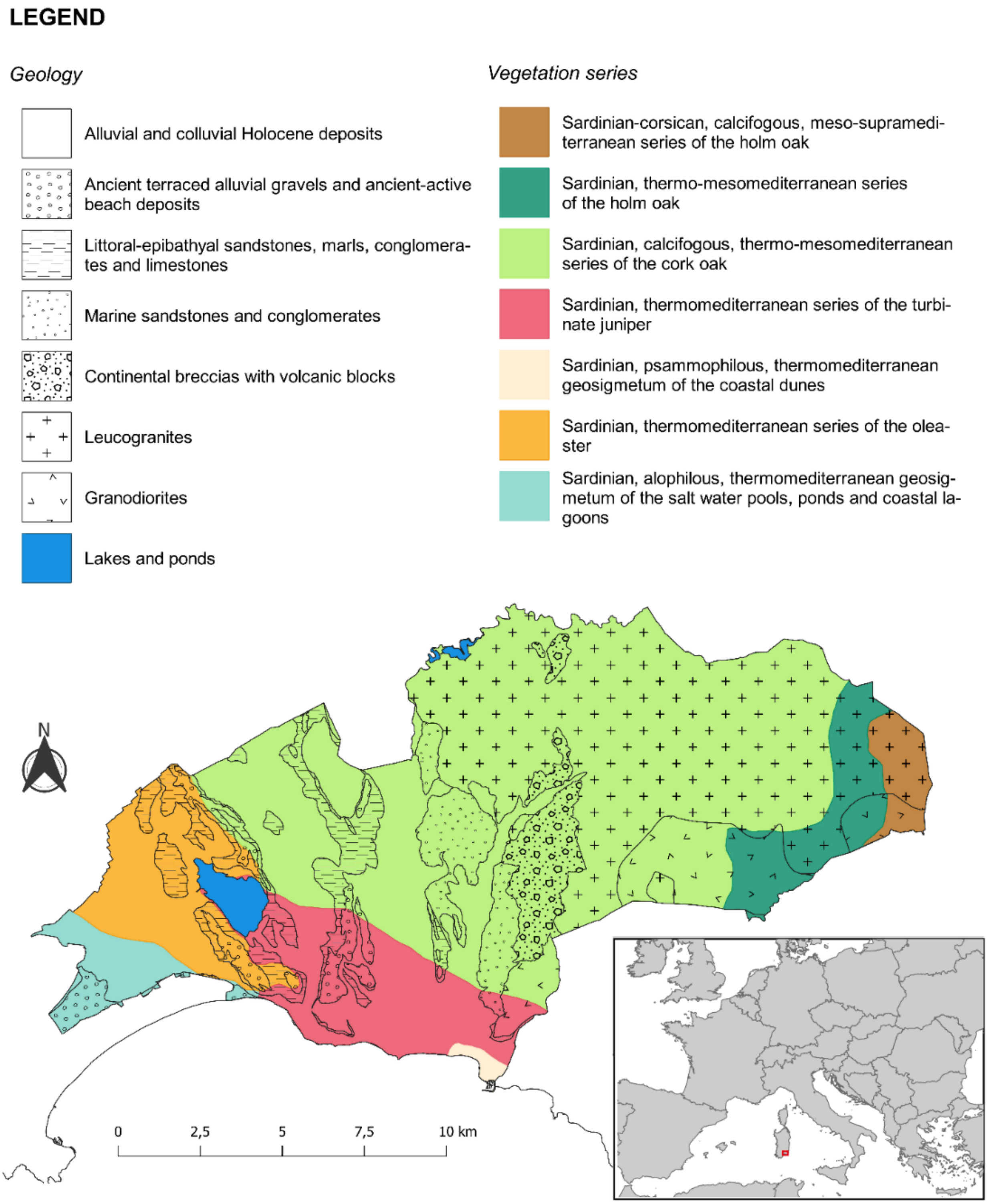
Considering that TOF are typically considered as large components of agroforestry, urban and rural forestry, and are linked to the activities of a large range of stakeholders (farmers, pastoralists, institutions linked to agriculture and rural development, people living in settlements and cities and institutions linked to urban management and development) (de Foresta et al., 2013; Pauleit et al., 2005), we expected to observe: (i) an increment of small woodlots outside forest in areas at the intermediate and high level of disturbance; (ii) a higher presence of alien plants in human-impacted land-use than in natural and semi-natural areas, due to planting practices adopted by people and land managers; (iii) significantly more heterogeneous structural attributes in unmanaged areas, compared to managed areas; (iv) more significant relationships among environmental filters and native contingent of the plant community than between environmental variables and alien contingent.

Materials and methods

Study area

The study was conducted in an area covering approximately 18,300 ha located in the eastern sector of the Metropolitan City of Cagliari (southern Sardinia, Italy), a medium-sized functional urban area characterized by three levels of human-impacted land-use and fragmentation degrees from natural areas to coastline zones (Palumbo et al., 2020). Part of the selected area is included in three Sites of Community Importance (ITB043055, ITB041106, ITB040022), and two regional parks ("Molentargius" and "Monti dei Sette Fratelli e Sarrabus"), partially overlapped (RAS, 2020). The area is characterized by two main physiographic units: a NE hill-mountain district composed by intrusive granitic litotypes of the Sardinian Variscan Basement (with a small part of sedimentary successions with clasts deriving from this outcrops) and by thin soils from neutral to acid (APAT, 2003; Barca et al., 2005); a SW plain-coastal district characterized by quaternary deposits (especially alluvial) and marine-transitional sedimentary successions, with thick and evolved soils from subacid to subalkaline (Fig. 1) (APAT, 2003; Barca et al., 2005; Carmignani et al., 2016). According to Canu et al. (2015), the investigated area is characterized by a Mediterranean pluviseasonal oceanic bioclimate and, in conformity with the variation in altitude (from sea level to 1018 m a.s.l.), a significant transition from lower thermomediterranean to the upper mesomediterranean thermotypes can be observed. As regards potential vegetation, the NE area presents a marked forest vocation (Fig. 1), with series mainly referring to thermo-mesomediterranean associations of evergreen, neutrophilous or acidophilous sclerophylls (all. *Fraxino orni-Quercion ilicis*, suball. *Clematido cirrhosae-Quercenion ilicis*) *sylvestris* (Bacchetta et al., 2009; Biondi et al., 2014; Mucina et al., 2016). They are represented by mature mesophilous woods, generally with *Quercus ilex* or *Q. suber* and shrub elements as *Erica arborea*, *Arbutus unedo*, *Phyllirea latifolia*, *Myrtus communis* and *Juniperus oxycedrus sylvestris* (Bacchetta et al., 2009; Biondi et al., 2014; Mucina et al., 2016). The high-shrub and pre-forest successions, distributed in the most thermo-xerophilous zone (all. *Juniperion turbinatae and all. Oleo sylvestris-Ceratonion siliquae*), represented by wild olive shrublands (with *Pistacia lentiscus*, *Euphorbia dendroides* and *Arisarum vulgare* in the herbaceous layer) and micro-woods of prostrate shrubs (dominated by *Juniperus turbinata* and *Olea europaea* var. *sylvestris* (Bacchetta et al., 2009; Biondi et al., 2014; Mucina et al., 2016). Due to the anthropic disturbance in the study area, these series often remain in the replacement stages of thermophilous shrublands, garrigues or perennial grasslands (Bacchetta et al., 2009). Near the coasts and the back-beach ponds halophilous and psammophilous plant communities are present (Bacchetta et al., 2009; Biondi et al., 2014; Camarda et al., 2016; Mucina et al., 2016).

Fig. 1. Map of geology and vegetation series of the study area. The arrangement of geological units follows the chronostratigraphic order (Carmignani et al. 2016). Vegetation series follows Bacchetta et al. (2009).



Small Woodlots Outside Forests (SWOFs) census and classification

A SWOF census was performed in the study area: to unambiguously identify the units, a SWOF unit was defined following the seven criteria proposed by de Foresta et al. (2013) and the definition provided by Italian National Forest Inventory (Baffetta et al., 2011; INFC, 2005), i.e., wooded land with an area between 0.05 to 0.5 hectares. The census was carried out via an intensive visual photointerpretation of aerial orthophotos with a cell resolution of 20 cm (RAS, 2016), a grain suitable to identify the smallest groups of trees. Following these criteria, each SWOF was manually delineated; then to validate the map we checked a random subsample (25%) in the field.

Once delimited the SWOFs, we overlaid a grid of 1 ha cell size on the study area to (i) classify the censused SWOFs according to the dominant land-use, and (ii) define the statistical population for the field sampling phase.

The grid was intersected with the first hierarchical level of regional land-use map (scale 1:25.000; RAS, 2008) in conformity with the European nomenclature system of the CORINE Land Cover (CLC; EEA, 2007): urban and artificial surfaces (cod. 1, CLC), amounting to 2318 ha; agricultural areas (cod. 2, CLC) corresponding to a surface of 6204 ha; natural and semi-natural areas (cod. 3, CLC) amounting to 8524 ha; wetlands (cod. 4, CLC) with a surface of 244 ha; water bodies (cod. 5, CLC) covering a surface of 58 ha.

Hence, each mapped SWOF was assigned to the dominant land-use (assign data by location): SWOFs in natural and semi-natural areas (NAT), SWOFs in agricultural areas (AGR), SWOFs in urban and artificial surfaces (URB), mixed SWOFs fell on two or more land-use (two or more grid cells).

To reduce the sampling effort and improve the homogeneity of the investigated population, from a total of 400 mapped and classified SWOFs, 42% of sites with a size of less than 0.1 hectares were excluded regardless of the class category, as well as the remaining SWOFs surrounded by a mixed land-use matrix (1.50%). A statistical population of $N = 201$ SWOF units (67 in URB, 70 in AGR, 64 in NAT), with a size ranging from 0.1 to 0.5 hectares, was defined. Then, a proportionally stratified random sampling was carried out to select a total of $n = 30$ SWOFs along the land-use gradient. As we have encountered several difficulties in accessing SWOFs located on private land, especially in urban ones, we sampled only 8 urban SWOFs. The remaining sites (up to 30) were equally assigned to the other two land-use types.

Field data collection

The 30 SWOFs randomly selected was subsampled with a systematic sampling design: SWOF centroids were used as middle points for linear transects, which were radiated from the centroid to the farthest corner of SWOFs boundaries. Five pseudo-replicate plots of 1 sqm per site, for a total of 150 plots in the whole study area, were systematically placed along each transect at a reciprocal distance proportional to the total length of the transect.

For each SWOF, the following attributes were considered: (i) occurrences and abundances of native and alien vascular plant, (ii) diameter at breast height (DBH).

Vascular plant occurrences and abundances were collected from April to August 2018. Abundances were recorded by applying a visual estimation of cover, expressed as a percentage, and then summed across the five plots in each transect per site. Every vertical layer (tree, shrub and herbaceous) were considered following the height thresholds proposed by Arrigoni (1996), Canullo and Falińska (2003), and Canullo et al. (2012): 1) tree layer, with woody forms growth higher than 5 m (e.g., *Eucaliptus, Pinus, Salix*); 2) shrub layer, with woody forms between 0.5 and 5 m (e.g., *Pistacia, Arbutus, Cistus*); 3) herbaceous layer, with herb forms from ground level up to 50 cm or suffrutices (e.g., *Oloptum*, *Trifolium*, *Ophioglossum*). We identified species using generalist floras

(Arrigoni, 2006; Pignatti, 1982; Tutin et al., 1964) and specialist monographs (Camarda and Valsecchi, 2008; de Martis et al., 1984; Marchetti, 2004; Ritter, 2014). The nomenclature follows Bartolucci et al. (2018) for native species and Galasso et al. (2018) for alien plants. *Pinus halepensis* subsp. *halepensis, Olea europaea* and *Ceratonia siliqua* were also included in the group of alien plants. In Sardinia, *P. halepensis* subsp. *halepensis* has recently been elevated to the native status from the national checklists (Bartolucci et al., 2020) in agreement with other authors (Arrigoni, 2006; Camarda and Valsecchi, 2008). However, the debate on nativity remains open (Pignatti, 2019) and in line with other authors (Arrigoni, 2006; Camarda and Valsecchi, 2008; Galasso et al., 2018), we considered this taxon as introduced and naturalized in the study area. Similarly, we considered *Olea europaea* and *Ceratonia siliqua* as naturalized archeophytes following the updated national checklists (Bartolucci et al., 2018; Galasso et al., 2018). Although they have long been considered native species (Arrigoni, 2006; Pignatti, 1982), their indigenous status is currently controversial (Camarda and Valsecchi, 2008; Pignatti, 2019) and some recent studies suggest their diffusion and spontaneization mediated by humans in some areas of the Mediterranean basin (Besnard and Casas, 2016; Médail et al., 2019; Ramón-Laca and Mabberley, 2004).

Six species not listed in the above-mentioned check-lists, for which the nativity status was unknown, were excluded from the analyses.

Trees and shrubs diameter at breast height of five individuals per plot were measured for a total of 25 measures per SWOF. Based on these measurements, we calculated the coefficient of variation of the diameter at breast height (DBH) as a measure of the structural diversity of SWOFs.

Environmental factors

Environmental factors were recorded at the site level and categorized into three main descriptor groups: (a) spatial-topographic factors (b) bioclimatic variables (c) and landscape measures (Table 1).

The first group consisted of variables describing spatial-topographic characteristics, such as geographic coordinate, elevation, inclination, exposition and distance to the coast, river, lake and lagoon. Geographic coordinates were recorded in each plot using a global positioning system (GPS) instrument and were expressed as angular units (degrees) in the WGS84 geographic coordinate system (EPSG code of 4326). Digital terrain model (DTM, resolution 10 m) was used to derive elevation, inclination, and exposition of each SWOF centroid, using ArcGIS 10.2.1 (ESRI, 2014). By means of ArcGIS 10.2.1 (ESRI, 2014), we also calculated three spatial variables considering the distance from each SWOF centroid to (i) the nearest coastline (coastal distance), (ii) the nearest river (river distance), (iii) the nearest artificial or natural lake and lagoon (lake distance).

The second descriptor group consisted of 19 biologically meaningful bioclimatic variables, related to temperature (BIO01-BIO07 and BIO10-BIO11), precipitation (BIO12-BIO17) or both temperature and precipitation (BIO08-BIO09 and BIO18-BIO19) (see Bazzato et al., 2021 for a complete description of each index).

The third descriptor group consisted of 17 metrics at the landscape-level (see McGarigal et al., 2002 for a complete description of each metric, Table 1), describing compositional and configurational characteristics of the surrounding landscape of each small woodlot. Landscape metrics were calculated within a 500 m buffer distance around each site, using the regional Land-use Map at the third hierarchical level of detail (scale 1:25.000; RAS, 2008) and Patch Analyst extension (Elkie et al., 1999; Rempel et al., 2012).

Table 1. List of the measured spatial-topographic and bioclimate parameters, and landscape metrics.

Descriptor group	**Variable name**	**Variable description**	**Type of variable**
Spatial-topographic	X	x geographical coordinate (degrees)	Spatial variable
	Y	y geographical coordinate (degrees)	Spatial variable
	Z	elevation (m)	Topographic variable
	Inclination	inclination (°)	Topographic variable
	Exposition	exposition (°)	Topographic variable
	Coast_dist	Distance from the coast (m)	Spatial variable
	Rivers_dist	Distance from the river (m)	Spatial variable
	Lake_dist	Distance from the lake and lagoon (m)	Spatial variable
Bioclimatic	BIO01	Annual Mean Temperature (°C)	Temperature-related variable
	BIO02	Mean Diurnal Range (Mean of monthly (max temp - min temp)) (°C)	Temperature-related variable
	BIO03	Isothermality (BIO2/BIO7) (x 100)	Temperature-related variable
	BIO04	Temperature Seasonality (standard deviation x 100)	Temperature-related variable
	BIO05	Maximum Temperature of Warmest Month (°C)	Temperature-related variable
	BIO06	Minimum Temperature of Coldest Month (°C)	Temperature-related variable
	BIO07	Temperature Annual Range (BIO5-BIO6) (°C)	Temperature-related variable
	BIO08	Mean Temperature of Wettest Quarter (°C)	Temperature-related and rainfall-related variable
	BIO09	Mean Temperature of Driest Quarter (°C)	Temperature-related and rainfall-related variable
	BIO10	Mean Temperature of Warmest Quarter (°C)	Temperature-related variable
	BIO11	Mean Temperature of Coldest Quarter (°C)	Temperature-related variable
	BIO12	Annual Precipitation (mm)	Rainfall-related variable
	BIO13	Precipitation of Wettest Month (mm)	Rainfall-related variable
	BIO14	Precipitation of Driest Month (mm)	Rainfall-related variable
	BIO15	Precipitation Seasonality (Coefficient of Variation)	Rainfall-related variable
	BIO16	Precipitation of Wettest Quarter (mm)	Rainfall-related variable
	BIO17	Precipitation of Driest Quarter (mm)	Rainfall-related variable
	BIO18	Precipitation of Warmest Quarter (mm)	Temperature-related and rainfall-related variable
	BIO19	Precipitation of Coldest Quarter (mm)	Temperature-related and rainfall-related variable
Landscape	SDI	Shannon's Diversity Index	Diversity Metric
	SEI	Shannon's Evenness Index	Diversity Metric
	AWMSI	Area Weighted Mean Shape Index	Shape Metric
	MSI	Mean Shape Index	Shape Metric
	MPAR	Mean Perimeter-Area Ratio	Shape Metric
	MPFD	Mean Patch Fractal Dimension	Shape Metric
	AWMPFD	Area Weighted Mean Patch Fractal Dimension	Shape Metric
	TE	Total Edge	Edge Metric
	ED	Edge Density	Edge Metric
	MPE	Mean Patch Edge	Edge Metric
	MPS	Mean Patch Size	Patch Density & Size Metric
	NumP	No. of Patches	Patch Density & Size Metric
	MedPS	Median Patch Size	Patch Density & Size Metric
	PSCoV	Patch Size Coefficient of Variance	Patch Density & Size Metric
	PSSD	Patch Size Standard Deviation	Patch Density & Size Metric
	TLA	Total Landscape Area	Area Metric
	CA	Class Area	Area Metric

Statistical analyses

Significant differences among land-use matrices were evaluated at the SWOF-level using permutational multivariate analysis of variance (PERMANOVA; Anderson, 2001) based on (i) Euclidean distance applied to species richness data of natives and aliens considered collectively; (ii) zero-adjusted Bray-Curtis dissimilarity (Clarke et al., 2006) on square-root transformed abundance data of each native and alien group considered separately.

Based on richness data, PERMANOVA (Anderson, 2001) was used to test the null hypothesis of no differences in the location (centroids) and/or spread among factors (Anderson, 2017; Anderson and Walsh, 2013): land-use matrices (three levels, fixed: natural and semi-natural areas, agricultural areas, urban and artificial surfaces), and taxa groups (two levels, fixed factor crossed with land-use matrices: natives and aliens). Post-hoc permutational pair-wise comparison tests with PERMANOVA *t* statistic and 999 permutations were then applied to investigate the significant interaction term “matrices x group” for species richness data, or the main term “matrices” for abundance data where groups were considered separately. For each source of variation, the pseudo multivariate variance components, expressed in percentage, were also calculated. Analyses were performed using 999 random permutations and the most conservative type III sum of squares for unbalanced designs (Anderson et al., 2008) in PRIMER v.6.1.12 software (Anderson et al., 2008; Clarke and Gorley, 2006).

Indicator species analysis (Cáceres et al., 2010; Dufrêne and Legendre, 1997) was used to explore the relationship between the native and alien contingent of the plant community and the matrices surrounding SWOFs, using the multipatt-multi-level pattern analysis function of the *indicspecies* R package (Cáceres and Legendre, 2009). Based on this analysis, the indicator species of each native and alien contingent for each matrix solely, or combinations of land-use matrices were calculated. Then, only significant species at a significance level of 0.05 with IndVal values greater than 0.5 were assessed, considering these values as the product of two conditional probabilities: specificity (component A), the positive predictive value of a species as indicator of the considered matrix; fidelity (component B), probability of finding a considered species in SWOFs belonging to the indicated land-use matrix (Cáceres et al., 2010).

To make a full distributional comparison of DBH structures among SWOFs surrounded by different land-use matrices, summary statistics were calculated at land-use level. Then, we adopted the coefficient of variation of diameter at breast height to test the difference in term of structural diversity among the land-use matrices, applying a Kruskal-Wallis test and pairwise comparisons with the multiple Wilcoxon tests, using the function of *kruskal.test* and *pairwise.wilcox.test* in R software, respectively (R core Team, 2020).

A principal component analysis (PCA) of (a) spatial-topographic factors, (b) bioclimatic variables and (c) landscape metrics was used to reduce environmental variables into a few independent and interpretable components (PCs) and describe the main gradient of land-use intensification surrounding SWOFs. Analyses were applied on standardized data (Borcard et al., 2011) of each descriptor group (spatial-topographic and climate parameters, and landscape metrics), using *rda* function in the *vegan* R package (Oksanen et al., 2019). The number of significant axes (PCs) was evaluated using the broken-stick criterion, a null model for random distribution of the variance among the axes were performed with the *evplot* function (Borcard et al., 2011). Spearman's rank correlations were then applied between the selected PCs and three groups of environmental variables and visualized through the scatter plot of the *psych* R package (Revelle, 2020).

The resulting sets of principal components were used consecutively, together with stand structural attributes, as predictors in the variation partition analyses (Borcard et al., 1992) to explore the effect of these variables on native and alien contingent of the plant community.

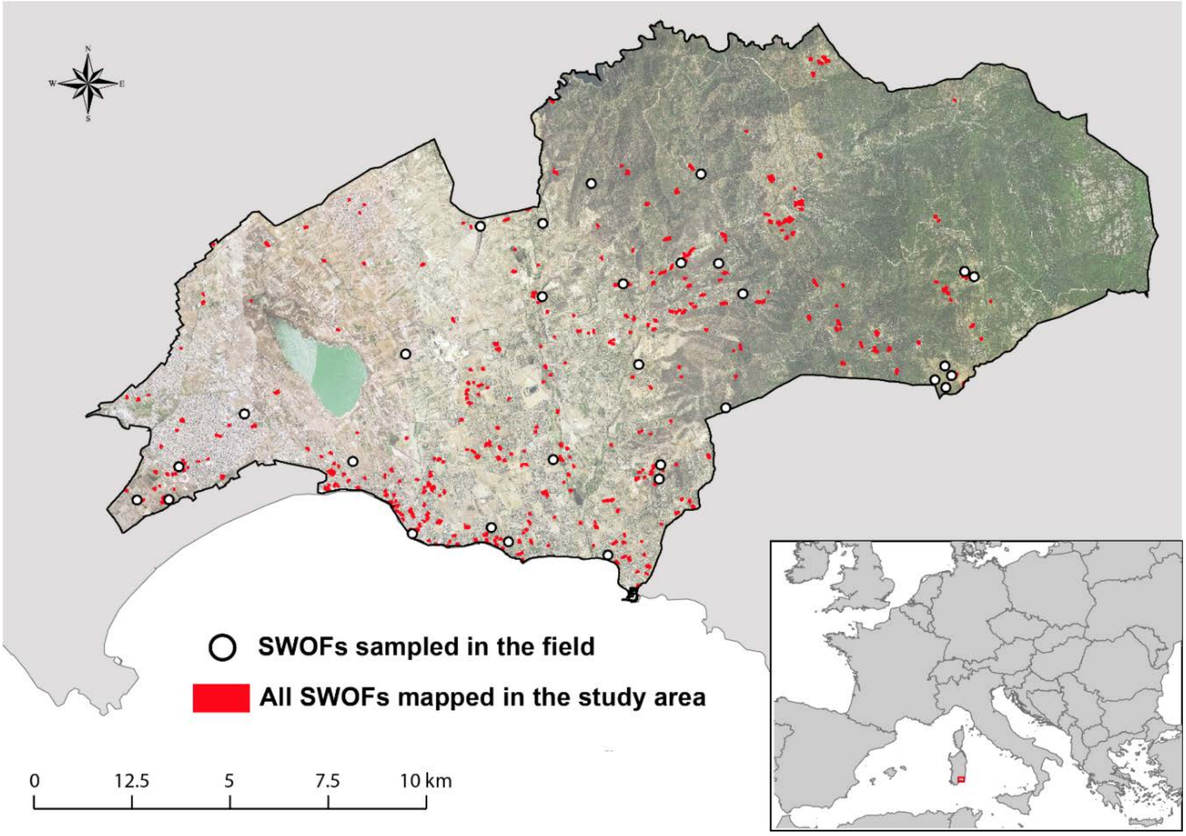
Based on Hellinger transformed cover data of native and alien plants, a variation partitioning analysis (Borcard et al., 1992) were performed using the *vegan* package (Oksanen et al., 2019). The significance of the individual fractions was tested by RDA analyses constraining 999 permutations within the land-use matrices (Peres-Neto et al., 2006).

Results

Small Woodlots Outside Forests (SWOFs) census and classification

The SWOF census led to 400 polygons representing the total number of SWOF in the whole study area (Fig. 2), which covered a total surface of 65.54 ha (about 0.38% of the study surface) with an average patch size of 1,638 m^2 and a coefficient of variation of 68% (Table 2).

Fig. 2. Distribution map of SWOFs mapped in the whole study area and sites sampled in the field.



The number and coverage of SWOFs increased along the land-use intensification gradient (Table 2). Compared to the other matrices, SWOFs in natural and semi-natural areas were fewer, covering almost 0.23% of the natural and seminatural territory with greater average patch size and a coefficient of variation of 64%. The total cover of SWOFs in agricultural and urban areas amounted respectively to 0.31% and 0.85% of the corresponded land-use territory, showing a similar average size and coefficient of variation (Table 2).

Table 2. Number of SWOF mapped in the study area and classified according to the corresponding land-use matrix: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB), mixed land-use areas. For each matrix, summary statistics of SWOF size (total coverage area; average, minimum and maximum area, coefficient of variation of patch size) are shown.

Land-use matrix	**Number of SWOF**		**AREA (m2)**								**CV (%)**	
			Tot		**Average**		**Min**		**Max**			
SWOF range	0.05 to 0.5 ha	0.1 to 0.5 ha	0.05 to 0.5 ha	0.1 to 0.5 ha	0.05 to 0.5 ha	0.1 to 0.5 ha	0.05 to 0.5 ha	0.1 to 0.5 ha	0.05 to 0.5 ha	0.1 to 0.5 ha	0.05 to 0.5 ha	0.1 to 0.5 ha
Natural and semi-natural areas	90	64	195089	176343	2168	2755	519	1023	4917	4917	64	45
Agricultural areas	123	70	192399	154408	1564	2206	507	1011	4775	4775	67	44
Urban and artificial areas	150	67	197235	134005	1315	2000	507	1006	5000	5000	66	44
Mixed land-use areas	37	31	70667	65754	1910	2121	643	1007	4419	4419	50	43
Total	**400**	**232**	**655390**	**530510**	**1638**	**2287**	**507**	**1006**	**5000**	**5000**	**68**	**46**

Considering only SWOFs with a size ranging from 0.1 to 0.5 hectares (Table 2), the percentage of cover concerning the corresponding land-use territory were similar in natural and agricultural areas (0.21% and 0.25%, respectively), and high in urban and artificial areas (0.58%), with a coefficient of variation lower than the totality of censused SWOFs (i.e., SWOFs with a size ranging from 0.05 to 0.5 ha, Table 2).

Differences in native and alien species richness and composition among land-use matrices and key related taxa

A different trend in the spread of native and alien species was observed between the three types of land-use matrices (Fig. 3; Annex S1, Table 2).

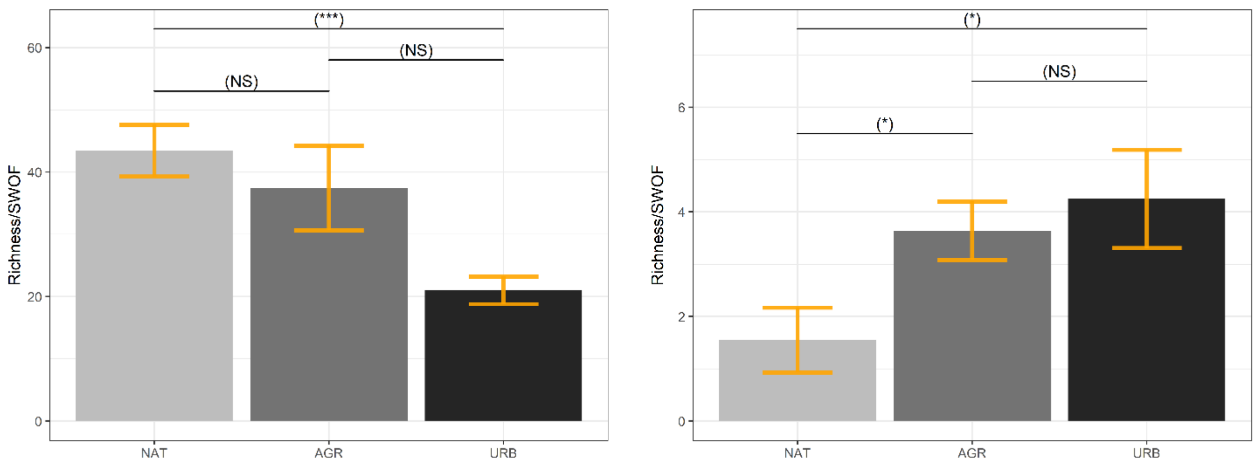
Overall, the native plant contingent was higher in all land-use matrices than the number of alien species (Annex S1, Table 1). Notably, from a total of 324 species sampled, 6.8% of these were alien. In the natural matrix, we found 95.1% of native plants and 4.9% of alien taxa; in the agricultural matrix, native plants amounted to 93.5% and alien plants to 6.5%; while finally, in the urban matrix, we found 85.4% of native taxa and 14.6% of alien plants.

The mean richness of native plants tended to decrease along the gradient of land-use intensification (from natural and semi-natural areas to urban ones), whereas aliens showed an opposite trend reaching the highest mean rate of expansion in term of richness in SWOF surrounded by urban areas (Fig. 3).

The highest values of the mean species abundance of native plants were in the natural SWOFs, followed by urban and agricultural SWOFs (Fig. 3). Regarding aliens, the higher values of abundance were recorded in the agricultural SWOFs compared to urban SWOFs and natural and semi-natural ones (Fig. 3).

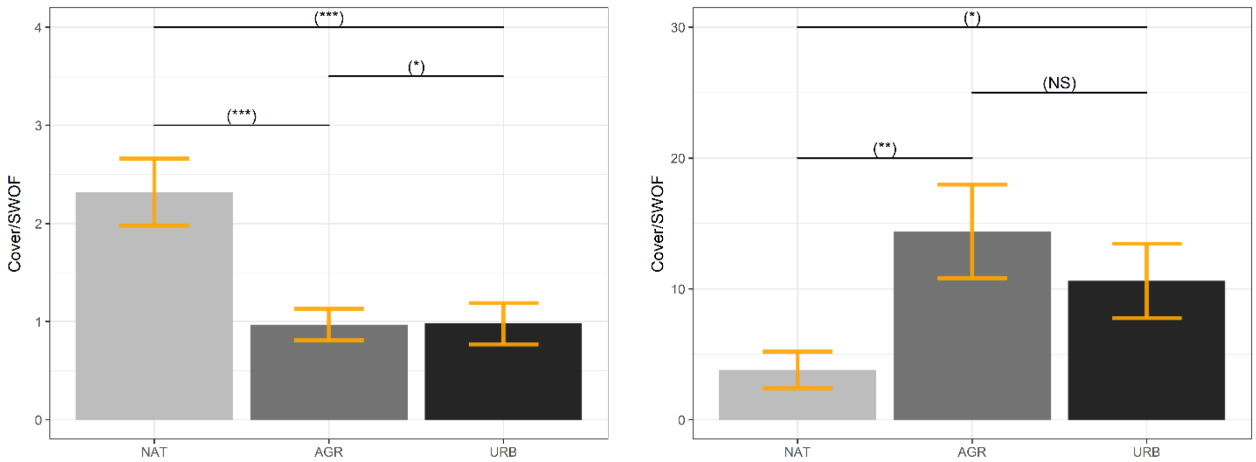
Fig. 3. Bar plots showing patterns of variation of richness and abundance (means ± SE) across the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). For each group of native and aliens, the mean value per SWOF, standard error bar, and statistically significant differences across each pair of land use, as indicated by PERMANOVA pair-wise tests, are shown. Significance codes: (***) $p \leq 0.001$, (**) $p \leq 0.01$, (*) $p \leq 0.05$, (NS) not significant.

(a) Native species richness **(b) Alien species richness**



(c) Native species richness abundance

(c) Native species abundance **(d) Alien species abundance**



Considering species richness, PERMANOVA results revealed that the nativity status was the strongest, with residuals, contributing the largest components of variation to the overall model (Annex S1, Table 3). The main effect of land-use matrices was less important than any other factors (Annex S1, Table 3). However, PERMANOVA clearly detected significant interactions of land-use matrices with the native and alien group (MA x GR; $p \leq 0.01$), suggesting different response patterns of richness across SWOFs surrounded by different land use (Annex S1, Table 3). Pairwise comparisons further support this general observation, underlining that species richness of native plants differed between SWOFs surrounded by natural and semi-natural areas vs urban SWOFs, but not across the other surrounding matrices (Fig. 3; Annex S1, Table 4). Conversely, significant contrasts among natural and semi-natural SWOFs and those surrounded by the human-impacted matrices were identified for the richness of alien plants (Fig. 3; Annex S1, Table 4).

Table 3. Results of the Indicator Species Analysis on the abundance data of native plants. For the species significantly associated with each matrix or combination of land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). IndVal values greater than 0.5, the two conditional probabilities (specificity, component A; fidelity, component B) and statistical significance lower than 0.05 are shown.

Land-use matrix	**Species**	**A**	**B**	**IndVal**	**p-Value**
NAT	*Rubia peregrina* L.	0.916	0.909	0.912	0.002**
	Asphodelus ramosus L. subsp. *ramosus*	0.868	0.909	0.888	0.001***
	Arbutus unedo L.	1.000	0.636	0.798	0.001***
	Cynosurus effusus Link	1.000	0.636	0.798	0.001***
	Rubus ulmifolius Schott	1.000	0.636	0.798	0.001***
	Briza maxima L.	0.811	0.727	0.768	0.01**
	Trifolium ligusticum Loisel.	1.000	0.546	0.739	0.003**
	Ornithopus compressus L.	0.864	0.546	0.686	0.015*
	Erica arborea L.	1.000	0.455	0.674	0.005**
	Scirpoides holoschoenus (L.) Soják	1.000	0.455	0.674	0.013*
	Carex distachya Desf.	0.996	0.455	0.673	0.015*
	Cistus monspeliensis L.	0.912	0.455	0.644	0.021*
	Aira elegantissima Schur subsp. *elegantissima*	1.000	0.364	0.603	0.019*
	Cistus salviifolius L.	1.000	0.364	0.603	0.043*
	Pulicaria odora (L.) Rchb.	1.000	0.364	0.603	0.044*
	Bromus hordeaceus L. subsp. *hordeaceus*	0.998	0.364	0.602	0.041*
AGR	*Oloptum miliaceum* (L.) Röser & H.R.Hamasha	0.866	0.818	0.842	0.006**
	Asparagus albus L.	0.965	0.636	0.784	0.003**
	Triticum vagans (Jord. & Fourr.) Greuter	0.876	0.364	0.564	0.046*
URB	*Lolium rigidum* Gaudin subsp. *rigidum*	0.960	0.625	0.774	0.01**
	Reichardia picroides (L.) Roth	1.000	0.500	0.707	0.001***
	Sonchus oleraceus L.	0.752	0.625	0.685	0.013*
	Hordeum murinum L.	0.968	0.375	0.602	0.05*
AGR+NAT	*Pistacia lentiscus* L.	0.999	0.682	0.825	0.014*
	Trifolium campestre Schreb.	1.000	0.546	0.739	0.03*
	Trifolium angustifolium L. subsp. *angustifolium*	1.000	0.500	0.707	0.037*
AGR+URB	*Sonchus tenerrimus* L.	0.995	1.000	0.998	0.001***
	Hedypnois rhagadioloides (L.) F.W.Schmidt	0.996	0.474	0.687	0.049*

When PERMANOVA analyses were conducted by using species composition data for each native and alien group separately, land-use matrices had a significant effect on the community composition of each group (MA $p \leq 0.001$; Annex S1, Table 5). However, the largest components of variation to the overall models were explained by residual (Annex S1, Table 5).

Results of pair-wise tests suggested that community composition of natives significantly diverged across all land-use types, whereas alien plants showed a significant contrast only between natural and semi-natural SWOFs and the SWOFs surrounded by the human-impacted land-uses (comparing results of PERMANOVA pair-wise tests, Fig. 3 and Annex S1, Table 6).

Indicator species analysis identified: i) 16 native indicator species exclusive of SWOFs located in the natural matrix; ii) 3 native indicator species and 1 alien indicator species exclusive of SWOFs located in the agricultural matrix; iii) 4 native indicator species and 1 alien indicator species exclusive of SWOFs located in the urban matrix. (Table 3 and Table 4).

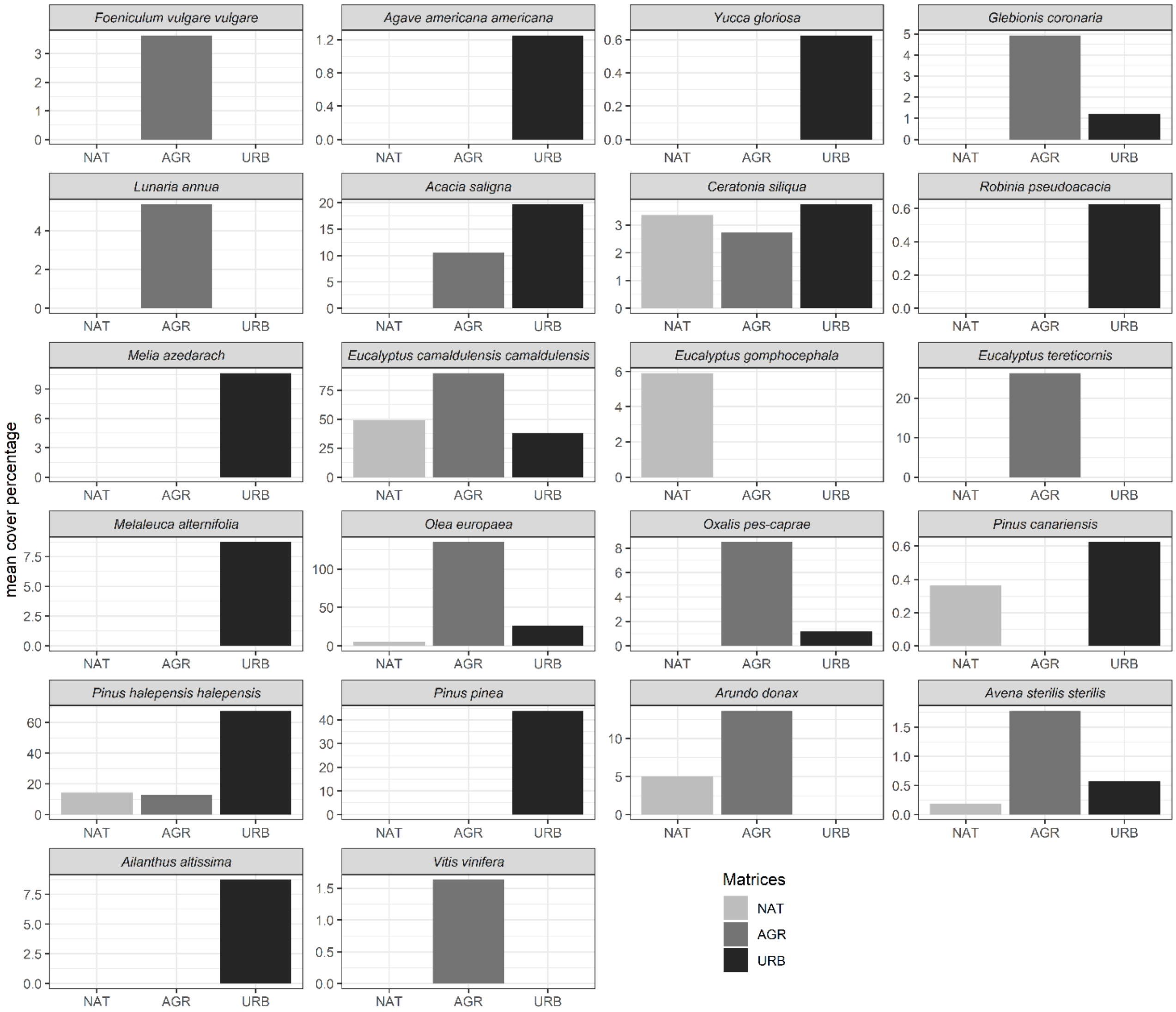
Among the native indicator species in the natural matrix, three species (*Rubia peregrina* L., *Asphodelus ramosus* L. subsp. *ramosus*, *Briza maxima* L.) showed high IndVal scores due to the high value of specificity and fidelity; thirteen species due to the high value of specificity (from 0.812 to 1.000) rather than fidelity (from 0.364 to 0.727). Only three native species were associated with the agricultural areas, showing high (> 0.780) or moderate (0.564) IndVal scores (Table 3). In particular, *Oloptum miliaceum* (L.) Röser & H.R. Hamasha was considered as a good indicator thanks to both specificity and fidelity, while the other two (*Asparagus albus* L. and *Triticum vagans* (Jord. & Fourr.) Greuter) due to specificity rather than fidelity. Similarly, the four native species associated with the urban land-use matrix reached high values of IndVal scores for the high values of specificity, ranging from 0.752 to 1.000 (Table 3). Considering the combination of agricultural and natural areas, all three species with high IndVal scores were considered as a good indicator for the high value of specificity, ranging from 0.999 to 1.000 (Table 3). Significantly, *Sonchus tenerrimus* L. as being associated with the combination of agricultural and urban matrices, reaching the highest IndVal score of the dataset thanks to the highest values of both specificity and fidelity; whereas *Hedypnois rhagadioloides* (L.) F.W. Schmidt showed a high IndVal score only due to specificity (Table 3).

For alien species, we sampled a few dominant species with high values of cover in the agricultural and urban land-use matrices, whereas the other alien species showed a negligible occurrence (Fig. 4).

In the SWOFs surrounded by agricultural land-use, the archaeophyte *O. europaea* and the ornamental *E. camaldulensis* subsp. *camaldulensis*, classified as invasive at regional level, were systematically spread, whereas the occurrence of other alien species was recorded only marginally (Fig. 4).

Pinus halepensis Mill. subsp. *halepensis* and *Pinus pinea* L. occurred as planted and naturalized aliens in SWOFs surrounded by urban areas, followed by species with an invasive status in the Region, such as *Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis* and *Acacia saligna* (Labill.) H.L. Wendl. (Fig. 4), a species of Union concern (Regulation (EU) 1143/2014, Commission Implementing Regulations (EU) 2016/1141 and 2017/1263).

Fig. 4. Contributions of individual alien species to the total abundance in SWOFs surrounded by natural and semi-natural land-use matrices. The mean cover percentage for each species and land-use matrix is shown.



The indicator species analysis of the alien abundance data identified the strong and significant association between *O. europaea* and *P. halepensis* subsp. *halepensis* with agricultural and urban matrices, respectively, showing high IndVal scores (Table 4): the former due to both high specificity (A = 0.810) and fidelity (B = 0.909), the latter due to high specificity (A = 0.712). Considering the combination of agricultural and urban SWOFs, the invasive *Oxalis pes-caprae* L. showed high IndVal scores due to high specificity (A = 0.999) rather than fidelity (B = 0.632) (Table 4).

Table 4. Results of the Indicator Species Analysis on the abundance data of alien plants. For the species significantly associated with each matrix or combination of land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). IndVal values greater than 0.5, the two conditional probabilities (specificity, component A; fidelity, component B) and statistical significance lower than 0.05 are shown.

Land-use matrix	**Species**	**A**	**B**	**IndVal**	**p-Value**
AGR	*Olea europaea* L.	0.810	0.909	0.858	0.004**
URB	*Pinus halepensis* Mill. subsp. *halepensis*	0.713	0.625	0.667	0.013*
AGR+URB	*Oxalis pes-caprae* L.	0.999	0.632	0.794	0.039*

Structural attribute differences among SWOFs surrounded by different land-use matrices

The mean DBH varied from 8.69 to 20.77 cm in SWOFs surrounded by different land-use (Table 5). The DBH of trees and shrubs in SWOFs located in natural and agricultural areas was more variable than in those located in urban areas (Table 5). In SWOFs located in natural and seminatural areas, the DBH was smaller than in agricultural and urban ones. The greatest asymmetry was shown by distributions from SWOFs of natural and seminatural area, while the skewness of those in human-impacted land-use was generally low (Table 5).

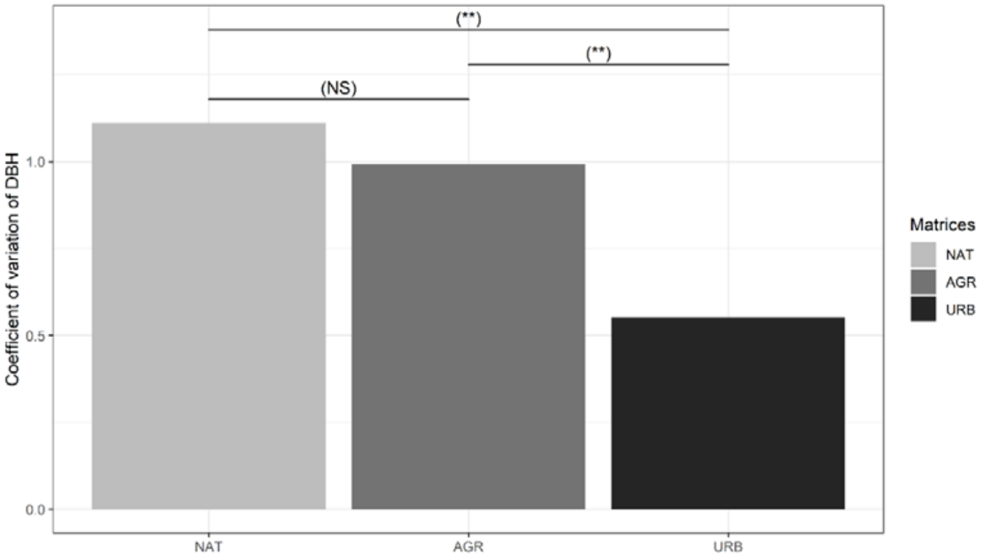
Table 5. Summary statistics of mean, standard deviation, coefficient variation and maximum values of tree and shrubs diameter at breast height (DBH) for each land-use matrix: natural and semi-natural areas (NAT). agricultural areas (AGR). urban and artificial surfaces (URB). Statistical note: mean species richness (µ); standard error (SE); minimum (min) and maximum (max) values.

Land-use matrix	Mean DBH (cm)	SD of DBH (cm)	CV of DB	Max DBH (cm)	Skewness	Kurtosis
NAT	8.69	9.65	1.11	63.98	2.91	10.43
AGR	11.48	11.40	0.99	59.21	1.81	3.25
URB	20.77	11.46	0.55	55.39	0.49	-0.27

Kruskal-Wallis test among small woodlots resulted that the land-use group means were significantly different for the considered structural attribute (i.e., coefficient of variation of diameter at breast height (DBH); chi-squared = 11.583, p-value = 0.0031).

Pairwise comparisons among land-use group means (Fig. 5) showed that the coefficient of variation of diameter at breast height (DBH) differed between natural and semi-natural SWOFs vs urban SWOFs, but also between urban SWOFs and agricultural SWOFs (Fig. 5). Conversely, no significant contrasts among SWOFs surrounded by natural and semi-natural areas, and SWOFs surrounded by agricultural areas were identified (Fig. 5).

Fig. 5. Paired differences of the coefficient of variation of diameter at breast height (DBH) among SWOFs surrounded by different land-use matrices. Significance of each paired sample Wilcoxon is shown with the following codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05, (NS) not significant.



Gradient of environmental factors

Based on spatial-topographic factors recorded in each SWOF, PCA resulted in a total of 8 PCs; comparing the eigenvalues to the broken-stick model, only the first 2 components were considered in further analysis (71% of the variance). PC1 axis of the spatial-topographic factors (PC1, 49% of the explained variation) was significatively positively related to longitudinal (x, $\rho = 0.85$), altitudinal (Z, $\rho = 0.91$) and coastal distance (Coast_dist, $\rho = 0.94$) variation ($p \leq 0.001$; Fig. 2, Annex S2). The second principal component (PC2), explaining 22% of the variation, was negatively related to lake and lagoon distance variables (Lake_dist, $\rho = -0.80$, $p \leq 0.001$; Fig. 2, Annex S2).

Overall, these two PCs described the decreasing of longitude, altitude, coastal distance, lake and lagoon distance from SWOFs located in natural and semi-natural areas to those in human-modified land-uses (Fig. 1; Annex S2).

Based on bioclimatic variables recorded in each SWOF, PCA resulted in a total of 18 PCs: following the broken-stick model, we considered in the further analysis only the first 2, accounting for 96% of the total variance. PC1 axis of the bioclimatic variables (PC1, 77% of the explained variation) was significatively and negatively associated ($p \leq 0.001$, Fig. 4, Annex S2) with variables describing Annual Mean Temperature (BIO1, $\rho = -0.82$), Mean Temperature of Coldest Quarter (BIO11, $\rho = -0.80$), and Mean Temperature of Wettest Quarter (BIO8, $\rho = -0.84$); it was also strongly positively related to variables describing Annual Precipitation (BIO12, $\rho = 0.97$), Precipitation of Wettest Month and Quarter (BIO13 and BIO16, both with $\rho = 0.98$), Precipitation of Driest Month and Quarter (BIO14 and BIO17, $\rho = 0.82$ and 0.88, respectively), Precipitation of Warmest and Coldest Quarter (BIO18 and BIO19, $\rho = 0.91$ and 0.97, respectively, $p \leq 0.001$; Fig. 4, Annex S2). The second axis (PC2) was positively related to the gradient of Precipitation Seasonality expressed as coefficient of variation (BIO15, $\rho = 0.82$, $p \leq 0.001$; Fig. 4, Annex S2).

Overall, these two PCs described the increase of mean temperature values and the decrease of precipitation values from SWOFs located in natural and semi-natural areas to those in human-modified land-uses (Fig. 3, Annex S2).

Based on landscape metrics calculated around each SWOF, PCA resulted in a total of 16 PCs: the first 3 components (accounted for 75% of the variance) were considered in further analysis (Fig. 5, Annex S2). PC1 axis of the landscape metrics (PC1, 43% of the explained variation) was negatively related to measures of Shannon's Diversity (SDI, $\rho = -0.87$), edge quantity and density (TE, $\rho = -0.92$; ED, $\rho = -0.98$) and patch number (NumP, $\rho = -0.96$), and positively associated with mean patch size (MPS, $\rho = 0.97$) and standard deviation of patch size (PSSD, $\rho = 0.87$). All of these showed a strong correlation with a statistical significance ($p \leq 0.001$) (see Fig. 6, Annex S2). The second principal component (PC2), accounting for 19% of the total variation, was positively related to Shannon's Evenness (SEI, $\rho = 0.85$) and negatively associated with patch size coefficient of variance (PSCoV, $\rho = -0.91$), both of these with a statistical significance ($p \leq 0.001$) (see Fig. 6, Annex S2). The third principal component (PC3), explaining 14% of the variation, was negatively related to measures of habitat shape complexity, such as area-weighted mean shape (AWMSI, $\rho = -0.55$, $p \leq 0.01$) and mean shape (MSI, $\rho = -0.61$, $p \leq 0.001$) (see Fig. 6, Annex S2).

Overall, these three PCs described the increase of fragmentation degree from woodlots surrounded by natural and semi-natural areas to those surrounded by agricultural and urban land-uses. Following this gradient, a reduction of mean patch size (MPS), patch size standard deviation (PSSD) and mean shape (MSI), as well as an increment of the number of patch (NumP), Shannon's

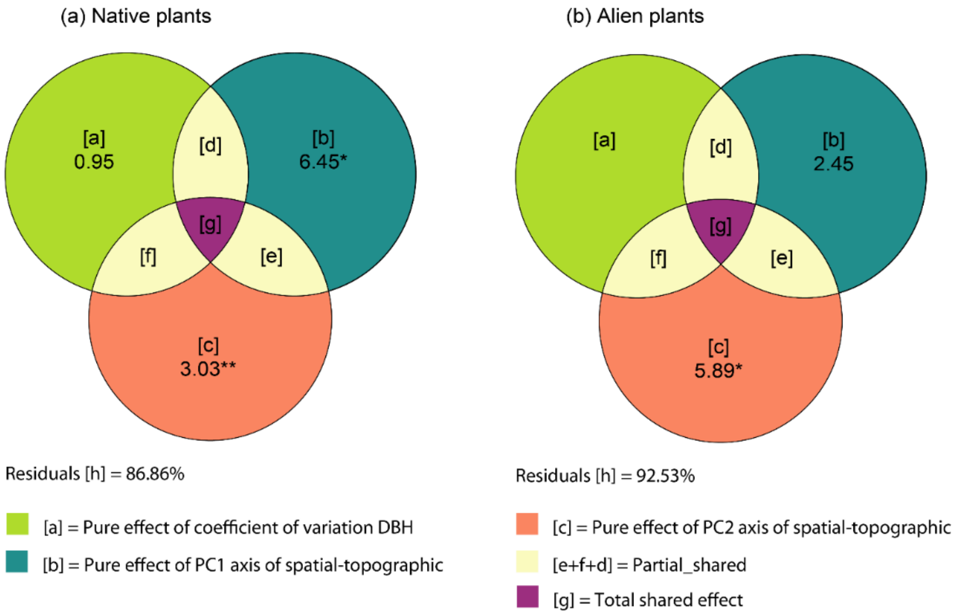
Diversity and Evenness (SDI and SEI), but also edge habitat (TD and ED) were observed from natural and semi-natural areas to human-modified land-uses (Fig. 5, Annex S2).

Effect of environmental factors and structural attributes on native and alien contingents

The variation partitioning revealed that the total variance explained by structural attribute and non-redundant spatial-topographic variables to the total variation of native and alien plants were significant for both groups (natives, 13.14%, F = 2.462, p = 0.002; aliens, 7.47%, F = 1.781, p = 0.046).

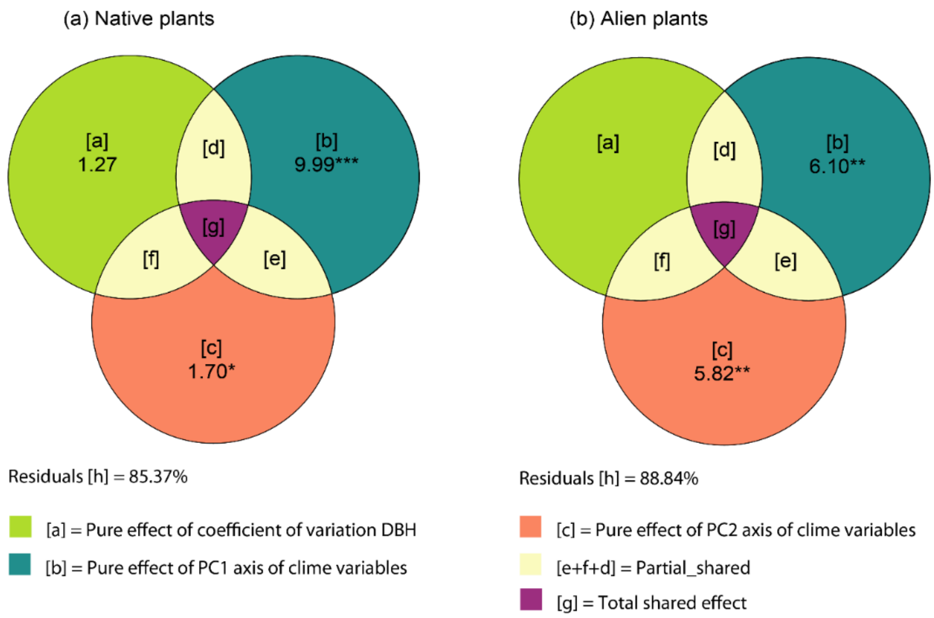
Considering the native contingent of the plant community, a large and significant portion of variation was attributable to the pure effects of the first (F = 3.003, p = 0.001) and second (F = 1.941, p = 0.005) PC axis of spatial-topographic variables (Fig. 6a). Considering the alien plants, only the second axis explained a significant portion of the total variation (F = 2.720, p = 0.016), whereas the other fractions were negligible (Fig. 6b).

Fig. 6. Result of the variance partitioning analysis, showing the contribution of SWOF structural attribute and non-redundant spatial-topographic variables to the variation of a) native and b) alien contingent of the plant community.



Considering SWOF structural attribute and bioclimatic variables, analyses showed that the total variance explained by the three set of predictors were significant for both native (14.63%, F = 2.656, p = 0.001) and alien (11.16%, F = 2.214, p = 0.014) plants. Pure effect of bioclimatic PCs explained a significant portion of the total variation both for native ([b] F = 4.161, p = 0.001; [c] F =1.539, p = 0.033) and alien ([b] F = 2.854, p = 0.007; [c] F = 2.768, p = 0.019) plants, while no independent effect was recognized for the coefficient of variation of DBH (Fig. 7).

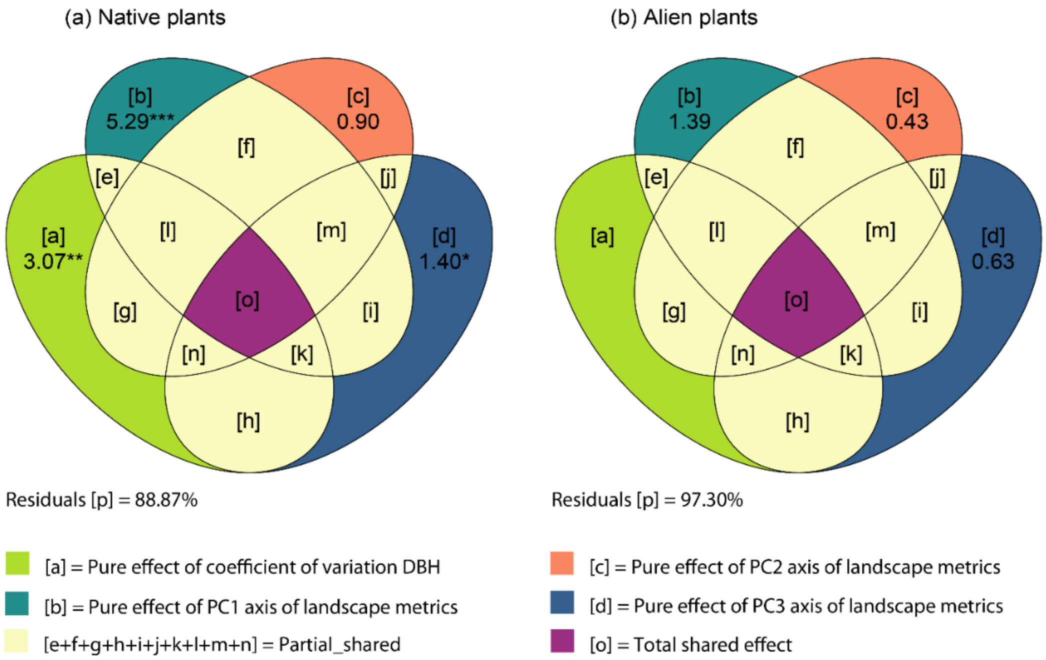
Fig. 7. Result of the variance partitioning analysis, showing the contribution of SWOF structural attribute and non-redundant bioclimatic variables to the variation of a) native and b) alien contingent of the plant community.



Considering SWOF structural attribute and landscape metrics, analyses showed that native plants were explained by the considered set of predictors (11.13%, F = 1.908, p = 0.016), while the global model for alien ones was not significant.

The pure effect of the coefficient of variation of DBH explained a significant part of the total variation of native contingent of the plant community (F = 1.899, p = 0.004). The variation attributed solely to the independent effect of PC1 axis of landscape measures was higher (F = 2.546, p = 0.001) than the pure effect of PC3 axis (F= 1.410, p = 0.048), while no independent effect was recognized for the second axis of these metrics (Fig. 8).

Fig. 8. Result of the variance partitioning analysis, showing the contribution of SWOF structural attribute and non-redundant landscape metrics to the variation of a) native and b) alien contingent of the plant community.



Discussion

Small Woodlots Outside Forests (SWOFs) census and classification

The wide distribution of TOF population across fragmented landscapes (Bellefontaine et al., 2002, 2001; Manning et al., 2006) makes a complete inventory and field census prohibitive (Marchetti et al., 2018a), underlining the importance of implementing and adopting a variety of methodological approaches to estimate and infer the TOF population parameters at large spatial scale (Corona, 2016; de Foresta et al., 2013; Kleinn, 2000).

Given the recognized importance of TOF (Bellefontaine et al., 2002, 2001; Manning et al., 2006) and the need for more comprehensive information regarding their spatial distribution and attributes (Schnell et al., 2015), we adopted a classical multiphase sampling design (pre-fieldwork phase and fieldwork phase) to study a particular type of TOF (i.e., SWOF) in a Mediterranean hot spot of biodiversity conservation (Marignani et al., 2017a, 2017b; Médail, 2017).

From the methodological point of view, the use of high spatial resolution data, such as the aerial photos here adopted in the first pre-fieldwork phase, is considered as a reliable method for collecting auxiliary information about these sparse and scattered resources (Baffetta et al., 2011; Lam et al., 2011; Marchetti et al., 2018a), and implementing stratified sampling designs for collecting TOF attributes in the field phase (Schnell et al., 2015).

Hierarchical classification schemes based on land use or land cover strata (Magdon et al., 2014) are considered as a very useful approach from a TOF inventory perspective, being suitable to guide sampling efforts of these woody resources (Tewari et al., 2014; Tomppo et al., 2014) and for categorizing TOF resources in respect to the land-use in which they grow (de Foresta et al., 2013; Kleinn, 2000).

The comparison of our results with surveys on out-of-forest trees conducted at European level is rather difficult, due to the different formal definitions adopted by the country or agency conducting an assessment (de Foresta et al., 2013). At the same time, these different formal definitions usually fixed to distinguish small from large patches can determine detrimental ambiguities in selecting the reference area associated with these resources (de Foresta et al., 2013; Sallustio et al., 2018), leading to possible misclassification with important consequences for assessment studies for their conservation. Considering the main survey conducted in France, which use the same classification system adopted by the present study, the cover of SWOFs was lower than that recorded and provided by French inventory (1.03% of the total territory) (Bellefontaine et al., 2001). Conversely, the percentage of SWOF cover recorded in this study resulted similar to that observed by Great Britain's National Forest Inventory (Brewer, A. et al., 2017), although woodlots are defined by considering a size between 0.1 and 2 ha, while in the present study the definition maintains the minimum surface threshold provided by FAO definition (de Foresta et al., 2013; FAO, 2010, 2006).

We obtained results that are in line with those at national (Sallustio et al., 2018) and regional level (Paletto et al., 2006), although slightly lower. At national level, the TOF cover almost 1.4%, with an average size of 0.1 ha (Sallustio et al., 2018). At regional level, the TOF cover in Molise Region amounted to 1.8% of the regional territory with an average patch size of 0.14 ha (Marchetti et al., 2018a). The total cover of TOF in central Italy amounted to 2.3% of the territory with an average size of 0.21 ha, even if the four regions considered showed a high cover variability, also between the type of TOF assessed (i.e., small woodlots or tree linear formations) (Paletto et al., 2006). Tree linear formations were clearly prevalent on that of small woodlots, which cover corresponded respectively to only 0.8%, 0.55% and 1.77% of the entire territories of central Italy, Lazio and Marche region (Paletto et al., 2006). However, the observed differences in terms of extension and average size between these four regions and our study can be justified by considering that these administrative areas were characterized by a particularly higher presence of TOF than the other ones, as explicitly reported by Paletto et al. (2006) and Sallustio et al. (2018).

As we expected, in our study the greatest number of SWOFs were located within the urban areas, while the other land-use types were characterized by a lower number of these woody resources. The percentage of SWOF cover increased following the gradient of land-use intensification, reaching the highest value in urban coastline zones. Coherently, the average patch size decreased along the same gradient, reaching the smallest value in the urban impacted land-use.

These results are consistent with the general spatial distribution trend of TOF reported by FAO (de Foresta et al., 2013). The increasing or decreasing of TOF cover are often linked to land-cover or land-use changes across the landscape (Novotný et al., 2017). In natural and semi-natural areas, TOF tend to be incorporated in recently expanded forest lands (Sallustio et al., 2018) due to the unification of small woodlots and solitary trees in new large elements, resulting in the conversion of trees outside forests into forest stands (Novotný et al., 2017). This is also consistent with the evident dynamics of Mediterranean forests where depopulation of the mountain regions is already associated with natural reforestation of natural and semi-natural areas (Carranza et al., 2020; Falcucci et al., 2007).

Similarly, the increasing of TOF cover in agricultural areas are often related to the abandonment of arable lands (Novotný et al., 2017). The tree densification as a consequence of the progressive abandonment of traditional farming and extensive grazing practices was described as a classical landscape pattern of Mediterranean land-use abandonment (San Roman Sanz et al., 2013). The changing in TOF cover, inversely relates to the loss of agricultural lands, was specifically observed for the Italian territory and was interpreted as an important trend of a possible extension of the green infrastructure network (Sallustio et al., 2018).

On the other hand, the presence of numerous small patches outside forest around built-up and within urban areas (Novotný et al., 2017) has been indicated as an important element for the mitigation of threats and pressures coming from the surrounding land uses and processes (Sallustio et al., 2018).

Differences in native and alien plant species richness and composition among land-use matrices and key related taxa

Considering the importance of maintaining native species and communities, as well as the heterogeneity and complexity of structures at the stand-level, focusing on TOF, especially in human-modified land-use (Marchetti et al., 2018a, 2018b; Paletto et al., 2006), could promote greater ecosystem functioning and stability (Angelini et al., 2015; Caviedes and Ibarra, 2017).

To the best of our knowledge, no studies have studied the pattern of richness and abundance of native and alien plants, exploring the key related taxa in Mediterranean SWOFs surrounded by different land-use type.

In this regard, studies on the presence and abundance of potential indicator species of native and alien taxa could represent an important instrument for managers and ecologists to judge the success or failure of management regimes designed to sustain biological diversity (Lindenmayer et al., 2000), also in trees outside forests. We used species richness and abundance data in SWOFs to determine how the different land-use matrix affects the diversity patterns and which potential indicator species could be associated with different management practices.

As expected, species richness showed a different pattern both for native and alien plants between SWOFs surrounded by natural areas and those in human-modified land-use. Species richness of native plants was higher in SWOFs located in the natural and semi-natural matrix than in those located in agricultural and urban matrices, most affected by anthropic disturbance. Moreover, the results of this study also indicated that the spread of alien plants along the gradient of land-use intensification varies markedly across the land-use matrices, confirming the general reports of other studies (Dostálek et al., 2016; Hobbs, 2000; Pauchard and Alaback, 2004; Sax and Brown, 2000). We found that the human impact, considered in general terms (i.e., agricultural and urban uses), clearly affected the number of alien species, leading to an increase of richness in urban and agricultural SWOFs compared to the natural ones. SWOFs located in natural areas were characterized by a few alien taxa that grow spontaneously and a high contribution of cultivated species such as *Eucalyptus* spp. Conversely, SWOFs in agricultural and urban areas were characterized by a high contribution of naturalized species, such as *Olea europaea* var. *sylvestris* a species recently classified as a naturalized archaeophyte in Sardinia (Bartolucci et al., 2018). Nevertheless, it is important to note that the number of alien species was small compared to the number of native plants. In Sardinia, the region interested by the highest increase in alien records from 2010 to 2018 (Galasso et al., 2018), the overall percentage of non-native species over the native flora accounted for 20.23% (Galasso et al., 2018), whereas, considering all the SWOFs sampled, we found that alien species accounted only for 6.8% of the total plant diversity.

The differences between SWOFs located in different land-use matrices were more relevant considering the community composition of native plants, which responded more sensitively to the anthropogenic disturbance gradient than the single alien plants.

The identification of indicator species partially confirmed the observed pattern, showing a high number of native indicator species compared to alien ones in all three land-use matrices.

The high number of bio-indicators detected in natural SWOFs for native species and the exclusivity of many of these demonstrated the peculiarity of this flora compared to the most disturbed areas such as agricultural and urbanized sites. Species such as *Rubia peregrina*, *Arbutus unedo, Erica arborea*, *Carex distachya*, *Cistus monspeliensis*, *Cistus salviifolius* characterized the successional or degradation stages of the potential vegetation that naturally developed in the study area (Bacchetta et al., 2009). This result enhances the high conservation and landscape value of SWOFs located in natural areas, even if sometimes dominated by an artificially implanted alien tree component (e.g., mainly of *Eucalyptus* plantation). Moreover, SWOFs in the natural matrix did not present key alien species being characterized by a particularly low presence of allochthonous species which resulted completely absent in 55% of natural SWOFs analyzed.

In SWOFs surrounded by agricultural matrix, *Oloptum miliaceum* (Rosati et al., 2020), *Asparagus albus* and *Triticum vagans* characterized the native communities as indicator species. Among these, *Asparagus albus* is generally associated with thermophilic micro-woods of *Olea europaea* var. *sylvestrys* (i.e., alien indicator species for agricultural SWOFs) which characterize mature and degraded stages of thermomediterranean turbinate juniper series (*Oleo Juniperetum turbinatae*), strongly affected by the impact of agropastoral activities (Bacchetta et al., 2009; Blasi et al., 2010). The association of *Asparagus* and *Olea* demonstrated the versatility of agricultural environments, which, on the one hand, host species of allochthonous origin, on the other hand, supported populations and natural vegetation dynamics. Moreover, agricultural SWOFs shared a good number of native indicator species with natural and urban SWOFs, and alien plants with the urban SWOFs.

The establishment of shared indicator species among these matrices denoted three different ecological groups: 1) natural native species (NAT+AGR); 2) generalist native species (URB+AGR); 3) alien species (URB+AGR). The first group included species, such as *Trifolium campestre*, *Trifolium angustifolium* and *Pistacia lentiscus*, which outlined the permanence of natural populations on SWOFs located in agricultural land-use, in the form of early (or degraded) communities such as meadows, clearings or low-shrub scrub (Bacchetta et al., 2009; Pignatti, 2019).

The second and third group were characterized by the establishment of mixed communities human-associated: synanthropic natives and invasive aliens. Among these, *Sonchus tenerrimus* and *Hedypnois rhagadioloides* characterized the native complex of those agricultural and urban SWOFs dominated by trampling, cutting, eutrophy or abandonment. In the same areas, a close (but not exclusive) association between the presence of the invasive *Oxalis pes-caprae* and agricultural and urban SWOFs were detected. Even if indicated as a strong threat to native Mediterranean ecosystems (Roy et al., 2020), in our study area the invasive *O. pes-caprae* was only present in one site in the natural and seminatural matrix, with low coverage. The distribution of this species is generally limited by the increase of elevation, irrespective of soil type, which together with the scarcity of suitable habitat and low propagule pressure (Ross et al., 2008) could explain its low presence in our natural and semi-natural SWOFs, especially in those located at higher altitude.

Considering the urban SWOFs and native communities, several generalist indicator species, such as *Hordeum murinum*, *Lolium rigidum*, *Reichardia picroides* or *Sonchus oleraceus*, were found. These species are typically widespread in a wide variety of habitats including areas subject to anthropogenic influence such as ruins, gardens and streets (Arrigoni, 2006; Pignatti, 2019).

On the other side, *Pinus halepensis* was detected as alien indicator species for urban SWOFs where it was introduced for ornamental, recreational or aesthetic purposes or other urban planning works and reforestation purposes (Arrigoni, 2006; Camarda and Valsecchi, 2008), together with other tree species, such as *Pinus pinea*, *Acacia saligna* and *Eucalyptus camaldulensis*.

Structural attribute differences among SWOFs surrounded by different land-use matrices

Structure-based indicators can be adopted to provide new knowledge and improve the effectiveness of current management strategies (Lindenmayer et al., 2000). Several studies indicated that the increase of heterogeneity and complexity of forest structure is related to higher biological diversity levels (Moning et al., 2009; Sullivan et al., 2013; e.g., Taboada et al., 2008).

In managed areas, plant structural attributes are modified by land managers which, generally, through clear-cutting and thinning lead to a regular, even-aged stand structure (Pach and Podlaski, 2015), and a decrease in the long-term abundance of deadwood (Bergeron et al., 2011), altering the biodiversity compared with irregular, unmanaged and uneven-aged woodland (Hansen et al., 1991; McComb et al., 1993).

The results of this study revealed a more diversified DBH variation in natural and agricultural SWOFs than in high-managed urban SWOFs: natural vegetation dynamics are ongoing both in natural sites and agricultural habitats, confirming the suggestions provided by indicator species and the general reforestation pattern observed in Mediterranean forests (Carranza et al., 2020; Falcucci et al., 2007) and farmlands (San Roman Sanz et al., 2013).

The detected differences are consistent with previous findings (Pach and Podlaski, 2015; see Rouvinen and Kuuluvainen, 2005), in which the variation of the structural attribute is linked to the effect and change of management practices adopted during the years (Rouvinen and Kuuluvainen, 2005).

Effect of environmental factors and structural attributes on native and alien contingents

Plant community composition can respond in very different and complex ways to the multitude of environmental conditions (Kichenin et al., 2013; Kumordzi et al., 2015) being the result of different filters, biotic and abiotic ones (Lavorel and Garnier, 2002).

Different studies highlighted the influence of spatial-topographic factors on plant community composition and diversity, such as slope (Dearborn and Danby, 2017) and latitude/altitude gradients (Xu et al., 2017). Similarly, differences in plant communities depend on species responses to bioclimatic factors, such as variation in temperature and precipitation (Guerin et al., 2019).

It is also well-recognized that fragmentation and planting practices can affect species richness (Decocq et al., 2004; Halpern and Spies, 1995), as well species composition (Becerra and Simonetti, 2013), leading to consequences such as community composition nestedness, with different and sometimes opposite patterns across fragmented landscapes, considering native and alien plants (Guirado et al., 2006; Rojas et al., 2011).

In our results, each group of predictors (spatial-topographic, bioclimatic and landscape-level variables) contributed to explaining the total variation in the composition and cover of native species, whereas only climatic factors and spatial-topographic filters were implied in explaining the compositional variation for alien species.

In particular, our analyses showed that spatial-topographical gradients were important in determining changes in native plant communities. Following the land-use intensification gradient, variations in composition and abundances of native plants turned out to be mainly related to a decrease in longitude, altitude and coastal distance, and a much lesser extent, to the distance from water bodies. As regards the compositional variability of non-native species, only lagoon and lake distance seemed to be relevant, with a pattern that outlines and reflects the gradient of human pressure (i.e., agricultural and urban SWOFs were closer to lagoon and lakes).

Considering the composition of native plant communities from SWOFs surrounded by natural areas to those surrounded by more anthropized areas, the gradient described by the main set of bioclimatic indicators, explained a good amount of variation, followed by the precipitation seasonality that explained a more negligible fraction. In particular, the transition from colder, rainier and with higher temperature ranges environments in the NE sector to the urbanized ones in the SW sector, reflect the differences in native community composition observed along the land-use intensification gradient. Similarly, but to a lesser extent, the same gradient described by these bioclimatic conditions appeared to be important in determining the compositional and abundance variations for non-native species. This link could be explained with the contribution of species like *Ceratonia siliqua* and *Olea europaea*: in fact, those archaeophytes were favored by the anthropic disturbance and the warm climate, being naturally distributed in the most thermophilic vegetation types (Bacchetta et al., 2009). The invasive neophytes *Oxalis pes-caprae* and *Acacia saligna* prefer coastal areas, and are more competitive in environments with higher disturbance (e.g., *Oxalis* is favored in the dispersion of its cloves through plowing (Del Vecchio et al., 2013; Pignatti, 2019; Ross et al., 2008). In addition, the influence of an opposite gradient, described by the seasonality of precipitation, appeared relevant to explain compositional variations of non-native species.

Moreover, our results suggest that native and alien plants respond differently to fragmentation, as aliens were not affected at all, while natives did. In particular, we highlighted that almost all adopted landscape metrics, representing the degree of fragmentation caused by the urbanization gradient, affected native plant community composition and that the response to the coefficient of variation of DBH was significant when analyzed together with landscape metrics. In this case, these findings highlighted the close relationship between the composition of native plant communities and the structural complexity of small patches, which was higher in natural and agricultural environments than in urban ones. All this, in addition to underlining the importance of patches in natural and agricultural environments, can be of help for planners of urban green areas, where anthropogenic disturbance could be partially compensated by a careful choice of species for urban reforestation to reconstruct the structural complexity observed in the natural matrices.

The alien contingent was apparently not influenced by fragmentation: this can be explained by the abundant presence of archaeophytes, perfectly integrated into native ecosystems. In this regard, previous studies suggested that landscape filters can be very important for the phase of the arrival and establishment of alien species, whereas other factors (e.g., biotic variables) gain more importance to regulate the abundance of these species, once they are present within the area (Lázaro-Lobo and Ervin, 2020).

Conclusion

TOF represent important multifunctional resources, providing fundamental ecosystem services at both global and local scales. However, the characteristics and dynamics of TOF coverage on different land-use types are poorly known and studies at local scale appear to be limited (Price et al., 2017; Schnell et al., 2015), especially considering the different types of TOF (i.e., scattered trees, small woodlots - SWOFs, trees lines).

This study, based on a classical two-phase sampling scheme (Schnell et al., 2015), has allowed the selection of a sample of 30 (7.5%) out of 400 SWOF elements, located in different land-use matrices, in which their attributes were collected and analyzed.

In respect to the land-use type, we demonstrated that small woodlots do not represent a negligible component of natural and human-impacted land-uses: the percentage of SWOF cover increased following the gradient of land-use intensification, reaching the highest value in urban coastline zones, while the average patch size decreased along the same gradient, reaching the smallest value in the urban land-use. These findings, consistently with the evident dynamics of Mediterranean forests (Carranza et al., 2020; Falcucci et al., 2007) and agricultural areas (San Roman Sanz et al., 2013), highlighted the chance to use SWOFs as possible extensions of the green infrastructure network both in natural and human-impacted land-uses (Sallustio et al., 2018).

Our results also support evidence that, although fragmentation has severe consequences, small patches like SWOFs may support a good level of naturality when surrounded by human-altered matrix and particularly, by agricultural landscapes. The observed level of naturality of agricultural landscapes, both in terms of community composition and stand-structure variation, can be considered as an opportunity for natural conversion of abandoned agricultural land to forest and consequent rewilding ecosystems (Navarro and Pereira, 2015). Notably, we observed that, along a land-use intensification gradient, the contribution of native species to the total diversity was much higher than that of alien contingent, which was a negligible component concerning the overall plant community. As expected, alien and native species responded differently and were differently affected by several structural attribute and environmental filters.

Understanding how these responses are related to each other and with different factors could improve our abilities to manage the environment. Moreover, understand which are the risk factors in human-shaped environments and the sites most threatened by, for example, invasive alien species, can help direct addressing conservation efforts both on a local and regional scale (Benedetti and Morelli, 2017).

In this framework, monitoring small woodlots outside forest should be considered a priority by landscape and urban planners, as well as by conservation researchers to promote efficient conservation efforts in understanding and preserving these small patches (Lindenmayer, 2019; Wintle et al., 2019).

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Author contributions

Erika Bazzato: Conceptualization, Methodology, Software, Formal Analysis, Investigation, Data Curation, Visualization, Writing-original draft, Writing – review & editing.

Erik Lallai: Conceptualization, Investigation, Data Curation, Visualization, Writing-original draft, Writing – review & editing

Elisa Serra: Conceptualization, Methodology, Writing-original draft, Writing – review & editing.

Maria Teresa Melis: Conceptualization, Methodology, Validation, Writing – review & editing, Supervision.

Michela Marignani: Conceptualization, Methodology, Validation, Writing – review & editing, Supervision

CHAPTER 2 - FLORISTIC AND FAUNISTIC COMPOSITION

The main aim of this chapter was to characterize the floristic and faunistic composition of the considered taxonomic groups by publishing new records for the island (vascular plants, Subchapter 2.1, paper 3; beetles, Subchapter 2.2, paper 4; spiders, Subchapter 2.3, paper 5; ants, Subchapter 2.4, paper 6) (Rosati et al. 2020; Schifani et al. 2021; Bazzato et al. submitted; Caria et al. accepted) and for Italy (spiders, Subchapter 2.3, paper 5) (Caria et al. accepted) collected during the fieldwork phase.

Subchapter 2.1: vascular plants

Paper 3

Rosati L., Fascetti S., Romano V.A., Potenza G., Lapenna M.R., Capano A., Nicoletti, P., Farris E., de Lange P.J., Del Vico E., Facioni L., Fanfarillo E., Lattanzi E., Cano-Ortiz A., Marignani M., Fogu M.C., **Bazzato E.**, Lallai E., Laface V.L.A., Musarella C.M., Spampinato G., Mei G., Misano G., Salerno G., Esposito A., Stinca A., 2020. New Chorological Data for the Italian Vascular Flora. Diversity, 12(1): 22. https://doi.org/10.3390/d12010022

Subchapter 2.2: coleoptera

Paper 4

Bazzato E., Marignani M., Ancona C., Caria M., Cillo D., Serra E. (submitted). First record of *Ozognathus cornutus* (Leconte, 1859) (Coleoptera: Ptinidae) from Sardinia, Italy. Submitted to Redia.

Subchapter 2.3: spiders

Paper 5

Caria M., Pantini P., Alamanni F., Ancona C., Cillo D., **Bazzato E**., (accepted). New records and distributional data for the Sardinian spider fauna (Arachnida, Araneae). Accepted byFragmenta entomologica

Subchapter 2.4: ants

Paper 6

Schifani E., Nalini E., Gentile G., Aleffi M., Alamanni F., Ancona C., Caria M., Cillo D., **Bazzato E.**, 2021. Ants of Sardinia: an updated checklist based on new faunistic, morphological and biogeographical notes. Redia 104, 21–35. https://doi.org/10.19263/REDIA-104.21.03.

Subchapter 2.1: vascular plants

New chorological data for the Italian vascular flora

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Abstract

Despite the importance for scientific and conservation purposes, the knowledge of the Italian territory is far from exhaustive. New chorological data for 87 vascular taxa regarding the central-southern part of Italy and its two main islands (Sicilia and Sardegna) are presented. Among these taxa, *Epilobium nummularifolium*, *Metrosideros excelsa*, and *Salvinia minima* are recorded as casual aliens for the first time in Europe (excluding Azores and Madeira for *M. excelsa*), while two native species (*Cyclamen balearicum* and *Polygala rupestris*) are new or confirmation records for Italy, respectively. Furthermore several taxa are new or confirmed at regional level. Finally, *Lathyrus cirrhosus*, *Urginea fugax*, and *Linum tenuifolium* are excluded from Italy, continental and peninsular Italy and Sardegna, respectively.

Keywords: alien species; endemic plants; checklist; new floristic records; plant diversity

Rosati L., Fascetti S., Romano V.A., Potenza G., Lapenna M.R., Capano A., Nicoletti, P., Farris E., de Lange P.J., Del Vico E., Facioni L., Fanfarillo E., Lattanzi E., Cano-Ortiz A., Marignani M., Fogu M.C., **Bazzato E.**, Lallai E., Laface V.L.A., Musarella C.M., Spampinato G., Mei G., Misano G., Salerno G., Esposito A., Stinca A., 2020. New Chorological Data for the Italian Vascular Flora. Diversity, 12(1): 22. https://doi.org/10.3390/d12010022

Introduction

Over the last 20 years floristic studies of the central-southern Italian Peninsula have greatly improved our knowledge of the flora of this region [1-15] and some syntheses at the national scale [16-22] have been made available.

However, despite the importance for scientific and conservation purposes, our knowledge of the Italian territory is far from exhaustive. In particular, in some regions of Italy, knowledge about the vascular flora remains inadequate usually because the geographic distributions of many species are poorly understood resulting in gaps in our knowledge of their actual extent [23]. This is the so-called “Wallacean shortfall”, that is considered one of the crucial sources of uncertainty in biodiversity inventories, limiting both biosystematic and ecological analyses of these [24,25] and as well as resulting in some exceptional discoveries when critical gaps are surveyed, such as the finding of a new taxon of *Gymnospermium* (Berberidaceae) in an unexplored mountain range between Basilicata and Campania [26,27]. Further biological invasions represent a major threat to biodiversity [28,29], particularly as new naturalizations of alien vascular plants are continuously happening, and are usually poorly documented within regional flora treatments. This is particularly the case on Mediterranean islands [30]. As a consequence, it is important that frequent assessment of regional floras are undertaken, not only to monitor established populations (indigenous and naturalized), but also to update distribution data to enable early warning of declines in indigenous or endemic species, and to document new arrivals of invasive plants, as an aid to better manage the invasive ones, particularly those species included in the European list of EU concern [31].

In this paper we document new observations of plants for the central-southern Italian Peninsula and the two main insular regions (Figure 1). Our records include both native and alien non-cultivated species with a careful assessment of the status of naturalization for the latter. For each taxon, relevant information about ecology and distribution are also provided.

Materials and Methods

The floristic data are based on field investigation carried out by the authors in the study area, as well as on herbaria and literature surveys. The collected or examined materials are preserved in public (ANC, AK, HLUC, PORUN-Herb. Stinca, REGGIO, SS; acronyms according to Thiers [32]) or private herbaria. In the floristic list (Appendix A), taxa are arranged in alphabetical order. Nomenclature, taxa delimitation and notes on the regional distribution follow the recent checklists of the Italian vascular flora [16,17] and their updates (33-40). Nomenclature of the species that had not been previously reported from Italy follows World Flora Online [41]. The taxa recorded in this paper were identified using Flora Europaea [42-46], Flora d’Italia [18-21,47] and some monographic works cited in the species' note listed in Appendix A.

For each species the following information is provided: basionym and most relevant synonyms; plant family; life form, attributed on the basis of field observations using the categories of Flora d’Italia [47]; native range; period of introduction (archaeophyte or neophyte); data report in the study area; current invasiveness status for each region, assessed by population monitoring over time according to the terminology of Pyšek et al. [48]; date and discovery localities with details on the location (municipality, administrative province), habitat, altitude, decimal degrees geographic coordinates (datum WGS84); collector(s) (*legit*), author(s) of the identification (*determinavit*) and, eventually, the identity confirmation (*confirmavit*); herbarium where specimen is kept; additional notes. Species not documented with a herbarium specimen are showed in Figure 2.

Results

Two native and four alien species are recorded for the first time in Italy; among the aliens, three species deserve more attention since they are new for Europe. Overall, 87 taxa of floristic interest for the Italian flora are herein reported, 50 are native and 37 represent alien taxa. Among the native plants seven are endemic taxa of the Italian territory.

Native taxa

Cyclamen balearicum and *Polygala rupestris* discovered in Sardegna are recorded for the first time in Italy, while *Euphorbia stricta* found in Basilicata is recorded as new for peninsular Italy (i.e., the regions south to Toscana and Marche, Sicilia and Sardegna excluded). *Lathyrus cirrhosus,* previously reported by [49], is here excluded from the Sardinian and Italian flora.

In the same way, *Linum tenuifolium* reported from Sardegna [50] and *Urginea fugax* from Basilicata [51] are not confirmed throughout this study. These species seem to have been recorded by mistake and are here excluded from the floras of these regions.

Thirty-three taxa are new for the regional floras of the study area: six were found in Lazio (*Dichoropetalum carvifolium-chabraei*, *Geranium pusillum*, *Hieracium pallescens s.l.*, *Lathyrus inconspicuus, Oeosporangium tinaei* and *Veronica spicata* subsp. *spicata*), six in Campania (*Bupleurum rollii, Kickxia spuria* subsp. *spuria, Lupinus albus* subsp. *graecus*, *Plantago argentea* subsp. *argentea*, *Potamogeton pusillus* and *Rumex maritimus*), 17 in Basilicata (*Alyssum diffusum* subsp. *garganicum, Bellevalia boissieri, Convolvulus siculus* subsp. *siculus, Daucus aureus, Epipactis schubertiorum*, *Eragrostis barrelieri* subsp. *barrelieri*, *Euphorbia stricta*, *Filago asterisciflora*, *Galanthus reginae-olgae* subsp. *vernalis, Isoetes duriei*, *Melampyrum barbatum* subsp. *carstiense, Ophrys crabronifera, Ophrys marmorata, Ophrys sipontensis, Quercus petraea* subsp. *austrotyrrhenica*, *Rosa subcollina* and *Sempervivum tectorum*) and four in Sardegna (*Asplenium viride, Himantoglossum hircinum, Isopyrum thalictroides* and *Oloptum miliaceum*).

In addition two hybrids within the genus *Rosa* are reported in Basilicata for the first time (*R.* × *bishopii* and *R. montana* × *R. villosa*) and one (*Populus* × *canescens*) in Sardegna. We also confirm ten taxa as present in the regions covered by this paper: *Prangos ferulacea* in Lazio; *Pedicularis petiolaris, Potamogeton pusillus, Puccinellia festuciformis* subsp. *festuciformis* and *Viola kitaibeliana* in Campania; *Trifolium spumosum* in Basilicata; *Nigella arvensis* subsp. *glaucescens* in Basilicata; *Melampyrum barbatum* subsp. *carstiense* and *Nigella arvensis* subsp. *glaucescens* in Puglia, *Muscari parviflorum* and *Ophrys speculum* in Calabria region. Finally, for one species (*Impatiens noli-tangere*), reported for the first time in Basilicata in the recent checklist of the italian vascular flora [16] on the basis of our unpublished data, full details about the record are provided.

Alien taxa

Three alien species are recognized new for Europe (excluding Azores and Madeira for *M. excelsa*) and for Italy: *Epilobium nummularifolium, Metrosideros excelsa,* found in Sardegna, and *Salvinia minima,* found in Calabria. In the same regions, we found *Coprosma repens, Pittosporum crassifolium (*Sardegna*)* recorded for the first time in Italy, and *Commelina erecta,* discovered in Calabria, recorded for the first time in Italy outside Sicilia. In addition to those species, thirty-one new alien vascular species are reported at regional level (two of these are present in two regions): two in Lazio (*Aubrieta columnae* subsp. *italica* and *Campanula poscharskyana*); two in Campania

(*Conringia orientalis* and *Pinus nigra* subsp. *laricio*); twelve in Basilicata *(Anredera cordifolia, Chamaerops humilis, Euphorbia nutans, Fagopyrum esculentum, Ficus microcarpa, Gleditsia triacanthos, Kalanchoë ×houghtonii, Lonicera japonica, Paspalum notatum, Phoenix canariensis, Pinus nigra* subsp. *laricio* and *Tagetes erecta*); twelve in Calabria (*Asclepias physocarpa*, *Cenchrus longisetus*, *Commelina erecta*, *Nicandra physalodes*, *Nothoscordum gracile*, *Oxalis debilis*, *Oxalis latifolia*, *Phyllostachys aurea*, *Physalis peruviana*, *Plumbago auriculata*, *Solandra maxima* and *Wisteria sinensis*); one in Sicilia (*Solandra maxima*); four in Sardegna (*Cordyline australis*, *Euphorbia nutans*, *Glandularia tenera* and *Lantana montevidensis*). Furthermore the presence of *Phyla nodiflora* in Marche, *Fagopyrum esculentum* in Calabria (herein recorded as new also in Basilicata) and *Tetragonia tetragonoides* in Sardegna is confirmed.

Figure 1. Research area with administrative Italian regions and locations of the new records of vascular plants reported in this study. We considered “continental“ Italy the regions north to Toscana and Marche, “peninsular Italy” the other regions of Italian peninsula and “insular Italy” the Sardegna (Sardinia) and Sicilia (Sicily) regions.

Discussion

We reported 50 native taxa for the regions covered in this paper, including several endemic Italian taxa that are range extensions from their previous reported limits. The discovery in Sardegna of *Cyclamen balearicum* and *Polygala rupestris*, two species that had until now not been recorded for Italy, is particularly relevant because their presence provides additional support for the biogeographical link between Sardegna and the W-Mediterranean areas [52-54].

In the same way, the numerous neophytes reported for the first time in the regional floras once again confirm that alien species are continuously invading the environment. Although the majority of the recorded species have been evaluated, with respect to their naturalization degrees as “casual”, their monitoring over time is considered of primary importance in order to be able to effectively intervene if these species prove to be highly invasive. One especially noteworthy discovery is *Salvinia minima*, a species known to be invasive in most of the Southern United States, where it is a significant threat to aquatic systems [55]. *Salvinia minima* is widely available for purchase for aquaria and water-gardens in Europe. Accordingly, we think that this species established in Italy from the aquatic plant trade, probably as an accidental introduction. The high number of floristic records of international and regional relevance provided , after a very short time from the publication of the latest Italian vascular flora checklist [16,17], even if not derived from a systematic sampling activity, testify of the importance to continuously implement our knowledge of plant diversity, especially for conservation purposes, studying and reviewing herbaria collections, coupled with frequent field surveys, with special attention, to achieve an early warning, to monitor introduction and naturalization of alien taxa.

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We are gretful to: G. Gottsclich for examining an *Hieracium* specimen; F. Ripullone and the project “Indagine conoscitiva e valorizzazione dell’area del bosco comunale Finocchio” for the supporting a botanical survey in the territory of Rotondella (Basilicata region).

Author contributions

Conceptualization, L.R., A.S., and E.FAR.; investigation, L.R., VA.R., G.P., MR.L., A.CA., P.C., E.F., PJ.dL., E.D., L.F., E.FAR., E.L., A-CO., M.M., MC.F., E.B., E.L., VLA.L., CM.M., G.SP., G.ME., G.MI., G.SA., A.E., A.S. writing—original draft preparation, L.R., A.S., CM.M., PJ.dL.and E.FAR.; review, all the Authors.

Subchapter 2.2: coleoptera

First record of *Ozognathus cornutus* (leconte, 1859) (Coleoptera: Ptinidae) from Sardinia, Italy

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Abstract

Ozognathus cornutus (LeConte, 1859) is recorded on the invasive alien tree *Robinia pseudocacia* L. (Fabaceae) in an urban area in Italy. The species has already been reported in two other Italian regions, but this is the first record for the Sardinian fauna. Due to the great adaptability of this alien species and the increasingly cosmopolitan trend, we suggest monitoring it with caution for early detection and to plan an appropriate rapid management response.

Keywords: Bostrichoidea, Ernobiinae, Alien species, *Robinia pseudoacacia*

Bazzato E., Marignani M., Ancona C., Caria M., Cillo D., Serra E. (submitted). First record of *Ozognathus cornutus* (Leconte, 1859) (Coleoptera: Ptinidae) from Sardinia, Italy. Submitted to Redia.

Introduction

The genus *Ozognathus* LeConte, 1861 (Bostrichoidea, Ptinidae, Ernobiinae) currently includes twelve described species from the Nearctic and Neotropical regions (Zahradník & Mifsud, 2005).

Among these, *Ozognathus cornutus* (LeConte, 1859) was described in California (USA) (LeConte, 1859, as *Anobium cornutum*); its type specimen is deposited in the California Academy of Sciences (White, 1982). Confirmed in California (White, 1982), in the areas adjacent to Mexico, in South America (e.g., in Chile; Honor & Rothmann, 2017), as well as in Réunion in the Indian Ocean (Lemagnen, 2013), this species recently spread to Australia (Sydney, New South Wales) (Plant Health Australia, 2019), New Zealand (Bercedo et al., 2005), Israel (Miłkowski, 2019), Tunisia (Zahradník & Mifsud, 2005) and, recently, in Europe. It has been recorded in Latvia (around Ulbroka) (Telnov et al., 2016), Great Britain (Eccles) (Stenhouse, 2017), Germany (Tübingen), Switzerland (Zurich) (Germann & Schmidt, 2017; Chittaro & Sanchez, 2019), southern France (Pourcieux) (Allemand et al., 2008), Spain (Cadiz, Catalonia, Alicante, Valencia) (Bercedo et al., 2005; Viñolas & Verdugo, 2012, Trócoli et al., 2020), Madeira Island (Funchal) (Zahradník & Mifsud, 2005), Gibraltar (GONHS, 2020), Canary Islands (Island of La Palma and Island of Tenerife) (Viñolas et al., 2018), and Malta (Marsa, TalMuxar, Żejtun) (Zahradník & Mifsud, 2005). The first published Italian records were reported from Sicily (Palermo) and Campania (Portici) regions by Cusimano et al. (2014) and subsequently reconfirmed, without further details, for Sicily by Sidoti et al. (2016).

We provide the first records of *Ozognatus cornutus* (LeConte, 1859) in Sardinia based on an intensive sampling conducted in Southern-Sardinia from April 2018 to December 2019.

Materials and methods

Study area

Sardinia, one Mediterranean hotspot for biodiversity conservation (Marignani et al., 2017a, b; Médail, 2017), located in the middle of the Mediterranean Sea, is the second-largest Mediterranean island, after Sicily, covering a surface area of around 24,000 km^2 (Palumbo et al., 2020). Thanks to its variety of landforms, complex orographic patterns (with hilly lands, plateaus, mountain and plains), heterogeneous geological substrata and climate variability (Bazzato et al., 2021), the island is characterized by high levels of biodiversity and different vegetation types (Bacchetta et al., 2009).

We collected the species during a field campaign aimed to investigate the impact of land-use matrices on plants and arthropod communities of Small Woodlots Outside Forests (hereafter, SWOFs) in the Metropolitan City of Cagliari (Palumbo et al., 2020; E. Bazzato, PhD dissertation 2021). We randomly selected 30 SWOFs (Fig. I) ranging from 0.1 to 0.5 hectares, by means of a stratified random sampling design in proportion to the number of target SWOFs present in each land-use stratum: 11 sites in natural and semi-natural (NAT) and agricultural (AGR) areas, 8 in urban and artificial surfaces (URB). All sampled sites were listed following the site code assigned based on the correspondent land-use stratum (Tab. 1).

Data collection

In the centroid of each SWOF, we placed at 5−10 meters above ground a Cross-vanes Window Flight Trap (CWFT, Fig II) for a total of 30 traps. Traps were active starting from July to October for the first year (2018), and from June to December for the second year (2019). Traps were re-triggered every 30−40 days (except for the last period ranging from August to December 2019) with ethylene glycol as non-attractive liquid preservative. All specimens from each site and trap were sorted and stored by the authors (EB, MC, CA). The determination is still in progress.

Identification and nomenclature

Specimens were examined using an Optika SZM-T stereomicroscope. They were morphologically identified by the authors (EB, CA, DC) using photographs reported by Zahradník & Mifsud (2005), as well as dichotomous key provided by Fall (1905). Habitus photograph was taken with a Pentax K7 digital camera attached to a Optika SZM-T stereoscope (Fig. III). Dry specimens were prepared and deposited in the private collection of the first and fifth authors (EB & DC).

The species has been reported on several host plant species (Tab. 2): nomenclature of host plant species follows Bartolucci et al. (2018, 2020), Galasso et al. (2018, 2020), Freiberg et al. (2020), and Rosati et al. (2020).

Fig. I - Study area located in the Metropolitan City of Cagliari (Southern Sardinia, Italy), characterized by a gradient of land-use intensification from natural and semi-natural areas to urbanized coastline zones.

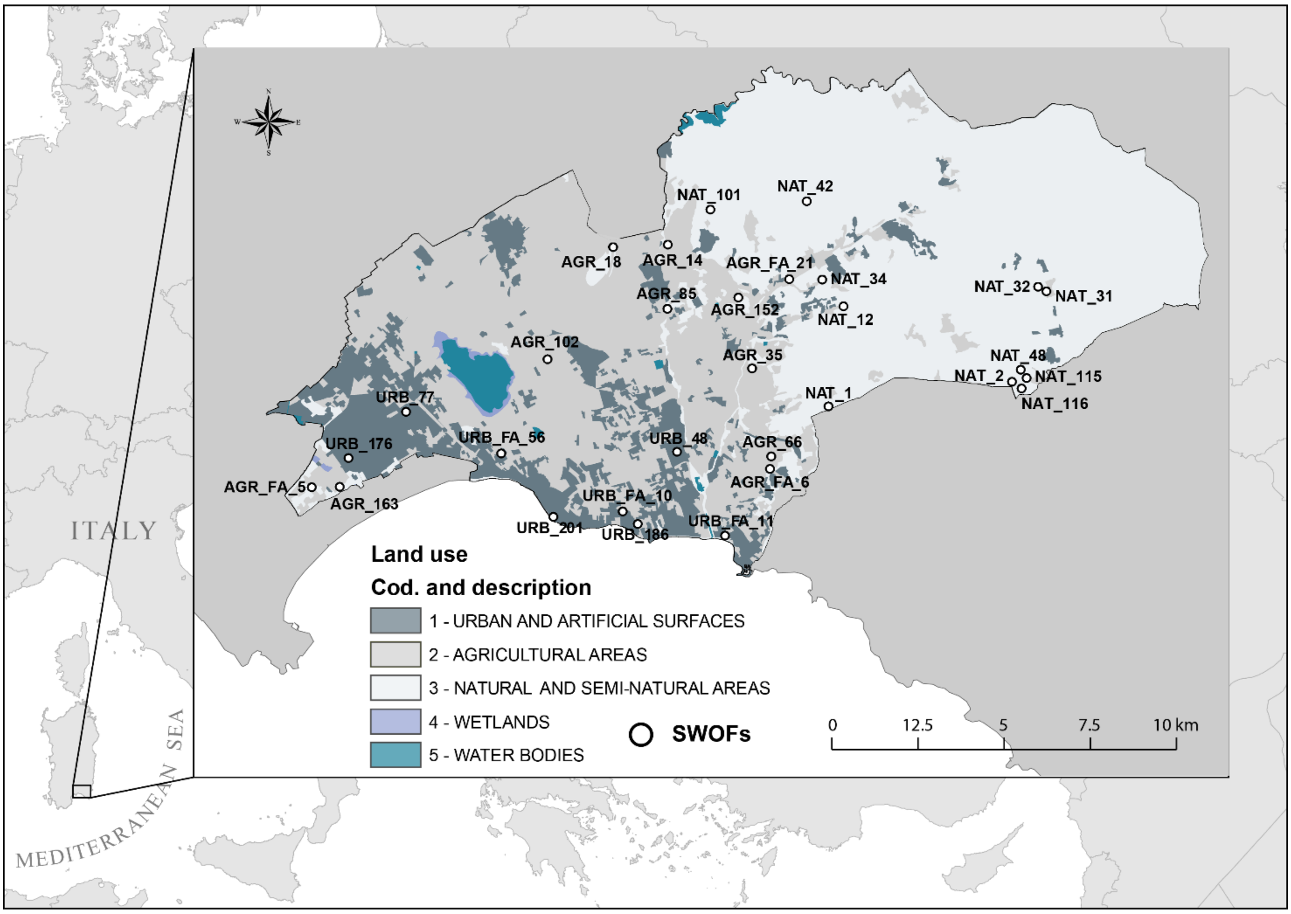


Fig. II - Cross-vanes Window Flight Trap (CWFT) placed on *Robinia pseudoacacia* within the SWOF located in Quartu Sant’Elena (Is Arenas, Via Pizzetti; site code URB 176) (photo by E. Bazzato).

Table 1 - List of the 30 Small Woodlots Outside Forests investigated along the gradient of land-use intensification. Municipality, locality, coordinates (expressed as metric units in the Monte Mario/Italy Zone 1 reference system, EPSG 3003), altitude (m a.s.l.) and plant species hosting the cross-vanes window flight trap are listed.

Site code	Municipality	Locality	E	N	Altitude	Family	Plant species
AGR_14	Maracalagonis	Corongiu, Sirigragiu	1524102	4348373	81	Myrtaceae	*Eucalyptus tereticornis* Sm.
AGR_18	Maracalagonis	Corongiu, Carroghedda	1522637	4348319	85	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
AGR_35	Quartucciu	Piscina Nuxedda	1526348	4345063	52	Oleaceae	*Olea europaea* L.
AGR_66	Quartu S.E.	Cani Nieddu, Frapponti,	1526881	4342688	84	Oleaceae	*Olea europaea* L.
AGR_85	Maracalagonis	Corongiu	1524097	4346644	60	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
AGR_102	Quartu S.E.	Simbirizzi, Sa Guardia Lada	1520859	4345293	35	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
AGR_152	Maracalagonis	Gruxi Lillius, Bacca Aruis	1525980	4346952	99	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
AGR_163	Quartu S.E.	Stagno di Quartu, C. D'Aquila	1515340	4341892	3	Oleaceae	*Olea europaea* L.
AGR_FA_6	Quartu S.E.	Str. Comunale Cani Nieddu	1526824	4342368	68	Oleaceae	*Olea europaea* L.
AGR_FA_21	Maracalagonis	Riu Piscina Nuxedda	1527337	4347432	80	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
NAT_1	Quartucciu	Corti de Perda	1528394	4344041	120	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
NAT_2	Maracalagonis	Sette Fratelli, Codoleddu	1533610	4344508	700	Salicaceae	*Salix atrocinerea* Brot. subsp. *atrocinerea*
NAT_12	Maracalagonis	Riu Monte Nieddu	1528748	4346698	140	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*

NAT_31	Sinnai	Sette Fratelli, Monte Cresia	1534230	4347117	663	Ericaceae	*Arbutus unedo* L.
NAT_32	Sinnai	Sette Fratelli, Monte Cresia	1534003	4347255	677	Ericaceae	*Arbutus unedo* L.
NAT_34	Maracalagonis	Villagio dei Gigli	1528227	4347442	120	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
NAT_42	Sinnai	Burranca	1527821	4349531	150	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
NAT_48	Maracalagonis	Sette Fratelli, Codoleddu	1533816	4344788	714	Salicaceae	*Salix atrocinerea* Brot. subsp. *atrocinerea*
NAT_101	Maracalagonis	Corongiu, Sedda Brandanu	1525242	4349318	140	Salicaceae	*Populus canescens* (Aiton) Sm.
NAT_115	Maracalagonis	Sette Fratelli, Codoleddu	1533989	4344598	706	Salicaceae	*Salix atrocinerea* Brot. subsp. *atrocinerea*
NAT_116	Maracalagonis	Sette Fratelli, Codoleddu	1533830	4344308	700	Fagaceae	*Quercus suber* L.
AGR_FA_5	Quartu S.E.	Stagno di Quartu, Bingia Spada	1514588	4341874	5	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
URB_48	Quartu S.E.	Via delle Bouganvillee	1524342	4342823	22	Tamaricaceae	*Tamarix canariensis* Willd.
URB_77	Quartu S.E.	Sant'Antonio, Via Belgio	1517111	4343897	9	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
URB_176	Quartu S.E.	Is Arenas, Via Pizzetti	1515564	4342648	6	Fabaceae	*Robinia pseudoacacia* L.
URB_186	Quartu S.E.	Sant'Andrea, Via Rimini	1523304	4340905	4	Pinaceae	*Pinus halepensis* Mill. subsp. *halepensis*
URB_201	Quartu S.E.	Foxi, Via IschiaAll	1521044	4341091	1	Oleaceae	*Olea europaea* L.
URB_FA_10	Quartu S.E.	Porticciolo, Via Riccione	1522893	4341230	11	Pinaceae	*Pinus pinea* L.
URB_FA_11	Quartu S.E.	Via Lago di Varese	1525605	4340594	2	Myrtaceae	*Eucalyptus camaldulensis* Dehnh. subsp. *camaldulensis*
URB_FA_56	Quartu S.E.	Margine Rosso, Via Valenzia	1519664	4342794	37	Pinaceae	*Pinus halepensis* Mill. subsp. *halepensis*

Results and discussion

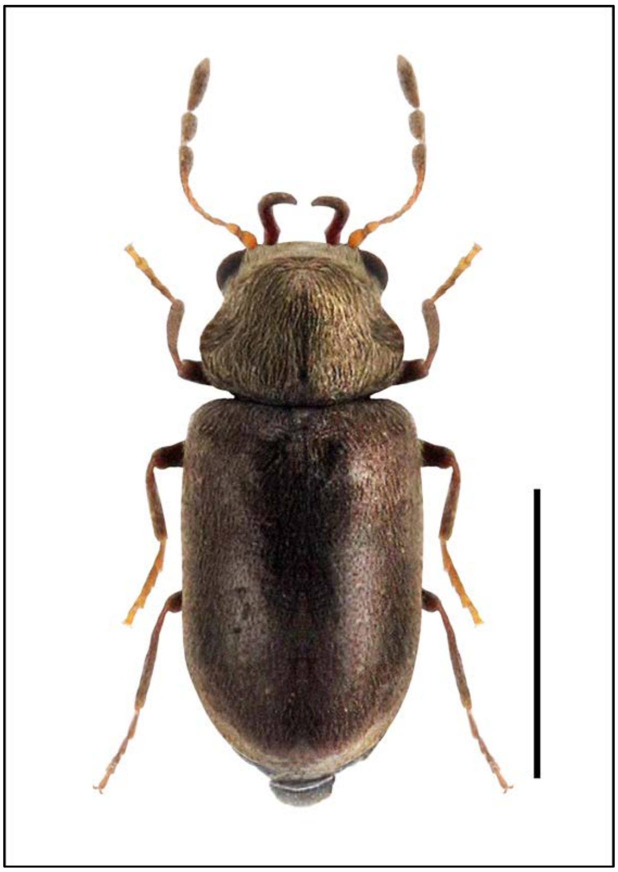
New species records

Ozognathus cornutus (LeConte, 1859) (Fig. III, ♂)

Examined material: SARDINIA: Quartu Sant’Elena (Cagliari province), Is Arenas, Via Pizzetti, site code URB 176: 19.VII–19.VIII.2019, 1 ♂ window trap on *Robinia pseudoacacia*, E. Bazzato, M. Caria & C. Ancona legerunt, det. E. Bazzato & C. Ancona; 19.VIII–03.XII.2019, 1 ♂ window trap on *Robinia pseudoacacia*, E. Bazzato, M. Caria & C. Ancona legit, det. E. Bazzato & D. Cillo.

The biology of *O. cornutus* is poorly known (Stenhouse, 2017; Vinolas, 2017), although recently its distribution range increased considerably in the temperate areas of the Euro-Mediterranean region, with the possibility of a further spread in Mediterranean territories in a relatively short time (Zahradník & Mifsud, 2005; Vinolas, 2017; Miłkowski, 2019). This species is known as polyphagous and easily adapts to a multitude of climates and a wide variety of microbiotopes (dried fruit, galls produced by insects), herbaceous plants, as well as in the bark and wood of various deciduous and coniferous trees (Stenhouse, 2017; Vinolas, 2017). The species seems to spread easily: in Switzerland, some specimens hatched from a head of garlic (*Allium sativum)* from Sicily bought on a market in Zurich (Chittaro & Sanchez 2019; Germann & Schmidt, 2017).

Fig. III – Habitus of *Ozognathus cornutus* (LeConte, 1859) ♂ (photo by C. Ancona; scale bar 1 mm).



Given its known adaptability, it is not surprising that *O. cornutus* has been found in different part of plants and different species. In particular, previous records were mainly attributable to Asteraceae family (Tab. 2), for a total of six species belonging to six genera (*Achillea* L., *Argyranthemum* Webb, *Baccharis* L., *Helianthus* L., *Scolymus* L., *Silybum* Vaill.) and secondly, to Fagaceae family, with four species belonging to a genus (*Quercus* L.). Furthermore, other families fairly represented by these records are Araucariaceae and Rosaceae, each with three species (Tab. 2).

To a lesser extent, with one or two species, Acanthanceae, Aizoaceae, Amaryllidiaceae, Arecaceae, Ebenaceae, Euphorbiaceae, Fabaceae, Lauraceae, Lythraceae, Moraceae, Oleaceae, Passifloraceae, Pinaceae, Rutaceae, Proteaceae and Simmondsiaceae families are mentioned (Tab. 2).

We recorded samples from 30 cross-vanes window flight traps placed in 30 individual trees belonging to 8 families, for a total of 9 genera and 11 different species considered (Tab. 1). Data reported were gathered in 30 randomly selected small woodlots outside forest along an area with an increasing urbanization gradient: notably, we found two individuals of *O. cornutus* only in one of those traps, placed on the invasive alien tree *Robinia pseudocacia* L., native to eastern North America, (Westbrooks, 1998; Rejmánek & Richardson, 2013), within a SWOF located in a public garden of the urban area, at the extreme of our gradient of land-use intensification.

Table 2 - List of host plant species and families reported for *O. cornutus.*

Family	**Current accepted host plant name**	**Reference and host plant name originally reported**
Acanthanceae	*Acanthus mollis* L.	Lüer, 2020, as *Acanthus mollis* L.
Aizoaceae	*Carpobrotus edulis* (L.) N.E.Br.	Knapp, 2014, as *Carpobrotus edulis* (L.) N.E.Brown, 1926
Amaryllidaceae	*Allium sativum* L.	Germann & Schmidt, 2017, as *Allium sativum*
Apiaceae	*Eryngium campestre* L.	Yus Ramos et al., 2019, as *Eryngium campestre* L.
Apiaceae	*Foeniculum vulgare* Mill.	Bercedo et al., 2005, as *Foeniculum vulgare*
Araliaceae	*Schefflera arboricola* (Hayata) Merr.	Trócoli et al., 2020, as *Schefflera arboricola* (Hayata) Merr. 1916
Araucariaceae	*Araucaria angustifolia* (Bertol.) Kuntze	Lüer, 2020, as *Araucaria angustifolia* (Bertol.) Kuntze
Araucariaceae	*Araucaria bidwillii* Hook.	Lüer, 2020, as *Araucaria bidwillii* Hook.
Araucariaceae	*Araucaria heterophylla* (Salisb.) Franco	Lüer, 2020, as *Araucaria heterophylla* (Salisb.) Franco
Arecaceae	*Phoenix dactylifera* L.	Trócoli et al., 2020, as *Phoenix dactylifera* L. 1753
Asteraceae	//	Bercedo et al., 2005, as Cardo
Asteraceae	*Achillea millefolium* L.	Trócoli et al., 2021, as *Achillea millefolium* L., 1753
Asteraceae	*Argyranthemum* spp.	García et al., 2016, as *Argyranthemum* spp
Asteraceae	*Baccharis pilularis* D.C.	Tilden, 1951, as *Baccharis pilularis* De Candolle
Asteraceae	*Helianthus annuus* L.	Lüer, 2020, as *Helianthus annuus* L.
Asteraceae	*Scolymus hispanicus* L.	Bercedo et al., 2005, as *Scolymus hispanicus*
Asteraceae	*Silybum marianum* (L.) Gaertn.	García et al., 2016, as *Silybum marianum* (L.) Gaertn.
Betulaceae	*Alnus* sp.	Viñolas & Verdugo, 2011, as *Alnus* sp.
Ebenaceae	*Diospyros kaki* L.f.	Lüer, 2020, as *Diospyros kaki* L. f.
Euphorbiaceae	*Euphorbia characias* L.	Trócoli et al., 2020, as *Euphorbia characias* L. 1753
Fabaceae	*Retama monosperma* (L.) Boiss.	Bercedo et al., 2005, as *Retama monosperma*; García et al., 2016, as *Retama rhodorhizoides* Webb & Berthel.
Fabaceae	***Robinia pseudoacacia* L.**	**Present work**
Fabaceae	*Tamarindus indica* L.	Trócoli et al., 2020, as *Tamarindus indica* L. 1753
Fagaceae	*Quercus agrifolia* Née	Trócoli et al., 2020, as *Quercus agrifolia* Neé, 1801
Fagaceae	*Quercus crassipes* Kunth	Viñolas, 2017, as *Quercus crassipes* Bonpl.
Fagaceae	*Quercus obtusata* Kunth	Viñolas, 2017, as *Quercus obtusata* Bonpl.
Fagaceae	*Quercus suber* L.	Bercedo et al., 2005, as *Quercus suber* L.
Lauraceae	*Persea americana* Mill.	Pence, 1950; Ebeling, 1959, as Avocado
Lythraceae	*Punica granatum* L.	Lüer, 2020, as *Punica granatum* L.
Moraceae	*Ficus carica* L.	Bercedo et al., 2005, as *Ficus carica*; Miłkowski, 2019, as *Ficus carica* L.
Oleaceae	*Fraxinus angustifolia* subsp. *syriaca* (Boiss.) Yalt.	Miłkowski, 2019, as *Fraxinus angustifolia* subsp. *syriaca* (Boiss.)
Passifloraceae	*Passiflora caerulea* L.	Lüer, 2020, as *Passiflora caerulea* L.
Pinaceae	*Pinus* sp.	White, 1982; Philips, 2002, as Pine
Proteaceae	*Macadamia* sp.	USDA, 1970, as Macadamia
Rosaceae	*Prunus dulcis* (Mill.) D.A. Webb	Allemand et al., 2008, as *Prunus dulcis*
Rosaceae	*Rhaphiolepis bibas* (Lour.) Galasso & Banfi	USDA, 1964, as *Locus tree*; Trócoli et al., 2020, as *Eriobotrya japonica* (Thunb.) Lindl.
Rosaceae	*Rosa* sp.	Lüer, 2020, as *Rosa* sp.
Rutaceae	*Citrus limon* (L.) Osbeck	Pence, 1950, as Lemon
Simmondsiaceae	*Simmondsia chinensis* (Link) C.K.Schneid.	Pinto & Frommer, 1980, as *Simmondsia chinensis* (Link) Schneider

Our finding, as well as being the first record from Sardinia island, reports an association with an invasive alien plant species not yet reported in other studies. Furthermore, although *O. cornutus* is able to colonize different environmental condition and climates, finding it only in an urban area suggests that its introduction could have occurred accidentally and in recent times, mainly due to the international trade in fruit, vegetables and alien plants (Cusimano et al., 2014).

Data on the presence of alien species are important since early detection and rapid response are key components for the successful management of Invasive Alien Species (Coughlan et al., 2020): our data suggest a relatively recent introduction on the island and the hypothesis of a possible rapid expansion of this species into other areas.

Conclusions

Monitoring of newly introduced species is of great importance for preventing new biological invasions, which can cause damage to biodiversity, economy and human well-being (Genovesi & Shine, 2004). As for now, *O. cornutus* is considered to be harmful to heritage works (Manachini, 2017); nevertheless, although there is no particular evidence in the literature of phytosanitary emergencies caused by *O. cornutus* (Pence, 1950; Bercedo et al., 2005), due to the behavior of the species (polyphagia, adaptability, marked propensity to passive transport), its rapid expansion in other urban areas up to natural ones cannot be excluded. Hence, this species deserves to be monitored with caution and, in case of discovery, it would be good practice to inform the competent bodies of the sector by providing location data to monitor the dispersal capacity of the species and evaluate the expansion of its distribution range.

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Author contributions

Conception and design of field research: Bazzato E.; material preparation and data collection: Bazzato E., Caria M. & Ancona C.; determination of species: Bazzato E., Ancona C., Cillo C.; conception and design of paper: Bazzato E., Serra E.; drafting the article: Bazzato E., Serra E., Ancona C., Marignani M.; revising the article for important intellectual content: Cillo D., Marignani M.

All authors contributed to read and approve the final version of the manuscript.

Subchapter 2.3: spiders

New records and distributional data for the Sardinian spider fauna (Arachnida, Araneae)

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Abstract

We present new distributional data of 24 spider species in Sardinia, including 7 endemics and 1 poorly recorded. Sixteen species from 7 families and six species from 2 families are recorded for the first time in Sardinia and Italy, respectively. Among these, the reported presence of *Cepheia longiseta* and five other species allows us to include Synaphridae family and five genera (*Anagraphis*, *Hahnia*, *Minyriolus*, *Thaumatoncus*, *Trabea*) in the Sardinian spider checklist, and two genera (*Anagraphis* and *Thaumatoncus*) in the Italian spider checklist. Relevant faunistic and distribution notes of poorly collected species are also provided.

Keywords: biogeography, endemism, Synaphridae, Sardinia, Italy

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Introduction

Spiders are very abundant and highly diversified generalist predators widespread in almost every terrestrial environment of the entire world (Foelix 2011; Turnbull 1973; Wise 1993). For these reasons, they are important elements in natural or even artificial systems, for example, due to their significance in biological pest control (Sunderland 1999).

Even though the spider fauna of Italy is relatively well catalogued, certain regions in the South remain poorly studied, and other studies in the Mediterranean areas are needed (Pantini & Isaia 2019).

For Sardinia, the second-largest island of the Mediterranean basin, a comprehensive checklist of spider fauna was published by Pantini et al. (2013) and included 495 taxa. Later on, 21 papers about Sardinian spider fauna, or including Sardinian material, have been published. In 2013, after the publication of the catalogue, Bolzern et al. (2013) and Planas et al. (2013) published phylogenetic and molecular data with taxonomical implications respectively for the *Tegenaria-Malthonica* complex and the genus *Lycosa*, whereas Colombo & Manunza (2013) reported the first case of malacophagy in *Cteniza sauvagesi* (Rossi, 1788). The next year, the Sardinian endemic mygalomorph *Amblyocarenum nuragicum* Decae, Colombo & Manunza, 2014 has been described. Later on, Logunov (2015) reported the first records of the salticids *Macaroeris flavicomis* (Simon, 1884) and *Pseudomogrus gavdos* (Logunov & Marusik, 2003), and Bosmans & Colombo (2015) contributed recording 14 new species and describing the linyphiid *Centromerus isaiai* Bosmans, 2015. In the same year, the Sardinian endemic linyphiid *Centromerus marciai* Bosmans & Gasparo (2015) and the Sardinian endemic mygalomorph *Nemesia asterix* Decae & Huber (2017) have been described. Thereafter, Bosmans et al. (2018) recorded *Haplodrassus rhodanicus* (Simon, 1914) and *H. typhon* (Simon, 1878) and described *H. securifer* Bosmans & Abrous, 2018 based on Apulian, Sardinian and Sicilian specimens. In the same year, the gnaphosid spider *Marinarozelotes huberti* (Platnick & Murphy, 1984) was recorded by Mulas & Ruiu (2018). Furthermore, Lunghi (2018) published a paper about the ecology of *Meta bourneti* Simon, 1922 from Monte Albo. Lastly, a revision of the genus *Zodarion* allowed Bosmans et al. (2019) to describe two new endemic species, *Zodarion pantaleonii* Bosmans & Pantini, 2019 and *Z. pseduonigriceps* Bosmans & Pantini, 2019.

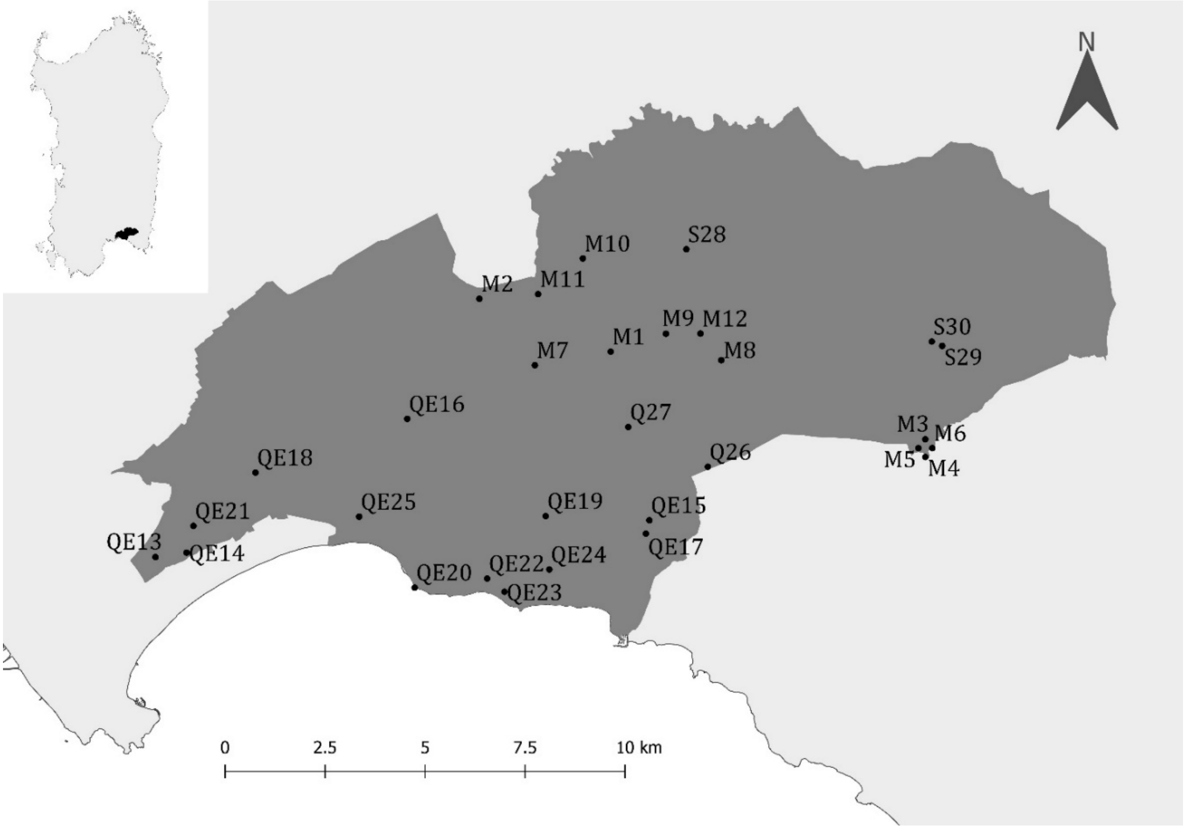
After these numerous studies, a synthesis of all available published scientific information brought the total number of Sardinian spiders up to 522 species (Pantini & Isaia 2019).

With this paper, we provide new distributional data of 24 taxa from 12 families for the Sardinian spider fauna. Sixteen species from 7 families and six species from 2 families are recorded for the first time in Sardinia and Italy, respectively. Among these, the first record of *Cepheia longiseta* and five other species allows us to include Synaphridae family and five genera (*Anagraphis, Hahnia*, *Minyriolus*, *Thaumatoncus*, *Trabea*) in the Sardinian spider checklist, and two genera (*Anagraphis* and *Thaumatoncus*) in the Italian spider checklist. Relevant faunistic and distribution notes of poorly collected species are also provided.

Thanks to some intensive samplings conducted in Sardinia from April 2018 to May 2019, our paper provides new contributions to the knowledge of spider species distribution in the Sardinian island, but also in the Italian territory.

Materials and methods

Fig. 1 – Map of the localities where the specimens have been collected; the area marked in grey correspond with the study area where the pitfall traps have been placed.



Most of the records in the present paper originate from sampling efforts in southern Sardinia (Fig. 1) aimed at investigating the impact of land-use matrices on plants and arthropod communities in 30 small woodlots outside forests (Bazzato et al. in preparation). The 30 different localities fall within four municipalities of the Metropolitan City of Cagliari (Table 1; Fig.1), a medium-sized functional urban area characterized by three levels of fragmentation degrees (Palumbo et al. 2020).

Specimens were collected by pitfall traps. Five pitfall traps were located in each small woodlot, for a total of 150. Traps were active starting from April 2018 to May 2019 and were replaced every 30-40 days. Additional few records, collected by hand, came from the private collection of the first author. The material was identified using an Optika SZM-T stereomicroscope 45x.

For each specimen the following data are reported: scientific name and distributional data according to the World Spider Catalogue (2021); material examined, including site abbreviations (Table 1), date, number of specimens and collection abbreviations (reported below) where specimens are preserved; some faunistic notes. Unless otherwise indicated, specimens have been collected by Alamanni F., Ancona C., Bazzato E. and Caria M.

Collection abbreviations

MSNB: Museo Civico di Scienze Naturali “E. Caffi”, Bergamo (Italy).

ZFMK: Zoological research Museum Alexander Koenig (ZFMK), Bonn (Germany).

MCC: Michele Caria, Guasila (Italy).

Table 1 – List of the localities where the specimens have been collected; coordinates are expressed following the WGS84 system. Altitude is expressed in m.

ID	Municipality	Locality	Coordinates	Altitude
M1	Maracalagonis	Bacca Aruis, Gruxi Lillius	39.271N, 9.300E	100
M2	Maracalagonis	Carroghedda, Corongiu	39.283N, 9.262E	85
M3	Maracalagonis	Codoleddu, Burranca	39.251N, 9.391E	715
M4	Maracalagonis	Codoleddu, Sette Fratelli	39.247N, 9.391E	700
M5	Maracalagonis	Codoleddu, Sette Fratelli	39.249N, 9.389E	700
M6	Maracalagonis	Codoleddu, Sette Fratelli	39.249N, 9.393E	705
M7	Maracalagonis	Coronigu	39.268N, 9.278E	60
M8	Maracalagonis	Riu Monte Nieddu	39.269N, 9.332E	140
M9	Maracalagonis	Riu Piscina Nuxedda	39.275N, 9.316E	80
M10	Maracalagonis	Sedda Brandanu, Corongiu	39.292N, 9.292E	140
M11	Maracalagonis	Sirigragiu, Corongiu	39.284N, 9.279E	80
M12	Maracalagonis	Villaggio dei Gigli	39.275N, 9.326E	120
QE13	Quartu Sant'Elena	Bingia Spada, Stagno di Quartu	39.225N, 9.168E	5
QE14	Quartu Sant'Elena	C. D'Aquila, Stagno di Quartu	39.226N, 9.177E	5
QE15	Quartu Sant'Elena	Frapponti, Cani Nieddu	39.233N, 9.311E	85
QE16	Quartu Sant'Elena	Sa Guardia Lada, Simbirizzi	39.256N, 9.241E	35
QE17	Quartu Sant'Elena	St. comunale Cani Nieddu	39.230N, 9.310E	70
QE18	Quartu Sant'Elena	Via Beglio, Sant'Antonio	39.244N, 9.197E	10
QE19	Quartu Sant'Elena	Via delle Bouganvillee	39.234N, 9.281E	20
QE20	Quartu Sant'Elena	Via Ischia, Foxi	39.218N, 9.243E	0
QE21	Quartu Sant'Elena	Via Pezzetti, Is Arenas	39.232N, 9.179E	5
QE22	Quartu Sant'Elena	Via Riccione, Porticciolo	39.220N, 9.264E	10
QE23	Quartu Sant'Elena	Via Rimini, Sant'Andrea	39.217N. 9.269E	5
QE24	Quartu Sant'Elena	Via Rio Piscinas, Scoa Moentis	39.222N, 9.282E	10
QE25	Quartu Sant'Elena	Via Valenzia, Margine Rosso	39.234N, 9.227E	35
Q26	Quartucciu	Corti de Perda	39.245N, 9.328E	120
Q27	Quartucciu	Piscina Nuxedda	39.254N, 9.305E	50
S28	Sinnai	Burranca	39.294N, 9.322E	150
S29	Sinnai	Monte Cresia, Sette Fratelli	39.272N, 9.396E	665
S30	Sinnai	Monte Cresia, Sette Fratelli	39.273N, 9.393E	675

Results

The list of new recorded species is resumed in Table 2.

Table 2 – List of the new recorded species in Sardinia and in Italy (fr: first record for the territory).

	Sardinia	Italy
Gnaphosidae		
Anagraphis ochracea (L. Koch, 1876)	fr	fr
Civizelotes solstitialis (Levy, 1998)	fr	fr
Drassodes serratichelis (Roewer, 1928)	fr	fr
Leptodrassus albidus Simon, 1914	fr	
Nomisia excerpta (O. Pickard-Cambridge, 1872)	fr	
Marinarozelotes lyonneti (Audouin, 1826)	fr	
Hahniidae		
Hahnia pusilla C. L. Koch, 1841	fr	
Linyphiidae		
Gongylidiellum murcidum Simon, 1884	fr	
Linyphia maura Thorell, 1875	fr	fr
Minyriolus medusa (Simon, 1881)	fr	
Thaumatoncus indicator Simon, 1884	fr	fr
Trichoncus aurantiipes Simon, 1884	fr	fr
Liocranidae		

Cybaeodes cf. *marinae* Di Franco, 1989	fr	
Lycosidae		
Trabea paradoxa Simon, 1876	fr	
Salticidae		
Salticus cingulatus (Panzer, 1797)	fr	
Synaphridae		
Cepheia longiseta (Simon, 1881)	fr	

SPECIES RECORDS

DYSDERIDAE C. L. Koch, 1837

Harpactea sardoa Alicata, 1966

Material examined. Q26, 13.VI-18.VII.2018, 4♂♂ (MSNB); S28, 12.VI-19.VII.2018, 1♂ (MSNB).

Distribution. Italy (Lazio and Sardinia).

Comments. Species described on Sardinian specimens from Monte Funnau (Dorgali, Nuoro). After its description, Brignoli (1979b) recorded it in a woodland in Monte Circeo (Latina, Lazio) and in the Isola Ventotene (Lazio). Later on, it was been recorded by Pantini et al. (2013) and Bosmans & Colombo (2015) in some Sardinian holm oak land.

GNAPHOSIDAE Pocock, 1898

Anagraphis ochracea (L. Koch, 1876)

Material examined. QE14, 06.VI-18.VII.2018, 2♂♂ (MSNB); QE18, 18.VII-22.VIII.2018, 1♂ (MSNB); QE21, 18.VII-22.VIII.2018, 1♀ (ZFMK).

Distribution. Albania, North Macedonia, Greece, Turkey.

Comments. The genus is here recorded for the first time. With these records, we relocate its western distribution limit from Balkans to Sardinia. The left palp of the male is illustrated in Fig. 2.

Civizelotes solstitialis (Levy, 1998)

Material examined. QE14, 18.VII-22.VIII.2018, 1♀ (MSNB).

Distribution. Bulgaria, Greece, Crete, Turkey, Israel, Iran.

Comments. The species is here recorded for the first time. Our record is the westernmost of this species. The epigyne is illustrated in Fig. 3.

Drassodes serratichelis (Roewer, 1928)

Material examined. QE16, 14.V-18.VII.2018, 1♂ (MSNB); municipality of Guasila (Province of South Sardinia), hand collecting, 13.VI.2019 and 15.VI.2019, 2 ♂ M. Caria leg. (MCC).

Distribution. Spain (Majorca), Greece, Turkey, Ukraine, Israel?. Introduced to the USA.

Comments. The species is here recorded for the first time.

Leptodrassus albidus Simon, 1914

Material examined. QE14, 18.VII-22.VIII.2018, 1♂ (MSNB).

Distribution. Azores, Canary Is., Spain to Greece, Turkey, Israel.

Comments. The species is here recorded for the first time. In Italy, it is known only in Capraia Isola (Toscana) (Di Franco & Pantini, 2000) and in the Sicilian localities Oasi del Simeto (e.g., Di Franco 2001) and Platani river (Pantini & Isaia 2008).

Nomisia excerpta (O. Pickard-Cambridge, 1872)

Material examined. M12, 24.V-18.VII.2018, 1♀ (MSNB), 18.VII-22.VIII.2018 1♀ (ZFMK); QE15, 10.V-18.VII.2018, 1♂ (ZFMK), 22.VIII-24.IX.2018, 1♀ (ZFMK); QE17, hand collecting, 5.VI.2018, 1♂ M. Caria leg. (MCC); Q26, hand collecting, 13.VI.2018, 1♂ M. Caria leg. (MCC).

Distribution. Canary Is. to the Middle East.

Comments. The species is here recorded for the first time. In Italy, it was recorded only twice in Calabria (Ijland & van Helsdingen 2016, 2019).

Marinarozelotes lyonneti (Audouin, 1826)

Material examined. M2, 11.V-18.VII.2018, 9♂♂5♀♀ (MSNB), 18.VII-22.VIII.2018, 1♀ (ZFMK); M7, 11.V-18.VII.2018, 1♂ (MSNB), 18.VII-22.VIII.2018, 2♀♀ (ZFMK); M9, 25.V-18.VII.2018, 1♂ (MSNB), 18.VII-22.VIII.2018, 2♀♀ (ZFMK), 22.VIII-24.IX.2018, 1♀ (ZFMK); M11, 14.V-16.VII.2018, 1♂ (MSNB); QE13, 18.VII-22.VIII.2018, 1♂ (ZFMK); QE16, 14.V-16.VII.2018, 4♂♂5♀♀ (MSNB), 3♀♀ (MCC); QE18, 9.V-18.VII.2018, 1♀ (MSNB); QE19, 10.V-18.VII.2018, 2♀♀ (MSNB), 18.VII-22.VIII.2018, 1♀ (ZFMK); QE24, VI-18.VII.2018, 1♂ (MSNB); QE25, 30.V-18.VII.2018, 1♀ (MSNB).

Distribution. Macaronesia, Mediterranean to Central Asia. Introduced to the USA, Mexico, Peru, Brazil.

Comments. The species is here recorded for the first time. In Italy, it is currently recorded only in the Delta del Po (Veneto) (Platnick & Murphy 1984) and in the Sicilian locality Oasi del Simeto (Di Franco 2001). This Mediterranean species shows a widespread distributional range which is possibly related to human activity and to its wide ecological requirements.

Zelotes sardus (Canestrini, 1873)

Material examined. M1, 22.VIII-24.IX.2018, 4♂♂ (ZFMK); M2, 11.V-18.VII.2018, 4♀♀ (MSNB), 18.VII-22.VIII.2018, 1♀ (ZFMK), 22.VIII-24.IX.2018, 2♂♂ (ZFMK); M3, 22.VIII-24.IX.2018, 1♂ (ZFMK); M4, 26.VI-18.VII.2018, 1♀ (MSNB), 22.VIII-24.IX.2018, 2♂♂ (ZFMK); M6, 24.IX-29.X.2018, 1♂ (ZFMK); M7, 11.V-18.VII.2018, 2♀♀ (MSNB); M8, 22.VIII-24.IX.2018, 1♂ (ZFMK), 24.IX-29.X.2018, 1♂ (ZFMK); M9, 22.VIII-24.IX.2018, 5♂♂1♀ (ZFMK); M11, 18.VII-22.VIII.2018, 1♀ (ZFMK), 22.VIII-24.IX.2018, 2♂♂ (ZFMK), 03.XII-8.I.2019, 1♀ (ZFMK); QE15, 10.V-18.VII.2018, 1♀ (MSNB), 18.VII-22.VIII.2018, 2♀♀ (ZFMK); QE16, 14.V-16.VII.2018, 2♀♀ (MSNB), 18.VII-22.VIII.2018, 2♀♀ (ZFMK); QE19, 10.V-18.VII.2018, 2♀♀ (MSNB); QE23, 29.X-03.XII.2018, 1♂ (ZFMK); QE24, 2.VI-18.VII.2018, 1♀ (MSNB), 24.IX-

29.X.2018, 1♂ (ZFMK); QE25, 30.V-18.VII.2018, 4♀♀ (MSNB), 24.IX-29.X.2018, 1♂ (ZFMK); Q26, 18.VII-22.VIII.2018, 1♀ (ZFMK); S29, 20.VI-19.VII.2018, 1♀ (MSNB), 18.VII-22.VIII.2018, 1♂ (ZFMK); S30, 15.VI-19.VII.2018, 1♀ (MSNB).

Distribution. France, Italy.

Comments. *Z. sardus* is known in Sardinia, Capraia Isola (Tuscany) (Di Franco & Pantini 2000), and Corsica (Simon 1914).

HAHNIIDAE Bertkau, 1878

Hahnia pusilla C. L. Koch, 1841

Material examined. M5, 26.VI-18.VII.2018, 1♂ (MSNB), 22.VIII-24.IX.2018, 1♂ (ZFMK); M6, 24.IX-29.X.2018, 1♀ (ZFMK).

Distribution. Europe, Russia (Europe to South Siberia).

Comments. The genus is here recorded for the first time. This species is distributed in West and Central Palearctic (Kovblyuk et al. 2017). In Italy, it is well-known in the northern part and in Puglia with only one record (Brignoli 1973; Pantini & Isaia 2019).

LINYPHIIDAE Blackwall, 1859

Gongylidiellum murcidum Simon, 1884

Material examined. M5, 03.XII-8.I.2019, 1♂ (MSNB).

Distribution. Europe, Turkey, Russia (Europe to West Siberia), Iran, Turkmenistan, Japan.

Comments. The genus is here recorded for the first time. *G. murcidium* is linked to places with high humidity levels (Nentwig et al. 2021). Indeed, the environment where we had collected the specimen is characterized by a superficial aquifer, that allowed the formation of temporary ponds in rainy months.

Linyphia maura Thorell, 1875

Material examined. M3, 03.XII-8.I.2019, 1♀ (MSNB); QE14, 03.XII-8.I.2019, 1♀ (MSNB); QE16, 03.XII-8.I.2019, 1♀ (MSNB); QE19, 10.V-18.VII.2018, 3♀♀ (MSNB).

Distribution. Western Mediterranean.

Comments. The species is here recorded for the first time. Records from Italy (Caporiacco 1950) was re-examined by van Helsdingen (1969) and assigned to *Linyphia triangularis*. This allowed omitting this species from Italian araneofauna (Pantini & Isaia 2019). With these records we re-include the species in the Italian fauna.

Minyriolus medusa (Simon, 1881)

Material examined. QE21, 29.X-03.XII.2018, 1♀; QE23, 29.X-03.XII.2018, 1♂ (MSNB); Q27, 24.IX-29.X.2018, 2♂♂ (MSNB), 29.X-03.XII.2018, 1♂ (MSNB).

Distribution. France, Italy, Austria, Croatia.

Comments. The genus is here recorded for the first time. In Italy, it is known in the Isola del Giglio (Tuscany) (De Dalmas 1922), and in a few other localities in Veneto and in Trentino-Alto Adige (Pantini & Isaia 2019).

Thaumatoncus indicator Simon, 1884

Material examined. QE21, 18.VII-22.VIII.2018, 1♂ (MSNB).

Distribution. Spain, France, Algeria, Tunisia

Comments. The genus is here recorded for the first time. The prosoma of the male is illustrated in Fig. 4.

Trichoncus aurantiipes Simon, 1884

Material examined. M2, 18.VII-22.VIII.2018, 2♂♂ (MSNB).

Distribution. Portugal, Morocco, Algeria, Tunisia.

Comments. The species is here recorded for the first time. It is a western Mediterranean species and is also recorded in the Balearic Islands (GBIF 2021b). The left palp of the male is illustrated in Fig. 5.

Palliduphantes angustiformis (Simon, 1884)

Material examined. M1, 29.X-03.XII.2018, 1♂1♀ (ZFMK); M2, 24.IX-29.X.2018, 1♀ (ZFMK), 03.XII-8.I.2019, 2♂♂3♀♀ (ZFMK); M3, 22.VIII-24.IX.2018, 1♂2♀♀ (ZFMK), 03.XII-8.I.2019, 1♀ (ZFMK); M4, 26.VI-18.VII.2018, 1♂ (MSNB), 24.IX-29.X.2018, 1♀ (ZFMK), 29.X-03.XII.2018, 1♀ (ZFMK), 03.XII-8.I.2019, 1♀ (ZFMK); M5, 26.VI-18.VII.2018, 1♂ (MSNB); M7, 11.V-18.VII.2018, 1♀ (MSNB), 24.IX-29.X.2018, 4♀♀ (ZFMK), 29.X-03.XII.2018, 1♀ (ZFMK), 03.XII-8.I.2019, 1♂3♀♀ (ZFMK); M8, 18.VII-22.VIII.2018, 1♀ (MSNB), 22.VIII-24.IX.2018, 1♀ (ZFMK), 24.IX-29.X.2018, 1♀ (ZFMK), 29.X-03.XII.2018, 1♂ (ZFMK); M9, 22.VIII-24.IX.2018, 1♀ (ZFMK); M10, 01.VI-18.VII.2018, 1♀ (MSNB), 22.VIII-24.IX.2018, 1♀ (ZFMK), 03.XII-8.I.2019, 1♀ (ZFMK); M11, 18.VII-22.VIII.2018, 2♀♀ (ZFMK), 22.VIII-24.IX.2018, 1♀ (ZFMK), 24.IX-29.X.2018, 1♂ (ZFMK); M12, 22.VIII-24.IX.2018, 1♀ (ZFMK), 24.IX-29.X.2018, 1♀ (ZFMK); QE13, 24.IX-29.X.2018, 1♀ (ZFMK); QE14, 22.VIII-24.IX.2018, 1♂1♀ (ZFMK); QE15, 24.IX-29.X.2018, 2♀♀ (ZFMK), 03.XII-8.I.2019, 2♂♂1♀ (ZFMK); QE16, 29.X-03.XII.2018, 1♀ (ZFMK); QE17, 29.X-03.XII.2018, 1♀ (ZFMK); QE18, 29.X-03.XII.2018, 1♀ (ZFMK), 03.XII-8.I.2019, 1♂ (ZFMK); QE19, 03.XII-8.I.2019, 1♂ (ZFMK); QE20, 24.IX-29.X.2018, 1♀ (ZFMK); QE21, 24.IX-29.X.2018, 1♀ (ZFMK), 03.XII-8.I.2019, 2♂♂ (ZFMK); QE22, 5.VI-18.VII.2018, 1♀ (MSNB), 24.IX-29.X.2018, 1♂ (ZFMK), 03.XII-8.I.2019, 1♂1♀ (ZFMK); QE23, 24.IX-29.X.2018, 1♀ (ZFMK), 29.X-03.XII.2018, 2♀♀ (ZFMK), 03.XII-8.I.2019, 4♂♂2♀♀ (ZFMK); QE24, 24.IX-29.X.2018 (ZFMK), 1♂, 03.XII-8.I.2019, 1♂2♀♀ (ZFMK); QE25, 18.VII-22.VIII.2018, 1♂ (ZFMK), 22.VIII-24.IX.2018, 1♀ (ZFMK), 24.IX-29.X.2018, 1♂ (ZFMK), 03.XII-8.I.2019, 2♂♂5♀♀ (ZFMK); Q26, 29.X-03.XII.2018, 1♀ (ZFMK); Q27, 22.VIII-24.IX.2018, 1♀ (ZFMK), 24.IX-29.X.2018, 1♀ (ZFMK), 29.X-03.XII.2018, 1♂2♀♀ (ZFMK); S28, 24.IX-29.X.2018, 1♀ (ZFMK); S29, 22.VIII-24.IX.2018, 1♂♀1 (ZFMK), 24.IX-29.X.2018, 3♀♀ (ZFMK).

Distribution. France (incl. Corsica), Italy (Sardinia).

Comments. Species endemic of the Tyrrhenian area, where is well recorded in Sardinia and Corsica.

LIOCRANIDAE Simon, 1897

Liocranum giersbergi Kraus, 1955

Material examined. M4, 24.IX-29.X.2018, 2♀♀ (ZFMK); S28, 18.VII-22.VIII.2018, 1♀ (ZFMK); S30, 15.VI-19.VII.2018, 1♀ (MSNB), 24.IX-29.X.2018, 1♀ (ZFMK), 29.X-03.XII.2018, 1♂1♀ (ZFMK), 03.XII-8.I.2019, 1♂ (MSNB).

Distribution. Italy (Sardinia).

Comments. Species known only in Sardinia, where is currently well-recorded. It has been described based on a female specimen from Bonorva, while the male has been described by Wunderlich (1995a) based on a specimen from Baunei.

Cybaeodes marinae Di Franco, 1989

Material examined. M8, 24.IX-29.X.2018, 1♂ (MSNB); M12, 24.IX-29.X.2018, 1♂ (MSNB); QE15, 22.VIII-24.IX.2018, 1♂ (MSNB); QE17, 22.VIII-24.IX.2018, 1♂♀ (MSNB); QE19, 29.X-03.XII.2018, 2♂♂ (MSNB); Q27, 22.VIII-24.IX.2018, 1♂ (MSNB); S29, 24.IX-29.X.2018, 2♂♂ (MSNB); S30, 29.X-03.XII.2018, 1♀ (MSNB).

Distribution. Italy.

Comments. The species is here recorded for the first time. It is actually considered an Italian endemic species. It was described based on specimens from Lazio and Sicily. Next, it was recorded in Calabria (Di Franco & Benfatto 2002).

LYCOSIDAE Sundevall, 1833

Trabea paradoxa Simon, 1876

Material examined. M7, 11.V-18.VII.2018, 2♂♂1♀ (MSNB).

Distribution. Southern Europe, Turkey.

Comments. The genus is here recorded for the first time. *T. paradoxa* seems to be a rare strictly Mediterranean species associated with the seaside environment (Tongiorgi 1968; Russell-Smith 1982; Lecigne 2017). In Italy, it has been recorded in seaside localities of Friuli-Venezia Giulia, Veneto, Campania, Calabria, Sicily and in the Tuscan Archipelago (Pantini & Isaisa 2019).

NEMESIIDAE Simon, 1889

Amblyocarenum nuragicum Decae, Colombo & Manunza, 2014

Material examined. QE23, 29.X-03.XII.2018, 1♀ (MSNB); S30, 22.VIII-24.IX.2018, 2♂♂ (ZFMK).

Distribution. Italy (Sardinia).

Comments. The species is endemic to Sardinia. Currently, it has been recorded in Maristella (Province of Sassari), Nuoro and Guspini (Decae et al., 2014).

SALTICIDAE Blackwall, 1841

Salticus cingulatus (Panzer, 1797)

Material examined. M5, 26.VI-18.VII.2018, 1♀ (MSNB).

Distribution. Europe, Turkey, Iran, Russia (Europe to Far East), Kazakhstan, Mongolia.

Comments. The species is here recorded for the first time.

SYNAPHRIDAE Wunderlich, 1986

Cepheia longiseta (Simon, 1881)

Material examined. M2, 18.VII-22.VIII.2018, 1♂ (MSNB).

Distribution. Southern Europe.

Comments. The family is here recorded for the first time. *Cepheia* is a monotypic genus that includes only *C. longiseta*, a poorly recorded species distributed in southern Portugal, southern Spain, Balearic Islands, southern France, southern Austria and northern Italy (Lopardo & Hormiga 2007). In Italy, it has been recorded only once in Liguria (Bertkau 1890) and twice in Trentino-Alto Adige (Thaler & Noflatscher 1990; Lopardo & Hormiga 2007).

THOMISIDAE Sundevall, 1833

Bassaniodes sardiniensis (Wunderlich, 1995)

Material examined. M4, 26.VI-18.VII.2018, 2♂♂ (MSNB).

Distribution. Sardinia.

Comments. Species described on a male specimen collected in Sorgono (Nuoro). Holotype and a second male specimen from Baunei (Nuoro) not reported in Wunderlich (1995b) are currently preserved in Senckenberg Naturmuseum Frankfurt (WSC, 2021). These are the second records of this species after its original description. The left palp of the male is illustrated in Fig. 6. The female is currently unknown.

ULOBORIDAE Thorell, 1869

Polenecia producta (Simon, 1873)

Material examined. QE15, 10.V-18.VII.2018, 1♀ (ZFMK); QE17, hand collecting, 5.VI.2018, 1♂ M. Caria leg. (MCC).

Distribution. Mediterranean to Azerbaijan.

Comments. Simon (1873) described this species and reported it in Corsica and Syria, but did not report the exact locations. Moreover, he did not directly collect the Syrian specimens, but they were sent to him by an acquaintance. In the Iberian Peninsula *P. producta* has been recorded a few times in both Spain (Fernández Galiano 1910; Franganillo 1925; Barrientos et al. 1985; Crespo et al. 2018; Muñoz-Maciá 2019) and Portugal (Cardoso 2004; Cardoso et al. 2008; Crespo et al. 2010) but the amount of record is scarce. These records are mapped by de Biurrun et al. (2019) and other Iberian records are reported in GBIF (2021a). In Italy, it was recorded only once by Brignoli (1979a), that collected a male specimen from Bortigiadas (Sassari) in 1966. Therefore, this is the second record of this species in Italy. However, records in mainland Italy and Sicily are still lacking. In the same paper, the author reported it in Lebanon too. Finally, it was recorded in Azerbaijan, where there are only three records (Dunin 1988; Guseinov 1999). Other records of this species are also reported in GBIF (2021a) for Agadir.

ZODARIIDAE Thorell, 1881

Zodarion pseudonigriceps Bosmans & Pantini, 2019

Material examined. M9, 25.V-18.VII.2018, 1♂2♀♀; M10, 01.VI-18.VII.2018, 4♀♀; Q26, 13.VI-18.VII.2018, 4♀♀, 22.VIII-24.IX.2018, 1♂, 24.IX-29.X.2018, 1♂1♀; S28, 12.VI-19.VII.2018, 10♀♀.

Distribution. Italy (Sardinia).

Comments. Species recently described on specimens from Montevecchio (Guspini) and Ingurtosu (Arbus). These are the first records of this species after its description.

Fig. 2 – *Anagraphis ochracea* (L. Koch, 1876): ♂ palp ventral view (A) and retrolateral view (B). Scale bar: 0.3 mm.

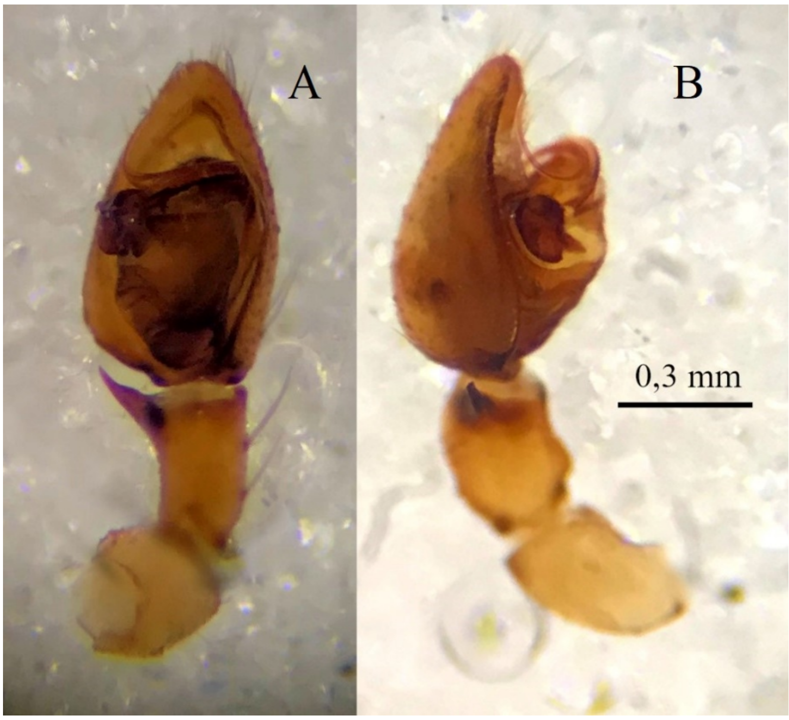


Fig. 3 – *Civizelotes solstitialis* (Levy, 1998): ♀ epigyne ventral view. Scale bar: 0.3 mm.

Fig. 4 – *Thaumatoncus indicator* Simon, 1884: ♂ prosoma lateral view. Scale bar: 0.3 mm.

Fig. 5 – *Trichoncus aurantiipes* Simon, 1884: ♂ palp retrolateral view (A), prolateral view (B), ventral view (C) and dorsal view (D). Scale bar: 0.3 mm.

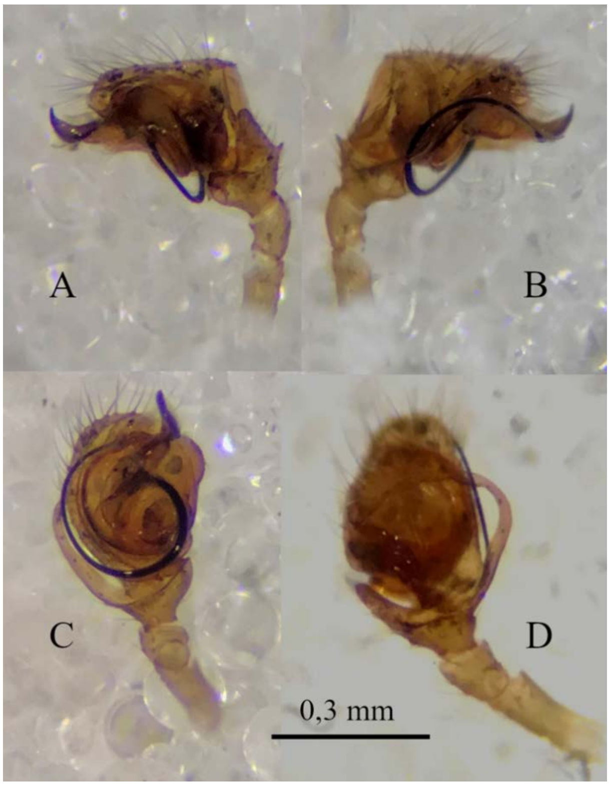


Fig. 6 – *Bassaniodes sardiniensis* (Wunderlich, 1995): ♂ palp ventral view (A), retrolateral view (B) and dorsal view (C). Scale bar: 0.5 mm.

Conclusion

We reported new distributional data of 24 taxa from 12 families for the Sardinian spider fauna, including several endemic Sardinian species.

The discovery in Sardinia of *Cepheia longiseta* and five other species that had until now not been recorded, respectively, for Sardinia and Italy, is particularly relevant because their presence allows us to include Synaphridae family and five genera (*Anagraphis*, *Hahnia*, *Minyriolus*, *Thaumatoncus*, *Trabea*) in the Sardinian spider checklist, and two genera (*Anagraphis* and *Thaumatoncus*) in the Italian spider checklist.

In the same way, the numerous species reported for the first time in the regional and national araneofauna once again confirm that lower attention has been given to the faunistic knowledge of the spider fauna of southern Italy. Two especially noteworthy findings are *Bassaniodes sardiniensis* and *Polenecia producta*, considering that the last records of the first one dates back to 23 years ago, whereas the last records of the second one in its whole known range of distribution date back to the end of the last century. For these reasons, these newly records assume great importance for the knowledge of these poorly collected species. The finding of *Zodarion pseudonigriceps* is also noteworthy, since this species has been described just one year ago (Bosmans et al. 2019).

The high number of species records of regional and national relevance provided, after a very short time from the publication of the latest Italian spider checklist (Pantini & Isaia 2019), suggesting that knowledge about Sardinian spider fauna is far from being satisfying and can be easily increased by carrying out targeted faunistic or field ecological studies. Local field surveys can lead to filling gaps in the distribution of poorly collected species, and to implement our knowledge of diversity at the regional and national levels, especially in Mediterranean areas were the high heterogeneity of climate and topographic complexity (Bazzato et al. 2021) have determined high levels of biodiversity and a wide variety of Potential Natural Vegetation (Farris et al. 2010; Bacchetta et al. 2009) and environment.

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Author contributions

Conception and design of field research: Bazzato E.; material preparation and data collection: Bazzato E., Caria M., Ancona C., Alamanni F.; determination of species: Caria M., Pantini P.; conception and design of paper: Caria M., Pantini P., Bazzato E.; drafting the article: Caria M., Bazzato E., Pantini P.; revising the article for important intellectual content: Pantini P.

All authors contributed to read and approve the final version of the manuscript.

Subchapter 2.4: ants

Ants of Sardinia: an updated checklist based on new faunistic, morphological and biogeographical notes

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Abstract

Sardinia is the second largest island in the Mediterranean region, receiving significant attention due to its interesting fauna and flora. The last checklist of Sardinian ants was published more than a decade ago, and, since then, it got outdated by numerous taxonomic and faunistic novelties. As a result of recent collecting efforts across the island, we present the first Sardinian records of *Messor ibericus* Santschi, 1931, *Solenopsis lusitanica* Emery, 1915 (new to Italy), *Temnothorax aveli* Bondroit, 1918 and *Tetramorium atratulum* (Schenck, 1852), while proposing to consider *Solenopsis fugax* (Latreille, 1798) and *Temnothorax affinis* (Mayr, 1855) as absent. We report for the first time a parasite-host association between *Tetramorium atratulum* and *Tetramorium semilaeve* André, 1883, and the conspicuous presence of ergatogynes within a *Solenopsis* colony (*S. lusitanica*). Morphological insights on the little-known *S. lusitanica* and *S. orbula* Emery, 1875 are also discussed. We combined the new findings and previous literature data into an updated checklist of 77 taxa and discuss a first biogeographic analysis of the Sardinian ants aided by chorotypes. Eurasian, European, Euro-Mediterranean and West-Mediterranean taxa are the numerically prevalent groups, while the overall number of species is significantly lower than in the other large Mediterranean islands. Considerable knowledge gaps still remain and some species are known to require additional taxonomic investigation.

Keywords: mirmecofauna; *Solenopsis*; ergatogynes; Tuscan Archipelago.

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Introduction

The Mediterranean basin is an important diversity hotspot worldwide (MÉDAIL & QUÉZEL, 1997; MÉDAIL, 2017), hosting a very large number of unique ant species (BOROWIEC, 2014). Sardinia is the second largest Mediterranean island, covering about 24,000 km^2 and being only slightly smaller (~1.7 km^2) than Sicily. Thanks to its variety of landforms, complex orographic patterns (with hilly lands, plateaus, mountain and plains), heterogeneous geological substrata and climate variability (BAZZATO et al., 2021), the island is characterized by high levels of biodiversity and it is broadly known to host a significant endemic component in its fauna and flora (BACCETTI, 1983; GRILL et al., 2007) in addition to a wide variety of Potential Natural Vegetations (FARRIS et al., 2010; BACCHETTA et al., 2009). Ants (Hymenoptera, Formicidae) are one of the ecologically more impactful insect groups in both natural and anthropogenic ecosystems (HÖLLDOBLER & WILSON, 1990; LACH et al., 2010), including Mediterranean forests and agroecosystems across Italy and Sardinia (e.g., LOI et al. 2012; CAMPOLO et al., 2015; CASTRACANI et al., 2017; GIANNETTI et al., 2019; SCHIFANI et al., 2020a). The oldest checklist of the Sardinian ant fauna was published by EMERY (1915) and included 47 taxa. Later on, this number increased to 57 in the Italian checklist published by BARONI URBANI (1971), and to 68 according to POLDI et al. (1995). However, the latest checklist, published by VERDINELLI et al. (2007), brought the total number up to 70. Since then, several other species were recorded as a result of few faunistic surveys (RIGATO & TONI, 2011; LOI, 2013), suggesting that further efforts are still needed to uncover the island's true myrmecological diversity. In addition, relevant changes occurred due to taxonomic revisions involving species present in Sardinia (e.g., SEIFERT et al., 2017) and presently, no updated checklist of the island is available.

Different independent collecting efforts on the Sardinian ant fauna were conducted by the authors of this paper, which resulted in a significant amount of new species records and the collection of new or little-known forms and a new parasite-host association in ants. In addition, a review of the relevant taxonomic and faunistic literature allowed us to compile a new and updated checklist to summarize current faunistic knowledge over the island's ants, providing a key source of information to facilitate future investigations on Sardinian and Western-Mediterranean ants and their biogeography.

Materials and methods

Collecting efforts were conducted in the years 2017-2019 by employing both direct sampling and pitfall traps filled with wine-vinegar saturated by sodium chloride as preservation method (see AGOSTI et al., 2001; BRANDMAYR et al., 2005) across four of the five administrative regions of Sardinia: Metropolitan City of Cagliari (CA) (see PALUMBO et al., 2020), and the provinces of Nuoro (NU), Oristano (OR) and South Sardinia (SU). In addition, we consulted material stored at the Milan Natural History Museum (Museo Civico di Storia Naturale, Milan, Italy - MSNM) and in authors personal collections, from Sardinia and also from neighboring regions whenever relevant. All specimens were identified under stereomicroscopes. Relevant taxonomic sources are mentioned for each taxon. Whenever geographic coordinates of the sampling sites are given, error range is estimated to be < 15 m.

Morphometric characters presented in this paper for *Solenopsis* refer to the morphometries used by GALKOWSKI et al. (2009), but French acronyms were abandoned in favour of the English-based acronyms proposed by SEIFERT (2018). Therefore, the following characters and acronyms were used (French equivalents are indicated in brackets): CW, maximum head width, across the eyes

(LaT); CL, maximum head length in median line (LoT); SL, maximum scape length as a straight line, excluding the articular condyle and its neck (LoSc); EL, maximum diameter of the compound eyes, including unpigmented ommatidia (Diam. oeil); ML, maximum length of the mesosoma (LoM); MW, maximum width of the mesosoma (LaM); Mh, maximum height of the mesosoma (HM); PeH, maximum height of the petiole (HP); PPH, maximum height of the postpetiole (HPP); PW, maximum width of the petiole (LaP); PPW, maximum width of the postpetiole (LPP). These measurements were obtained using the software ImageJ (SCHNEIDER et al. 2012) and high-quality pictures of the specimens taken at up to 10x magnification using a Canon EOS 1300D camera and micro photography lens.

To facilitate a first biogeographic analysis of the checklist, we attempted to rely on the most broadly utilized chorotypes model proposed by VIGNA TAGLIANTI et al. (1999). However, in the framework of ant biogeography, we found highly limiting to miss chorotypes defining Maghrebian and South-Western European distributions, both disappearing under the wider Western-Mediterranean chorotype *sensu* VIGNA TAGLIANTI et al. (1999). PARENZAN (1994) proposed to use the acronyms NAW and ESW to refer to these distributions, so we follow his nomenclature for these two types. As a result, the following chorotypes were used: ASE, Asiatic-European; EME, East-Mediterranean; ESW, South-West European; EUR, European; CEM, Central Asiatic-European-Mediterranean; MED, Mediterranean; NAF, North-African; NAW, South-Western Mediterranean; SACO, Sardo-Corsican endemic; SARD, Sardinian endemic; SEU, South-European; TEM, Turano-European-Mediterranean; TUE, Turano-European; TYRR, Tyrrhenian endemic; WME, West-Mediterranean. Chorotypes were assigned to each taxon mainly on the basis of species distribution data from AntMaps (see JANICKI et al., 2016; GUÉNARD et al., 2017).

New species records

Messor Forel, 1890

Messor ibericus Santschi, 1931

Examined material: SARDINIA: All the specimens are stored in the University of Cagliari collection deposited in the Zoological Research Museum Alexander Koenig (Bonn, Germany): Sa Guardia Lada, Simbrizzi, Quartu Sant’Elena (CA), 16.VII-03.XII.2018, 12 workers, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Sirigragiu, Corongiu, Maracalagonis (CA), 22.VIII.2018, 3 workers, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. C. D'Aquila, Stagno di Quartu, Quartu Sant’Elena (CA), 16.VII.2018-II.2019, 4 workers, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Riu Piscina Nuxedda, Maracalagonis (CA), 17.VII-25.IX.2018, 9 workers, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Bingia Spada, Stagno di Quartu, Quartu Sant’Elena (CA), VII-VIII.2018, 2 workers, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Via Belgio, Sant’Antonio, Quartu Sant’Elena (CA), II.2019, 1 worker, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani.

Remarks: This species appears to be sole representative of the *Messor structor* complex in the Western Mediterranean region, where it was known under the name *M. structor* until recently (Steiner et al. 2018). All Sardinian specimens we examined were expectedly identified as *M. ibericus* according to the discriminant function proposed by STEINER et al. (2018). Under these conditions, all previous records of *M. structor* from the island should provisionally be attributed to *M. ibericus*, removing *M. structor* from the Sardinian checklist. According to the few verified data

(STEINER et al., 2018; SCHIFANI & ALICATA, 2018; SCHÄR et al., 2020), *M. ibericus* may be the sole species of this complex also in the rest of Italy, while some recent mentions for the peninsula (RIGATO & WETTERER, 2018; SCUPOLA, 2018) refer to identifications prior to the taxonomic revision of STEINER et al. (2018).

Solenopsis Westwood, 1840

Solenopsis lusitanica Emery, 1915

Examined material: SARDINIA: Domus de Maria (SU), 38.945720, 8.813108, 05.IX.2018, 63 males, 4 queens, 20 ergatogynes, 210 minor workers, 143 major workers, E. Nalini legit, det. E. Nalini & E. Schifani, E. Nalini collection. Iglesias (SU), 19.IX-03.X.2006, 1 queen and 1 male, G. Chessa legit, det. E. Nalini, Museo Civico di Storia Naturale di Milano (MSNM) – published in Rigato & Toni (2011). Narcao (SU), 02.X.2018, 1 queen and 1 male, E. Nalini legit, det. E. Nalini, E. Nalini collection. Pantaleo (SU), 39.090699, 8.802370, 09.IX.2018, 27 males, 28 queens, 98 minor workers, 18 major workers (two colonies), E. Nalini legit, det. E. Nalini, E. Nalini collection.

In addition to the abovementioned Sardinian material, further Italian specimens belonging to the same morphospecies were found from the Tuscan Archipelago, representing a significant novelty that we also report on:Isola d'Elba, 17.IX-03.X.2000, 1 queen and 2 males, P. Scaramozzino legit, det. E. Nalini, MSNM. Isola di Montecristo, 1 queen (15-26.IX.1999) and male (15-25.IX.2000), F. Strumia legit, det. E. Nalini, MSNM. Isola di Pianosa, 15-26.IX.2000, 1 queen and 2 males, P. Scaramozzino legit, det. E. Nalini, MSNM.

Remarks: This species is new to Italy (POLDI et al. 1995). However, attribution of the examined material to this taxon bears an inevitable degree of uncertainty until a proper taxonomic revision of the group is produced. GALKOWSKI et al. (2009) started to revise the nomenclatural chaos that reigns over the difficult taxonomy of Mediterranean *Solenopsis*, a genus that includes a wide amount of very poorly described taxa whose identity is difficult to delimit. The *S. lusitanica* group appears distinguishable among the *Solenopsis* of South-Western Europe by morphological characters provided by GALKOWSKI et al. (2009). At the same time, no distinction is actually possible between *S. lusitanica* and the other valid taxa of the same group, *S. balachowsky* Bernard, 1959 and *S. gallica* Santschi, 1934, both considered of uncertain identity (GALKOWSKI et al., 2009). For this reason, and since *S. lusitanica* is also the sole of the three currently considered present in any nearby region (see the Corsican checklists by BLATRIX et al., 2018; 2020) we provisionally name the Sardinian *Solenopsis* material belonging to the *S. lusitanica* group as *S. lusitanica* until a proper taxonomic revision assesses the validity of *S. balachowsky* and *S. gallica*. Further information on this identification is given in the morphological section of this paper. Considering the extreme uncertainty around the true identity of *S. fugax* (Latreille, 1798) until recently (GALKOWSKI et al. 2009), it is unsurprising that all the Sardinian material previously identified as *S. fugax* we managed to examine (see RIGATO & TONI, 2011) corresponds to *S. lusitanica* instead. In a similar fashion, recent attempts to find *S. fugax* in Sicily yielded no results, as only a form tentatively named *S. latro* Forel, 1894 was recovered by SCHÄR et al. (2020). While we have verified records of *S. fugax* from other regions of Italy (at least across the Po Plain - Emilia-Romagna: SCHIFANI et al., 2020a; Lombardy: CASTRACANI et al., 2020, and also Trezzo sull'Adda (MI), 45.612021, 9.522218, 10.IX.2019, E. Nalini leg., E. Nalini personal collection; Veneto: Bovolone (VR), 27.VIII.2014, E. Nalini leg. and Spinea (VE), 14.IX.2017, D. Vallotto leg., E. Nalini personal collection), we provisionally propose to remove this species from the Sardinian checklist. The findings of *S. lusitanica* in the Tuscan Archipelago (biogeographically close to both

Corsica and the Tuscan coast, see FATTORINI, 2009; DAPPORTO et al., 2017) may suggest its presence on mainland Italy, perhaps unnoticed due to misidentifications with *S. fugax*. A proper assessment over the identity and distribution of the Italian *Solenopsis* spp. is evidently required, but it depends on a satisfactory resolution of at least some of the taxonomic problems affecting this genus in the Mediterranean region.

Temnothorax Mayr, 1861

Temnothorax aveli (Bondroit, 1918)

Examined material: SARDINIA: Piscinas, Arbus (SU), 0 m, 39.5404, 8.4521, 25.V.2006, P. Cornacchia, M. Bardiani, D. Birtele & D. Whitmore legit, 1 worker labeled as *Temnothorax affinis*, det. E. Schifani, Bosco Fontana Natural Reserve ant collection (Lombardy) - published in Rigato & Toni (2011).

Remarks: Current uncertainty over the separation of *T. aveli* from *T. italicus* (Consani, 1952) complicates naming of the Sardinian population: populations from France (including Corsica) and Iberia have been traditionally identified as *T. aveli* (or as one of its junior synonyms, see for example BUSCHINGER et al., 1988; COLLINGWOOD & PRINCE, 1998; ARNAN et al., 2006; HERNÁNDEZ CUBA et al., 2006; PLATEAUX & CAGNIANT, 2012; ESPADALER et al., 2013; TINAUT, 2016; BLATRIX et al., 2018), while those of the Italian peninsula and North-Western Balkans as *T. italicus* (e.g., BRAČKO, 2006; 2017; SCHULZ et al., 2006; CASTRACANI et al., 2010; RIGATO & WETTERER, 2018; SCUPOLA, 2018; GIANNETTI et al., 2019; SCHIFANI et al., 2020a). *Temnothorax aveli* was described from France (BONDROIT 1918), while *T. italicus* from Central/Northern Italy (CONSANI & ZANGHERI 1952). However, none of the few qualitative characters provided by CONSANI & ZANGHERI (1952) to separate *T. italicus* from *T. aveli* appears fully reliable following a comparison of French *T. aveli* (type material and other material generously shared with us by C. Galkowski and R. Blatrix) and Italian material of *T. italicus* (published in GIANNETTI et al., 2019; SCHIFANI et al., 2020a), and the two taxa have been suggested to be potential synonyms (SCUPOLA, 2018). While an attempt to shed light over this taxonomic uncertainty is ongoing, we provisionally decided to use the name *T. aveli* due to the lack of darkened antennal clubs typical of *T. italicus* according to CONSANI & ZANGHERI (1952). The examined specimen had previously been identified as *T. affinis* (RIGATO & TONI, 2011): while there are evident similarities in shape and color between *T. affinis* and *T. aveli/T. italicus*, *T. affinis* is characterized by an evidently much coarser body sculpture (SEIFERT, 2018) and by a much less prominent subpetiolar process. As a result of our new identification, which was further aided by direct comparison with several *T. affinis* workers from Italy and Central Europe, *T. affinis* is provisionally excluded from the Sardinian fauna.

Tetramorium Mayr, 1855

Tetramorium atratulum (Schenk, 1952)

Examined material: SARDINIA: Dorgali (NU), 21.VIII.2018, 1 queen and 1 male from a *T. semilaeve* nest, M. Plumari legit, det. V. Gentile, V. Gentile collection. Dorgali (NU), 21.VIII.2018, 1 queen and 1 male from a *T. semilaeve* nest, M. Plumari leg., det. V. Gentile, M. Plumari collection. Narcao (SU), 39.1670, 8.6628, 2.IX.2018, 1 queen and 1 male from a *T. semilaeve* nest, E. Nalini legit, det. E. Nalini, E. Nalini collection.

Remarks: Inquiline social parasite ants as *T. atratulum* are very easily overlooked during faunistic surveys, sometimes severely hinder a correct understanding of their rarity, distribution and conservation status (ESPADALER & LÓPEZ-SORIA, 1991; SCHIFANI, 2017). Although still treated as Vulnerable in the IUCN Red List (SOCIAL INSECTS SPECIALIST GROUP, 1996), *T. atratulum* is one of the few exceptions: records of this species are quite numerous and its geographic range widely extends from the West Palearctic, where it is native, to the Neartic region, where it was introduced alongside one of its host species, *Tetramorium immigrans* Santschi, 1927 (DASH & SANCHEZ, 2009: SEIFERT, 2018; ZHANG et al., 2019). Interestingly, there are many different *Tetramorium* host species that *T. atratulum* is known to exploit. Three belong to the *T. caespitum* complex (WAGNER et al., 2017), one to the *T. chefketi* species complex and one to the *T. ferox* species complex (SANETRA et al., 1999; for complexes definitions see CSŐSZ et al., 2005; CSŐSZ & SCHULZ, 2010). The most interesting aspect of this first Sardinian record is that in all findings the host species was *T. semilaeve* André, 1883, which does not belong to any of these complexes and was never recorded as a host for *T. atratulum* before. SANETRA et al. (1999) had speculated that *T. semilaeve* could be a host for *T. atratulum* in Italy but no findings ever occurred. It is worth mentioning that the Sardinian populations of *T. semilaeve,* somewhat similarly to those of Calabria and Sicily (Fig. I; SCHÄR et al. 2020), at least chromatically sometimes deviate from the definition given by BOROWIEC et al. (2015) ("never dark brown"). In addition, none of the known host species of *T. atratulum* is known to occur in Sardinia, with the sole exception of *T. immigrans* which is probably introduced on the island (see WAGNER et al., 2017; CASTRACANI et al., 2020).

Morphological notes

Solenopsis lusitanica Emery, 1915

There are at least two reasons that make worth offering a morphological overview of the Sardinian material we identified as *S. lusitanica*. First is obviously the scarceness of information currently available over the morphological identity of this taxon and the taxonomic confusion which reigns over the *S. lusitanica* complex, hindering a proper biological and biogeographical understanding. Second, the extraordinary finding of 20 ergatogynes within one of the examined *S. lusitanica* colonies (Fig. II), which represents an unexpected and very rare case within *Solenopsis*.

As mentioned above, the original description of *S. lusitanica* is almost completely useless to its identification (see EMERY, 1915) and the sole information available was provided by Galkowski et al. (2009), consisting in a brief description and morphometric characterization of one queen and one male specimen (alongside 1 queen and 1 male of *S. balachowsky*). However, we compared our material with the definition and morphometric data provided by GALKOWSKI et al. (2009) and to some *S. lusitanica* specimens from Spain kindly sent to us by C. Galkowski. In order to do so, we recorded the 12 morphometric characters used by GALKOWSKI et al. (2009) on 31 specimens from 3 colonies (Tab. 1). Despite past confusion, *Solenopsis lusitanica* males and queens are much smaller than those of *S. fugax*, and queens lack the longitudinal rugae running from the frons to the ocelli in *S. fugax* (Fig. II). Distinction of workers appears to be considerably more difficult, especially for minors. Major workers have darkened heads (often slightly darkened near the vertex in minors too), and both minors and majors appear to be less hairy than *S. fugax* (Fig. III). It is worth noting that the type worker of *S. lusitanica gaetula* Santschi, 1936 (AntWeb CASENT0913907), an even more ambiguously defined taxon from Morocco (SANTSCHI 1936), is clearly outside any imaginable intraspecific variation of the form we examined, and may instead be more closely related to the *S. orbula* complex.

Regarding the collected ergatogynes, these showed a significant morphological diversification, encompassing individuals more closely resembling queens and others more closely resembling workers (Fig. III). Ergatogynes represent one of the several different morphological mosaics found in ants (SCHIFANI et al., 2020b). They are classified as either intercastes (i.e., rarely generated hybrid phenotypes usually unable to reproduce) or ergatoids (a distinct kind of specialized reproducers which may be even more common or replace queens in some species) (PEETERS, 1991). The very high number of ergatogynes found within a single nest in our case timidly suggests that they may represent functional ergatoids, however documentation of ergatogynes in *Solenopsis* is almost non-existent, recommending prudence in taking interpretations (GLANCEY et al., 1980). Further investigation should aim to see whether ergatogynes are routineously produced by the colonies of this species.

Tab. 1. Morphometric characters of examined *Solenopsis lusitanica* specimens. Definition of morphometric characters follows Galkowski et al. (2009), nomenclature adapted to the English standard proposed by Seifert (2018) and French equivalents in parentheses when possible. Values are given in µm (mean ± standard deviation (minimum – maximum).

	males **(7 specimens, 3 colonies)**	**queens** **(6 specimens, 3 colonies)**	**ergatogynes** **(5 specimens, 1 colony)**	**minor workers** **(7 specimens, 1 colony)**	**major workers** **(6 specimens, 1 colony)**
CW (LaT)	600±18 (570 - 621)	808±23 (785 - 845)	566±34 (539 - 623)	400±12 (381 - 418)	475±36 (439 - 537)
CL (LoT)	470±33 (429 - 528)	745±27 (713 - 779)	600±31 (570 - 643)	475±12 (456 - 488)	544±21 (511 - 571)
SL (LoSc)	126±7 (113 - 133)	512±13 (493 - 529)	370±24 (329 - 392)	308±8 (294 - 319)	347±14 (335 - 368)
SL/HW	0.21±0.01 (0.18 - 0.22)	0.63±0.01 (0.61 - 0.65)	0.66±0.07 (0.52 - 0.59)	0.77±0.03 (0.70 - 0.80)	0.73±0.03 (0.68 - 0.78)
SL/HL	0.27±0.03 (0.22 - 0.31)	0.69±0.03 (0.65 - 0.74)	0.62±0.06 (0.51 - 0.66)	0.65±0.02 (0.61 - 0.77)	0.64±0.01 (0.63 - 0.64)
CW/CL	1.28±0.07 (1.18 - 1.40)	1.08±0.05 (1.01 - 1.16)	0.94±0.02 (0.92 - 0.97)	0.84±0.01 (0.83 - 0.86)	0.87±0.04 (0.86 - 0.94)
EL (Diam œil)	230±5 (223 - 236)	217±9 (207 - 232)	75±11 (60 - 87)	40±4 (34 - 45)	53±7 (47 - 65)
Ocellus diameter	78±5 (69 - 84)	83±17 (68 - 117)	-	-	-
ML (LoM)	1235±70 (1154 - 1332)	1502±55 (1410 – 1563)	710±37 (649 - 747)	481±19 (456 - 508)	565±35 (536 - 606)
MW (LaM)	841±38 (789 - 880)	826±48 (751 - 874)	370±32 (323 - 412)	252±6 (246 - 260)	291±19 (266 - 307)
MH (HM)	795±42 (714 - 840)	943±55 (870 - 992)	441±57 (349 - 483)	-	-
PeH (HP)	215±12 (195 - 227)	372±21 (339 - 400)	211±19 (182 - 229)	161±3 (157 - 166)	186±5 (181 - 193)
PPH (HPP)	261±25 (220 - 286)	364±24 (320 - 384)	188±24 (150 - 213)	124±9 (114 - 137)	153±9 (138 - 165)
PW (LaP)	230±16 (206 - 254)	326±21 (308 - 366)	181±15 (161 - 197)	127±3 (124 - 132)	150±9 (138 - 164)
PPW (LPP)	265±6 (259 - 275)	367±17 (338 - 380)	196±14 (183 - 217)	142±7 (135 - 157)	158±10 (149 - 177)

Solenopsis orbula Emery, 1875

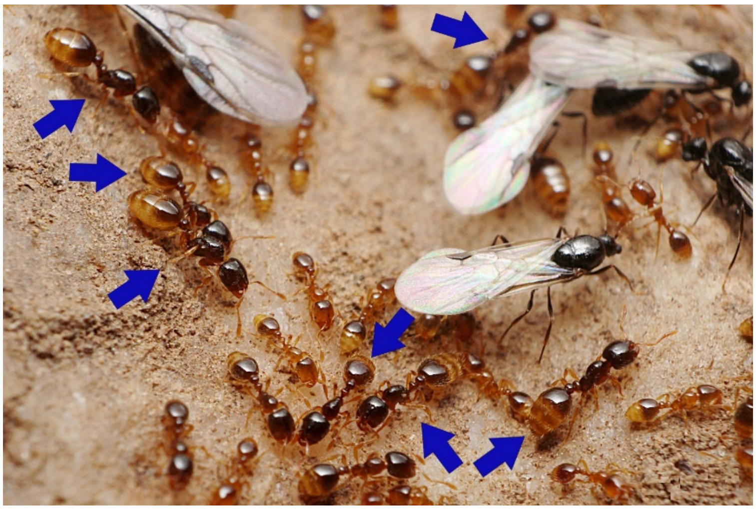
Examined material: SARDINIA: Mari Ermi, Cabras (OR), 05.VII.2017, 2 queens and 1 male, Emilio Villani legit, det. V. Gentile, V. Gentile collection. Specimens stored at the in the University of Cagliari collection deposited in the deposited in the Zoological Research Museum Alexander Koenig (Bonn, Germany): Sirigragiu, Corongiu, Maracalagonis (CA), 1.VI-18.VII.2018, 2 workers, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Piscina Nuxedda, Quartucciu (CA), 18.VII-21.VIII.2018, 2 workers, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Str. Comunale Cani Nieddu, Quartu Sant'Elena (CA), 17.VII-21.VIII.2018, 1 worker, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Riu Monte Nieddu, Maracalagonis (CA), 17.VII-21.VIII.2018, 1 worker, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Via Lago di Varese, Quartu Sant'Elena (CA), I.2019, 1 worker, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Via Pizzetti, Is Arenas, Quartu Sant'Elena (CA), 16.VII-29.X.2018, 3 workers, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani. Via Riccione, Porticciolo, Quartu Sant'Elena (CA), 20.VIII-30.X.2018, 28 workers, E. Bazzato, D. Cillo, M. Caria & Cesare Ancona legit, det. E. Schifani.

Fig. I. *Tetramorium semilaeve* worker from a Sardinian colony hosting *Tetramorium atratulum*. 1: lateral view, 2: dorsal view, 3: head view. Scale bar: 0.5 mm. Photos by Elia Nalini.

The identity of this peculiar species has long been vague. It was originally described from Corsica on the basis of its worker caste (EMERY, 1875a; a separate description also in EMERY, 1875b) and then ANDRÉ (1883) provided a brief description of a *S. orbula* queen from Algeria. Later on, FOREL (1905) described the subspecies *S. orbula terniensis* Forel, 1905 from some Algerian workers and a Spanish queen. However, differences between the two subspecies remain barely defined and only described for the worker caste (e.g., EMERY, 1909; 1916). As a result, the status of ssp. *terniensis* continued to be dubious (e.g., MEI, 1995), while significant morphological data from the type population of Corsica were more recently provided by GALKOWSKI et al. (2010) for all three castes. Despite these long-lasting taxonomic uncertainties, the name *S. orbula* s.l. and *S. orbula terniensis* have been widely used in the Mediterranean region: in Sardinia (EMERY 1916, ssp. *orbula*), Sicilian Channel (EMERY, 1916, ssp. *terniensis*), peninsular Italy (POLDI, 1992), Maltese Islands (SCHEMBRI & COLLINGWOOD, 1981), mainland France (CASEVITZ-WEULERSSE & GALKOWSKI, 2009), Spain (COLLINGWOOD & YARROW, 1969), Tunisia (FOREL, 1890; 1905, as ssp. *terniensis* in the latter), Lybia (FINZI, 1940, ssp. *terniensis*), Egypt (GRANDI, 1935, ssp. *terniensis*), Israel (VONSHAK & IONESCU-HIRSCH, 2009, ssp. *terniensis*), Lebanon (TOHMÉ, 1969), Syria (TOHMÉ & TOHMÉ, 1980), Turkey (AKTAÇ, 1988), Crete and in some Aegean Islands (SALATA & BOROWIEC, 2018).

Moreover, in North Africa, several other vaguely defined taxa show similar morphological traits to *S. orbula*: *S. atlantis* Santschi, 1934, *S. longiceps* Forel, 1907, its subspecies *S. longiceps barbara* Santschi, 1934 (FOREL 1907; SANTSCHI 1934), in addition to the aforementioned *S. lusitanica gaetula*. The only North-African species similar to *S. orbula* but at the same time safely distinguishable on the basis of the existing literature is the Egyptian *S. cooperi* Donisthorpe, 1947 thanks to the efforts of SHARAF et al. (2009). Due to biogeographical reasons, the conspecificity of the Sardinian population with that of Corsica has never been questioned (EMERY, 1916; VERDINELLI et al., 2007). On the other hand, at least some of the abovementioned Mediterranean records probably belong to different species. For example, TOHMÉ & TOHMÉ (1980) described in detail a Syrian 'variety' of *S. orbula* which clearly looks morphologically different from the Corsican *S. orbula* (also see GALKOWSKI et al., 2009). A quite different form of *Solenopsis* from Afghanistan was also described as a subspecies of *S. orbula* by PISARSKI (1967), and only later recognized as a different taxon, *S. knuti* Pisarski, 1967 (DLUSSKY & RADCHENKO, 1994). In the case of most records though, doubts cannot be solved from the scarce indications published.

Fig. II. *Solenopsis lusitanica* colony from Sardinia. Blue arrows indicate ergatogynes, while the rest of wingless individuals are workers of different size and winged individuals are one queen (upper right side) and four males (on the left). Photo by Elia Nalini.



The material we collected in Sardinia included swarming sexuals collected in July (the same period reported for nuptial flights in Tuscany by POLDI (1992)) and allows us to highlight a number of peculiar morphological characteristics of this species, hopefully providing some useful indication for the study of other Mediterranean populations until a proper taxonomic revision of the group is produced. The following morphological traits seem particularly distinctive of this taxon (Fig. IV):

- Small-sized queen caste characterized by a remarkably thin mesosoma (visibly much thinner than the head), an elongated rectangular-shaped head, a mostly dark-colored body contrasting with a yellowish gaster and feeble sculpture.
- Polymorphic workers with elongated, rectangular-shaped heads and a relatively characteristic mesosoma profile with a high and long propodeum. The occiput is not excavated as in species like *S. cooperi*. Most of the workers are uniformly yellow but the largest workers possess a contrasting dark head. This characteristic is visually impressive and clearly distinctive, but seems surprisingly neglected in the existing literature where is almost never reported (e.g., not in GALKOWSKI et al., 2010). The Corsican population possess the same habitus of the Sardinian

specimens (see https://www.myrmecofourmis.fr/Solenopsis-orbula, accessed: 23.06.2020) and so does the Maltese one according to BARONI URBANI (1968). A comparison with pictures from the Balearic Islands (kindly shared with us by JOSEALBERTO FERNÁNDEZ) also appear to testify there the presence of the exact same form.

Finally, male morphology (Fig. III) may also be quite distinctive, but the lack of information over the morphological traits of most other species does not allow us to elaborate further.

Fig. III. *Solenopsis lusitanica* from Sardinia. 1-3: male, 4-6: queen, 7-12: ergatogynes, 13-15: major worker, 16-18: minor worker. Scale bars: 0.5 mm. Photos by Elia Nalini.

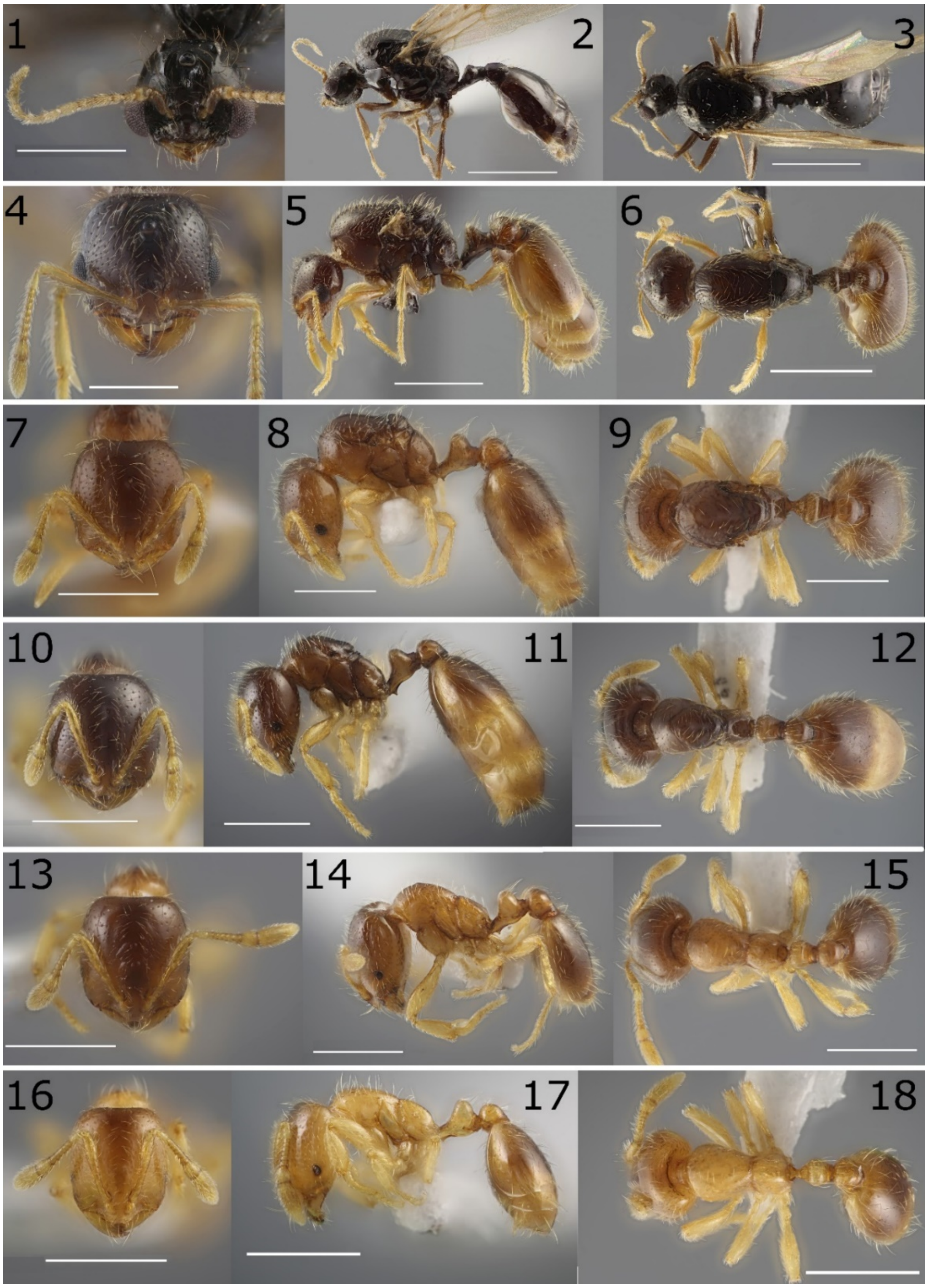
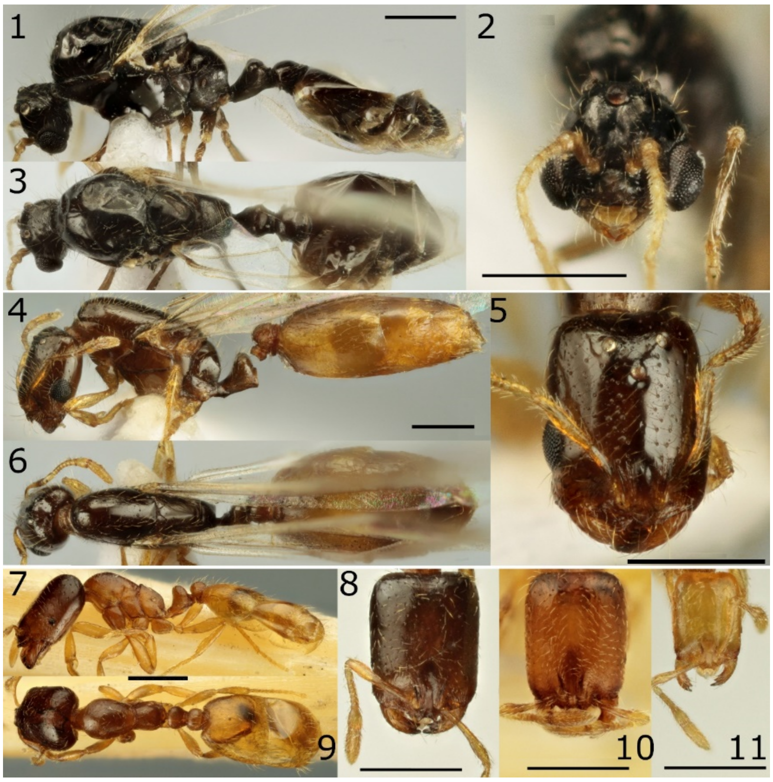


Fig. IV. *Solenopsis orbula* from Sardinia. Up to bottom: male, queen, workers. Scale bars: 0.5 mm. 1-3: male, 4-6: queen, 7-9: major worker, 10: medium-sized worker, 11: minor worker. Photos by Enrico Schifani.



Checklist and biogeography

By reviewing the existing literature, and especially those new records or taxonomic changes that were published since 2007, and in consideration of the novelties proposed in this paper, we list a total of 77 taxa (Tab. 2). In comparison to the checklist by VERDINELLI et al. (2007), we add 10 new species, 4 of which are the result of data presented in this study. Moreover, 5 other species are removed. Most of the species forming the Sardinian ant fauna belong to the subfamily Myrmicinae Lepeletier de Saint-Fargeau, 1835 (57%), about one third to Formicinae Latreille, 1809 (30%) while Dolichoderinae Forel, 1878 and Ponerinae Lepeletier de Saint-Fargeau, 1835 are much smaller groups (about 5% each) and finally Leptanillinae Emery, 1910 represent less than 3%.

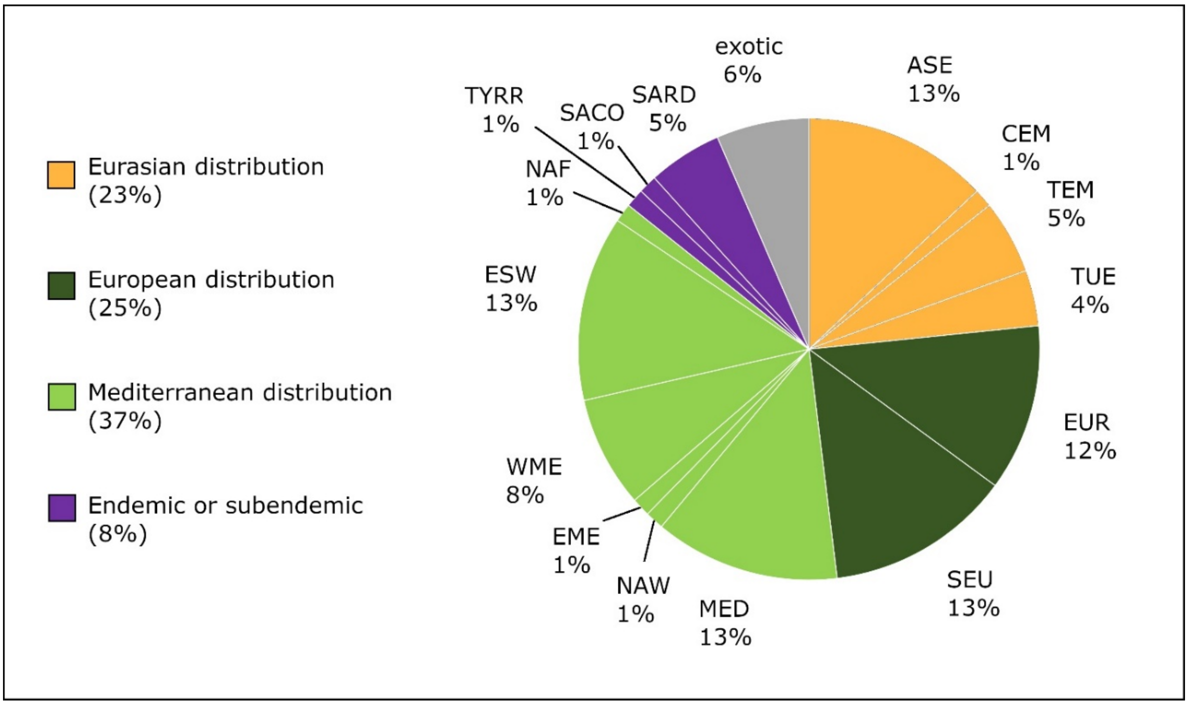
Tapinoma magnum Mayr, 1861 is added on the basis of the records published by SEIFERT et al. (2016). This species was previously considered under the name *Tapinoma nigerrimum* Nylander, 1856 in Italy (POLDI et al., 1995). VERDINELLI et al. (2007) had not included *T. nigerrimum* in their checklist without providing any explanation, however records of this species on the island exist at least since a century (EMERY, 1914). On the other hand, the presence of the rarely collected inquiline social parasite *Plagiolepis xene* Stärcke, 1936, whose host is *P. pygmaea* (Latreille, 1798), was discovered recently (LOI, 2013). *Camponotus universitatis* Forel, 1890, *Formica clara* Forel, 1886 *Lasius platythorax* Seifert, 1991 and *Tapinoma madeirense* Forel, 1895 were recorded for the first time by RIGATO & TONI (2011) (alongside *T. affinis*, but see the new species records section). The presence of *T. madeirense* in Sardinia (recorded by RIGATO & TONI, 2011) was then implicitly questioned by the following description of its cryptic species *T. subboreale* Seifert, 2012 from continental Europe (SEIFERT, 2012) but data later published by SEIFERT et al. (2016) confirmed the presence of the true *T. madeirense* in Sardinia. On the other hand, *Aphaenogaster fiorii* Emery, 1915 and *A. subterranea* (Latreille, 1798) were considered absent in Sardinia by Alicata & Schifani (2019) and Galkowski et al. (2019) respectively. Moreover, Rigato & Toni (2011) suggested that all past records of *Formica rufibarbis* Fabricius, 1793 from Sardinia should be attributed to *F. clara* instead. In addition, *Ponera sulcitana* Stefani, 1970 (a supposed endemism) was declared to be a junior synonym of the tramp species *Hypoponera punctatissima* (Roger, 1859) (Bolton & Fisher, 2011) and *Leptothorax exilis specularis* Emery, 1916 a junior synonym of *Temnothorax exilis* (Salata et al., 2018). Finally, Wagner et al. (2017) recorded *T. immigrans* and Seifert (2020) recorded *Lasius grandis* Forel, 1909 for the first time.

Tab. 2. Checklist of Sardinian ants and chorotypes of each species. Subfamilies are indicated by letters: D (Dolichoderinae), F (Formicinae), L (Leptanillinae) and M (Myrmicinae).

Sf	Species	Ch	Sf	Species	Ch
D	*Linepithema humile* (Mayr, 1868)	exotic	M	*Monomorium subopacum* (Smith, F., 1858)	MED
D	*Tapinoma madeirense* Forel, 1895	ESW	M	*Myrmecina graminicola* (Latreille, 1802)	EUR
D	*Tapinoma magnum* Mayr, 1861	WME	M	*Myrmecina melonii* Rigato, 1999	SARD
D	*Tapinoma simrothi* Krausse, 1911	MED	M	*Myrmica spinosior* Santschi, 1931	ESW
F	*Camponotus aethiops* (Latreille, 1798)	SEU	M	*Pheidole pallidula* (Nylander, 1849)	ESW
F	*Camponotus fallax* (Nylander, 1856)	TEM	M	*Solenopsis lusitanica* Emery, 1915	WME
F	*Camponotus gestroi* Emery, 1878	MED	M	*Solenopsis orbula* Emery, 1875	MED
F	*Camponotus lateralis* (Olivier, 1792)	MED	M	*Stenamma debile* (Foerster, 1850)	EUR
F	*Camponotus piceus* (Leach, 1825)	ESW	M	*Stenamma sardoum* Emery, 1915	SARD
F	*Camponotus universitatis* Forel, 1890	SEU	M	*Stenamma striatulum* Emery, 1895	SEU
F	*Camponotus vagus* (Scopoli, 1763)	CEM	M	*Strongylognathus testaceus* (Schenck, 1852)	TUE
F	*Colobopsis truncata* (Spinola, 1808)	TEM	M	*Strumigenys argiola* (Emery, 1869)	SEU
F	*Formica clara* Forel, 1886	ASE	M	*Strumigenys baudueri* (Emery, 1875)	SEU
F	*Formica cunicularia* Latreille, 1798	ASE	M	*Strumigenys membranifera* Emery, 1869	exotic
F	*Formica lugubris* Zetterstedt, 1838	exotic	M	*Strumigenys tenuipilis* Emery, 1915	SEU
F	*Lasius bicornis* (Foerster, 1850)	ASE	M	*Temnothorax aveli* (Bondroit, 1918)	ESW
F	*Lasius brunneus* (Latreille, 1798)	ASE	M	*Temnothorax exilis* (Emery, 1869)	SEU
F	*Lasius flavus* (Fabricius, 1782)	ASE	M	*Temnothorax kraussei* (Emery, 1915)	SEU
F	*Lasius emarginatus* (Olivier, 1792)	EUR	M	*Temnothorax lichtensteini* (Bondroit, 1918)	SEU
F	*Lasius grandis* Forel, 1909	ESW	M	*Temnothorax mediterraneus* Ward et al. 2014	ESW
F	*Lasius lasioides* (Emery, 1869)	MED	M	*Temnothorax nylanderi* (Foerster, 1850)	EUR
F	*Lasius niger* (Linnaeus, 1758)	ASE	M	*Temnothorax parvulus* (Schenck, 1852)	EUR
F	*Lasius paralienus* Seifert, 1992	EUR	M	*Temnothorax ravouxi* (André, 1896)	EUR
F	*Lasius platythorax* Seifert, 1991	ASE	M	*Temnothorax recedens* (Nylander, 1856)	MED
F	*Plagiolepis pallescens* Forel, 1889	ASE	M	*Temnothorax sardous* (Santschi, 1909)	SARD
F	*Plagiolepis pygmaea* (Latreille, 1798)	ASE	M	*Temnothorax tuberum* (Fabricius, 1775)	ASE
F	*Plagiolepis xene* Stärcke, 1936	ESW	M	*Temnothorax unifasciatus* (Latreille, 1798)	TEM
L	*Leptanilla doderoi* Emery, 1915	SARD	M	*Tetramorium atratulum* (Schenck, 1852)	TUE
L	*Leptanilla revelierii* Emery, 1870	WME	M	*Tetramorium brevicorne* Bondroit, 1918	SACO
M	*Aphaenogaster ichnusa* Santschi, 1925	ESW	M	*Tetramorium caespitum* (Linnaeus, 1758)	EUR
M	*Aphaenogaster sardoa* Mayr, 1853	NAW	M	*Tetramorium immigrans* Santschi, 1927	exotic
M	*Aphaenogaster senilis* Mayr, 1853	ESW	M	*Tetramorium meridionale* Emery, 1870	MED
M	*Aphanogaster spinosa* Emery, 1878	TYRR	M	*Tetramorium semilaeve* André, 1883	WME
M	*Cardiocondyla mauritanica* Forel, 1890	NAF	M	*Cryptopone ochracea* (Mayr, 1855)	TUE
M	*Crematogaster scutellaris* (Olivier, 1792)	WME	P	*Hypoponera eduardi* (Forel, 1894)	MED
M	*Messor capitatus* (Latreille, 1798)	WME	P	*Hypoponera punctatissima* (Roger, 1859)	exotic
M	*Messor ibericus* Santschi, 1931	SEU	P	*Ponera coarctata* (Latreille, 1802)	TUE
M	*Messor minor* (André, 1883)	MED	P	*Ponera testacea* Emery, 1895	EUR
M	*Messor wasmanni* Krausse, 1910	EME			

Biogeographically (see Fig. V), the Sardinian ant fauna contains a large number of widely-distributed species (48%): 23% of the Sardinian species fall into the Palearctic distribution category (ASE+CEM+TEM+TUE) and 25% in the European category (EUR+SEU) *sensu* Vigna Taglianti et al. (1999). It may be worth mentioning that widely distributed European species account for a smaller proportion of taxa among Sardinian ants in comparison to what they do in the context of other faunistic assemblages (Stoch & Vigna Taglianti, 2006). Mediterranean species are only slightly less numerous, amounting to about 45% of Sardinia's ant species. These are dominated by species with a Western Mediterranean distribution amounting to 32% of the fauna (WME, 8%; ESW, 13%; SARD: 5%; NAW, TYRR and SACO combined: 6%) and circum-Mediterranean species (13%, MED). The distinction between Maghrebian and South-Western European species is highly informative, as 10 species belong to the latter category and only 1 to the first. The prevalence of Sardinian endemics to Sardo-Corsican ones is 4:1 and well-reflects the general traits of the Sardinian biogeography (BACCETTI, 1983). Among endemic taxa, it is worth noting that *S. sardoum* was until recently erroneously considered to extend its distribution to mainland Europe (see RIGATO, 2011). Eastern-Mediterranean influences are about completely absent as expected (SCHIFANI & ALICATA, 2019). Exotic species are not particularly numerous (6%), although the status of some taxa such as *C. mauritanica* or even *L. niger* is unclear and targeted investigations in anthropogenic and urban areas are likely to discover additional taxa (SCHIFANI, 2019; BLATRIX et al. 2020).

Fig. V. Biogeographic components of the Sardinian ant fauna according to chorotypes.



Conclusions

The present study offers several faunistic novelties and a comprehensive summary over the Sardinian ant fauna, in addition to morphological information of general interest for the difficult study of the Mediterranean *Solenopsis*. While the understanding of some ant genera in Sardinia seems quite satisfactory (perhaps even for the enigmatic genus *Leptanilla*, see LEO & FANCELLO, 1990), the overall picture is definitely far from being conclusive. The number of 79 taxa currently considered to be present is not particularly high. In comparison, Sicily, which is about the same size of Sardinia, is grossly estimated to be inhabited by about 150 species (SCHIFANI & ALICATA, 2018) and the nearby Corsica, less than one third of the size of Sardinia, hosts 91 species according to the latest checklist (BLATRIX et al., 2018; 2020). Even the much smaller island of Crete is thought to be home to 100 taxa (SALATA et al., 2020), while there is no recent estimate for Cyprus. The reason behind this comparatively smaller number is most likely explained in part by still insufficient investigation. The considerable number of new species records recently produced by relatively limited investigation efforts reinforce this idea. At the same time, a role was likely also played by higher biogeographic isolation of Sardinia in comparison with both Corsica and Sicily. Viable connections between Corsica and Tuscany and between Sicily and Calabria are important to explain their current ant fauna assemblages. Many continental species are not found in Sardinia but inhabit Corsica, Sicily or both (e.g., *Myrmica sabuleti* Meinert, 1861, *Aphaenogaster italica* Bondroit, 1918, *A. subterranea* - see VERDINELLI et al., 2007; SCHIFANI & ALICATA, 2018; SEIFERT, 2018; GALKOWSKI et al., 2019). In addition, Sicily high number of species may be explained by Maghrebian and Balkan influxes only marginally able to reach Sardinia (ALICATA & SCHIFANI, 2019; SCHIFANI & ALICATA, 2019; CENTORAME et al., 2020). In particular, *Aphaenogaster sardoa* and *Myrmecina melonii* are the only two elements of the Sardinian fauna that testify an ancient colonization of Southern-Tyrrenian species (SCHIFANI et al., 2020c). On the other hand, it is notable that both Corsica and Sardinia entirely lack the subfamily Proceratiinae, which is found elsewhere in all neighbouring Mediterranean regions.

Taxonomic uncertainty is already quite evident for some species inhabiting Sardinia (in addition to those mentioned in this paper, see for example the Sardinian *Formica cunicularia* Latreille, 1798 according to SEIFERT & SCHULZ, 2009). Moreover, BLATRIX et al. (2020) proposed to consider the Corsican *T. unifasciatus* populations as a cryptic species (*T. cordieri* (Bondroit, 1918)) due to spines length and mtDNA differences and suggested that the local form chromatically similar to *L. emarginatus* and morphometrically clustering within *L. grandis* (see SEIFERT 2020) may be an undescribed cryptic species. At the same time, there are a number of old species records that we maintained in the list but that clearly need to be verified due to the taxonomic advancements of the last years. Good examples can be found for the genera *Lasius*, *Ponera* and *Temnothorax* which witnessed major taxonomic improvements over the last few decades (e.g., SEIFERT, 2020; CSŐSZ & SEIFERT, 2003; CSŐSZ et al., 2015). The old, isolated finding of *H. punctatissima* could potentially represent instead *H. ergatandria* (Forel, 1893), a cryptic tramp species whose presence in Italy has never been checked for despite being recorded across Europe (SEIFERT, 2013).

While many novelties are expected from further investigation, the figure regarding the main biogeographical traits of the Sardinian ant fauna that were identified in this paper is unlikely to change dramatically. The present checklist will offer a useful and solid basis to direct future researches.

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Author contributions

ES: Conceptualization, Methodology, Data Curation, Visualization, Writing-original draft, Writing – review & editing.

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CHAPTER 3 – CROSS-TAXON CONGRUENCE

The main aim of this chapter was to assess the diversity patterns of several taxonomical groups - vascular plants and six groups of ground-dwelling arthropods (pseudoscorpions, spiders, darkling, rove and ground beetles, ants) collected in the fieldwork phase to evaluate cross-taxon congruence in species richness and composition, and explore the effect of environmental variables (spatial-topographic, bioclimatic and landscape-level variables) and biological interactions on taxa concordance (Subchapter 3.1, paper 7).

Subchapter 3.1: taxa concordance

Paper 7

Bazzato E., Caria M., Lallai E., Schifani E., Cillo D., Maccherini S., Bacaro G., Marignani M., (prepared). Cross-taxon congruence in small woodlots outside forest along a land-use intensification gradient.

Subchapter 3.1: taxa concordance

Cross-taxon congruence in small woodlots outside forest along a land-use intensification gradient

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Abstract

Landscape modifications are among the most important direct drivers of ecosystem service changes at global, national and local scales. Small remnant patches of human-modified land-use provide fundamental ecosystem services and contribute to the human wellbeing.

We explored the importance of small woodlots outside forest for biodiversity conservation in a Mediterranean gradient of land-use intensification by using a cross-taxon approach focused on vascular plants and six ground-dwelling invertebrates: pseudoscorpions, spiders, but also darkling, rove and ground beetles, and ants.

We investigated how assemblages of these different taxonomic groups, respond to different landscape contexts surrounding the small patches out-of-forest. Our findings showed that landscape context exerted a stronger effect on species composition than species richness, highlighting the need to consider a suitable measure for interpreting ecological processes.

We found that land-use intensification led to a general homogenization on species richness and beta diversity patterns of almost of considered taxa (except pseudoscorpions, spiders and plants). However, we also found a relatively high level of richness and abundance of ground-dwelling invertebrates in small woodlots outside forest surrounded by land-uses at high or intermediate disturbance, underlining the need to preserve these remaining patches in human-fragmented landscape. This study also showed how the highly complex and variable interplay of environmental drivers and biological interactions influenced the cross-taxon congruence. Environmental filtering, direct and indirect biotic interactions resulted as important drivers to determine taxon diversity patterns and congruence among vascular plants and invertebrates of high trophic-level, but also among ground-dwelling taxa groups.

Although little is known about the biological interactions that drive cross-taxon congruence of diversity, the inclusion of other trophic levels, together with lower trophic level, can provide useful indications for planning and management conservation efforts of small woodlots outside forest in fragmented landscapes. Understanding how the different drivers influence the assemblage patterns of different taxonomic groups is essential to comprehend the role of small woodlots outside forest supporting high levels of biodiversity and providing important ecosystem services.

Keywords: vascular plants, invertebrates, Multi-taxa, diversity patterns, biological interaction

Bazzato E., Caria M., Lallai E., Schifani E., Cillo D., Maccherini S., Bacaro G., Marignani M., (prepared). Cross-taxon congruence in small woodlots outside forest along a land-use intensification gradient.

Introduction

Landscape modifications are among the most important direct drivers of ecosystem service changes at global, national and local scales. A large percentage of remnant forestland is projected to be converted to other uses by 2050, due to agriculture and urban sprawl (Millennium Ecosystem Assessment 2005). The expansion of agriculture activities and urban infrastructures have occurred over most of the Mediterranean region, and especially the Mediterranean basin (Falcucci et al. 2007; Marignani et al. 2017a) with a constantly increasing rate of land consumption and different sprawling patterns at the local scale (Strollo et al. 2020).

Land-use intensification (Newbold et al. 2015), habitat loss and fragmentation (Fischer and Lindenmayer 2007) are some of the most significant threats to biodiversity and key topics in landscape ecology (Wilson et al. 2016). Fragmentation leads to the reduction of continuous tracts of habitat to smaller and more isolated remnant patches separated by a matrix of human-transformed land cover (Haddad et al. 2015). Although large, intact and well-connected patches are vitally important for the maintenance of ecological processes and biodiversity conservation (Lindenmayer 2019), small isolated remnant patches and appropriate matrix management can be considered as valuable complements (Fischer and Lindenmayer 2002). Several recent reviews and meta-analyses have demonstrated the high conservation value and importance of small remnant patches, especially in human-modified landscapes (Fischer and Lindenmayer 2002; Le Roux et al. 2015; Tulloch et al. 2016; Lindenmayer 2019). Increasing attention has been paid to sampling strategies of Trees Outside Forests (TOF; FAO 2010; for Italy, Baffetta et al. 2011; Corona et al. 2011; Sallustio et al. 2018): small patches with an area less than 0.5 ha and different spatial patterns, including scattered individual trees, linear arboreal elements and wood residual fragments, more specifically called Small Woodlots Outside Forests (hereafter SWOF; Italian National Forest Inventory; http://www.infc.it), distributed along watercourses, canals, roads and highways or, in general, over all-natural and human-modified land-uses (FAO 2010; de Foresta et al. 2013). They play an important role in climate mitigation, soil and water resources protection, agricultural crop protection and, thanks to their wide distribution, promote the conservation of biodiversity by representing ecological corridors and offering the living fences for numerous animal and plant species within fragmented landscapes (Bellefontaine et al. 2001; Manning et al. 2006). Even though human-modified and fragmented landscapes are currently considered complex and heterogeneous environments (Fischer and Lindenmayer 2006; Fahrig 2017), these small patches are excluded from most connectivity analyses and conservation initiatives (Cadavid-Florez et al. 2020). Considering that policymakers, land planners, and conservation organizations generally focus their efforts on large, intact and well-connected areas by underestimating the importance of these fragments (Wintle et al. 2019), more empirical studies on taxa inhabiting small patches are urgently needed to support and promote their conservation (Lindenmayer 2019). Given this urgent need, several authors have pointed out the important role of scattered trees for biodiversity conservation in commercial forests (Mazurek and Zielinski 2004; Matveinen-Huju et al. 2006), agricultural areas (Dunn 2000; Harvey et al. 2006; Manning et al. 2006; DeMars et al. 2010; Fischer et al. 2010; Frizzo and Vasconcelos 2013) and urban landscapes (Yasuda and Koike 2009; Stagoll et al. 2012). However, only a few studies analysed and compared the abundance and richness of multiple taxonomic groups across different areas and landscape contexts supporting scattered trees (Frizzo and Vasconcelos 2013; Azihou et al. 2013; Le Roux et al. 2018; Prevedello et al. 2018).

While robust knowledge on biodiversity patterns across different landscape contexts is needed to guide conservation decision-making and mitigate the effects of human-induced changes, not every biological group can be cost-effectively measured (Lindenmayer and Likens 2011; Ware et al.

2018). Collecting data on many taxonomic groups is generally time and financially consuming due to the high sampling effort required (Favreau et al. 2006). Given the considerable resources needed, often not available, a growing number of studies are quantifying, exploring and summarizing knowledge on cross-taxon congruence across a wide range of geographical regions in natural or human-modified landscapes (Hess et al. 2006; Rodrigues and Brooks 2007; Westgate et al. 2014, 2017). Cross-taxon surrogate approaches are sustained by the assumption that a subset of taxa (i.e., surrogates) can be representative of biodiversity of other taxa (i.e., targets), or broader patterns of biodiversity (Westgate et al. 2014; Ware et al. 2018). Basic surrogate taxa requirements include a well-known taxonomy, relatively stable systematics, a widely studied ecology, congruent response or sensitivity to conditions and environmental gradients with the target group (Oliver et al. 1998; Zara et al. 2021). Recent reviews have indicated that species richness congruence is stronger at large spatial scales and next to the equator, while species composition congruence is high at large extents and grain sizes (Westgate et al. 2014, 2017). The difficulties in identifying congruence in biodiversity patterns could be attributed to the spatial and temporal scales (Westgate et al. 2014, 2017), to differential responses of groups to environmental variables (Heino et al. 2009), as well as to the differences in the measure of diversity and analytical methods adopted (Gioria et al. 2011).

Studies on coarse scales are useful for the early stages of conservation planning, nevertheless, local or regional scale studies are needed for planning and management conservation efforts (Chiarucci et al. 2011; Santi et al. 2016; Burrascano et al. 2018).

Surprisingly enough, few studies in the Mediterranean areas are conducted (Larsen et al. 2012) and we are not aware of any study focused on the cross-taxa congruence of Mediterranean SWOFs.

To fill these gaps, in this study we compared diversity patterns of vascular plants and six groups of ground-dwelling arthropods (pseudoscorpions, spiders, darkling, rove and ground beetles, ants) across different areas supporting SWOFs in a Mediterranean fragmented landscape.

We focused the study on vascular plants and arthropods as potential biological indicator taxa, considering that they constitute more than 80% of all currently described species (Stork 2018), covering a wide range of diet specializations and contribute to important ecosystem processes (Yang and Gratton 2014) with combined responses to multiple stressors (Noriega et al. 2018).

We specifically asked (1) does the land cover matrix exert an influence on the diversity pattern in terms of species richness and composition? (2) which are the effect of environmental variables and biological interactions on taxa concordance in SWOFs located along the gradient of land-use intensification?

Considering that cross-taxon congruence can be driven by different mechanisms including (i) similar response to the same environmental gradient, (ii) responses to different, but correlated environmental gradients, (iii) common loss of diversity and (iv) biotic interactions (Gaston 1996; Heino 2010), we hypothesized to observe:

a concordant variation of taxonomic groups, with an increase of diversity in SWOFs located in natural and semi-natural areas compared to those in human-modified land-uses;

a conditional effect of environmental variables on taxa concordance in species composition, as it could derive simply by shared responses to the same stressor environmental gradient;

a pure effect of environmental variables and/or biological interactions on the variation of taxa communities in SWOFs located along the strong land-use gradient.

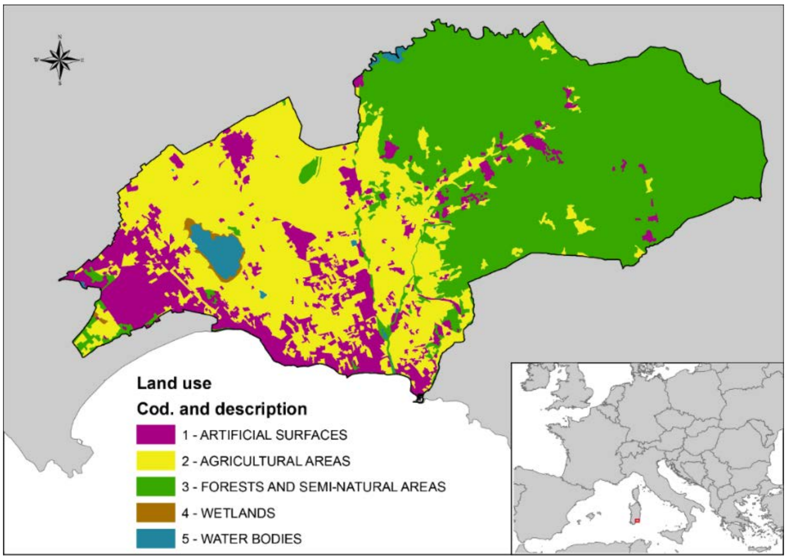
Although identifying the main determinants of cross-taxon congruence is complicated, simultaneously examining the responses of multiple taxonomic groups to the same stressor environmental gradient could contribute to better understand the effect of human-disturbance on SWOFs diversity patterns (Wintle et al. 2019; Lindenmayer 2019), with theoretical and applied implications for small woodlots outside forest management. In a perspective of the increasing need for Nature-based Solutions, quantitative tests of these expectations using multiple taxonomic groups would also provide evidence to explicitly considered SWOFs in future conservation programs and justify conservation efforts (Wintle et al. 2019; Lindenmayer 2019), particularly in Mediterranean human-modified landscapes, one of the hotspots for biodiversity conservation (Marignani et al. 2017a, b; Médail 2017).

Materials and methods

Study area

The study area is located in the Metropolitan Area of Cagliari (Southern Sardinia, Italy), a medium-sized functional urban area characterized by three levels of fragmentation degrees (Palumbo et al. 2020) covering 18,300 hectares (Fig. 1). The study was conducted in an area characterized by a gradient of land-use intensification that runs roughly in an east-south/ direction from natural areas to urbanized zones (natural and semi-natural, agriculture and artificial areas) and an elevation gradient from 0 m a.s.l. to 700 m a.s.l.. The climate is typically Mediterranean, with a strong seasonality characterized by dry and hot summers and relatively rainy and mild winters. The area is ascribable to the Mediterranean pluvioseasonal oceanic macrobioclimate, with one class of continentality (strong euoceanic), four thermotypic horizons (from lower themomediterranean to upper mesomediterranean) and five ombrothermic horizons (from lower dry to lower humid), resulting in a combination of 11 isobioclimates (Canu et al. 2015).

Fig. 1. Study area located in the Metropolitan Area of Cagliari, Southern Sardinia, characterized by a gradient of land-use intensification from hilly natural areas to urbanised coastline zones (natural and semi-natural, agriculture and urban areas).



Sampling design

Using photointerpretation of digital colour orthophotos, we identified and mapped all the SWOFs ranging from 0.05 to 0.5 hectares. Hence, SWOFs were assigned to the dominant landscape contexts, according to the first hierarchical level of regional land-use map (RAS 2020): natural and semi-natural areas (NAT), agricultural areas (AGR), and urban and artificial surfaces (URB). SWOFs with a size of less than 0.1 hectare (about 42%) were excluded, as well as those surrounded by a mixed land-use matrix (about 1.50%). From a total of 201 detected SWOFs (67 in URB, 70 in AGR, 64 in NAT), a proportionally stratified random sampling was carried out to select a total of 30 SWOFs along the land-use gradient. Unfortunately, due to the difficulties to survey in private gardens and similar areas, located in the URB matrix, only 8 urban SWOFs were sampled and the remaining sites (up to 30) were equally assigned to the other two land use types (11 sites in NAT and AGR).

SWOFs selected in each land-use stratum were sampled with a systematic sampling design. For each SWOF, we used the centroid as central middle point and built a linear transect, which was radiated from the centroid to the farthest sides of SWOF's boundaries

Data collection

We targeted vascular plants and six groups of ground-dwelling invertebrates: pseudoscorpions (Arachnida, Pseudoscorpiones), spiders (Arachnida, Araneae), darkling beetles (Insecta, Coleoptera, Tenebrionidae), rove beetles (Insecta, Coleoptera, Staphylinidae), ground beetles (Insecta, Coleoptera, Carabidae), and ants (Insecta, Hymenopera, Formicidae). For insects, we decided to consider taxa at the family level rather than the order level to take into account ecologically coherent phylogenetic groups (Evans et al. 2019; Oberprieler et al. 2020).

Vascular plant occurrences and abundances were recorded from April to August 2018. Abundances were evaluated as a cover percentage within five replicate plots of 1 sqm in each transect per site and then were summed across the five plots in each transect per site (i.e., SWOF).

Ground-dwelling invertebrates were collected by means of pitfall traps. Pitfall traps are considered a standard, cost-effective and reliable method for sampling mobile, surface-dwelling arthropods (Yi et al. 2012; Skvarla et al. 2014). Following Brandmayr et al. (2005), traps were made by small plastic vessels, 9 cm in diameter and 11 cm deep, with a small hole near the top to allow the drainage of rainwater. Five pitfall traps per transect were located in the centre of each plot. Each trap was filled with wine-vinegar saturated by sodium chloride as preservation method. Ground-dwelling invertebrates were collected for a year (from April 2018 to May 2019) to catch the highest biological activity of each group. The traps were emptied every 30-40 days; thus, nine trap-emptyings made up a year-sample. Since some traps were found overturned or tampered, 101 out of the 1350 placed traps (5 traps for each of the 30 sampling sites, for nine sampling periods) were not included in the analysis. Occurrences and abundances collected by 1249 traps were summed along the time periods and considered in the study.

Environmental factors

Three distinct predictor sets were considered at the site level: (i) spatial-topographic factors (ii) bioclimatic variables (iii) and landscape measures (Table 1).

The first set of environmental predictors consisted of variables describing spatial-topographic characteristics. Geographic coordinates, expressed as angular units (degrees) in the WGS84 geographic coordinate system (EPSG code of 4326), were recorded in each plot/trap through a global positioning system (GPS) instrument. Elevation, inclination, and exposition were derived from the Digital Terrain Model (DTM, resolution 10 m), using ArcGIS 10.2.1 (ESRI 2014). ArcGIS 10.2.1 (ESRI 2014) was also used to calculate three spatial distances from the centroid of each SWOF to the nearest coastline (coastal distance), to the nearest river (river distance), and the nearest artificial or natural lake and lagoon (lake distance).

The second set of environmental predictors consisted of 19 biologically meaningful bioclimatic variables (Bazzato et al. 2021), related to temperature (BIO01-BIO07 and BIO10-BIO11), precipitation (BIO12-BIO17), and bioclimatic variables related to both temperature and precipitation (BIO08-BIO09 and BIO18-BIO19).

The third set of environmental predictors consisted of 17 metrics at the landscape-level, describing compositional and configurational features of the surrounding landscape of each site (see McGarigal et al. 2002 for a complete description of each metric, Table 1). Landscape metrics were calculated within 500 m buffer distance of each SWOF, using a regional Land-use Map at the third hierarchical level of detail (CORINE legend, scale 1:25.000; RAS 2008) and Patch Analyst extension (Elkie et al. 1999; Rempel et al. 2012).

Table 1. List of the measured spatial-topographic and bioclimate parameters, and landscape metrics.

Predictor set	Variable name	Variable description	Type of variable
Spatial-topographic	X	x geographical coordinate (degrees)	Spatial variable
	Y	y geographical coordinate (degrees)	Spatial variable
	Z	elevation (m)	Topographic variable
	Inclination	inclination (°)	Topographic variable
	Exposition	exposition (°)	Topographic variable
	Coast_dist	Distance from the coast (m)	Spatial variable
	Rivers_dist	Distance from the river (m)	Spatial variable
	Lake_dist	Distance from the lake and lagoon (m)	Spatial variable
Bioclimatic	BIO01	Annual Mean Temperature (°C)	Temperature-related variable
	BIO02	Mean Diurnal Range (Mean of monthly (max temp - min temp)) (°C)	Temperature-related variable
	BIO03	Isothermality (BIO2/BIO7) (x 100)	Temperature-related variable
	BIO04	Temperature Seasonality (standard deviation x 100)	Temperature-related variable
	BIO05	Maximum Temperature of Warmest Month (°C)	Temperature-related variable
	BIO06	Minimum Temperature of Coldest Month (°C)	Temperature-related variable
	BIO07	Temperature Annual Range (BIO5-BIO6) (°C)	Temperature-related variable
	BIO08	Mean Temperature of Wettest Quarter (°C)	Temperature-related and rainfall-related variable
	BIO09	Mean Temperature of Driest Quarter (°C)	Temperature-related and rainfall-related variable
	BIO10	Mean Temperature of Warmest Quarter (°C)	Temperature-related variable
	BIO11	Mean Temperature of Coldest Quarter (°C)	Temperature-related variable
	BIO12	Annual Precipitation (mm)	Rainfall-related variable
	BIO13	Precipitation of Wettest Month (mm)	Rainfall-related variable
	BIO14	Precipitation of Driest Month (mm)	Rainfall-related variable
	BIO15	Precipitation Seasonality (Coefficient of Variation)	Rainfall-related variable
	BIO16	Precipitation of Wettest Quarter (mm)	Rainfall-related variable
	BIO17	Precipitation of Driest Quarter (mm)	Rainfall-related variable
	BIO18	Precipitation of Warmest Quarter (mm)	Temperature-related and rainfall-related variable

	BIO19	Precipitation of Coldest Quarter (mm)	Temperature-related and rainfall-related variable
Landscape	SDI	Shannon's Diversity Index	Diversity Metric
	SEI	Shannon's Evenness Index	Diversity Metric
	AWMSI	Area Weighted Mean Shape Index	Shape Metric
	MSI	Mean Shape Index	Shape Metric
	MPAR	Mean Perimeter-Area Ratio	Shape Metric
	MPFD	Mean Patch Fractal Dimension	Shape Metric
	AWMPFD	Area Weighted Mean Patch Fractal Dimension	Shape Metric
	TE	Total Edge	Edge Metric
	ED	Edge Density	Edge Metric
	MPE	Mean Patch Edge	Edge Metric
	MPS	Mean Patch Size	Patch Density & Size Metric
	NumP	No. of Patches	Patch Density & Size Metric
	MedPS	Median Patch Size	Patch Density & Size Metric
	PSCoV	Patch Size Coefficient of Variance	Patch Density & Size Metric
	PSSD	Patch Size Standard Deviation	Patch Density & Size Metric
	TLA	Total Landscape Area	Area Metric
	CA	Class Area	Area Metric

Statistical analyses

Patterns in species richness for seven taxonomic groups were calculated using site-based rarefaction curves (Chiarucci et al. 2008) to compare datasets considering the same sampling effort (Gotelli and Colwell 2001; Bacaro et al. 2016). Site-based rarefaction curves were performed collectively for seven taxonomic groups in the whole study area and each land-use matrix by means of the exact method (Kobayashi 1982) and 9999 permutations in the *specaccum* function of the *vegan* package (Oksanen et al. 2019).

Diversity pattern in species richness and composition among land-use matrices

Cross-taxon congruence in species richness and composition among land-use matrices was carried out at the SWOF-level using (1) permutational multivariate analyses-of-variance and (2) non-metric multidimensional scaling (NMDS), and (3) differences in beta diversity.

We performed the analyses using Euclidean distance on species richness data and zero-adjusted Bray-Curtis dissimilarity on square-root transformed abundance data of each taxonomic group. To take into account samples entirely defaunated (Clarke et al. 2006), the Bray-Curtis coefficients were zero-adjusted adding a ‘dummy species’ to the original abundance data where no species were recorded.

Based on richness data, permutational multivariate analysis-of-variance (PERMANOVA; Anderson 2001) was used to test the null hypothesis of no differences in the location (centroids) and/or spread among groups (Anderson and Walsh 2013; Anderson 2017): land-use matrices (three levels, fixed: natural and semi-natural areas, agricultural areas, urban and artificial surfaces), and taxonomic groups (fixed factor, seven levels crossed with land-use matrices). The significant interaction terms were then investigated using post-hoc permutational pair-wise comparison tests with PERMANOVA *t* statistic and 999 permutations. For each source of variation, we also calculated the pseudo multivariate variance components expressed in percentage. All tests were performed using 999 random permutations and the most conservative type III sum of squares for unbalanced designs (Anderson et al. 2008) in PRIMER v.6.1.12 software (Clarke and Gorley 2006) with the additional add-on package PERMANOVA+ v.1.0.2 (Anderson et al. 2008).

Multivariate analyses-of-variance (PERMANOVA; Anderson 2001) were also repeated separately for each taxonomic group by using species abundance data.

Differences in beta diversity among land-use matrices were tested with the *betadispersion2* R function (see Bacaro et al. 2013) and 9999 permutations. This function was used to test the null hypothesis of no differences in multivariate dispersion within groups (Anderson 2006), avoiding mixing within-group dissimilarities with between-group dissimilarities (see Bacaro et al. 2013). Comparisons between pairs of group mean dispersions were evaluated by Tukey honestly significant difference (HSD) tests for each pair of land-use matrix (Anderson 2006).

The tested differences among land use matrices were visualized by non-metric multidimensional scaling (NMDS) using the *metaMDS* function of the *vegan* package (Oksanen et al. 2019).

Effect of environment variables and biological interactions on cross-taxa concordance

We used Mantel Test to detect the linear independence of groups (Legendre and Fortin 2010) and the differences in their location, their dispersion, or some other distributional quality among taxonomic groups (Anderson and Walsh 2013); then, we used the Partial Mantel Test to simultaneously control the (co)variation associated with environmental variables (Guillot and Rousset 2013).

Results of the Mantel Test and Partial Mantel Test were compared to verify if after removing the conditional effect of environmental variables, the observed significant taxa concordance remained between all pairs of taxa.

For all tests, the monotonic relationships were evaluated based on zero-adjusted Bray-Curtis dissimilarity on square-root transformed abundance data and the Euclidean distance matrix of environmental predictors sets by using the Spearman rank correlation (Legendre and Legendre 2012). Given the hierarchically stratified sampling design adopted (i.e., each land-use strata encompassed more than one SWOF), both of these two types of tests were computed in the whole study area, constraining 999 permutations at the land-use level, using the argument ‘strata’ in the *mantel* and *mantel.partial* functions of the *vegan* package (Oksanen et al. 2019). Since sites within a land-use stratum are expected to be more similar to each other, this permutation approach returned a conservative estimate of the significance of the results (Oksanen et al. 2019).

To appraise the unique influence of environmental variables and/or biological interactions on the variation of taxa communities in SWOFs located along the strong land-use gradient, a variation partitioning analysis (Borcard et al. 1992) for each pair of taxa (i.e., 42 combinations) were applied using non-redundant environmental factors (divided into three main predictors sets, see Table 1).

To reduce the multicollinearity in each predictor set and describe the main gradient along the land-use gradient of the studied SWOFs, ten principal component analyses (PCA) were conducted by using *rda* function in the *vegan* package (Oksanen et al. 2019): one for each of the seven taxonomic groups based on Hellinger transformed data and one for each of the three environmental predictor sets (spatial-topographic and climate parameters, and landscape metrics) scaled to zero mean and unit variance (Borcard et al. 2011). The significance of individual principal component axes was evaluated using the broken-stick criterion through the *evplot* function (Borcard et al. 2011). Based on this criterion, the scores from principal components were used as predictors in the variation partition analyses (Annex S3): the first two principal components of pseudoscorpions (Pseudoscorpiones), the first three axes of vascular plants the first four axes of spiders (Araneae), and darkling beetles (Tenebrionidae), the first six principal components of ground beetles (Carabidae) and ants (Formicidae), the first seven axes of rove beetles (Staphylinidae); the first two axes of spatial-topographic and climate parameters, and the first three axes of landscape metrics.

Variation partitioning analyses were performed in the *vegan* package (Oksanen et al. 2019) based on Hellinger transformed data and adjusted R^2 statistics (Peres-Neto et al. 2006; Legendre and Legendre 2012). The significance of the fractions was tested by RDA analyses constraining 999 permutations within the land-use strata (Peres-Neto et al. 2006).

The proportion of total variation in each taxonomic group were therefore partitioned into different fractions (Økland 2003): [a] pure effect of another taxonomic group, [b] pure effect of spatial-topographic variables, [c] pure effect of climate variables, [d] pure effect of landscape measures, [e+f+g+h+i+j+k+l+m+n] partial shared effects of two/three set of factors, [o] total shared effect of all predictor sets, [p] unexplained variation.

Results

A total of 330 species of vascular plants and 66,412 individuals of ground-dwelling invertebrates grouped into six taxonomic groups were collected (Annex S1, Table 1): 390 individuals belonging to 13 species of pseudoscorpions; 2,821 spiders assigned to 106 species; 1,084 darkling beetles of 22 species; 7,215 rove beetles of 55 species; 2,777 ground beetles assigned to 38 species; and 52,125 ants identified to 35 species. During the field work several new species have been identified: new records for the island (vascular plants; spiders; ants) (Rosati et al. 2020; Schifani et al. 2021; Caria et al. accepted) and for Italy (spiders) (Caria et al. accepted).

Almost all rarefaction curves, except those of vascular plants and to lesser extent spiders, reached an asymptote indicating that the sampling provided a good coverage of the species present in the whole study area and in each land-use matrix (Fig. 2 e Fig. 3).

Fig. 2. Rarefaction curves of richness of each taxonomic group: pseudoscorpions (Pseudoscorpiones), spiders (Araneae), darkling beetles (Tenebrionidae), rove beetles (Staphylinidae), ground beetles (Carabidae), ants (Formicidae), and vascular plants. The number of SWOFs collected in the whole study area is shown.

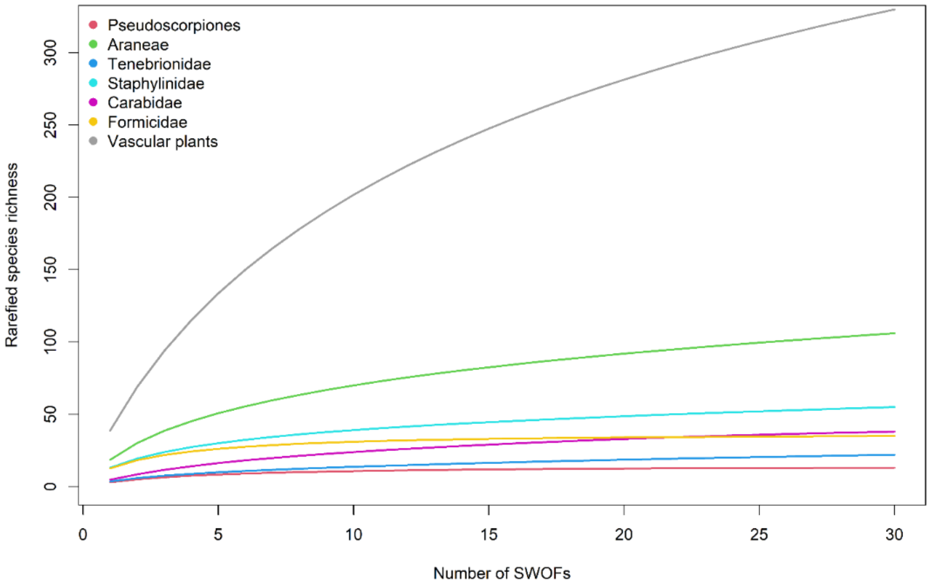
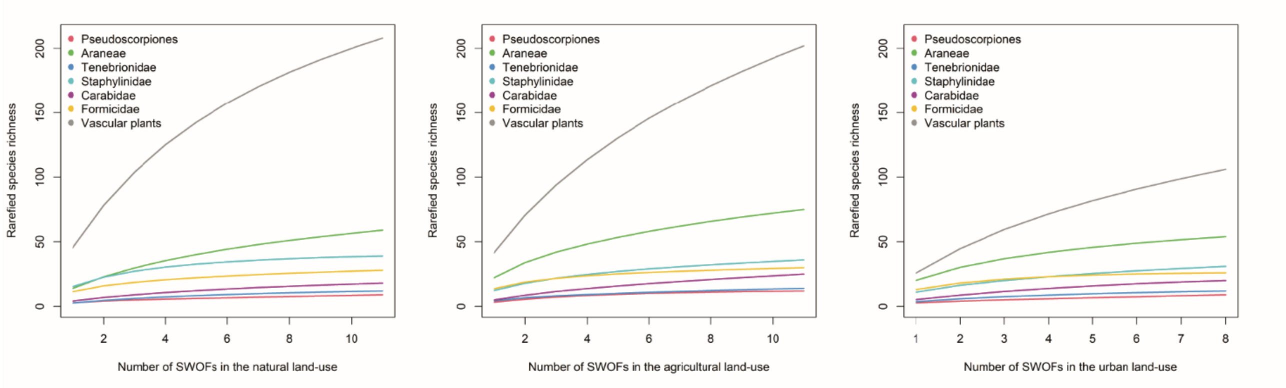


Fig. 3. Rarefaction curves of richness of each taxonomic group: pseudoscorpions (Pseudoscorpiones), spiders (Araneae), darkling beetles (Tenebrionidae), rove beetles (Staphylinidae), ground beetles (Carabidae), ants (Formicidae), and vascular plants. The number of SWOFs collected in each land use is shown.



Diversity pattern in species richness and composition among land-use matrices

Contrasting response patterns of species richness and abundance between the three levels of human-modified land-uses for seven different taxa were observed (Fig. 4; Annex S1, Table 2).

The mean number of pseudoscorpions, spiders and ant species were higher in the agricultural SWOFs than in the urban and natural SWOFs, but their abundance increased following the gradient of land-use intensification (from natural and semi-natural areas to urban ones) (Fig. 4). The highest values of the mean species richness and abundance of darkling beetles were in the agricultural SWOFs, followed by urban and natural SWOFs (Fig. 4). Regarding rove beetles, they showed the highest richness and abundance in natural and semi-natural SWOFs, whereas the mean values were similar in agricultural and urban SWOFs (Fig. 4).

The mean richness and abundance of ground beetles tended to increase along the gradient of land-use intensification (from natural and semi-natural areas to urban ones), whereas vascular plants showed an opposite trend both for richness and abundance (Fig. 4).

Considering species richness, PERMANOVA results revealed that taxonomic groups were the strongest, with residuals, contributing the largest components of variation to the overall model (Annex S2, Table 1). The main effect of land use matrices was less important than any other factors (Annex S2, Table 1). However, PERMANOVA clearly detected significant interactions of land use matrices with taxonomic groups (MA x GR; $p \leq 0.01$), suggesting specific response patterns of richness across SWOFs surrounded by different land use (Annex S2, Table 1). Pairwise comparisons further support this general observation, underlining that species richness of spiders differed between agricultural vs natural and semi-natural areas, but also across natural and semi-natural areas vs urban and artificial surfaces (Fig. 4; Annex S2, Table 2). Significant contrasts among natural and semi-natural areas vs urban and artificial surfaces were also identified for the richness of vascular plants. No differences in species richness were evident for any other taxonomic groups (pseudoscorpions, beetles and ants) (Fig. 4; Annex S2, Table 2).

When PERMANOVA analyses were conducted by using species composition data for each taxonomic group separately (Fig. 4), land use matrices had a significant effect on community composition of each group, except for rove beetles (MA $0.001 \leq p \leq 0.5$; Annex S2, Table 3). However, the largest components of variation to the overall models were explained by residual for all taxa (Annex S2, Table 3). In term of beta diversity, average dissimilarity from individual observation samples to their group centroid proved to be significantly different among land use matrices only for pseudoscorpions and spiders (Table 2). On average, beta diversity was slightly higher in SWOFs surrounded by agricultural areas for pseudoscorpions, and in SWOFs surrounded by natural and semi-natural areas for spiders, whereas the other taxonomic groups showed similar and not significant values across land use matrices (Table 2).

Fig. 4. Bar plots showing patterns of variation of richness and abundance (means ± SE) across the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). For each group, the mean value per SWOF, standard error bar, and statistically significant differences across each pair of land use, as indicated by PERMANOVA pair-wise tests (1) and BETA Tukey’ s post hoc test. (2) are shown. Significance codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05, (NS) not significant.

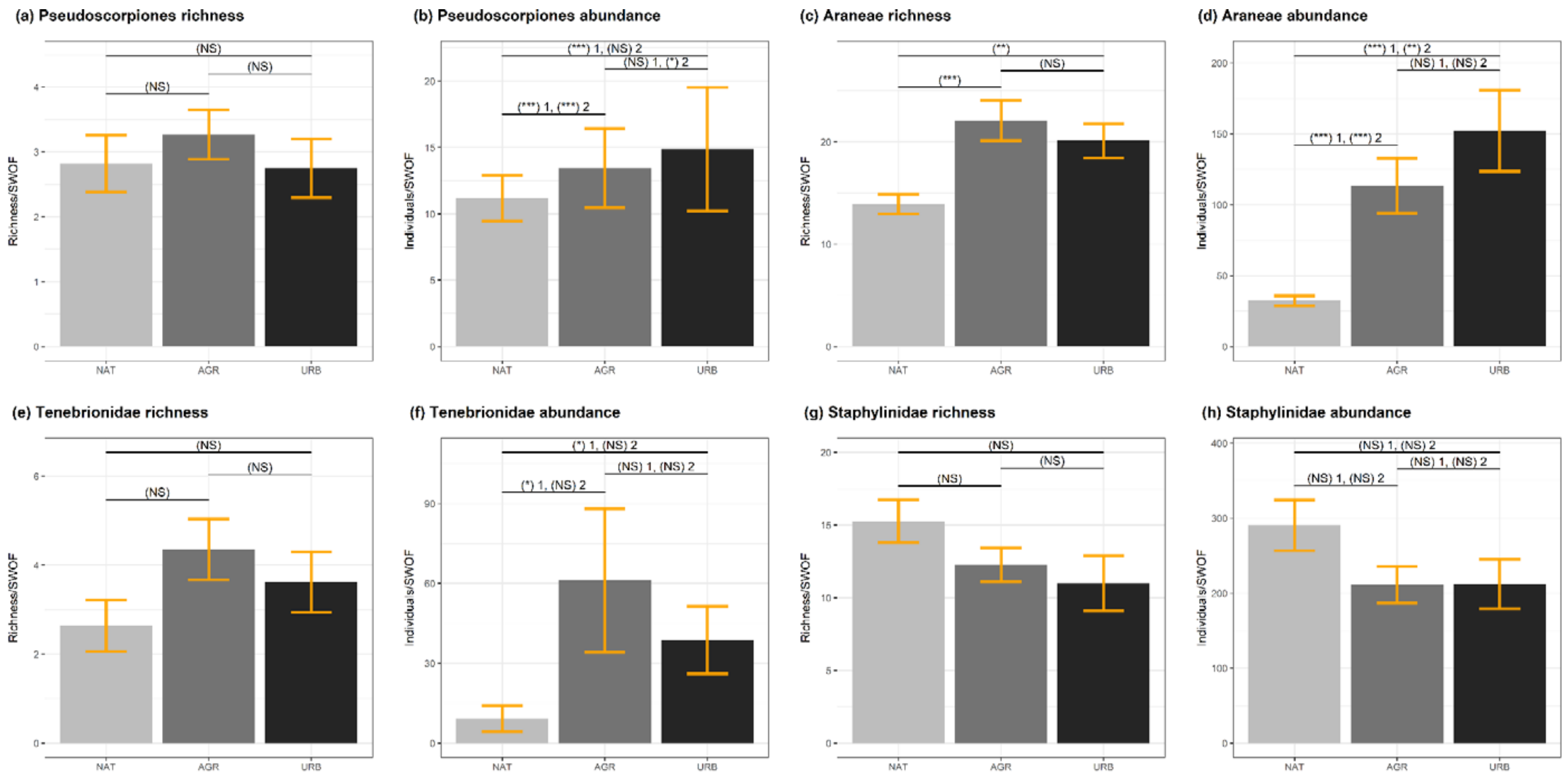
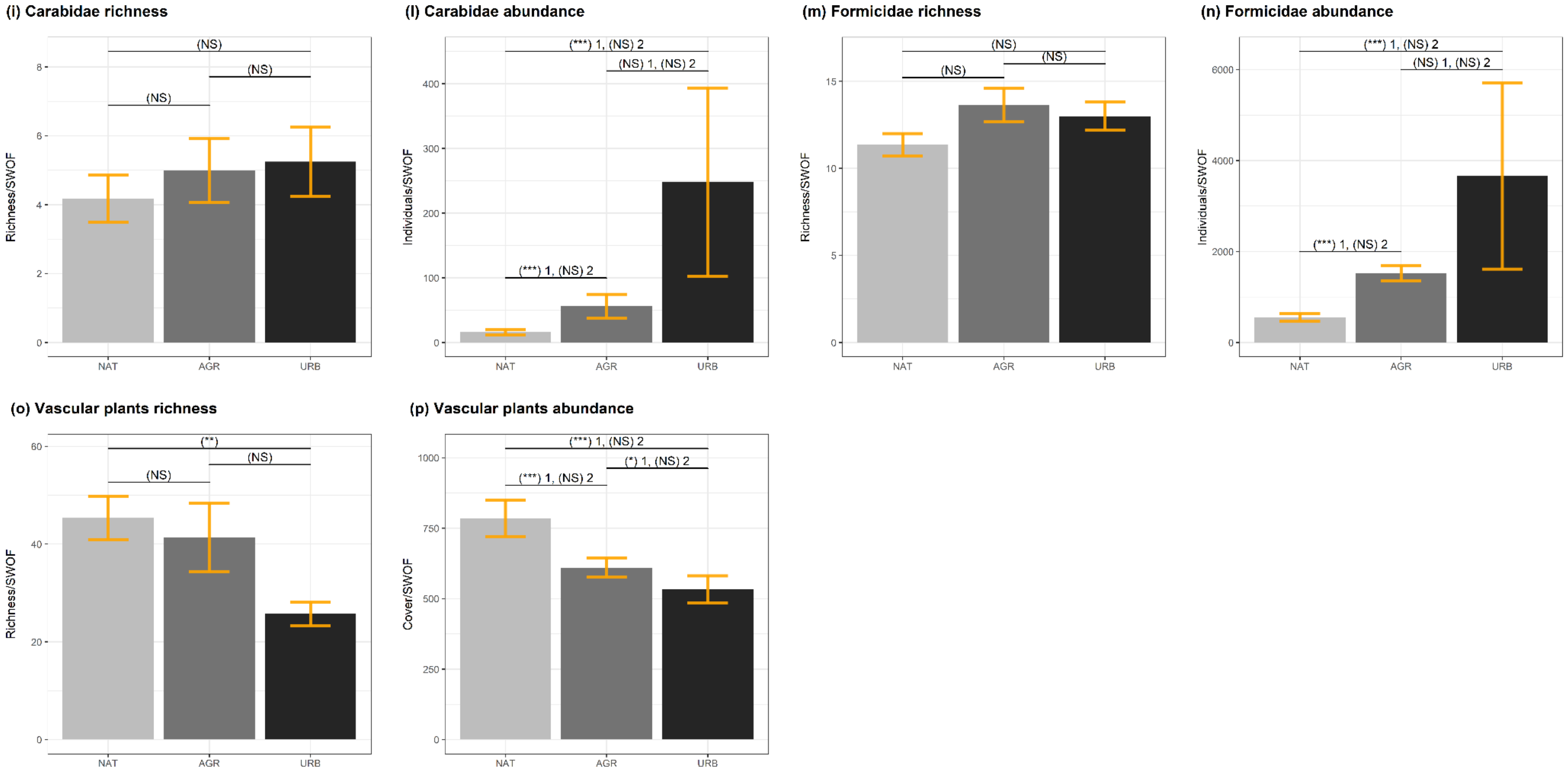


Fig. 4. Bar plots showing patterns of variation of richness and abundance (means ± SE) across the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). For each group, the mean value per SWOF, standard error bar, and statistically significant differences across each pair of land use, as indicated by PERMANOVA pair-wise tests (1) and BETA Tukey’ s post hoc test. (2) are shown. Significance codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05, (NS) not significant.



(i) Carabidae richness

(l) Carabidae abundance

(m) Formicidae richness

(n) Formicidae abundance

(o) Vascular plants richness

(p) Vascular plants abundance

Table 2. Mean beta diversity among land-use matrices calculated for each taxonomic group, separately, in considered 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). Taxonomic groups: pseudoscorpions (Pseudoscorpiones), spiders (Araneae), darkling beetles (Tenebrionidae), rove beetles (Staphylinidae), ground beetles (Carabidae), ants (Formicidae), and vascular plants.

Taxonomic group	**F and p-value**	**NAT**	**AGR**	**URB**
Pseudoscorpiones	11.99***	0.58	0.78	0.65
Araneae	12.17***	0.69	0.59	0.62
Tenebrionidae	1.43	0.62	0.62	0.68
Staphylinidae	1.98	0.48	0.44	0.45
Carabidae	2.11	0.73	0.80	0.74
Formicidae	1.68	0.56	0.58	0.63
Vascular plants	2.96	0.79	0.76	0.82

These general results were supported by Nonmetric multidimensional scaling (NMDS) ordination plots that identified the shift of group centroids among land-use types for almost all taxa, as well as high overlaps of 95% confidences ellipses between land-use matrices (Annex S2, Fig. 1).

Results of pair-wise tests suggested that community composition of all taxonomic groups, except rove beetles, in SWOFs surrounded by agricultural areas were significantly different from those in natural and semi-natural areas for the shift of their structure, and in the case of pseudoscorpions and spiders also for their heterogeneity (comparing results of PERMANOVA pair-wise tests (1) and BETA Tukey’s post hoc test. (2), Fig. 4). No statistically significant contrasts on community structures and heterogeneity of all ground-dwelling invertebrates (except pseudoscorpions) were detected among agricultural areas and urban/artificial surfaces, but remarkably community structure of vascular plants and beta diversity of pseudoscorpions differed across these two land-use matrices (Fig. 4). No significant difference in dispersion was observed between urban/artificial surfaces and natural/semi-natural areas considering community data of pseudoscorpions, darkling and ground beetles, ants, and vascular plants, indicating differences were due to the changing of their structure (Fig. 4) and not to the increase in within-group variance (Fig. 4). On the contrary, spider communities differed across these land-use types both for the change of structure and heterogeneity (Fig. 4). Not surprisingly (given Fig. 1, Annex S2), no statistically significant differences both in term of community structure and heterogeneity were detected for rove beetles among land-use matrices (Fig. 4; Annex S2, Fig. 1).

Effect of environment variables and biological interactions on cross-taxa concordance

Mantel tests among all pairs of seven taxa identified 17 significant positive correlations out of 21 pairwise comparisons (Table 3). All taxonomic groups had significant correlations with more than half other taxa (≥ 3): pseudoscorpions and vascular plants had the highest number (6) of significant correlations, followed by darkling beetles and ants (5). The strength of the pairwise correlations were generally low (min. value $\rho = 0.01$) to moderate (max. value $\rho = 0.61$). In particular, there was moderate significant correlations ($0.50 \leq \rho \leq 0.61$) between vascular plants and spiders, vascular plants and ants, spiders and ants; and low correlations ($\rho < 0.43$) between the remaining taxa.

All taxa (ground beetles excluded) were significantly correlated with environmental factors: the highest values of coefficient were found for vascular plants (ρ = 0.62, p ≤ 0.01) and spiders (ρ = 0.61, p ≤ 0.01), followed by ants (ρ = 0.45, p ≤ 0.05), while the other groups showed generally low values (ρ ≤ 3) (Table 3).

Table 3. Correlation coefficients between pairwise distance of environmental variables (Euclidean distance) and community data (Bray-Curtis dissimilarity) of taxonomic groups, calculated by using Mantel Test (Spearman rank correlation) in considered 30 SWOFs located in different land-use matrices. Significance codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

Taxonomic group	Araneae	Tenebrionidae	Staphylinidae	Carabidae	Formicidae	Vascular plants	Environment
Pseudoscorpiones	0.32**	0.18*	0.19*	0.32**	0.33**	0.42***	0.27*
Araneae	-	0.31***	0.2	0.43***	0.50***	0.61***	0.61**
Tenebrionidae		-	0.01	0.31***	0.22*	0.36***	0.28**
Staphylinidae			-	0.09	0.26*	0.28*	0.32*
Carabidae				-	0.23	0.37*	0.25
Formicidae					-	0.53***	0.45*
Vascular plants						-	0.62**

Table 4. Correlation coefficients between pairwise distance of community data (Bray-Curtis dissimilarity) of taxonomic groups, calculated by using Partial Mantel Test (Spearman rank correlation) in considered 30 SWOFs located in different land-use matrices when the environmental effect was removed. Data in bold indicate significant p-values: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

Taxonomic group	Araneae	Tenebrionidae	Staphylinidae	Carabidae	Formicidae	Vascular plants
Pseudoscorpiones	0.21	0.12	0.11*	0.27*	0.25*	0.33***
Araneae	-	0.18**	0.00	0.37***	0.32***	0.37***
Tenebrionidae		-	-0.09	0.26***	0.11	0.25**
Staphylinidae			-	0.01	0.14	0.11
Carabidae				-	0.14	0.28**
Formicidae					-	0.36**

After removing the effect of environmental variables on taxa congruence in partial Mantel tests, only 12 of the 21 pairwise correlations appeared weaker but still significant (Table 4). Vascular plants maintained the highest number (5) of significant correlations with other taxa, followed by pseudoscorpions, spiders and ground beetles (4); rove and darkling beetles and ants remained correlated with a low number of other taxa (≤ 2) (Table 4). More precisely, the strength of the inter-group congruences were similar (0.32 ≤ ρ ≤ 0.37) between vascular plants and spiders, vascular plants and ants, ground beetles and spiders, but also between vascular plants and pseudoscorpions, spiders and ants; lower strength (0.25 ≤ ρ ≤ 0.28) were found between ground beetles and vascular plants, ground beetles and pseudoscorpions, ground and darkling beetles, pseudoscorpions and ants, darkling beetles and vascular plants.

Results of 42 variation partitioning analyses showed that although part of the variability in the community structures remained unexplained, the explanatory capacity of predictor variables was generally high in most analyses (Annex S4, fraction [abcdefghijklmno]). The relative importance of each set of predictors for structuring taxa communities along the land-use gradient differed markedly (Fig. 5).

Darkling beetles and vascular plants were able to explain a large and significant degree of variance in pseudoscorpion communities (8.35%, p = 0.033; 10.42%, p = 0.013, respectively; fraction [a]); a considerable and significant degree of variance was also explained by climate variables (6.67%, p = 0.018) and landscape metrics (6.34%, p = 0.028) when darkling beetles and vascular plants (fraction [c] and [d]) were considered as explanatory taxonomic group, respectively.

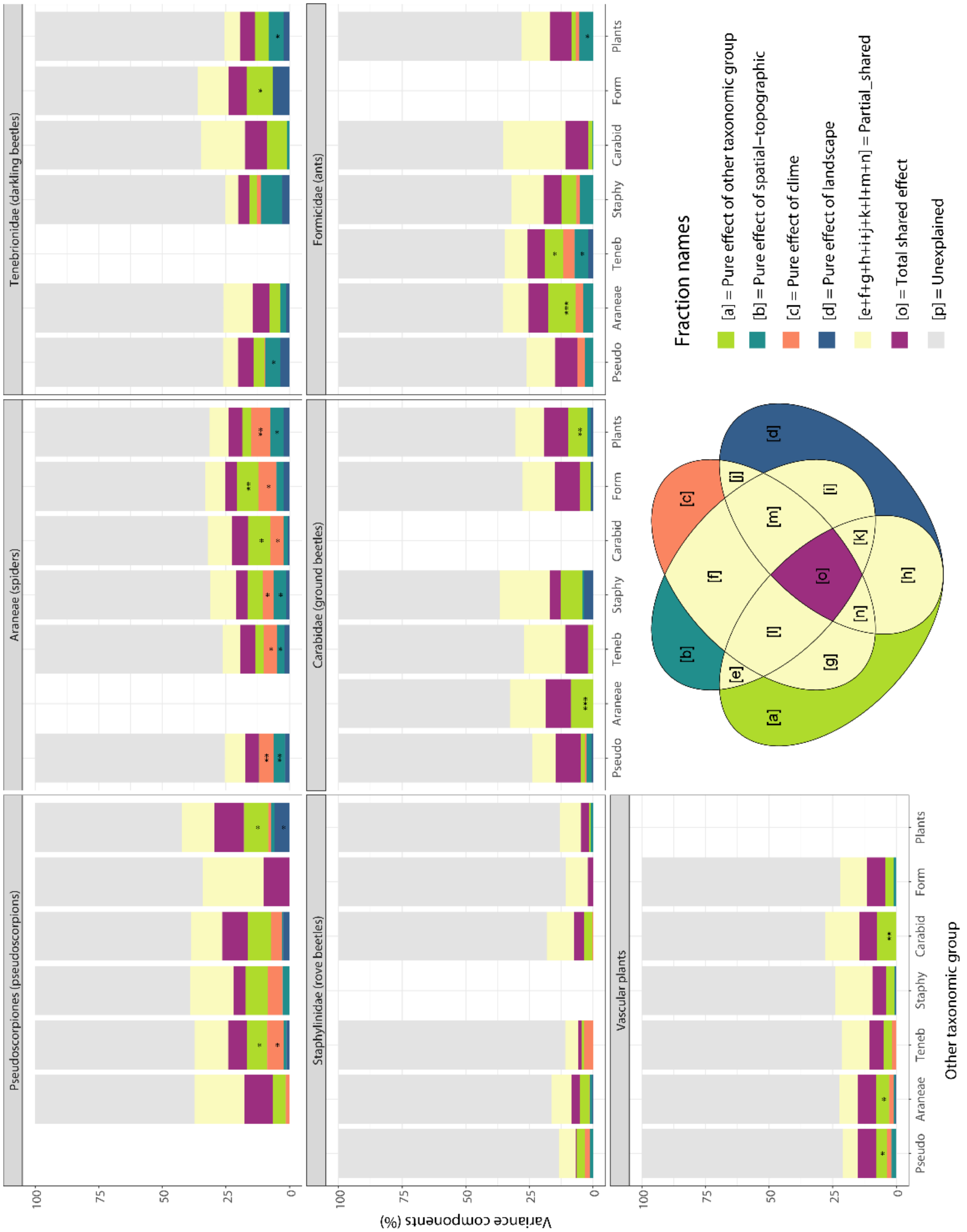
Variation in the spider communities were more explained by ground beetles and ants (9.21%, p = 0.017; 9.02%, p = 0.008, respectively; fraction [a]), than spatial-topographic and climate variables: the variation attributed solely to spatial-topographic variables (Fig. 5, fraction [b]) was 4.83% (p = 0.010), 2.98% (p = 0.049), 5.36% (p = 0.023), 5.77% (p = 0.015), when pseudoscorpions, darkling and rove beetles, and vascular plants were set as independent variable, respectively; the unique contribution of clime (fraction [c]) varied from 4.62% to 8.60% ($0.003 \leq p \leq 0.046$) according to the taxonomic group considered as predictor.

In darkling beetles, only ants (11.71%, p = 0.035) explained a significant degree of variance, followed by spatial-topographic variables when pseudoscorpions and vascular plants were set as independent predictors (6.28%, p = 0.040; 6.16%, p = 0.044, respectively; fraction [b]).

When the ant communities were considered as dependent variables, most of the variation were attributed to the pure effect of another taxonomic group (spiders, 10.88%, p = 0.001; darkling beetles, 7.50%, p = 0.014) and to unique effect of spatial-topographic variables: spatial-topographic variables (fraction [b]) alone explained 5.61% (p = 0.030) and 5.55% (p = 0.046) of variance when darkling beetles and vascular plants were used as explanatory taxonomic group, respectively.

The unique effect of environmental predictor sets (spatial-topographic and climate variables, landscape metrics) was found to be not significant for ground beetles and vascular plants ($p \geq 0.05$), while most of the variation in these two groups were attributed to the pure effect of other biological groups: spiders (9.16%, p = 0.001) and vascular plants (7.90%, p = 0.005) appeared as the strongest predictor, explaining the greatest and significant proportion of variation in ground beetle communities; pseudoscorpions, spiders and ground beetles (4.33%, p = 0.017; 5.24%, p = 0.031; 7.90%, p = 0.006, respectively; fraction [a]) were important independent components in the variation of vascular plants.

Fig. 5. Partitioning of variation in community composition of each taxonomic group recorded along the land-use gradient. Dependent variables (Bray-Curtis dissimilarity) are shown in the multi-panel plot: pseudoscorpions (Pseudoscorpiones), spiders (Araneae), darkling beetles (Tenebrionidae), rove beetles (Staphylinidae), ground beetles (Carabidae), ants (Formicidae), and vascular plants. Bar chart categories show the unexplained variation and pure or combined effect of the independent variables (scores from the ten principal components): another taxonomic group (column group defined by a categorical variable), spatial-topographic variables, bioclimate parameters, and landscape metrics. On the bottom, Venn's diagram shows the name of each fraction: [a] pure effect of another taxonomic group, [b] pure effect of spatial-topographic variables, [c] pure effect of climate variables, [d] pure effect of landscape measures, [e+f+g+h+i+j+k+l+m+n] partial shared effects of two/three set of factors, [o] total shared effect of all predictor sets, [p] unexplained variation. In the bar chart, fractions with negative values of adjusted R^2 were interpreted as zeros and were not shown in the diagram. Significance codes: (***) $p \leq 0.001$, (**) $p \leq 0.01$, (*) $p \leq 0.05$.


Pseudoscorpiones (pseudoscorpions)
Araneae (spiders)
Tenebrionidae (darkling beetles)
Staphylinidae (rove beetles)
Carabidae (ground beetles)
Formicidae (ants)
Vascular plants
Variance components (%)
100
75
50
25
0
Pseudo
Araneae
Teneb
Staphy
Carabid
Form
Plants
Other taxonomic group
[a]
[b]
[c]
[d]
[e]
[f]
[g]
[h]
[i]
[j]
[k]
[l]
[m]
[n]
[o]
Fraction names
[a] = Pure effect of other taxonomic group
[b] = Pure effect of spatial–topographic
[c] = Pure effect of clime
[d] = Pure effect of landscape
[e+f+g+h+i+j+k+l+m+n] = Partial_shared
[o] = Total shared effect
[p] = Unexplained

Discussion

We considered a gradient of increasing human disturbance, from SWOFs surrounded by natural and seminatural areas (preserved habitats with dead and decaying trees, significant cover of leaf litter, shrubs and herbs) to urban small woodlots (shrub layer regularly removed, highest trampling by human visitors).

Results highlighted the importance to preserve these small fragments also in the human-modified landscapes and not only in remaining natural areas, and the role of environmental factors and biological interactions to cross-taxon congruence.

Aside from studies on the role of scattered trees as keystone structures or biodiversity foci of landscapes (Manning et al. 2006; Fischer et al. 2010), we're not aware of any other study that assesses cross-taxa congruence across different land-use matrices supporting SWOFs in a Mediterranean fragmented landscape.

Our study confirms the central role of SWOFs for biodiversity conservation and the influence of landscape context on SWOFs biodiversity of different taxonomical groups, as observed in landscapes supporting scattered trees (Azihou et al. 2013; Le Roux et al. 2018; Prevedello et al. 2018).

Diversity pattern in species richness and composition among land-use matrices

The specific response patterns of taxa to land-use is controversial: the effect of land-use intensity on diversity patterns was described as neutral, positive or negative according to the analyzed taxa (Jones and Leather 2013; Gosling et al. 2016), trophic level (Seibold et al. 2019), the intensity of urbanization (McKinney 2008), as well as the spatial scale of analysis (Piano et al. 2020).

In accordance with studies demonstrating an increasing species richness with moderate urbanization (suburban areas; see McKinney 2008), we observed that agricultural and urban SWOFs sustained a relatively high richness or abundance of ground-dwelling invertebrates, compared to natural and semi-natural SWOFs. To our knowledge, this is the first study comparing patterns of species richness and community composition among land-use matrix for pseudoscorpions, corroborating a stronger effect on species composition than on species richness.

On the contrary, rove beetles and vascular plants showed an opposite trend, showing an increase of alfa-diversity from urban to natural and semi-natural SWOFs.

Most of the literature focused on the impact of extensive urbanization on species richness, in particular for plants (McKinney 2008; Gosling et al. 2016): in our study species richness and composition data of taxonomic groups respond differently to dominant matrices surrounding SWOFs, with a stronger effect on species composition rather than on species richness, consistently with other studies on beetles and vascular plants in human-modified landscapes.

For example, a contrasting pattern of richness and abundance was observed for different families of beetles (rove, scarabs and ground beetles) in large woodland fragments surrounded by pastures and Pine plantations of Australia (Yong et al. 2020). Similarly, Aggemyr et al. (2018) demonstrated that responses of plant species richness and composition differed significantly to local and landscape context in fragmented landscapes. Species composition can reveal more complex patterns than species richness alone, highlighting the importance to choose a suitable measure for interpreting ecological processes. In fact, the use of low-informative measures (e.g., total species richness and abundance) to evaluate the effects of human impact, may conduct to insufficient or even misleading descriptions of the changes of ecological communities, supporting the use of high-informative measures (e.g., assemblage composition and structure), both in meta-analyses and in primary studies (Hekkala and Roberge 2018).

In terms of beta diversity, except pseudoscorpions and spiders, taxa did not show significant differences across land-use matrices. These results seemed to suggest that land-use change might lead to a general homogenization of species richness and beta diversity patterns of almost all considered taxa. The homogenization driven by land-use intensification are consistent with previous studies on species richness (McKinney 2006) and beta diversity patterns (Buhk et al. 2017), but was observed also on species traits from other studies across Europe (Gámez-Virués et al. 2015; Carmona et al. 2020) and in the same environmental gradient of this study (Bazzato et al. submitted).

A moderate increase in land-use intensification could lead to a homogenization of groups at different trophic levels (Gossner et al. 2016): in our study, all ground-dwelling invertebrates did not diverge in terms of community structures across SWOFs surrounding by matrices at high or intermediate disturbance (urban and agricultural areas), while only vascular plant communities changed their compositional structures.

The observed stronger effect in higher trophic levels (e.g., spiders) might be linked to the increase of human disturbance, the strong dependence on the lower trophic level (e.g., plants; Lindenmayer et al. 2005) and the consequent multiple interacting factors (Scherber 2015). This could be especially true for spiders, known to be sensitive to environmental change, but also strongly dependent on lower trophic groups and their fluctuations (Cardoso et al. 2011).

SWOFs in natural and semi-natural areas were characterized by native trees and shrubs (e.g., *Arbutus unedo*, *Erica arborea*, *Erica terminalis*, *Salix atrocinerea* subsp.

atrocinerea), whereas SWOFs in agricultural and urban areas showed a simplified vertical structure, impoverished shrub layer, with a significant presence of alien species (e.g., *Eucalyptus camaldulensis camaldulensis*) and cultivated ones (e.g., *Pinus halepensis halepensis* and *Olea europaea*).

The community composition of primary producers (Ebeling et al. 2020), as well as its vertical structure (Schuldt et al. 2019), may have affected arthropod assemblages by altering light penetration and microclimate (Driscoll et al. 2013) and offering different resources, habitat spaces and niches for the groups of other trophic levels (Langellotto and Denno 2004).

The increase of human-disturbance and the changing of plant community composition may have promoted a loss of specialized species of other taxa, favoring good disperser (Niemelä 2001) or generalist and open-habitat species in modified land-uses (Gámez-Virués et al. 2015; Gossner et al. 2016). In fact, groups with active or high movement ability may have more chances of (re)colonizing surrounding areas, keeping viable populations and reducing the compositional differences, than groups with lower or passive dispersal capacities, such as plants (Soininen et al. 2007; Silva et al. 2017).

The cross-habitat movements of organisms is more evident in complex patch-matrix landscapes (Tscharntke et al. 2012) associated with a high amount of edge habitat maintained by human processes (Magura et al. 2017). In the study area, we observed a reduction of patch size (MPS and MedPS) and an increment of edge habitat (TD and ED) from natural and semi-natural areas to urban ones. In our landscape context, in human-modified land-uses (agricultural and urban areas) potential colonists from the overall species pool inhabiting the surrounding matrix (open-habitat and generalist species) can move through the edges and penetrate to the core of small woodlots (Magura et al. 2017) or vice versa (Tscharntke et al. 2012).

The ordination analyses confirm these patterns at least for some invertebrate groups.

Concerning darkling beetles, a group of saprophagous that feed on a wide variety of both vegetal and animal organic remains (Fattorini 2013; Fattorini et al. 2015), the assemblages of SWOFs in agricultural and urban areas were dominated by two species, *Stenosis sardoa sardoa* and *Tentyria grossa sardiniensis*, related to farmlands, but also to maquis, matorral, garrigue and thermo-Mediterranean bushes (Ruffo and Stoch 2006). Agricultural SWOFs resulted particularly suitable for *Crypticus gibbulus* and *Pimelia goryi goryi*, two species present in a variety of environments and altitudes (Aliquò et al. 2006). Conversely, SWOFs surrounded by natural and semi-natural areas were associated with *Asida androgyna*, a habitat specialist species, strictly range-localized in the local forested areas (Leo 2012).

Similarly, SWOFs in agricultural and urban areas were mainly associated with habitat-generalist and/or good disperser ground beetle species, whereas small woodlots in natural and semi-natural areas were characterized by habitat-specialist, and in some cases by poor disperser species.

It has been recognized that the capacities of dispersion of ground beetles, associated with the wing morphological trait, is important to the survival of this group in human-modified areas (Niemelä 2001). The number of good dispersers (macropterous species) could decrease with the patch size thanks to their better capacities to maintain the population in small and isolated patches, than poor dispersers (brachypterous species) (Niemelä 2001). Thanks to their capacities and differentiation, several studies on ground beetles explained that disturbed habitats support higher mobile species potentially colonizing and adapted to fluctuating environments, than natural areas (Gobbi and Fontaneto 2008; Gobbi et al. 2015). Ground beetle assemblages of SWOFs surrounded by agricultural and urban areas were mainly dominated by (i) *Laemostenus complanatus*, a macropterous species (Pizzolotto et al., 2008), with a typically functional wing morphology for good dispersal (Casale 1988), occurring in waste ground, parks and gardens along the coast (Brandmayr et al. 2005); (ii) *Calathus cinctus* a wing-dimorphic species that occupies temporary habitats, showing a high potential for flight and a high turnover of populations (Aukema 1990). On the other side, other dominant species of agricultural SWOFs were *Percus strictus oberleitneri*, a species that despite having short wings (brachypterous) (Fiori 1972) is commonly found in grass and lowlands (Ruffo and Stoch 2006) and *Carabus morbillosus constantinus*, a common species presents at all altitudes and in a wide variety of habitats including farmland, grassland, parks and gardens (Chatenet 2005). In those SWOFs, we also found *Amara aenea*, an opportunistic species related to open and disturbed habitats, including cultivated fields (Mazzei et al. 2015), whose presence in similar areas is related to the edge effect due to the proximity of open lands (Baini et al. 2014).

Conversely, ground beetle assemblages of SWOFs in natural and semi-natural areas were dominated by poor dispersal species with short wings (brachypterous, personally verified) related to natural and well-preserved mountainous habitats, such as *Percus strictus ellipticus* (Fiori 1972) and *Laemostenus carinatus* (Chatenet 2005). The finding of other macropterous species, such as *Agonum* gr. *viduum*, *Paranchus albipes* and *Zabrus ignavus ignavus* (Allegro and Chiarabaglio 2016; Pizzolotto et al. 2016) in natural and semi-natural SWOFs, could be explained by considering the frequent events of flooding occurring in some sampled areas located in mountain areas. In particular, *Paranchus albipes* is considered a ubiquitous riverside species (Brandmayr et al. 2009), colonizing wet places, including artificial caves and semi-saline environments (Anderson et al. 2000).

Concerning ants, a few euryecious species were detected in most SWOFs, irrespective of their levels of disturbance (*Camponotus lateralis*, *Crematogaster scutellaris*, *Lasius niger*, *Pheidole pallidula*, *Plagiolepis pygmaea*, *Solenopsis lusitanica* and *Tetramorium semilaeve*; see Seifert 2018). The social parasite *Plagiolepis xene*, sometimes suspected of poor dispersal capability and fragmented populations (Trontti et al. 2006; but see Mardulyn et al. 2014), was collected across the whole disturbance gradient alongside its host *P. pygmaea*. The presence of several other less commonly encountered species was apparently not related to the disturbance level. On the other hand, the most natural sites were characterized by the presence of several species associated with moister, cooler conditions or leaf litter (*Aphaenogaster ichnusa*, *Myrmica spinosior*, *Stenamma debile*, *Temnothorax tuberum*; see Seifert 2018; Galkowski et al. 2019). Also, the Tyrrhenian endemic *Aphaenogaster spinosa* was mostly restricted to these sites and replaced by *Aphaenogaster senilis*, which belongs to the same species group (Boer 2013), where the disturbance was higher. At the opposite end, *Tetramorium immigrans* (most-likely an introduced species in Sardinia) and *Tapinoma magnum*, both well-known to thrive in highly disturbed habitats and cities (Wagner et al. 2017; Seifert 2017, 2018; Castracani et al. 2020), were almost exclusively encountered in urban sites. Quite interestingly, *Tetramorium meridionale*, a Mediterranean species whose ecology is still poorly documented (Sanetra et al. 1999), appeared to specifically avoid urban sites instead. However, a large number of other species was absent or rare among the most undisturbed sites and widely present in the others: these are a broad and heterogeneous group of ants, in general ecologically linked to open or thermophilous habitats, ranging from the endemic *Temnothorax sardous* to the invasive alien *Linepithema humile* (argentine ant), but also including *Hypoponera eduardi*, *Tapinoma madeirense*, the above mentioned *A. senilis* and to some extent the species from the specialized granivorous genus *Messor*. In comparison with the other investigated groups, it is worth mentioning that some of the sites of this study are sufficiently close to each other so that a single ant colony can extend over more than one of them: this is not only true for *L. humile*, whose supercolonies can cover several kilometers, but also quite evident for species such as *C. scutellaris* or *T. magnum* (Seifert 2018). Dispersal ranges of ant sexuals responsible for colony foundation are also mostly undocumented, even though they are estimated to reach up to hundreds of kilometers in some species (Seifert 2018). However, the peculiar lack of flight dispersal capability of the foundress queens of *A. senilis* and *A. spinosa* (Boer 2013) did not appear to result in markedly different distribution patterns in comparison to the other ant species. The social lifestyle and plurennial nature of ant colonies could enhance their resilience in habitat patches which have been subject to recent transformations, making their distribution less sensitive during short-term monitoring.

Similarly to our results, previous studies on spiders (Gallé and Torma 2009), spiders and ground beetles (Lacasella et al. 2015; Magura et al. 2017), rove and ground beetles (Knapp et al. 2013) showed that forest remaining patches with edge habitat maintained by human processes were colonized by generalist or matrix-species, while patches with edge maintained by natural processes prevent the invasion of matrix (open-habitat) species or were characterized by specialists species.

On the other hand, it has been described that the increase of edge habitat in a complex landscape could lead to an improvement of functional connectivity amongst habitats, or otherwise, to an inhibition thereof if edges function as barriers (Tscharntke et al. 2012).

In small woodlots patches surrounded by human-modified land-uses, with a large amount of edge habitat, limited dispersal species could be more vulnerable to habitat loss and fragmentation than the better dispersing organisms (Niemelä 2001; Cushman and Landguth 2012). As in the case of darkling beetles, a group that typically have poor dispersal ability (Fattorini 2013; Fattorini et al. 2015). For example, psammo-halobious members of Pimeliinae, also recorded in this study, are typically flightless compared with other xylophilous species in this group (Fattorini 2020). Thanks to this ecological trait, members of this family represent excellent biogeographical markers that have been repeatedly used to explore the biogeography of Mediterranean islands and identify biodiversity hotspots (Fattorini 2006; Fattorini et al. 2015). Basing on vulnerability, geographical distribution, habitat specificity and abundance, darkling beetles have been used to identify priority urban green areas for biodiversity conservation (Fattorini 2014a). Green areas have been recognized as important refugial areas for their conservation (Fattorini 2014a). In addition, it has been underlined that large patches in natural areas may not automatically preserve a good number of species of darkling beetles and that large patches might not be better for their conservation than several smaller ones (Fattorini 2020).

If on the one hand, patterns of all ground-dwelling invertebrates across urban and agricultural seemed to suggest the homogenization of communities, on the other hand, the high level of richness and abundance of taxa in small woodlots outside forest surrounded by both of these land-use types underline the need to preserve remaining patches in human-modified land-use to avoid that homogenization drives a generalized biodiversity loss (McKinney 2006; Buhk et al. 2017) and extinction of entire communities in the long-term (Gámez-Virués et al. 2015).

Notwithstanding that the impacts of human disturbance are neither temporary nor fully avoidable (Araia et al. 2020), solutions to preserve species and communities with the inclusion of wildlife in agricultural (Simons and Weisser 2017) and urban areas (Apfelbeck et al. 2019, 2020) are possible and needed (Capotorti et al. 2020).

Effect of environment variables and biological interactions on cross-taxa concordance

The effectiveness of cross-taxon congruence is still controversial, and generalizations for their practical utilization are not uniquely supported by ecological studies due to contrasting results among studies (Bevilacqua et al. 2012; Barbato et al. 2019; Bacaro et al. 2019).

Contrasting results on the effectiveness of cross-taxon congruence could be attributed to the spatial and temporal scales (Westgate et al. 2014, 2017), responses of groups to environmental variables (Heino et al. 2009) and analytical methods adopted (Gioria et al. 2011).

Although these contrasting results are not encouraging (Margules and Pressey 2000), local or regional scale studies could contribute to understanding the main drivers of taxonomic surrogacy for their practical utilization in planning and management conservation efforts (Chiarucci et al. 2011; Santi et al. 2016; Burrascano et al. 2018).

A general understanding of the main environmental gradients that affect species composition (Gjerde et al. 2007; Sætersdal and Gjerde 2011) is needed to use cross-taxon congruence in conservation planning.

The role of environmental factors on cross-taxon congruence has been debated by a wide range of studies (Duan et al. 2016). Some authors have underlined that different taxa could show independent response patterns to environmental factor changes by leading to a poor cross-taxon congruence (Lovell et al. 2007; Bagella et al. 2011) or a potential success of the use of taxonomic surrogates (Gioria et al. 2010; Toranza and Arim 2010; Barbato et al. 2019).

Environmental heterogeneity (Margules and Pressey 2000) and gradients (e.g., latitudinal, altitudinal, climate gradient) (Warman et al. 2004; Toranza and Arim 2010) are drivers of variation in species diversity patterns (Stein et al. 2014) and can be considered as the major cause of the cross-taxon congruence (Sætersdal and Gjerde 2011).

However, they are not the only ones (Kraft et al. 2015; Magura et al. 2018). Environmental filtering, direct and indirect biotic interactions act together to determine taxon diversity patterns (Duan et al. 2016; Magura et al. 2018): environmental factors might represent the first filter for the presence of species and communities, but species and communities adapted to a certain environmental condition could be further filtered out by biotic interactions (Duan et al. 2016; Magura et al. 2018). For example, the differences in competitive ability for the resources use and the differences in predation susceptibility can determine a fewer species coexistence, so that assemblages with more similar and more related taxa can be produced (Magura et al. 2018).

Consequently, cross-taxon congruence could depend even on biological interactions, including trophic and non-trophic interactions (e.g., competition, facilitation), also within the same trophic level (Morlon et al. 2014; Özkan et al. 2014), especially at finer scale (Toranza and Arim 2010). At finer spatial scale, the role of biotic interactions in determining congruence between different groups may be stronger than environmental factors, which could become more important at larger spatial scale (Duan et al. 2016).

Results of this study corroborate the influence of environmental conditions for cross-taxon congruence, highlighting the role of highly complex and the variable interplay of environmental drivers and biological interactions in the explored taxon-pairs congruence.

After removing the effect of environmental factors on taxa congruence, a drop in the inter-group congruences was observed and a high number of pairwise correlations appeared weaker, but still significant.

Specifically, vascular plants maintained the highest number of significant correlations with other taxa (all ground-dwelling invertebrates, except rove beetles), followed by pseudoscorpions, spiders and ground beetles.

The role of vascular plants as a surrogate of other taxonomic groups has been extensively studied both in natural and human-modified areas (Burrascano et al. 2018). For instance, plants have proven to be adequate indicators for some vertebrates (Blasi et al. 2010; Eglington et al. 2012), spiders (Schoeman et al. 2020), parasitoid wasps (Fraser et al. 2009), ants (Zara et al. 2021), ground beetles (Yanahan and Taylor 2014; Duan et al. 2016), dung beetle (Schoeman et al. 2020), butterflies (Maccherini et al. 2009; Santi et al. 2010). Woody vegetation was recently recognized as a surrogate of darkling beetle communities probably thanks to the strong relationship of this invertebrate group with plant debris (Schoeman et al. 2020).

Our study revealed that pseudoscorpions, spiders and ground beetles could be considered good potential surrogates of other arthropods or plants. Interestingly, the role of the pseudoscorpions as potential surrogates of other groups (here for rove and ground beetles, ants and plants), as well as the role of plants for pseudoscorpions, seems to be unexplored from other studies on cross-taxon congruence.

Spiders and ground beetles can effectively be used as indicators of other taxa (for spiders, Churchill 1997; for ground beetles, Kotze et al. 2011; Gerlach et al. 2013), even in human-disturbed habitat (Uehara-Prado et al. 2009). In our study spiders were linked to darkling and ground beetles, ants and plants, while ground beetles to pseudoscorpions, spiders, darkling beetles and plants.

Unbelievably, all environmental predictor sets (spatial-topographic and climate factors and landscape metrics) were found not significant for ground beetles and vascular plants, despite significant relationships were described by a wide range of studies. For example, ground beetles were indicated as early-warning signalers of climate change, thanks to their capacities to adopt efficient adaptation strategies under biotic and abiotic changes, also at multiple spatial scales (Kotze et al. 2011). Population sizes typically show year-to-year variations of seasonality depending on weather conditions (Niemelä 2001; Rainio and Niemelä 2003), and specific adaptations of ground beetles life-cycles to seasonal fluctuations in water levels were commonly observed and described (Moran et al. 2012). At smaller scales, their communities are affected by microhabitat variations (Niemela et al. 1992), showing micro-spatial changes of assemblages even within a few meters depending on microclimatic parameters (e.g., light intensity, temperature, rainfall), soil features (e.g., pH, moisture, soil organic matter content, soil porosity, soil rock fragments, etc.), vegetation cover and leaf litter cover (see for example Antvogel and Bonn 2001; Uboni et al. 2019). There is, however, not enough research to determine how these variables are more informative at a certain scale and condition to identify causes behind differences in richness and abundance (Niemelä 1997; Moraes et al. 2013).

Even if our study has shown the differences across SWOFs surrounded by undisturbed and disturbed areas were particularly well-differentiated for ground beetle assemblages, with a high presence of species richness and abundance in the latter ones (coherently with Brandmayr et al. 2005; Latty et al. 2006; Uehara-Prado et al. 2009), only biological relationships with plants and spiders appeared to be significantly important to explain the variance of their communities along the land-use gradient.

In the same way, results of this study showed that after accounting for the effects of the three sets of environmental variables (spatial-topographic and climate parameters, and landscape metrics) and community composition of each group, (i) vascular plants *per se* explained only a significant amount of variation in pseudoscorpions and ground beetle assemblages, (ii) pseudoscorpions, spiders and ground beetles were important independent components in the variation of community composition of vascular plants.

These results suggested that many arthropod organisms depend, directly or not, on vegetation functions, such as the provision of diversified trophic resources and appropriate physical habitats for survival and reproduction (Gardarin et al. 2018), as well as plants are indirectly affected by the presence of other organisms (Schuldt et al. 2018).

It has been recognized as insects and other small invertebrates by transforming biomass and altering nutrient cycling and decomposition rates contribute to litter inputs to the soil lead to potentially complex indirect effects on plant communities (Schuldt et al. 2018). Predatory arthropods can affect trophic cascades in ‘brown’ food webs (Yang and Gratton 2014) and can be important agents of biological control by maintaining the population density of other organisms (Walton et al. 2012).

In this regard, it has been described that carabid predators play a key role in the biological control of other organisms and in lowering several cropland pest populations (De Heij and Willenborg 2020), considering that they can assume a wide range of prey, such as springtails (Collembola; Birkhofer et al. 2011), aphids (Winder et al. 2001), slugs (Symondson et al. 2002), as well as Lepidoptera larvae (Suenaga and Hamamura 1998; Clarke and Gorley 2006). Among the recorded predators, carabid species such as the two subspecies of *Percus* (*P. strictus oberleitneri* and *P. strictus ellipticus*) are generalist predators (Ruffo and Stoch 2006); *Agonum* gr. *viduum* is recognized as a predator of springtails and earthworms (Brandmayr et al. 2005); *Licinus punctatulus* and *Carabus morbillosus constantinus*, being a specialized predator of slugs (the second one with the prevalence of the *Helix* genus) stand out among them (Brandmayr et al. 2005; Chatenet 2005).

Apart from predator species (obligate or predominant zoophages), other species are phytophages, or specifically granivorous, and cover an important role for pre- and post-dispersal seed of herbaceous plants (Talarico et al. 2016). For instance, Honek et al. (2013) revealed that carabids were the most important seed predators of the temperate crop areas, and for these capacities, ground beetles have been recognized as biological control agents of weeds in the agricultural lands (Gaines and Gratton 2010; Kulkarni et al. 2015). Different genera of carabid beetles, such as *Amara* and *Harpalus,* have been specifically mentioned as weed seed bank-regulators (Bohan et al. 2011) to the reducing of weed seedling emergence (White et al. 2007). Among the recorded phytophagy or granivorous species, (i) the two members of Zabrini - *Amara aenea* and *Zabrus ignavus ignavus* - are spermatophagous with predatory behavior still regularly present and can eat a large spectrum of different kinds of seeds (Brandmayr et al. 2005; Talarico et al. 2016); (ii) adult species of the granivorous *Acinopus picipes* are associated with the seeds of *Daucus carota*, *Foeniculum vulgare*, and *Triticum aestivum* (Talarico et al. 2016).

Considering the wide range of feeding behaviours from zoophagy to granivorous species recorded in this study, the biological interaction detected from analyses among plants and invertebrates seems to be widely justified.

Considering the other interactions, accounting for the effects of the three set of environmental variables (spatial-topographic and climate parameters, and landscape metrics) (i) darkling beetles and climate factors explained a significant variation of pseudoscorpions communities, as well as landscape metrics when vascular plants were considered as one of independent variables; (ii) ground beetles and ants, as well as climate and spatial topographic variables, were important independent effect of spider communities; (iii) only spider, as well as the above mentioned plants, explained a significant variation of ground beetles; (iv) only ants and spatial topographic factors (when pseudoscorpions and vascular plants were set as independent explanatory variables) significantly affect the community composition of darkling beetles; (v) spiders and darkling beetles, as well as spatial topographic variables (when darkling beetles and plants were set as independent explanatory variables), explained a significant variation of ant communities.

The variation of pseudoscorpion communities depending on the vascular plants is supported by other studies, showing the sensitivity of this group to the types of vegetation (Jiménez-Hernández et al. 2020) and amount of leaf litter (Dennis et al. 2001). The accumulation of plant litter seems to favour the populations of pseudoscorpions by creating suitable temperature and humidity conditions, as well as shelter and food necessary for their survival and reproduction (Aguiar et al. 2006). In this regard, the high density of darkling beetles which feed on a wide variety of vegetal and animal organic remains (Fattorini 2013; Fattorini et al. 2015) might create a favourable condition for the pseudoscorpion communities, justifying the observed ecological interaction amongst these two groups found in our study. Although, this ecological interaction appears not to have been documented by other studies and should be further investigated. Despite relationships between pseudoscorpions and environmental factors are frequently unclear, as different results are observed even in the same place (Aguiar et al. 2006), the close relation with climate factors has been proposed to explain their abundance variation (Villarreal et al. 2019): abundance tends to increase during the dry season (Adis and Mahnert 1990, 1993; Aguiar et al. 2006; Villarreal et al. 2019); the low water availability in the dry season force some species to leave their refuges to search more specific microhabitats adapted to their survival; as a consequence, a high number of specimens can be sampled (Eijsackers 2001; Aguiar et al. 2006; Villarreal et al. 2019). The trend observed by previous studies might help to explain the results obtained in our study, in which climate variables significantly explained the variation of pseudoscorpion communities along the climate gradient, as also recently observed by Jiménez-Hernández et al. (2020). Furthermore, the high abundances found in urban and agricultural SWOFs could be justified by the low availability of precipitations together with the high temperatures present in these areas, compared to the natural ones. Conversely, the low presence of this group in natural and semi-natural SWOFs may be associated with the greater impact of precipitations and seasonal flooding present in these areas compared to the other ones on the biota (Battirola et al. 2017). Adis and Mahnert (1990) observed a strong synchronization between the species phenology and flood periods, so pronounced that pseudoscorpion communities in non-flooded areas were typically dominated by terricolous species compared to flooded areas with a high density of arboreal species or species migrated to tree trunks. A similar impact in relation to the seasonal flooding was also observed in other arachnids, such as soil spiders (Battirola et al. 2010). Given the scarcity of ecological studies on pseudoscorpions (Villarreal et al. 2019), the relation of pseudoscorpion communities and landscape metrics is difficult to explain, even considering the experiment conducted in agricultural areas by Bell et al. (1999), where species richness and abundance were linked to management, age margins and distance to the boundary habitats.

Similarly, the significant degree of variance of spider communities explained by climate factors is supported by previous studies. For example, variation of temperatures and microclimatic conditions reduce species occurrence, leading to a general decrease of spider species richness (Viterbi et al. 2020), and in particular, microclimatic variables are recognized as key factors of spider communities (Sattler et al. 2010). According to our findings, Sattler et al. (2010) also described the relationship between spatial variables and spiders, indicating a significant variation of their communities in alpine timberline, in dry meadows, in intensive agricultural and urban areas. Although we did not find a direct influence of vegetation on the variation of spider communities, other studies conducted in Sardinia described an increase of microweb-building spider species in dependency both of microclimate factors and plant composition changes (Mannu et al. 2020).

Interestingly, our study confirms as different groups of soil arthropods, such as spiders and ants, and spiders and ground beetles, with potentially many shared preys (Moraes et al. 2013), are affected by each other and have reciprocal interaction, also in Mediterranean environments.

The significant degree of variance of spider communities explained by ant communities and *vice versa* are consistent with previous studies that have highlighted the biological interaction between these two groups. The variation in abundance of the two groups may be due to intraguild interaction, which can manifest itself both as predation between groups belonging to the same guild (IGP, intraguild predation; for further information see Polis et al. 1989), and as a competition for the same trophic resource. Intraguild interaction is therefore one of the most important factors that contribute to determining the structure and composition of communities (Polis et al. 1989; Wise 1995). In this regard, the omnivorousness of ants allows them to occupy several guilds at the same time, including those occupied by spiders and other predatory arthropods (Sanders and Platner 2006).

Spiders are generally disadvantaged - in terms of abundance and, to a limited extent, of species richness - by the presence of ants (Wise 1995; Halaj et al. 1997; Sanders and Platner 2006). Ants play a role in determining spider abundances: a significant increase of web-building spider abundance (Araneidae and Theridiidae) were observed in antless plots (Mestre et al. 2012), and the manipulation of the ant densities led to an abundance variation of araneic arboreal species (Halaj et al. 1997), probably due to the variation in the shared prey density (Halaj et al. 1997; Mestre et al. 2012). It was also observed that the removal of ants has determined a strong increase in the density of Linyphiidae; while the reduction in the abundance of *Formica cunicularia* and *F. fusca* was observed in conjunction with the increase in wandering spider abundances (Sanders and Platner 2006).

However, the effects of ant predation on spiders and *vice versa* are not uniform (Wise 1995; Halaj et al. 1997). It has also been recognized as the web-building spiders did not undergo abundance variations following the increase in the density of potential prey (Wise 1995;

Halaj et al. 1997). Spiders that actively hunt in foliage would be able to detect ants and change direction, and ants would not always be able to regard web-building spiders as potential prey (Wise 1995; Halaj et al. 1997).

There are also cases in which spiders are benefited by the presence of ants, such as the myrmecophile and myrmecomorphic species (which evolved by convergence in different families such as Salticidae, Gnaphosidae, Thomisidae, Clubionidae, Zodariidae etc.), but also the myrmecophagous species (which can be myrmecomorphs at the same time). The similarity of some spiders to ants is to be considered a form of Batesian mimicry rather than a form of aggressive mimicry towards the ants themselves. This is proven by the fact that myrmecomorphic spiders are avoided, just like the ants occupying the same environment, by other potential predators (e.g., other arthropods and birds) (Foelix 2011). However, myrmecomorphy as a Batesian mimicry would not be considered, in most cases, as an adaptive response to the intraguild predation mentioned above: the ants would use chemoreceptors to identify conspecifics and not visual signals (Herberstein 2011). In any case, the predatory specialization of some spiders towards ants and the development of specific adaptations to avoid predation by ants (which, as mentioned, do not correspond to all cases) is probably due to intraguild predation (IGP; Herberstein 2011).

Similarly, also the reciprocal interaction between spiders and ground beetles could be explained by considering the dynamics of competition for the resource acquisitions; even if such resource use might differ qualitatively and quantitatively between groups thanks to different functional specializations, and different biotic and abiotic requirements for reproduction (Moraes et al. 2013). Our results confirm that the survival and persistence of ground beetle assemblages in an environment strongly depend on food availability, be that plant (e.g., seeds and plant tissues) and animal (e.g., other invertebrates) matter, and the influence of different food conditions are probably more significant than currently acknowledged (Magura et al. 2018).

The food limitation (both for larval and adult) could influence ground beetle assemblages (Lenski 1984; Nelemans 1987) and strong intra-guild competition exists between ground beetles and other generalist predators, including spiders and ants (Lövei and Sunderland 1996). By quantifying food limitation for ground beetles and sheet-web spiders (Linyphiidae) in a cereal field, Bilde and Toft (1998) have revealed that food availability influences the predator populations led to drastic seasonal changes, at least for ground beetles, and that all tested predators were food-limited to some degree all the time. Magura et al. (2018) showed how the predation may affect ground beetle assemblages more in the rural sites than in urban ones, highlighting that the presence of alternative food resources in the urban areas can reduce the strength of competition, both the interspecific and intraguild one.

The co-existence of potential predators could also have indirect non-consumptive effects (De Heij and Willenborg 2020). For example, Charalabidis et al. (2017) showed how the level of choosiness of a seed-eating carabid beetle, *Harpalus affinis*, decreased as a function of either predation risk or competition, both between intraspecific or interspecific competitors. Also, ground beetle foraging activity decreased with indirect and direct mice predator cues by leading to an increase of carabid seed consumption and a reduction of carabid prey consumption, demonstrating a cascading effect over multiple trophic levels of the predator presence (Blubaugh et al. 2017). Birkhofer et al. (2011) found that structural differences in the crops led to different aggregations of generalist predators (spider and ground beetles) and decomposer prey (Collembola), with more complex habitats provided more refuges for the prey which resulted in a higher coexistence of predators, whereas structurally less complex managed fields were characterized by a lower coexistence of predators due to a decrease of collembolan prey availability, suggesting more intense intraguild interactions in structurally less complex habitats. These experiments highlighted as interspecific competitions, predator-mediated interactions (like intraguild predation), together with indirect non-consumptive effects, drive changes in ground beetle species with complex mechanisms (Blubaugh et al. 2017; Charalabidis et al. 2017), so much that they are considered the most important biotic interactions affecting the community of ground beetles (Magura et al. 2018).

It may therefore not be misleading that such inter-biotic mechanisms have also determined the different patterns of abundance in spider and ground beetle communities observed in our study among SWOFs surrounded by different land-use types in the Mediterranean, as already argued by Magura et al. (2018) and Birkhofer et al. (2011) in Temperate environments.

Despite the well-documented association between spider and ant species, other types of interactions with other macroorganisms, as well as microorganisms, related to ants have been somewhat overlooked (Bekker et al. 2018) and are receiving increasing attention only in recent years (Härkönen and Sorvari 2014; Parmentier et al. 2016; Parker 2016). These interactions often appear to be complex, and in many instances, are restricted to a single genus or species (Rocha et al. 2020). Among these, many species of darkling beetles exploiting the various resources of ant colonies were described, even if the recurrence of myrmecophily (the partial or complete dependence on ant colonies) in this family is relatively modest given the body size (Parker 2016). Despite this, darkling beetles, being principally detritivorous, are attracted to detritus that accumulates around the ant nests and associate with peripheral areas of colonies (Sánchez-Piñero and Gómez 1995; Parker 2016). Most myrmecophiles within the family belong to the subfamily Pimeliinae (e.g., Adelostomini, Cnemeplatiini, Cossyphodini, Stenosini), but include also some genera belonging to other subfamilies (e.g., Alleculinae, Lagriinae, Tenebrioninae, Diaperinae) with almost unknown and underexplored biology (Parker 2016; Rocha et al. 2020). Beyond these, a wide number of other members of the family have been recorded in apparently facultative interactions with ant colonies (Sánchez-Piñero and Gómez 1995; Parker 2016). Based on that information, the variation of darkling beetle communities depending on the ant communities observed in this study is not surprising. Even if, considering the dominant species found, it cannot be fully explained by the presence of obligate myrmecophiles, but probably only facultative myrmecophiles that tend to feed on debris around nests, and therefore may be exempt from the size constraint. Among these, *Crypticus gibbulus* is presumably able to exploit nest refusal or harvested food in some way by considering that it is not uncommon to find it among the frustules at the mouth of anthills (Aliquò et al. 2006). Aside from the member of Crypticini already mentioned, other phytosaprophagous or zoosaprophagous species, here recorded as dominant (e.g., *Stenosis sardoa sardoa*, Stenosini; *Tentyria grossa sardiniensis*, Tentyriini), might be related to ant-debris even if it is not documented elsewhere. Despite some studies reported the importance of climate factors (such as temperatures and aridity) as filtering drivers for termo-xerophilic species of darkling beetles (Fattorini 2014b; Fattorini et al. 2020), no relationships are detected in our study. While a significant degree of variance of darkling beetle communities was explained by spatial topographic factors, coherently with other studies conducted along other Mediterranean elevation gradients (Fattorini 2014b; Fattorini et al. 2020).

Conclusion

Our results emphasize the different influence of landscape context on Mediterranean SWOFs biodiversity of different taxonomical groups, covering various trophic levels.

Quantitative evidence shows a stronger effect of dominant matrices surrounding SWOFs on species composition rather than on species richness, highlighting a general homogenization of taxa diversity across matrices at high or intermediate disturbance. However, the high level of richness and abundance of taxa in small woodlots outside forest surrounded by both of these land-use types underline the need to preserve these remaining patches to avoid that homogenization drives a generalized biodiversity loss (McKinney 2006; Buhk et al. 2017) and extinction of entire communities in the long-term (Gámez-Virués et al. 2015).

Changes in the matrix surrounding woodland patches may alter habitat structure via correlated and simultaneously acting drivers, leading to cascading effects on different taxa communities, especially concerning dispersal, foraging resources and related dynamic interactions (Driscoll et al. 2013; Schuldt et al. 2018).

This study contributes to enhancing the knowledge on the highly complex and variable interplay of environmental drivers and biological interactions in explaining cross-taxon congruence in species composition.

While numerous studies have been reported on the importance of vascular plants in determining invertebrate community compositions, providing a wide range of resources, habitat spaces and niches (Langellotto and Denno 2004; Schuldt et al. 2019; Ebeling et al. 2020), the functions of higher trophic groups to other groups have garnered less attention (Schuldt et al. 2018).

However, our study draws attention to the importance of higher trophic-level organisms, positively related to the lower trophic level, highlighting as they can contribute to ecosystem functions and multifunctionality, as recently suggested by Schuldt et al. (2018).

The inclusion of other trophic levels, together with lower trophic level, is crucial to understanding the effects of biodiversity on ecosystem multifunctionality and cross-taxon relationships, by improving the practical application of ecological theory (Schuldt et al. 2015, 2018).

The magnitude of the observed cross-taxon relationships and the high number of significant biological interactions among ground-dwelling taxa groups observed in this study may be promising for future studies and application of invertebrate surrogates in the presence of strong stressor environmental gradient.

Considering invertebrate groups with their potentially diverse functional, trophic and non-trophic roles for the regulation of ecological processes (Gerlach et al. 2013), as well as the influence of abiotic factors, might allow identifying complementary surrogates for a wide range of other organisms (Schuldt et al. 2015), providing useful indications for planning and management conservation efforts (Gerlach et al. 2013).

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Author contributions

Conception and design of field research: Bazzato E.; material preparation and data collection: Bazzato E., Caria M. & Ancona C.; determination of species: Bazzato E., Caria M., Lallai E., Schifani E., Cillo C.; conception and design of paper: Bazzato E., Marignani M., Maccherini S., Bacaro G.; drafting the article: Bazzato E.; revising the article for important intellectual content: Marignani M., Maccherini S., Bacaro G.

CHAPTER 4 – PLANT FUNCTIONAL TRAITS

The ecological role of the herbaceous layer for maintaining the long-term stability, structure and function of the forest ecosystems is well-known (Campetella et al. 2020). It has been demonstrated that herbaceous layer (i) contributes to forest biodiversity which is largely a function of the herb-layer community, (ii) represents the site of initial competitive interactions for the regeneration of the dominant canopy tree species, (iii) affects the energy processes and nutrient cycling of the organic layer, (iv) shows multifaceted responses to various disturbances of both natural and anthropogenic origin (Gilliam 2007, 2014).

Despite the growing awareness of the special role of the herbaceous layer, the variability of its functional traits has received relatively little attention (Paź-Dyderska et al. 2020), probably also due to the time, labor and money needed for fieldwork. Plant functional traits — the morpho-physio-phenological features measurable at the individual plant level — influence the growth, reproduction and survival of species and populations (Violle et al. 2007). They reflect the outcome of evolutionary and community processes responding to environmental filters (Kattge et al. 2020), affect other trophic levels (Loranger et al. 2012, 2013; Lavorel et al. 2013) and ecosystem properties and derived benefits (Lavorel and Garnier 2002; Garnier and Navas 2012; Lavorel et al. 2015). Therefore, studies with trait-based approaches can be particularly important, especially considering that they ensure a better inter-taxon and inter-region comparability than traditional taxonomic approaches (Paź-Dyderska et al. 2020). Considering this importance, the main aim of this chapter was to study the functional traits of vascular plants in SWOFs surrounded by different land-use types, focusing on plant height (Subchapter 4.1, paper 11) and seed of herb layer communities (Subchapter 4.2, paper 8) collected in the fieldwork phase.

Subchapter 4.1: plant height of herb layer communities

Summary statistics of the database were published in the paper 11 (not included).

Paper 11 (not included)

Kattge J., Bönisch G., Díaz S., et al. 2020. TRY plant trait database – enhanced coverage and open access. Global Change Biology, 26(1): 119-188. https://doi.org/10.1111/gcb.14904

Subchapter 4.2: seed of herb layer communities

As an example, an analysis of data of seed mass of two congeneric species was provided.

Paper 8

Bazzato E., Serra E., Maccherini S., Marignani M. (submitted). Reduction of intraspecific seed mass variability along a land-use intensification gradient. Submitted to Ecological indicator

Subchapter 4.1: plant height of herb layer communities

Introduction

Plant height has been recognized as one of the primary drivers of light extinction down the canopy of plant species, explaining most of the variation in limited plant growth of competing individuals (Violle et al. 2009). Being taller than competing neighbors confers an advantage in the acquisition of light and is central to a species' carbon gain strategy (Westoby et al. 2002).

Because of this, the height of an individual should be considered in relation to the height of competing neighbors (Westoby et al. 2002).

Despite the ease of its measurement, the interpretation of plant height attributes should be done with caution, considering that this trait is highly dynamic and strongly dependent on plant ontogeny and disturbance regime (Garnier and Navas 2012).

In addition, vegetative plant height may be somewhat tricky to define for herbaceous species, when the plant bends, the inflorescence has significant photosynthetic portions or a very little photosynthetic area higher up as in the case of rosette species (Pérez-Harguindeguy et al. 2013), which produce a stalk at the end of an elusive vegetative stage (Garnier and Navas 2012).

In these cases, reproductive height at maturity can be the operational measure that solves the practical problems of herb traits sampling (Garnier and Navas 2012). It follows that the reproductive height might be considered as the most relevant trait to measure for broad-scale comparisons of herbaceous species differing in growth forms and habits (Garnier and Navas 2012).

Given this importance, the reproductive height of the herb layer communities was measured in each considered site at the plot level and the data was published in a global plant trait database (Kattge et al. 2020). A large amount of data has been gathered and analyses are still ongoing.

Materials and methods

Fieldwork was conducted in the 30 SWOFs along the gradient of land-use intensification only during a growing season, from May to September 2019.

When possible, 10 individual plants were considered for each species that represented 80% of the total biomass in each plot of 1 square meter (Pakeman and Quested 2007), for a total of 150 plots. The reproductive height of healthy mature herb plants was measured according to standardized protocols (Pérez-Harguindeguy et al. 2013).

Results

A total of 6,320 individual plants belonging to 172 species and 41 families were measured in the 150 plots along the gradient of land-use intensification.

Mean of the species studied ranged from 1.81 cm ± SE 0.29 (*Selaginella denticulata* (L.) Spring) to 102.66 cm ± SE 8.73 (*Rubus ulmifolius* Schott) (Table 1). The coefficient of variation ranged from 0.12 (*Asparagus horridus* L.) to 1.04 (*Oxalis pes-caprae* L.) (Table 1).

Table 1. Summary statistics of reproductive height for each species sampled in the study area along the gradient of land-use intensification. Mean, standard deviation (SE), coefficient of variation (CV), minimum (Min) and maximum (Max) values are shown.

Family	**Species**	**Mean**	**SE**	**CV**	**Min**	**Max**
Amaryllidaceae	*Allium subhirsutum* L. subsp. *subhirsutum*	19.53	0.82	0.27	7.90	28.00
Amaryllidaceae	*Allium triquetrum* L.	36.70	2.65	0.23	24.00	49.00
Anacardiaceae	*Pistacia lentiscus* L.	79.26	10.06	0.40	38.80	137.00
Apiaceae	*Daucus carota* L.	58.21	3.61	0.44	5.00	94.00
Apiaceae	*Eryngium campestre* L.	27.88	1.26	0.14	22.20	34.60
Apiaceae	*Foeniculum vulgare* Mill. subsp. *vulgare*	91.06	6.24	0.22	60.50	122.50
Apiaceae	*Oenanthe crocata* L.	65.57	4.40	0.21	45.00	90.00
Apiaceae	*Smyrnium olusatrum* L.	52.57	3.27	0.34	26.00	95.50
Apiaceae	*Thapsia garganica* L. subsp. *garganica*	71.47	4.91	0.34	11.50	133.50
Apiaceae	*Torilis africana* Spreng.	20.83	1.93	0.83	2.50	98.00
Apiaceae	*Torilis nodosa* (L.) Gaertn. subsp. *nodosa*	9.44	1.12	0.65	2.00	27.50
Araceae	*Arisarum vulgare* O.Targ.Tozz. subsp. *vulgare*	10.37	0.73	0.50	3.00	25.00
Asparagaceae	*Asparagus acutifolius* L.	41.01	1.28	0.58	4.70	139.00
Asparagaceae	*Asparagus albus* L.	53.18	2.53	0.40	19.00	104.50
Asparagaceae	*Asparagus horridus* L.	41.95	1.62	0.12	33.50	51.00
Asphodelaceae	*Asphodelus ramosus* L. subsp. *ramosus*	78.25	3.24	0.64	10.00	708.00
Aspleniaceae	*Asplenium onopteris* L.	12.35	1.26	0.32	6.00	16.00
Asteraceae	*Andryala integrifolia* L.	23.80	3.21	0.43	5.00	41.00
Asteraceae	*Carduus pycnocephalus* L. subsp. *pycnocephalus*	55.97	5.08	0.50	20.00	102.00
Asteraceae	*Carlina corymbosa* L.	31.50	2.88	0.29	23.00	47.50
Asteraceae	*Carlina gummifera* (L.) Less.	17.34	1.63	0.39	9.00	29.00
Asteraceae	*Centaurea melitensis* L.	43.26	2.62	0.19	32.00	55.00
Asteraceae	*Crepis vesicaria* L. subsp. *vesicaria*	21.21	1.11	0.17	15.20	26.20
Asteraceae	*Galactites tomentosus* Moench	38.31	1.28	0.30	8.50	61.50
Asteraceae	*Glebionis coronaria* (L.) Spach	36.81	1.89	0.28	20.00	64.00

Asteraceae	*Hedypnois rhagadioloides* (L.) F.W.Schmidt	20.57	1.00	0.41	4.00	43.50
Asteraceae	*Hippocrepis biflora* Spreng.	8.54	0.75	0.39	4.00	16.50
Asteraceae	*Hypochaeris achyrophorus* L.	19.17	2.15	1.02	3.80	107.00
Asteraceae	*Pallenis spinosa* (L.) Cass. subsp. *spinosa*	42.98	2.98	0.31	24.00	75.40
Asteraceae	*Phagnalon saxatile* (L.) Cass.	35.85	2.79	0.25	23.40	52.50
Asteraceae	*Pulicaria dysenterica* (L.) Bernh.	32.40	1.90	0.18	25.50	43.00
Asteraceae	*Pulicaria odora* (L.) Rchb.	42.27	2.60	0.34	13.50	76.00
Asteraceae	*Reichardia intermedia* (Sch.Bip.) Samp.	53.94	3.51	0.27	24.50	76.00
Asteraceae	*Sonchus tenerrimus* L.	39.62	0.88	0.38	6.00	94.00
Asteraceae	*Tolpis virgata* (Desf.) Bertol. subsp. *virgata*	49.94	2.25	0.25	29.00	72.50
Asteraceae	*Urospermum dalechampii* (L.) F.W.Schmidt	38.54	3.36	0.49	19.00	82.00
Asteraceae	*Urospermum picroides* (L.) Scop. ex F.W.Schmidt	19.61	1.00	0.37	7.50	39.50
Brassicaceae	*Hirschfeldia incana* (L.) Lagr.-Foss. subsp. i*ncana*	27.53	2.89	0.33	15.00	41.00
Brassicaceae	*Lunaria annua* L.	33.91	1.99	0.26	22.60	59.00
Brassicaceae	*Sisymbrium officinale* (L.) Scop.	28.10	1.26	0.14	22.00	35.50
Caryophyllaceae	*Petrorhagia dubia* (Raf.) G.López & Romo	22.60	2.41	0.58	4.50	52.00
Caryophyllaceae	*Polycarpon tetraphyllum* (L.) L. subsp. *diphyllum* (Cav.) O. Bolòs & Font Quer	3.27	0.29	0.40	1.00	6.00
Caryophyllaceae	*Silene gallica* L.	14.55	0.66	0.40	5.00	34.50
Caryophyllaceae	*Stellaria media* (L.) Vill. subsp. *media*	5.11	0.44	0.47	2.50	12.00
Chenopodiaceae	*Beta vulgaris* L. subsp. *maritima* (L.) Arcang.	46.15	3.17	0.29	23.00	71.00
Chenopodiaceae	*Chenopodiastrum murale* (L.) S.Fuentes. Uotila & Borsch	26.90	2.07	0.49	9.10	68.00
Cistaceae	*Cistus monspeliensis* L.	89.34	4.59	0.23	56.00	126.00
Cistaceae	*Cistus salviifolius* L.	58.70	3.18	0.17	43.00	78.00
Convolvulaceae	*Convolvulus althaeoides* L.	17.82	2.86	1.02	1.50	73.50
Convolvulaceae	*Convolvulus arvensis* L.	7.94	0.49	0.20	5.70	10.20
Crassulaceae	*Umbilicus rupestris* (Salisb.) Dandy	16.27	1.19	0.23	12.00	22.00
Cyperaceae	*Carex distachya* Desf.	17.09	0.89	0.44	2.50	41.50
Cyperaceae	*Carex distans* L.	23.91	6.86	0.95	10.00	90.00
Cyperaceae	*Carex hispida* Willd.	84.71	4.33	0.32	37.50	153.00
Cyperaceae	*Carex* sp.	13.56	2.07	0.48	6.40	24.80
Cyperaceae	*Cyperus badius* Desf.	66.65	2.59	0.12	58.60	83.00
Cyperaceae	*Isolepis cernua* (Vahl) Roem. & Schult.	6.23	0.44	0.22	4.20	8.30
Cyperaceae	*Scirpoides holoschoenus* (L.) Soják	76.66	4.01	0.29	40.00	111.50
Dennstaedtiaceae	*Pteridium aquilinum* (L.) Kuhn subsp. *aquilinum*	61.76	4.37	0.39	24.00	105.80
Dioscoreaceae	*Dioscorea communis* (L.) Caddick & Wilkin	50.06	8.94	0.98	4.80	173.50
Euphorbiaceae	*Euphorbia peplus* L.	5.13	0.32	0.39	2.50	12.50
Euphorbiaceae	*Mercurialis annua* L.	25.72	3.66	0.45	13.50	42.70
Fabaceae	*Astragalus pelecinus* (L.) Barneby subsp. p*elecinus*	5.61	0.63	0.36	3.30	9.80
Fabaceae	*Ervilia hirsuta* (L.) Opiz	11.89	2.25	0.60	3.00	26.00
Fabaceae	*Ervum tetraspermum* L.	19.62	5.22	0.84	6.10	62.00
Fabaceae	*Hymenocarpos circinnatus* (L.) Savi	9.83	1.02	0.33	6.00	15.00
Fabaceae	*Lathyrus ochrus* (L.) DC.	18.00	2.28	0.57	6.80	37.00
Fabaceae	*Lotus edulis* L.	10.28	1.26	0.46	5.00	21.50

Fabaceae	*Lotus ornithopodioides* L.	4.30			4.30	4.30
Fabaceae	*Medicago ciliaris* (L.) All.	7.77	0.76	0.31	5.40	12.60
Fabaceae	*Medicago minima* (L.) L.	9.84	1.12	0.51	3.50	21.00
Fabaceae	*Medicago polymorpha* L.	7.36	1.09	0.47	2.60	14.50
Fabaceae	*Medicago praecox* DC.	5.69	0.44	0.48	2.00	13.50
Fabaceae	*Medicago truncatula* Gaertn.	7.54	0.45	0.42	2.00	17.50
Fabaceae	*Ononis ornithopodioides* L.	14.76	0.59	0.25	6.80	21.00
Fabaceae	*Ononis viscosa* L. subsp. *breviflora* (DC.) Nyman	37.80	4.24	0.35	16.50	54.30
Fabaceae	*Ornithopus compressus* L.	12.36	0.64	0.49	3.80	33.50
Fabaceae	*Scorpiurus muricatus* L.	7.87	0.58	0.47	2.00	17.00
Fabaceae	*Trifolium angustifolium* L. subsp. *angustifolium*	18.48	2.04	0.60	3.50	42.50
Fabaceae	*Trifolium arvense* L. subsp. *arvense*	11.37	0.80	0.45	2.00	21.50
Fabaceae	*Trifolium campestre* Schreb.	9.61	0.39	0.53	2.50	28.50
Fabaceae	*Trifolium cherleri* L.	8.25	0.65	0.25	5.50	11.00
Fabaceae	*Trifolium glomeratum* L.	5.72	0.88	0.49	3.50	13.00
Fabaceae	*Trifolium ligusticum* Loisel.	9.20	0.79	0.38	3.50	16.00
Fabaceae	*Trifolium micranthum* Viv.	4.57	0.39	0.38	2.00	9.00
Fabaceae	*Trifolium spumosum* L.	7.05	0.82	0.37	3.00	11.50
Fabaceae	*Trifolium stellatum* L.	9.22	1.09	0.53	3.00	21.00
Fabaceae	*Trifolium subterraneum* L. subsp. *subterraneum*	14.86	3.30	0.99	2.00	57.50
Fabaceae	*Tripodion tetraphyllum* (L.) Fourr.	7.08	0.78	0.35	3.50	12.00
Fabaceae	*Vicia angustifolia* L.	18.66	2.32	0.39	10.30	32.30
Fabaceae	*Vicia disperma* DC.	12.44	1.91	0.53	5.50	24.50
Fabaceae	*Vicia sativa* L.	14.90	0.99	0.47	5.00	45.50
Geraniaceae	*Geranium columbinum* L.	22.65	1.06	0.15	17.50	29.50
Geraniaceae	*Geranium molle* L.	11.04	0.83	0.53	3.50	26.30
Geraniaceae	*Geranium purpureum* Vill.	23.92	1.27	0.58	2.50	59.20
Hypericaceae	*Hypericum hircinum* L. subsp. *hircinum*	80.08	7.11	0.28	50.00	111.00
Juncaceae	*Juncus articulatus* L. subsp. *articulatus*	35.85	3.98	0.35	17.00	55.00
Juncaceae	*Juncus effusus* L. subsp. *effusus*	95.73	3.84	0.13	70.20	110.00
Lamiaceae	*Lavandula stoechas* L. subsp. *stoechas*	42.67	1.82	0.30	22.50	73.00
Lamiaceae	*Mentha suaveolens* Ehrh. subsp. *insularis* (Req.) Greuter	60.95	6.09	0.32	18.00	85.00
Lamiaceae	*Micromeria graeca* (L.) Benth. ex Rchb. subsp. *graeca*	21.42	1.38	0.41	5.00	41.00
Lamiaceae	*Stachys glutinosa* L.	34.66	2.13	0.27	22.00	54.00
Lamiaceae	*Stachys major* (L.) Bartolucci & Peruzzi	41.22	3.14	0.59	7.20	107.00
Lamiaceae	*Thymbra capitata* (L.) Cav.	34.31	3.93	0.36	17.00	55.30
Linaceae	*Linum strictum* L.	13.88	0.83	0.37	3.50	25.00
Malvaceae	*Malva neglecta* Wallr.	33.52	4.05	0.54	10.00	97.00
Oleaceae	*Olea europaea* L.	54.00	4.19	0.25	39.00	78.00
Ophioglossaceae	*Ophioglossum vulgatum* L.	17.12	1.13	0.21	11.00	24.30
Orobanchaceae	*Bellardia viscosa* (L.) Fisch. & C.A.Mey.	17.82	1.56	0.28	11.00	27.00
Oxalidaceae	*Oxalis pes-caprae* L.	22.74	4.31	1.04	2.50	73.00
Plantaginaceae	*Misopates orontium* (L.) Raf.	20.88	2.71	0.41	10.50	37.50

Plantaginaceae	*Plantago bellardii* All. subsp. *bellardii*	8.78	0.60	0.22	5.50	11.00
Plantaginaceae	*Plantago lagopus* L.	20.02	0.92	0.33	9.00	36.00
Poaceae	*Agrostis stolonifera* L. subsp. *stolonifera*	47.12	2.89	0.43	16.50	99.50
Poaceae	*Aira cupaniana* Guss.	7.80	1.26	0.51	3.00	14.00
Poaceae	*Aira elegantissima* Schur subsp. *elegantissima*	10.30	0.84	0.45	3.40	19.50
Poaceae	*Anisantha diandra* (Roth) Tutin ex Tzvelev	28.90	1.94	0.21	18.00	39.00
Poaceae	*Anisantha madritensis* (L.) Nevski subsp. *madritensis*	19.16	0.94	0.49	4.60	45.50
Poaceae	*Anisantha sterilis* (L.) Nevski	25.84	2.34	0.50	11.00	58.00
Poaceae	*Avena barbata* Pott ex Link	41.62	1.68	0.36	4.50	92.00
Poaceae	*Avena sterilis* L. subsp. *sterilis*	49.58	3.36	0.43	21.50	116.50
Poaceae	*Brachypodium distachyon* (L.) P.Beauv.	15.73	0.61	0.53	2.50	46.00
Poaceae	*Briza maxima* L.	25.55	2.28	0.40	12.00	46.00
Poaceae	*Bromus hordeaceus* L. subsp. *hordeaceus*	68.47	2.08	0.19	44.00	97.00
Poaceae	*Catapodium balearicum* (Willk.) H.Scholz	20.38	1.98	0.43	4.00	36.50
Poaceae	*Catapodium rigidum* (L.) C.E.Hubb. subsp. *rigidum*	12.77	1.32	0.56	2.50	25.50
Poaceae	*Cynodon dactylon* (L.) Pers.	23.02	0.61	0.25	9.00	40.00
Poaceae	*Cynosurus echinatus* L.	32.26	1.20	0.39	9.00	71.00
Poaceae	*Cynosurus effusus* Link	21.67	1.30	0.42	4.50	45.00
Poaceae	*Dactylis glomerata* L. subsp. *hispanica* (Roth) Nyman	57.95	2.61	0.28	23.00	97.30
Poaceae	*Dasypyrum villosum* (L.) P.Candargy	31.68	1.95	0.28	11.50	46.00
Poaceae	*Festuca danthonii* Asch. & Graebn. subsp. *danthonii*	15.91	1.95	0.39	9.00	26.30
Poaceae	*Festuca ligustica* (All.) Bertol.	30.41	2.96	0.44	10.10	47.50
Poaceae	*Festuca sicula* C.Presl	61.23	2.82	0.15	49.00	82.00
Poaceae	*Gastridium ventricosum* (Gouan) Schinz & Thell.	59.45	7.64	0.51	13.00	94.00
Poaceae	*Holcus lanatus* L. subsp. *lanatus*	52.92	4.33	0.52	21.00	158.00
Poaceae	*Hordeum murinum* L.	22.75	0.85	0.31	9.60	47.00
Poaceae	*Hyparrhenia sinaica* (Delile) Llauradó ex G. López	53.37	3.06	0.18	36.00	67.00
Poaceae	*Lagurus ovatus* L. subsp. *ovatus*	19.62	1.08	0.46	3.50	45.50
Poaceae	*Lolium interruptum* (Desf.) Banfi. Galasso. Foggi. Kopecký & Ardenghi subsp. *interruptum*	98.94	4.70	0.21	67.00	141.50
Poaceae	*Lolium perenne* L.	34.12	3.55	0.33	19.30	50.00
Poaceae	*Lolium rigidum* Gaudin subsp. *rigidum*	49.02	2.94	0.27	29.20	74.00
Poaceae	*Melica ciliata* L. subsp. *magnolii* (Godr. & Gren.) K.Richt.	62.42	4.28	0.31	22.50	108.50
Poaceae	*Melica minuta* L. subsp. *latifolia* (Coss.) W.Hempel	57.14	8.48	0.45	24.50	112.00
Poaceae	*Oloptum miliaceum* (L.) Röser & H.R.Hamasha	62.96	1.44	0.31	10.50	107.00
Poaceae	*Oloptum thomasii* (Duby) Banfi & Galasso	78.11	8.12	0.87	25.00	605.00
Poaceae	*Stipellula capensis* (Thunb.) Röser & H.R.Hamasha	28.50	2.08	0.23	17.50	42.60
Poaceae	*Trisetaria panicea* (Lam.) Paunero	12.65	1.80	0.63	4.00	29.40
Poaceae	*Triticum neglectum* (Req. ex Bertol.) Greuter	23.00	2.80	0.39	9.50	36.00
Poaceae	*Triticum vagans* (Jord. & Fourr.) Greuter	21.80	1.62	0.33	13.50	39.00
Primulaceae	*Cyclamen repandum* Sm. subsp. *repandum*	5.14	0.49	0.43	1.50	10.50

Primulaceae	*Lysimachia arvensis* (L.) U.Manns & Anderb. subsp. *latifolia* (L.) Peruzzi	6.51	0.36	0.50	1.50	16.00
Primulaceae	*Lysimachia linum-stellatum* L.	3.94	0.26	0.21	2.80	5.50
Ranunculaceae	*Nigella damascena* L.	20.77	2.42	0.37	10.50	37.20
Ranunculaceae	*Ranunculus macrophyllus* Desf.	21.40	1.05	0.44	5.00	44.00
Ranunculaceae	*Ranunculus ophioglossifolius* Vill.	13.17	1.78	0.33	8.00	21.00
Rosaceae	*Poterium sanguisorba* L.	10.08	1.27	0.40	3.00	15.00
Rosaceae	*Rosa sempervirens* L.	37.95	4.97	0.41	10.00	53.00
Rosaceae	*Rubus ulmifolius* Schott	102.66	8.73	0.38	45.00	200.00
Rubiaceae	*Galium divaricatum* Lam.	19.14	3.64	0.60	6.50	39.50
Rubiaceae	*Galium murale* (L.) All.	15.19	2.59	0.54	5.00	28.00
Rubiaceae	*Rubia peregrina* L.	26.06	1.09	0.62	3.90	126.00
Rubiaceae	*Theligonum cynocrambe* L.	6.21	0.53	0.52	2.00	15.50
Santalaceae	*Osyris alba* L.	85.60	9.73	0.36	15.00	124.00
Selaginellaceae	*Selaginella denticulata* (L.) Spring	1.81	0.29	0.51	0.60	3.40
Smilacacee	*Smilax aspera* L.	55.51	11.36	0.65	19.00	138.00
Thymelaeaceae	*Daphne gnidium* L.	39.48	4.63	0.29	21.00	51.40
Thymelaeaceae	*Thymelaea hirsuta* (L.) Endl.	24.89	2.36	0.30	16.70	43.60
Urticaceae	*Parietaria lusitanica* L. subsp. *lusitanica*	5.45	0.73	0.42	2.00	8.50
Urticaceae	*Urtica membranacea* Poir.	37.12	2.74	0.33	18.00	61.50
Vitaceae	*Vitis vinifera* L.	54.11	4.31	0.25	35.50	77.00

Subchapter 4.2: seed of herb layer communities

Reduction of intraspecific seed mass variability along a land-use intensification gradient

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Abstract

The functional response of natural patches to surrounding land-use changes is strongly related to variations in functional traits of coexisting species. To exemplify the effects on species of a general pattern of land-use intensification mountains-coastland, we investigated the variation of a key plant trait - seed mass - in small woodlots located along a land-use intensification gradient for two common species (*Asparagus albus* and *Asparagus acutifolius*) in the Mediterranean areas. We aimed to investigate the responses of seed mass of species living in small patches along a land-use gradient of intensification and explore the relationship between its variation and different environmental filters. Along the gradient, *A. acutifolius* seed mass decreased from natural to urban matrix (higher to lower altitude), whereas *A. albus* seed mass increased along the same gradient, with larger seed in patches located in the urban matrix than in those within natural matrix. At intra-specific level, *A. acutifolius* seeds were significantly different at the extremes of the gradient (natural vs urban matrix), while *A. albus* showed significant differences both between natural and urban matrices and between agricultural and urban matrices, revealing more sensitiveness to land-use change. Environmental drivers on the seed mass of the two species showed an opposite trend of the two species in relation to the environmental variables analyzed. The matrix surrounding the small patches influenced seed mass variability: in the small patches surrounded by natural and agricultural matrices, we observed for both species a higher seed mass variability, being highest in the agricultural matrix, while we observed a limited variability in artificial areas, suggesting a homogenization in terms of seed mass in the urban matrix. Environmental drivers on the seed mass of the two species showed an opposite trend in relation to biotic, topographic and bioclimatic variables. We observed that for two common Mediterranean species, landscape matrix influenced one of the most important seed functional traits, leading to a reduction of intraspecific variability in artificial context. Understanding how and why these relations occur could improve our capacity to find adaptive strategies for environmental management.

Keywords: Urban homogenization, functional trait, Small Woodlots Outside Forests, *Asparagus albus*, *Asparagus acutifolius*, Sardinia

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Introduction

The Mediterranean basin is one of the most significantly altered hotspot on Earth (Myers et al., 2000) and therefore it has been classified as one of the most sensitive to climate change (Giorgi & Lionello, 2008). For several millennia this region has been intensively affected by human activities (Marignani et al. 2017a) and today only 4,7% of primary vegetation remains, with a landscape strongly shaped from this anthropogenic disturbance (Falcucci et al., 2007) with a constantly increasing rate of land consumption (Strollo et al., 2020). In the last 50 years, the human pressure along the coastal zones increased worldwide, with impacts such as degradation and loss of the littoral landscape resulting from an alteration of the natural spatial pattern (Malavasi et al., 2013; Bajocco et al., 2012; Marignani et al., 2017b) while mountain areas are abandoned and naturally reforested (Carranza et al., 2020; Falcucci et al., 2007).

In this threatened and fragmented landscape context, conservation efforts focused on the understanding and preservation of large habitat fragments, rather than the smaller ones (Cadavid-Florez et al., 2020; Fahrig, 2017); nevertheless, small patches of natural remnants can have an important ecological value in ecosystem functioning and services (Archibald et al., 2011; Capotorti et al., 2020; Maccherini et al., 2011). Small Woodlots Outside Forests (hereafter SWOFs; Italian National Forest Inventory; http://www.infc.it), a peculiar type of Trees Outside Forest (TOF; FAO, 2001), are defined as a group of trees with an area larger than 0.05 ha and less than 0.5 ha (FAO, 2010; FAO, 2013). These landscape patches, exactly likewise Trees Outside Forest, can play an important role for climate condition, wind or soil erosion mitigation (Bellefontaine et al., 2001), improve water quality through phytodepuration (Endreny, 2002), promote the conservation of biodiversity, reduce the effects of habitat fragmentation (Fischer et al., 2010; Gibbons et al., 2008) and represent ecological corridors, as well as habitats for numerous animal and plant species (Bellefontaine et al., 2001).

The functional response of small patches like SWOFs to land-use changes, as well as their capacity to provide ecosystem services, is strongly related to changes in functional traits of coexisting species. Several ecological filters, such as abiotic and biotic drivers, are also involved in this multi-scale relationship (Keddy, 1992; Vanneste et al., 2019; Violle et al., 2007). Large-scale environmental factors could affect ecological strategies, local environmental variations determine niche partitioning and finally, landscape factors determine species dispersion among patches and shape the distribution of the functional traits in community assemblages (Vanneste et al., 2019). Surprisingly enough, we are not aware of any study focused on the ecological value and functioning of SWOFs considering, for example, SWOFs located in different landscape matrix: although those small patches show to have a promising functional role, only a few specific studies focused on them (Baffetta et al., 2011; Corona et al., 2011; Sallustio et al., 2018).

To exemplify the effects on species of a general pattern of land-use intensification mountains-coastland, we investigated the variation of a key plant trait - seed mass - in SWOFs located along a land-use intensification gradient for two common species (*Asparagus albus* L. and *Asparagus acutifolius* L.) in the Mediterranean areas.

Seed mass is one of the most important plant traits controlling plant population dynamics and community structure (Guo et al., 2010; Leishman, 2001; Rees, 1995; Jiménez-Alfaro et al., 2016; Saatkamp et al., 2019), playing a crucial role in interspecific interactions (Carón et al., 2014; Westoby et al., 1996). Furthermore, seed mass is important in colonization process (Vanneste et al., 2019) due to his correlation with other traits such as germination rate (Shahi et al., 2015), seedling establishment and seedling persistence (Harper et al., 1970; Fenner & Thompson, 2005; Weiher et al., 1999). Beyond its ecological importance, seed mass is also easy to measure (Chelli et al., 2019; Kattge et al., 2020). The variation in this trait can occur at all ecological levels (Guo et al., 2010; Jiménez-Alfaro et al. 2016; Saatkamp et al 2019): within populations, within and across species (Harper et al., 1970; Moles et al., 2005; Westoby et al., 1996), among conspecific individuals, but also within individual plants, inflorescences and fruits (Fenner & Thompson, 2005). Seed mass' variations may have important ecological implication (Cordazzo, 2002), affecting seed predation (Thompson, 1987), dispersal (Greene & Johnson, 1993) and dormancy (Harel et al., 2011) but also plant height, growth form (Garnier & Navas, 2012) and competitive plant ability (Leishman, 2001).

We aimed to investigate the responses of seed mass of species living in SWOF along a land-use gradient of intensification and explore the relationship between its variation and different environmental filters. More specifically, regarding the response of seed mass of two congeneric Mediterranean species in SWOFs located along a land-use intensification gradient, we addressed the following questions:

(i) Does the land-use gradient influence seed mass and its variability at intra- and interspecific levels?

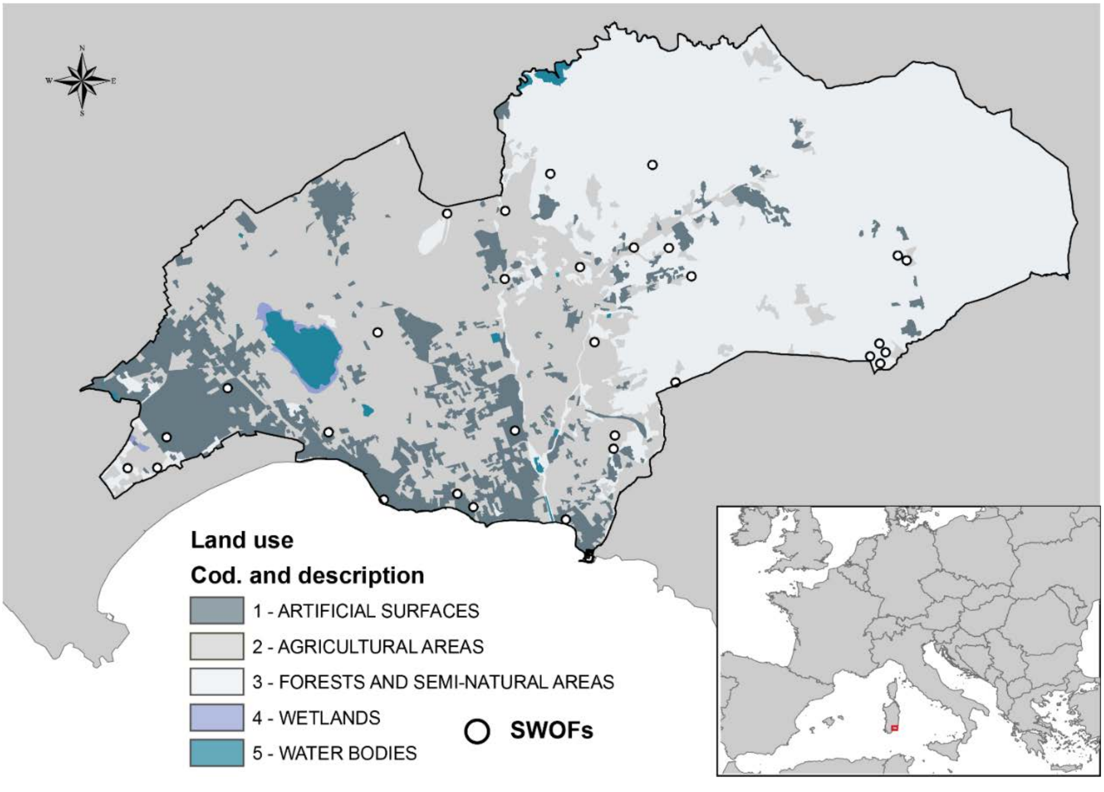
(ii) Which environmental variables mostly affect their seed mass?

Materials and methods

Study area and investigated species

The study was conducted in the Eastern Metropolitan City of Cagliari (Southern Sardinia, Italy), a medium-sized functional urban area characterized by three levels of fragmentation degrees (Bazzato et al., submitted; Palumbo et al., 2020). The study area covers a surface of about 18,300 hectares and includes the municipalities of Maracalagonis, Quartu Sant'Elena, Quartucciu, Settimo San Pietro and Sinnai. The area is characterized by a gradient of land-use intensification that runs roughly in an east-south/west direction from mountainous areas to coastline zones (forests and semi-natural areas, agricultural areas and artificial surfaces) (Figure 1) and an elevation gradient from 0 m a.s.l. to 700 m a.s.l.. Climate is typically Mediterranean, with a strong seasonality characterized by dry and hot summers and relatively rainy and mild winters. Canu et al. (2015) identify for this area the Mediterranean pluvioseasonal oceanic macrobioclimate, one class of continentality (strong euoceanic), four thermotypic horizons (from lower themomediterranean to upper mesomediterranean) and five ombrothermic horizons (from lower dry to lower humid), resulting in a combination of 11 isobioclimates. As regards potential vegetation, the area is characterized by thermo-mesomediterranean associations of evergreen, neutrophilous or acidophilous sclerophylls. In the upper areas, the most mesophilous woods with *Quercus ilex* or *Q. suber* and shrub elements as *Erica arborea, Arbutus unedo, Phyllirea latifolia, Myrtus communis* and *Juniperus oxycedrus* dominate. The high-shrub and pre-forest successions are distributed in the most thermo-xerophilous zone with wild olive shrublands (*Olea europaea* var. *sylvestris* with *Pistacia lentiscus, Juniperus turbinata* and *Euphorbia dendroides*) (Bacchetta et al., 2009). Due to the increasing anthropic disturbance, semi-natural areas are dominated by successional stages of thermophilous shrublands, garrigues or perennial grasslands (Figure 1).

Fig. 1. Study area located in the Metropolitan City of Cagliari, Southern Sardinia, characterized by an east-south/west direction gradient of land-use intensification, ranging from mountainous areas to coastline zones (forests and semi-natural, agricultural and artificial surfaces).



We studied *Asparagus acutifolius* L. and *Asparagus albus* L. (Liliaceae), two perennial plants of *Asparagus* genus, with different niche breadth and present all along the gradient of land-use intensification in our study area.

A. acutifolius is a perennial, herbaceous Steno-Mediterranean species (Ferrara et al., 2011), common in shrub communities, *Quercus ilex* woods, wood glades, edges, marginal areas, from sea level to 1000-1100 m a.s.l. (Urbani et al., 2007). *A. acutifolius* is common and widely distributed in the Mediterranean basin and its islands (Ferrara et al., 2011; Sica et al., 2005), where it has been used in the traditional diet since ancient times (Benincasa et al., 2007; Ferrara et al., 2011; Mantovani et al., 2016). This species adapts to shade and to extreme temperature ranges i.e., low winter and high summer temperatures (Mantovani et al., 2019). It is a frugal species, tolerant to drought and resistant to some pathogens (Kubota et al., 2012), adapting to marginal and arid lands (Lo Porto et al., 2019). Seeds of *A. acutifolius* are described with a strong dormancy and difficult germination (Katsenios et al., 2019).

A. albus is a spiny shrub with deciduous cladodes. Flowers are hermaphrodite and the fruits, ripen by fall, are dispersed by birds (Tirado & Pugnaire, 2003); it propagates by seed and vegetative methods as well. This plant is characteristic of Mediterranean maquis, linked to drylands, rocky soils and cliffs in subhumid and semiarid bioclimates, living from the sea level to 900-1000 m a.s.l. (Urbani et al., 2007; Vèla, 2018). *A. albus* is distributed throughout the islands and peninsulas of Southwestern Europe and Northwestern Africa (Véla, 2018), with a W-Steno-Mediterranean chorotype. In Italy, it is present only in the Southern regions (Campania, Calabria, Sicily and Sardinia; Bartolucci et al., 2018).

According to Ellenberg’s Indicator Values (EIV, Pignatti et al., 2005) and comparing the two *Asparagus* species, *A. acutifolius* is characterized by higher values of N and R, indicating a species being more competitive in nutrient-rich soils (N) and slightly more alkaline and calcareous conditions (R) while, *A. albus* is more thermophilous and light tolerant (T, L; Table 1).

Table 1. Ellenberg indices for the two *Asparagus* species investigated. Climatic variables: light conditions (L), temperatures (T), climatic continentality (C). Edaphic conditions: moisture (U), reaction (R), nutrient availability (N), salinity (S)

Species name	**L**	**T**	**C**	**U**	**R**	**N**	**S**
Asparagus acutifolius L.	6	9	4	2	5	5	0
Asparagus albus L.	8	10	3	2	4	2	0

Sample design of small woodlots outside forest

Data were collected by means of a multi-phase sampling design, considering the hierarchical and nested structure of the populations.

Using photointerpretation of digital color orthophotos (2016), we identified and mapped all the SWOFs, between 0.05 and 0.5 hectares. We classified the study area according to a land-use map (I hierarchical level, scale 1:25,000; RAS, 2017): artificial surfaces (URB), agricultural areas (AGR) and forest and semi-natural areas (NAT). Hence, according to the dominant land-use type surrounding the SWOFs, every SWOF was assigned to the corresponding land-use matrix (URB, AGR or NAT). We excluded SWOFs with a size of less than 0.1 hectare, as well as those surrounded by a mixed land-use matrix.

From a total of 201 detected SWOFs (67 in URB, 70 in AGR, 64 in NAT) we randomly selected 30 SWOFs (8 in URB, 11 in AGR and NAT) along the land-use gradient.

Within the 30 SWOFs we collected seeds and information on environmental, biotic and topographic drivers.

Seed data collection

We collected mature seeds from healthy adult plants of the two *Asparagus* species during the period October-December 2019, following Pérez-Harguindeguy et al. (2013). From a total of 30 SWOFs, we found seeds of at least one species in 20 SWOFs, while in the other four sites only *A. acutifolius* individuals with no seeds were recorded. In particular, we sampled *A. acutifolius* in 18 SWOFs (5 NAT, 9 AGR, 4 URB) and *A. albus* in 14 SWOFs (3 NAT, 8 AGR, 3 URB); seeds of both species were found in 12 out of 20 SWOFs (3 NAT, 6 AGR, 3 URB). When available, we collected a minimum of 10 seeds from each plant. Seeds were cleaned and air-dry stored; then, they were oven-dried at 80°C for at least 48 hours, or until equilibrium mass (Pérez-Harguindeguy et al., 2013). For each species, we weighted to 100 µg accuracy 10 seeds randomly chosen with 15 replicates from each SWOF and then seed mass per single seed was calculated.

Environmental predictors

To explore the effect of environmental predictors on seed mass variation, three distinct sets were considered: (i) biotic (ii) topographic (iii) and bioclimatic variables (Table 2). In each SWOF we conducted a systematic sampling: we used SWOF’s centroid as the middle point for a linear transect, stretching along the maximum SWOF’s length. Hence, we surveyed five plots of 1 sqm, equally distanced one from another, along each transect.

Table 2. List of biotic, topographic and bioclimatic variables used as environmental predictors. Data refer to sites (SWOF).

Predictor set	Variable name	Variable description
Biotic	Tree	% tree cover
	Shrub	% shrub cover
	Herb	% herb cover
	spec_rich	Number of vascular plant species richness
	H	Shannon diversity index ($H = \sum_{i=1}^{S} p_i \log(b)\, p_i$)
	CV_DBH	Coefficient of Variation of diameter at breast height (DBH)
Topographic	X	x geographical coordinate (m)
	Y	y geographical coordinate (m)
	Z	elevation (m)
	Inclination	inclination (°)
	Exposition	exposition (°)
	Coast_dist	Distance from the coastline (m)
Bioclimatic	BIO01	Annual Mean Temperature (°C)
	BIO05	Maximum Temperature of Warmest Month (°C)
	BIO06	Minimum Temperature of Coldest Month (°C)
	BIO08	Mean Temperature of Wettest Quarter (°C)
	BIO09	Mean Temperature of Driest Quarter (°C)
	BIO10	Mean Temperature of Warmest Quarter (°C)
	BIO11	Mean Temperature of Coldest Quarter (°C)
	BIO12	Annual Precipitation (mm)
	BIO13	Precipitation of Wettest Month (mm)
	BIO14	Precipitation of Driest Month (mm)
	BIO16	Precipitation of Wettest Quarter (mm)
	BIO17	Precipitation of Driest Quarter (mm)
	BIO18	Precipitation of Warmest Quarter (mm)
	BIO19	Precipitation of Coldest Quarter (mm)

Biotic and structural variables

The first set of environmental predictors consisted of ten variables describing the structural and biotic characteristics of the SWOFs (Table 2): trees, shrubs and herb cover, species richness, Shannon diversity index and the coefficient of variation of tree diameter at breast height (DBH).

Tree, shrub and herb coverage were visually estimated as a percentage at the plot level and then summed across the five plots in each transect per site (i.e., SWOF). For each SWOF, plant species richness and Shannon diversity index of tree, shrub and herb layer were calculated (R vegan package; Oksanen et al., 2018).

Tree and shrubs diameter at breast height of five individuals per plot were measured for a total of 25 measures per SWOF. Based on these measurements, we calculated the coefficient of variation of the diameter at breast height (DBH) as a measure of structural diversity of SWOFs.

Topographic variables

The second set of environmental predictors consisted of variables describing topographic characteristics (Table 2) calculated for each SWOF: x and y geographical coordinates are expressed as metric units in the Monte Mario/Italy Zone 1 reference system (EPSG 3003); elevation, inclination and exposition are derived from a DTM (10 m detail); coastal distance is expressed as the metric distance from the centroid of each SWOF to the nearest coastline.

Bioclimatic variables

The third set of environmental predictors consisted of 14 biologically meaningful bioclimatic variables (Bazzato et al., 2021) based on a long-term climate series derived from Canu et al. (2015). We considered bioclimatic variables related to temperature (BIO01, BIO5-BIO06, BIO10-BIO11), to precipitation (BIO12-BIO14, BIO16-BIO17) and bioclimatic variables related to both temperature and precipitation (BIO08-BIO09, BIO18-BIO19).

Statistical analyses

To investigate the difference of seed mass along grades of land-use intensification i.e. matrix (three levels, fixed: NATural, AGRicultural and URBan), among SWOFs (random factor, nested within matrix, 20 levels) and species (fixed factor, two levels: *A. albus, A. acutifolius* crossed with matrix and SWOF), we used a permutational multivariate analysis of variance (PERMANOVA; Anderson, 2001). The advantage of this method is the possibility to accommodate random effects, hierarchical and mixed models, unbalanced and asymmetrical designs, while simultaneously maintaining robust statistical properties in a distribution-free setting (Anderson, 2017). Analyses were conducted using data of 15 replicates (10 seeds each) per SWOF for each species. We used the Euclidean distance and performed the analysis with 999 permutations of residuals under a reduced model (Anderson, 2001). Significant terms ($p < 0.05$) were investigated using a *post-hoc* permutational pairwise comparison test with 999 permutations. We also calculated the pseudo multivariate variance component, expressed as percentages, for each source of variation. PERMANOVA was performed using the routine included in PRIMER 6+ software.

To identify the key factors that affect intraspecific seed mass variation of *A. albus* and *A. acutifolius*, we conducted an information-theoric approach of a candidate set of Generalized Linear Models (GLMs) for each response variable at SWOF level (i.e., mean seed mass calculated across all 15 replicates per SWOF for each species), using the three sets of environmental drivers as predictor variables and matrix (NAT, AGR, URB) as a fixed effect.

Before performing the analyses, all predictor variables were standardized (z-scores) using “decostand” R function (Oksanen et al., 2018), in order to use the magnitude of regression coefficients to rank their relative importance. To avoid multicollinearity within each set of environmental predictors for each species, we applied a variable-selection procedure using “vifstep” function (Naimi et al., 2014) with a cut-off value of Variance inflation factor VIF=3; only selected predictors were retained for the following steps (Table 3).

For each predictor group and each species, a subset of all candidate reduced models were automatically computed using “glmulti” R package (Calcagno, 2019). All candidate reduced models were ranked based on comparisons of AICc, a variant of Akaike’s Information Criterion corrected for small sample sizes and small ratio n/K (sample size and predictors) (<40; Burnham & Anderson, 2002). All models with a difference in $\Delta AICc \leq 2$ when compared to the best model (i.e., the one with lowest AICc value) were considered to have similar plausibility and display in results with their Akaike weights (w_i, ranging from 0 to 1) and their evidence ratios (w_1/w_i) in comparison to the highest-ranked model (w_1) (Burnham & Anderson, 2002). For all subsets of candidate models where the highest Akaike weight (w_1) was below 0.90 and all other models had a small difference of evidence ratios, we used model averaging to calculate the relative importance (RI, the sum of Akaike weights w_i across all models with 95% probability where the variable appeared) of each predictor and to generate parameter coefficients for the remaining predictors (Burnham & Anderson, 2002). Although does not exist a RI threshold, a high RI value of the predictor indicates that the variable is frequent among the candidate models with high Akaike weights (w_i), and it has a greater probability to be a component of the best model (Burnham & Anderson, 2002).

Table 3. Biotic, topographic and bioclimatic set of predictor variables selected by the variable-selection procedure (VIF=3 as cut-off value).

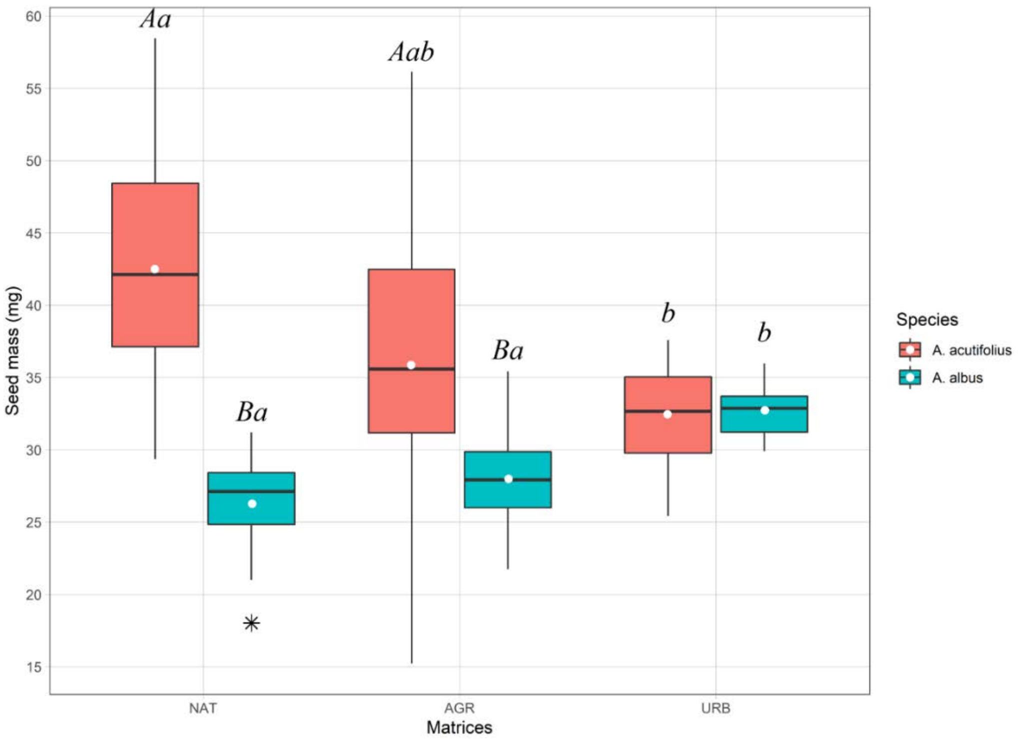
Asparagus acutifolius		***Asparagus albus***	
Predictors	VIF	Predictors	VIF
Biotic		Biotic	
H	1.1	H	1.1
Tree	1.5	Tree	1.7
Shrub	2.7	Shrub	2.8
Herb	1.4	Herb	1.3
CV of DBH	1.8	CV of DBH	1.8
Topographic		Topographic	
X	1.8	X	1.6
Y	1.2	Y	1.2
Inclination	2.1	Inclination	1.5
Exposition	1.7	Exposition	1.1
Bioclimatic		Bioclimatic	
BIO09	1.6	BIO08	1.1
BIO10	1.5	BIO09	1.1
BIO13	1.1		

Results

A total number of 11,513 seeds of *Asparagus* were collected in 20 SWOFs: 7,143 from 160 individuals of *A. acutifolius* and 3,597 from 149 individuals of *A. albus*.

Mean seed mass of *A. acutifolius* was 36.95 mg (SE = 0.54), in line with data available in the literature (ranging from 34.5 mg to 40.8 mg per seed; Kattge et al., 2020). It ranged from a maximum of 58.4 mg to a minimum of 15.2 mg per seed: the heavier seeds were recorded in SWOFs surrounded by a natural matrix, while the lighter seeds were found in SWOFs enclosed in an agricultural matrix. Mean seed mass for *A. albus* is 28.65 (SE = 0.24), coherent with available data (28.42 with a SE = 1.52; Kattge et al., 2020). The seed mass ranged from a maximum of 36 mg to a minimum of 18 mg per seed, with heavier seeds recorded in the urban matrix and the lighter ones in the natural land-use matrix. Boxplots of the seed mass variation, among different matrices, showed an opposite trend for the two species studied (Figure 2). Both median and mean values decreased in *A. acutifolius* along the land-use intensification gradient (from natural to urban matrix). In contrast, these values increased in *A. albus* along the same gradient. Furthermore, for both species, the less seed mass variability was found in SWOFs located surrounded by artificial surfaces (URB) and the highest in the agricultural ones (AGR).

Fig. 2. Variability of seed mass (mg) for *A. acutifolius* and *A. albus* in SWOFs surrounded by a different land-use matrix. Boxes span the 25th to the 75th percentile; whiskers span from 10th percentile to the 90th percentile. The bar across the box shows the median seed mass, the white point the mean seed mass and the star the outliers. Different capital letters indicate significant differences between species within each matrix; different lower-case letters indicate significant differences between matrix within each species.



PERMANOVA revealed that almost all sources of variation significantly affected seed mass for both species (Table 4). The largest component of variation was associated with the main effect of species, followed by the main effect of SWOF, the Matrix x Species interaction and SWOF x Species interaction (Table 4). Pairwise comparisons for the Matrix x Species interaction revealed that within NAT and AGR matrix, the contrasts between species were significant (Table 5). Moreover, within species the contrasts were all significant except for *A. acutifolius* among AGR vs. NAT and AGR vs. URB, and for *A. albus* among AGR vs. NAT (Table 5).

Table 4. PERMANOVA results on seed mass of *A. acutifolius* and *A. albus*. Significance codes: (*) p < 0.05, (***) p < 0.001

Source of variation	***df***	**MS**	***F***	**Variance components (%)**
Matrix	2	0.0225	0.31	0
Species	1	0.7033	20.73**	25.77
SWOF (Matrix)	17	0.0821	164.19***	23.91
Matrix x Species	2	0.2098	6.18*	22.42
SWOF (Matrix) x Species	9	0.0339	67.89***	18.93
Residual	448	0.0005		8.96
Total	479			

Table 5. Results of PERMANOVA pairwise test for the interaction Matrix x Species. NAT – Forested and semi-natural areas, URB – Artificial areas, AGR – Agricultural areas (*) p < 0.05

Groups	***t* statistic**
Matrix x species	
Within Matrix - Forested and semi-natural areas (NAT)	
A. acutifolius, A. albus	4.7182*
Within Matrix - Agricultural areas (AGR)	
A. acutifolius, A. albus	3.6557*
Within Matrix – Artificial areas (URB)	
A. acutifolius, A. albus	0.6509
Within species *A. acutifolius*	
AGR, NAT	1.2819
AGR, URB	0.6527
NAT, URB	2.4500*
Within species *A. albus*	
AGR, NAT	0.9102
AGR, URB	2.8562*
NAT, URB	3.8084*

Effect of environmental drivers on mean seed mass of *Asparagus acutifolius*

The model selection procedure applied to biotic predictors exploring mean seed mass of *A. acutifolius* identified two candidate reduced models (Table 6). The first one retained only the intercept ($w_i = 0.268$), meanwhile, the second one also included the coefficient of variation of DBH ($w_i = 0.100$), with a small difference in the evidence ratios between them (Table 6). From all possible candidate models of topographic predictors, only one best model was identified with a low Akaike weight ($w_i = 0.380$) (Table 6).

Considering bioclimatic predictor variables, two candidate reduced models were identified, but the Akaike weights of these models tended to be very small ($w_i << 0.90$) (Table 6), and the values of evidence ratios confirmed there was not enough support to the best AICc model (Table 6).

Table 6. Summary of the top regression models within 2 AICc units of the highest-ranked model predicting mean seed mass of *A. acutifolius* (ACUTI). The models are in descending order from the most to the least supported based on Akaike information criteria (AICc). For candidate subset, we reported difference in AIC value (Δi) from that of the best model; Akaike weights of the ith model (w_i), representing the probability that the *i*th model is the best fitting model in each candidate subset; evidence ratios (ER = w_1/w_i), comparing *i*th model (w_i) to the highest-ranked model (w_1).

Group of predictors	Candidate models	AICc	Δi	w_i	ER
Biotic	ACUTI ~ 1	-32.762	0.000	0.268	1.000
	ACUTI ~ 1 + CV_DBH	-30.788	1.974	0.100	2.683
Topographic	ACUTI ~ 1 + Inclination	-37.657	0.000	0.380	1.000
Bioclimatic	ACUTI ~ 1 + BIO13	-33.438	0.000	0.352	1.000
	ACUTI ~ 1	-32.762	0.676	0.251	1.402

Model-averaged estimates confirmed low relative importance of all biotic predictors to explain the variation in mean seed mass of *A. acutifolius* (RI < 0.2), reflecting their low weight (Table 7). Considering values of model-averaged parameter estimates and their relative importance based on the set of models within 95% confidence level, the variables most frequently found in candidates were inclination and exposition, with a high (RI = 0.7604) and low frequency (RI = 0.3582) respectively, showing positive associations with the variation of the mean seed mass (Table 7).

Average parameter coefficients showed a positive variation in mean seed mass associated with increasing precipitation of the wettest month (BIO13) that showed the highest frequency (RI = 0.5305) on the set of bioclimatic models within 95% confidence level (Table 7).

Table 7. Model-averaged estimates for predictors of mean seed mass of *A. acutifolius* response. The average parameter coefficients (β) from all candidate models with 95% probability are presented. The relative importance (RI) reflects the frequency with which a given predictor is found in the candidate models.

Group of predictors	Predictors of seed mass of *A. acutifolius*	β	Unconditional SE	RI
Biotic	(Intercept)	0.3681	0.0222	1.0000
	CV_DBH	0.0037	0.0084	0.1997
	H	0.0018	0.0055	0.1630

	Shrub	0.0024	0.0064	0.1625
	Tree	0.0009	0.0042	0.1475
	Herb	-0.0004	0.0038	0.1459
	MATRIXNAT	0.0085	0.0170	0.1265
	MATRIXURB	-0.0044	0.0115	0.1265
Topographic	(Intercept)	0.3695	0.0177	1.0000
	Inclination	0.0382	0.0273	0.7604
	Exposition	0.0118	0.0186	0.3582
	X	-0.0014	0.0057	0.1639
	Y	0.0009	0.0037	0.1370
Bioclimatic	(Intercept)	0.3687	0.0208	1.0000
	BIO13	0.0201	0.0237	0.5305
	BIO09	0.0010	0.0042	0.1463
	BIO10	-0.0001	0.0029	0.1333
	MATRIXNAT	0.0050	0.0105	0.0755
	MATRIXURB	-0.0026	0.0069	0.0755

Effect of environmental drivers on mean seed mass of *Asparagus albus*

The Akaike weights of the best-ranked regression models examining the effect of biotic predictors on mean seed mass of *A. albus* showed a high probability that the first model (w_1 = 0.457) was the best fitting model in comparison to the second one (Table 8).

Considering topographic and bioclimatic predictor variables, five candidate reduced models were identified within 2 AICc units of the highest-ranked model (Table 8). However, Akaike weights of this subset of reduced models tended to be very small (w_1 << 0.90) and the values of evidence ratios proved to be insufficient to support the best AICc model for each subset (Table 8), highlighting the need to estimate parameters by model-averaging.

Table 8. Summary of the top regression models within 2 AICc units of the highest-ranked model predicting mean seed mass of *A. albus* (ALBUS). The models are in descending order from the most to least supported based on Akaike information criteria (AICc). For candidate subset, we reported difference in AIC value (Δi) from that of the best model; Akaike weights of the ith model (w_i), representing the probability that the *i*th model is the best fitting model in each candidate subset; evidence ratios (ER = w_1/w_i), comparing *i*th model (w_i) to the highest-ranked model (w_1).

Group of predictors	**Candidate models**	**AICc**	**Δi**	**w_i**	**ER**
Biotic	ALBUS ~ 1 + Tree + CV_DBH	-70.216	0.000	0.457	1.000
	ALBUS ~ 1 + CV_DBH	-68.593	1.622	0.203	2.250
Topographic	ALBUS ~ 1 + MATRIX + X	-54.677	0.000	0.173	1.000
	ALBUS ~ 1 + MATRIX	-54.004	0.673	0.124	1.400
	ALBUS ~ 1 + X	-53.960	0.717	0.121	1.431
	ALBUS ~ 1 + Inclination	-53.720	0.957	0.107	1.613
	ALBUS ~ 1 + Y + Inclination	-52.757	1.920	0.066	2.611
Bioclimatic	ALBUS ~ 1 + BIO19	-55.656	0.000	0.208	1.000
	ALBUS ~ 1 + MATRIX + BIO19	-54.652	1.003	0.126	1.652
	ALBUS ~ 1 + BIO06 + BIO19	-54.459	1.196	0.115	1.819
	ALBUS ~ 1 + MATRIX	-54.004	1.652	0.091	2.284
	ALBUS ~ 1 + BIO10 + BIO19	-53.658	1.998	0.077	2.716

Model-averaged estimates for biotic predictors allowed us to identify two predictors with high relative importance, both retained in the highest-ranked model: coefficient of variation of DBH (RI = 1) and tree cover (RI = 0.61), which were associated with a negative and positive response of mean seed mass of *A. albus*, respectively (Table 8). Other biotic predictors – herb cover and Shannon diversity index - showed a generally low frequency on the set of models within 95% confidence level, reflecting their low weight (Table 8).

Considering topographic predictors, the variables most frequently found in reduced models were longitude and land-use matrices, also retained in the first three models, with a high (RI = 0.4452) or low (RI = 0.3580) frequency, respectively (Table 9). Average parameter coefficients showed that mean seed mass of this species responded negatively to the increasing of longitude. It was positively associated with the urban matrix and, at the end of the gradient, negatively associated with the natural matrix (Table 9). The remaining variables - inclination, latitude, and exposition – were less frequent than others and showed a negative relationship with the variation of mean seed mass (Table 9).

Model-averaged estimates for bioclimatic predictors showed that the precipitation of the coldest quarter (BIO19) was the variable most frequently found (RI = 0.6218) (Table 9), and it was also retained in four out of five models from the set of candidates (Table 8). Average parameter coefficients showed a positive variation in mean seed mass associated with a decline in the precipitation of the coldest quarter (BIO19) and in the mean temperature of the warmest quarter (BIO10) (Table 9). On the contrary, mean seed mass of *A. albus* responded positively to the increasing of minimum temperature of the coldest month (BIO06) and mean temperature of the driest quarter (BIO09), even if both variables had low relative importance (Table 9).

Table 9. Model-averaged estimates for predictors of mean seed mass of *A. albus* response. The average parameter coefficients (β) from all candidate models with 95% probability are presented. The relative importance (RI) reflects the frequency with which a given predictor is found in the candidate models.

Group of predictors	Predictors of seed mass of *A. albus*	β	Unconditional SE	RI
Biotic	(Intercept)	0.2865	0.0040	1.0000
	CV_DBH	-0.0283	0.0043	1.0000
	Tree	0.0056	0.0054	0.6134
	Herb	0.0011	0.0021	0.1984
	H	-0.0005	0.0011	0.0801
Topographic	(Intercept)	0.2836	0.0088	1.0000
	X	-0.0072	0.0093	0.4452
	MATRIXNAT	-0.0031	0.0089	0.3580
	MATRIXURB	0.0164	0.0226	0.3580
	Inclination	-0.0045	0.0072	0.2855
	Y	-0.0030	0.0053	0.2285
	Exposition	-0.0015	0.0030	0.1511
Bioclimatic	(Intercept)	0.2843	0.0084	1.0000
	BIO19	-0.0118	0.0110	0.6218
	MATRIXNAT	-0.0013	0.0070	0.2687

	MATRIXURB	0.0116	0.0180	0.2687
	BIO10	-0.0036	0.0061	0.2682
	BIO06	0.0033	0.0056	0.2553
	BIO09	0.0016	0.0033	0.1525

Discussion

Several studies highlighted the relationship between plant reproductive traits and different environmental variables; variation in seed traits, especially in seed mass, was usually related to environmental conditions such as climate, soil nutrients and moisture (Bhatt et al., 2019; Quesada et al., 1996), but can occur under other factors, such as environmental stress caused by land-use change and habitat fragmentation (Chen et al., 2020). Our study area presents a well-known pattern of land-use/transformation that shows a non-random spatial pattern, but exemplifies a global pattern: we observed a gradient of anthropic use intensification of the landscape, ranging from a low intensity in the higher elevation sites, to a medium intensity agricultural use in the hilly areas, and the highest intensity in the coastal and lowland areas (in Italy; see Strollo et al., 2020; Rosati et al., 2008).

Along this gradient, *A. acutifolius* seed mass decreased from natural to urban matrix (higher to lower altitude), whereas *A. albus* seed mass increased along the same gradient, with larger seed in SWOFs located in the urban matrix than in those within natural matrix. At intra-specific level, *A. acutifolius* seeds were significantly different at the extremes of the gradient (natural vs urban matrix), while *A. albus* showed significant differences both between natural and urban matrices and between agricultural and urban matrices, revealing more sensitiveness to land-use change.

Comparing the two species, at the interspecific level we observed that the matrix surrounding the SWOFs influenced seed mass variability: in the SWOFs surrounded by natural and agricultural matrices we observed for both species a higher seed mass variability, being highest in the agricultural matrix, while we observed a limited variability in artificial areas, suggesting a homogenization in terms of seed mass in the urban matrix. These results highlighted that land-use change might not only shape the mean seed mass of coexisting species, but also the variability of seed mass within and across species. Urban disturbance related to human activities promote homogenization in the physical environment (Chen et al., 2020; McKinney, 2006): the decrease of variability and the homogenization for this trait in the two species within SWOFs located in the urban matrix can be seen as a “biotic homogenization” (McKinney, 2006), a sort of levelling in terms of functional diversity, with both heavier and lighter seed mass being disadvantaged in urban context at inter-specific level (Williams et al., 2015).

The anthropic gradient that we observed in the study area is superimposed on a natural one, therefore we investigated if the differences we detected could be attributable to environmental variables, rather than to the land-use. Hence, as a second step, we decided to explore the environmental factors that mostly influenced the seed mass variation along the land-use intensification gradient. In this context, the ecological value of small isolated patches depends not only on environmental variables per se but also on how these environmental factors are shaped and modified by different anthropic land use.

A. acutifolius' seed mass is positively influenced by local factors, such as inclination and exposition, and by the precipitation of the wettest month (BIO13): in our land-use intensification gradient this corresponded to natural areas, usually steeper and rugged compared to the more transformed ones (i.e., agricultural and artificial areas), where uneven-structured *Quercus ilex* woods dominate, characterized by a lower mesomediterranean, upper subhumid, euoceanic weak bioclimate (Canu et al., 2015).

On the contrary, *A. albus* appeared to be more sensitive considering both the highest number of predictors selected from model selection procedure and the inclusion in these models of the land-use matrix. Along the land-use intensification gradient, seed mass was higher in SWOFs surrounded by artificial matrix and characterized by a larger tree cover and a simplified structure (i.e., reforestation and urban parks); nevertheless, even if more thermophilus and light tolerant compared to *A. acutifolius*, the mean temperature of summer could represent a limiting factor for *A. albus*, characterized by a more restricted ecological amplitude and a relatively reduced tolerance to environmental variations (Véla, 2018). Bhatt et al. (2019) hypothesized that the variability in seed mass of two species, recorded from population with similar climatic characteristics, can be attributed to resource availability among different populations. Accordingly, the positive effect of the urban land-use matrix for the seed mass in *A. albus* could also be related to urban nitrogen deposition, a phenomenon described for lichens communities (Llop et al., 2017) and plant-traits (Liu et al., 2020). In the future, soil samples could help us to validate this hypothesis.

Our results demonstrated the different effect of environmental drivers on the seed mass of the two species, showing an opposite trend in relation to biotic, topographic and bioclimatic variables. These opposite responses are not uncommon at intra and interspecific level. Carón et al. (2014, 2018) found species-specific responses of seed mass variation to environmental conditions in Central Andes: along a latitudinal and elevation gradient, seed mass of tree species is strongly influenced by the elevation, with seed mass of some species increasing with elevation and lower temperatures and the opposite behavior of the other species analyzed. In Carón et al. (2014) seed traits were studied along a latitudinal gradient in Europe; results showed that closely related *Acer* species exhibit divergent responses of seed mass to temperature variation, with *A. platanoides* more influenced by the climatic conditions than *A. pseudoplatanus*. Similarly, Soper Gorden et al. (2016), found an opposite variation pattern for seed mass of different species in relation to temperature and latitude in North America. For *Allium* species in the Tibetan plateau, Ge et al. (2020) reported a selection pressure on species with smaller seeds and less germination at higher altitude, showing that temperature, elevation and seed mass had independent effects on seed germinability.

Intraspecific variation and variability in seed mass along an environmental gradient could enhance the species’ ability to cope with the rapidly environmental changing (Wu et al., 2018). Furthermore, these differences can also be related to the variation of other traits such as seed shape and/or plant height and have implications in seeds germination percentage and other life-history stages. Different studies affirmed the relation between seed mass and germination rate (Shahi et al., 2015) and seedling establishment and persistence (Fenner & Thompson, 2005; Harper et al., 1970; Westoby, 1998; Weiher et al., 1999).

Conclusions

Environmental drivers that affect seed mass can differ between species and at the same time, the same drivers can act in a different direction and with different intensity, leading to conclusions that defy models with patterns generalization (Jiménez-Alfaro et al., 2016; Saatkamp et al., 2019). We observed that for two common Mediterranean species, landscape matrix influenced one of the most important seed functional traits, leading to a reduction of intraspecific variability in artificial context. Understanding how and why these relations occur could improve our capacity to find adaptive strategies for environmental management (Jiménez-Alfaro et al., 2016). In this framework, woodlots outside forest in landscape and urban planning and biodiversity management practices should be a priority and accordingly, we should enhance conservation efforts in understanding and preserving these small patches (Sallustio et al., 2018).

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Author contributions

Field research and statistical analysis: EB; Processing of seed: EB, ES. Research design: EB, ES, SM, MM; Advice on statistical analysis: SM, MM; Manuscript preparation: EB, ES, SM, MM.

CHAPTER 5 – ENVIRONMENTAL PLANNING

Confronting the global challenges of climate change, biodiversity loss, deforestation, air and water pollution, requires real commitment and effective cooperation across government levels (vertical coordination, multi-level governance), policy (horizontal) and administrative areas (territorial coordination) (Botequilha Leitão and Ahern 2002; Pearson et al. 2010). In particular, functional urban areas (metropolitan areas) can experiment collaboration, proposing an integrated and strategic planning to provide a rational and viable response to make cities greener, more natural or, in other words, more resilient to the global environmental change. Scientifically sound and flexible solutions inspired and supported by nature, simultaneously provide environmental, social and economic benefits and help build resilience (EU, 2020). These nature-based solutions, however, have a distinctive set of premises and involve the innovative application of knowledge about environmental, diversity patterns and organisms (EU, 2015).

In this context, the main aim of this chapter was to propose a methodological framework aiming to define a new indicator that can be used as a reference/benchmark for landscape monitoring and planning (Subchapter 5.1, paper 9) and describe the framework adopted by the Metropolitan City of Cagliari for the definition of its strategic plan (Subchapter 5.2, paper 10).

Subchapter 5.1: an index to plan functional urban areas

Paper 9

Bazzato E., Cillo D., Bacaro G., Marignani M. (prepared). Multiple approach leads to multi-solutions – an index to plan functional urban areas.

Subchapter 5.2: strategic plan of Metropolitan City of Cagliari

Paper 10

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Subchapter 5.1: an index to plan functional urban areas

Multiple approach leads to multi-solutions – an index to plan functional urban areas

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Abstract

Decision-makers and planners are called upon to make decisions that are based on many organisms and concepts, in addition to human wants and needs: deal with multiple priorities increases the request for scientifically sound and flexible solutions. We quantified composition and configuration features of the landscape to test how they affect the species richness in different functional groups of beetle species: threatened species; endemic species; alien, synanthropic and anthropophilic species; widespread species. The relationship between landscape-based metrics and species richness of each group were estimated and used to identify groups suitable to define a new indicator. Our results showed that landscape and species-based measures were correlated: highly fragmented habitats were typified by positive relationships with the richness of alien, synanthropic and anthropophilic species, and by negative relationships with the richness of threatened and endemic species, while we observed mixed responses in the richness of widespread species. Based on observed relationships, we proposed a Human Impact Indicator (HII) to quantify the degree of impact across local authorities within a functional urban area. We tested this methodological framework in a real planning jurisdiction located in a Mediterranean area (Sardinia, Italy). Our work aims to propose a multidisciplinary approach that recognized the complementary contributions of multiple levels of organization and propose a useful framework to develop nature-based solutions.

Keywords: Beetles; Landscape fragmentation; Monitoring land management; nature-based solutions

Bazzato E., Cillo D., Bacaro G., Marignani M. (prepared). Multiple approach leads to multi-solutions – an index to plan functional urban areas.

Introduction

In the last two centuries, the impact of human activities on land has grown enormously, altering the landscape with important ecological consequences (Tylianakis et al. 2008). Habitat fragmentation is one of the most urgent challenges facing both environmental planners (Carsjens and van Lier 2002) and ecologists (Haddad et al. 2015; Wilson et al. 2016): transformation, destruction, reduction and the consequent isolation of remaining patches of habitat directly affect population demographic structure, species dispersion dynamics, communities and the ecosystems' dynamics (Saunders et al. 1991; Fahrig 2003). Due to habitat fragmentation, specialist species decrease, whilst generalist or non-native species increase, leading to species turnover at the community level (Duan et al. 2019; Fahrig 2003).

Many authorities have already taken up the challenge to integrate existing EU nature legislation (e.g., 2020 Biodiversity Strategy), significantly contributing to sustaining nature-based solutions (Geneletti et al. 2017; Miller et al. 2009), and anchoring biodiversity objectives into key sectoral policies, programmes and projects to fill important gaps on sustainable environmental planning. In fact, to effectively tackle the need of meeting the multiple and often competing goals of land-use planning, we should promote coordination across government levels (vertical coordination, multi-level governance), policy (horizontal) and administrative areas (territorial coordination) (Botequilha Leitão and Ahern 2002; Pearson et al. 2010). In particular, functional urban areas (metropolitan areas) can experiment collaboration and joint planning with their municipalities beyond the administrative boundaries, proposing integrated and sustainable landscape planning (Brody et al. 2004).

In this context, nature-based solutions and ecological studies are becoming more important in understanding how current and future planning strategies and decisions will affect ecosystem services and biodiversity conservation (Bohnet and Smith 2007; Brody et al. 2004; Hoversten and Swaffield 2019). Multiple priorities increase the request for a scientifically sound and flexible solution (Marignani et al. 2017) and since planners and ecologists cannot measure everything of potential interest within a given planning area, choosing what to measure is critical. Many authors suggested the use of a system of quantitative measures describing multiple levels of organization and their main attributes (ecosystem, landscape, community, species and gene) (Poiani et al. 2000; Carignan and Villard 2002; Siddig et al. 2016).

At the landscape level, several metrics have been used as indicators of biodiversity (Walz, 2011; Schindler et al. 2015; Babí Almenar et al. 2019; Fahrig 2003): landscape heterogeneity, in term of composition and configuration of patches, determines the increase of the number of available niches for species (Kisel et al. 2011), thus increasing biodiversity.

At the species level, a wide variety of potential indicators was proposed (keystone or umbrella species, dispersal-limited species, resource-limited species, process-limited species, flagship species; see Carignan and Villard 2002). Indicators commonly used include several types of taxa, ranging from plants to microorganisms (Carignan and Villard 2002; Siddig et al. 2016). Arthropods are considered indicators of ecosystem integrity because are closely linked to the state of the environment (Dufrêne and Legendre 1997). This is particularly true for groups such as insects, which are estimated to include 5.5 million species globally (1.5 million of which are beetles, Stork et al. 2018), contributing to important ecosystem services (Losey and Vaughan 2006). From this perspective, insects could be used as reliable and sensitive indicators that echoes the interactions between human activity, landscape fragmentation and the natural environment. Insects respond to habitat fragmentation and habitat loss: for example, generalist species with broad feeding needs, more likely than specialist species, find sufficient resources within a fragment and have the highest chance of success in a fragmented landscape (Collinge 2000; Gibb and Hochuli 2002). Therefore, considering the wide distribution of insects, their numerical presence, and importance as sensitive indicators, it would be advantageous to integrate information on their status to support the management decisions.

Landscape-planning requires the use of reference values useful to effectively assess landscape conditions and to determine the positive or negative effects of changes in time (Hersperger et al. 2017); nevertheless, some gaps in the theory and data for defining quantified references values remain (La Rosa et al. 2016). To cope with the growing need to develop an easy assessment framework that reflects the ecosystem complexity, we propose an integrated multidisciplinary approach taking into account the complementary contributions of multiple levels of organization. Specifically, we identified two extremes along a continuum of approaches: (i) a pattern-oriented approach which considered several metrics at landscape level, (ii) a species-oriented approach, considering species richness as an aggregated measure at species level. A combination of these complementary approaches has already led to particularly useful insights into ecosystem functioning (Fischer and Lindenmayer 2007); but to the best of our knowledge, no study used a single indicator to integrate these two approaches, considering arthropods as indicators at the species level.

According to the proposed framework, we identified composition and configuration features related to the fragmentation status of our study area and we tested if landscape-based measures affect the species richness in different groups of beetle species: threatened species; endemic species; alien, synanthropic and anthropophilic species; widespread species.

We compared responses of these different groups to i) evaluate the relationship between species richness of each group and landscape-based measures and ii) define a new indicator that can be used as a reference/benchmark for landscape monitoring and planning. We tested this methodological framework in the Metropolitan City of Cagliari, a planning jurisdiction located on a Mediterranean island (Sardinia, Italy). Our work aims to propose a multidisciplinary approach that recognized the complementary contributions of multiple levels of organization and propose a useful tool for other functional urban areas and local planning authorities.

Materials and methods

Study area

The Metropolitan City of Cagliari (MCC hereafter) is a medium-sized functional urban area of about 125,000 hectares located in the Mediterranean basin (south Sardinia, Italy, fig.1). This territorial authority includes seventeen municipalities (Palumbo et al. 2020). Given its large extension, the territory is characterized by a complex orographic pattern, which typifies several landscapes placed on different geological substrata, generating heterogeneous landscapes and hosting a wide variety of natural habitats. Geology and geomorphology of the study area are quite heterogeneous for age and typology: 21 geological units have been recognized (RAS 2017), including granite complexes (Gerrei, Sarrabus, Sulcis-Iglesiente); two tectonic units (Sarrabus and Arburese); two volcanic districts (Sarroch and Siliqua); sedimentary successions from Oligocene and Miocene, Palaeogene and Post “Discordanza Sarda” (Sardinian Unconformity) and sediments of various origin (alluvial, aeolian, lacustrine, littoral and slope movement-related). Due to its geographic position, the climate is typically Mediterranean ranging from lower thermomediterranean to mesomediterranean thermotype (Canu et al. 2014). Vegetation is mainly composed of evergreen oak matorral, Oleo-lentisc brush, Meso-Mediterranean silicicolous garrigues and maquis and lagoons, which collectively comprise more than 70% of MCC.

Integration approach

Based on previous experiences (Blasi et al. 2008; Fischer and Lindenmayer 2007), the methodological framework recognized two extremes, i.e. pattern and species-oriented (Fig. 1). At landscape level (pattern), we analyzed land-use data to identify the composition and configuration features of landscape and define a fragmentation status of the territorial authorities (i.e., single municipalities). At species level, we analyzed beetle species-data to evaluate the species-richness of the indicator groups. Consequently, we combined the obtained outputs to assess the relationship between landscape-based and species-based measures. For each grid cell (see below), the species richness of each indicator group was correlated to the status of conservation and fragmentation using Pearson’s *r* correlation coefficient. Finally, we used this relationship to propose a new indicator able to combine and summarize landscape (pattern) and species-based measures.

Data processing and analysis were carried out using ArcGIS, Patch Analyst extension (Elkie et al. 1999; Rempel et al. 2012), and *vegan* (Oksanen et al. 2018), *gclus* (Hurley 2012) and *PerformanceAnalytics* (Brian and Peter 2018) R packages (RStudio Team 2016).

Landscape level

Landscape measures were based on land use composition and configuration to evaluate, respectively, the conservation status and metrics at landscape level to evaluate the fragmentation status of each territorial authority (i.e., single municipalities).

The conservation status was evaluated with the Index of Landscape Conservation (ILC) (Pizzolotto and Brandmayr 1996; Ferrari et al. 2008), using a reclassified Land Use Map at the third hierarchical level of detail (CORINE legend, scale 1:25.000; RAS 2017): (i) each land-use class was reclassified according to an anthropization coefficient ranging from 1 (high) to 9 (low anthropization; Corona et al. 2011); (ii) the reclassified land use classes were sorted by their increasing naturalness gradient; (iii) ILC were calculated for the entire MCC and single municipalities.

ILC values range between 0 and 1, where 0 corresponds to a high anthropization level and 1 corresponds to a high conservation status and naturalness.

We analyzed landscape configuration and spatial relationships among patches using eight metrics at landscape level for each municipality (see McGarigal et al. 2002, Table 1). Hence, we synthesized compositional and configurational features of landscape in a single measure defined as “fragmentation status”. Fragmentation status was evaluated by standardizing the original data matrices of *m* objects (territorial authorities, i.e. single municipalities) x *n* variables (ILC and metrics at landscape level) and performing a hierarchic clustering and a Principal Components Analysis (PCA), to reduce the original number of attributes into a smaller number of highly correlated factors on the one hand and to identify groups with higher similarity, on the other hand.

Agglomerative hierarchical cluster analysis was done using the complete linkage method, and Euclidean distances as a measure of dissimilarity. Hence, we projected the clusters on the scatterplot of the PCA summarizing compositional and configurational metrics.

To portray patterns of fragmentation and conservation status of each municipality in the study area, fragmentation status values and ILC were classified in three groups (high, medium and low value), using the Jenks Natural Breaks Classification that optimizes the arrangement of a set of values, reducing the variance within classes and maximizing the variance among them (Jenks 1967).

Species level

We selected 40 beetles species divided into four indicator groups (Table 2), identified according to the species' autoecology, chorology, ecological and functional importance, and their conservation/management concern, derived from the literature (Aliquò et al. 2007; Audisio et al. 2014; Ballerio et al. 2014; Brandmayr et al. 2005; Curletti et al. 2003; Ruffo and Stoch 2006), and personal observation. We classified as threatened the species listed on the Italian Red List of saproxylic beetles (Audisio et al. 2014) which are recognized to be strongly menaced by habitat loss and fragmentation. Common species, in terms of abundance and distribution, were classified as widespread species. We included in the endemic species group only taxa that are endemic to the Sardinia and Corsica islands, though they are not included in the Italian Red List of saproxylic beetles (Audisio et al. 2014). We also considered the species which are usually linked to the human-altered land-uses: alien, synanthropic and anthropophilic species (hereafter anthropogenic species).

Species occurrences were derived from public and private collections. Collectors' data were integrated with published lists (Latella et al. 2006). We gathered 3205 records (individuals) of 40 species, collected by fifty collectors from 1900 to 2017: seven individuals (0.2% relative occurrence) were collected before 1950; 660 individuals (20.6% relative occurrence) between 1951 and 1999; 2538 individuals (79.2% relative occurrence) collected from 2000 to 2017. Species occurrences collected by many collectors interested in several insect groups in almost one century can be assumed representative considering that, on average, the whole area has been reached by collectors. Each species occurrence was implemented in a geodatabase and georeferenced following the levels of accuracy used in CKmap project (Latella et al. 2006). For data analyses, the study area was subdivided into 387 contiguous grid cells of 4 km^2: georeferenced records were univocally assigned to a grid cell and each grid cell were univocally assigned to a single municipality, using a spatial join polygon based on cell centroid. We used grid cells with aggregated point information to assess the species richness of each group; hence, species richness' values were subdivided into three classes (high, medium and low richness following the same criterion discussed above for fragmentation status; Jenks 1967) to produce thematic maps.

The human impact indicator (HII)

To synthesized in one index the landscape-based and species-based measures, we proposed a new indicator based on the distance between the real values and the reference values (positive and negative) of each type of measure (landscape-based and species-based measures). We considered as a positive reference value a desirable state at landscape and at species level: for example, at landscape scale, we define as "positive reference value" the maximum value of conservation status or the minimum value for fragmentation status while, for species, the positive reference is represented by the highest value of richness of threatened species or by the lowest value of anthropogenic and alien species richness in collected data. In the same way, a “negative reference value” corresponded to the minimum conservation status or maximum fragmentation status value or, for species, to the lowest richness of threatened species or to the highest richness of anthropogenic and alien species richness in collected data. Landscape and species measures were considered equally important, covering complementary aspects; therefore, we gave equal weights, within and among them. This maintains the indicator neutral to discretionary choices but, at the same time, it offers the possibility of adaptation for other planning areas.

The Human Impact Indicator (HII) was calculated as:

$$HII = \frac{P_{landscape} - R_{landscape}}{P_{landscape} - N_{landscape}} + \frac{P_{species} - R_{species}}{P_{species} - N_{species}}$$

where P were the positive reference values for each type of measure (landscape stands for landscape-based measures; species stands for species-based measures); N were the negative reference values for landscape and species; R were the real values for each type of measure.

HII ranges between 0 and 2, where 0 implied a low level of human impact and 2 implied a high level of human impact. The index is conventionally set to 0 when P = N.

Based on the results of the correlation of species richness of each indicator group vs status of conservation and fragmentation, we calculated HII using the values of conservation status (landscape) and richness of threatened species (species); to test the flexibility of the index, we calculated HII using conservation and fragmentation status (landscape) vs richness of anthropogenic and alien species (species).

Results

Landscape level

The conservation status at MCC level showed quite a high value (ILC=0.63). However, a more detailed analysis showed a heterogeneous degree of naturalness and conservation status at the municipalities scale, ranging from 0.20 to 0.83.

As regards the landscape fragmentation, landscape-based metrics were synthesized in two principal components that accounted for 79% of the total variance: the first component (47% of the total variance) was negatively correlated with the conservation status (ILC and landscape composition) and the size and edge metrics (MPS and MPE). The second component (32% of the total variance) was positively correlated with the parameters describing the shape of polygons (AWMSI e AWMPFD).

The projection on the scatterplot of PCA of the three groups identified by hierarchical clustering, allowed the groupings to be distinguished according to the correlated factors (Fig. 2).

The first cluster brought together municipalities (Villa San Pietro, Sarroch, Maracalagonis, Sinnai, Uta, Pula and Assemini) showing high conservation status index; high average size of polygons (MPS) for the presence of large patches; the high average edges of polygons (MPE), typical of landscapes characterized by patches with complex shapes suggesting natural boundaries. The second cluster brought together municipalities (Sestu, Capoterra, Quartu Sant'Elena, Quartucciu, Monserrato, Elmas, Decimomannu and Cagliari) characterised by the low average size of polygons (MPS) due to the presence of small patches; low average edges of polygons (MPE), typical of landscapes characterized by patches with regular shapes, confirmed by the low values obtained for the shape index weighted on the average area (AWMSI); uniformity in the patch size (low values of PSCov). The third group included the municipalities of Selargius and Settimo San Pietro, which showed a low average size of the polygons (MPS) for the presence of small patches; the complex and irregular shape of the patches as suggested by the high values assumed by the form indexes (AWMSI, AWMPFD and MPAR) and a low conservation status.

Analysis at landscape level confirmed the presence of three different levels of fragmentation (Fig. 2): a low level of fragmentation (Cluster 1) for the municipalities characterized by patches of larger average size and more natural environment with a good level of nature conservation; a medium level of fragmentation (Cluster 2) defined by patches with uniformly smaller dimensions and more anthropized environments; a high level of fragmentation (Cluster 3) defined by patches of small dimensions, complex shape and a low conservation status (ILC). Conservation and fragmentation status maps showed the value of the status across municipalities (Fig. 3).

Species level

Overall, we observed more threatened species (50.0% of species, 38.2% of individuals) than any other group. However, widespread species (20.0% of species, 24.3% individuals), endemic species (15.0% of species, 22.4% of individuals), and anthropogenic and alien species (15.0% of species and 15.1% of individuals) were also well represented.

At the municipality level, the number of records ranges from nine to a maximum of 1110 individuals (Table 3). Total species richness per cell showed up similarly variation at the municipality scale, having a low value (1-6 species) for the municipalities in which very few data were present (e.g., municipalities of Decimomannu, Monserrato, Settimo San Pietro, Villa San Pietro), and a maximum value of 33 out of 40 selected species (for the municipalities of Maracalagonis and Sinnai) (Table 3). However, almost all groups showed a low species richness, with an average number of 21 individuals per species and 5 species per cell (Fig. 4).

The relationship between landscape-based and species-based measures showed different results for the selected indicator groups. Richness of threatened species was weakly positively correlated with conservation status ($r = 0.13$, $p < 0.05$) and negatively correlated with fragmentation status ($r = -0.11$, $p > 0.05$). An opposite trend was observed for richness of endemic species, negatively correlated with conservation status ($r = -0.11$, $p > 0.05$) and positively correlated with fragmentation status ($r = 0.22$, $p < 0,01$). Richness of anthropogenic and alien species resulted correlated negatively with conservation status ($r = -0.26$, $p < 0.01$) and positively with fragmentation status ($r = 0.26$, $p < 0.001$). Widespread species resulted not significantly related neither to conservation status ($r = -0.021$, $p > 0.05$) nor to fragmentation status ($r = 0.057$, $p > 0.05$).

The Human Impact Indicator (HII)

Considering the significant correlation, we tested HII based on the distance between the real values and the reference values (positive and negative) at landscape and species level: conservation status (ILC) and richness of threatened species; conservation and fragmentation status and richness of anthropogenic and alien species. Cell-by-cell indicator score assigned to administrative boundaries reflected a different level of human impact among the municipalities (Fig. 5 and Fig. 6). According to the HII, the municipality of Sinnai defined the best local value towards which other local authorities should tend to improve their sustainable planning for the conservation of threatened species diversity and to reduce the level of human impact, as confirmed by the low value of the index when evaluating the impact associated to alien and anthropogenic species (Fig. 6). On the opposite, the highest average HII score, obtained for Monserrato, Selargius, Elmas and Settimo municipalities, resulted from the highest fragmentation status and richness of anthropogenic and alien species (Fig. 6), confirming the results obtained with clustering and PCA (Fig. 3).

Discussions

In similar studies conducted in other Mediterranean regions, the conservation status of the landscape was lower ($ILC = 0.37 \pm 0.18$), in contrast to higher ILC value observed in the Temperate region ($ILC= 0.69 \pm 0.18$, Blasi et al., 2008). The difference Mediterranean vs Temperate can be related to the general reduction of natural and seminatural areas observed in the Mediterranean (Bajocco et al. 2012; Zoppi et al. 2015) and the increasing forest cover in Temperate regions (Falcucci et al. 2007). The rather high value of the conservation status observed in the Metropolitan City of Cagliari ($ILC = 0.63$) was not in line with those studies, and it is justified by the presence of several natural and preserved sites. Nevertheless, this high value of conservation status is not evenly distributed within MCC, but at the municipality scale, we observed a gradient of land-use intensification from the peripheral areas to the Central part of the Metropolitan City of Cagliari (Elmas, Monserrato, Selargius); this gradient was confirmed by the fragmentation status assessment. In fact, cluster analysis and PCA confirmed a clear difference among local municipalities based on their composition and configuration parameters describing the conservation status, size, edge and shape of polygons. Results can be useful especially to the municipalities characterized by an intermediate level between two extremes conditions: focusing on the municipalities showing a lower fragmentation status, we can suggest a local and achievable fragmentation status objective (low fragmentation reference) to those in need of restoration or protection-oriented strategies for the semi-natural and natural environment (intermediate fragmentation). To manage and sustain both human needs and conservation urgencies, land planners need to identify a reference point, especially if the relationship landscape heterogeneity–biodiversity is unknown (Seiferling et al. 2014); to overcome this limitation, the measured relationship between species richness and fragmentation status revealed that some groups could be suitable for this task.

In particular, species richness of threatened species resulted correlated with the conservation status, while anthropogenic and alien species resulted correlated to both type of measures at landscape level. Increasing the landscape conservation status, the number of threatened species increases and the number of anthropogenic and alien species decreases; coherently, in the most fragmented landscapes anthropogenic and alien species increased. On the contrary, the richness of threatened species did not appear to be influenced by any variation of the fragmentation status. Coherently, in many studies, the effect of spatial configuration on saproxylic beetles remains unclear, because dispersal sources are poorly identified (Seibold et al. 2017). Furthermore, the effect of isolation could depend on inter-patch distances or to a different perception of species, which are difficult to unravel in correlational studies (Fahrig 2013). It is also significant to note that some species included in the “threatened species group”, defined according to the Red List of saproxylic beetles, show a generalist autoecology. For example, the saproxylic beetle *Capnodis tenebrionis* (Linnaeus, 1761) (Coleoptera, Buprestidae) is a generalist and primary guest on various trees’ roots and trunks, that can be found also in sub-urban or agricultural environment (see Curletti et al. 2003) while *Probaticus ebeninus ebeninus* (Villa, 1838) (Coleoptera, Tenebrionidae) shows a distribution pattern which comprises urban and sub-urban areas (see Aliquò et al. 2007). Some species of Cetonids (*Protaetia* spp.), included in the Red List, are quite common in the edge of woodland areas or in forested zones, but they also colonize urban or sub-urban green areas. In the same way, the unexpected response of endemic species could be explained considering the autoecology of these species that can adapt to fragmented landscapes, although they are characterized by a restricted distribution. As it was expected, widespread species showed mixed responses, being able to exploit a wide range of resources (Bazzato et al. 2019).

For land managers, a useful way to explore the main attributes of landscape is to examine the richness and the distribution pattern of species, or composition changes with increasing habitat fragmentation (Seiferling et al. 2014). This issue underlines the importance of the synthetic indicators which can compare between ‘‘fragmented’’ and ‘‘reference status" (Parrot 2010); those comparisons are generally required by decision-makers (Acosta-Alba and Van der Werf 2011; Hersperger et al. 2017). Human Impact Indicator (HII) sought to fill gaps evidenced by La Rosa et al. (2016) in the definition of quantitative measures of references values, considering as reference values the extremes, positive and negative, observed in the whole planning area at landscape and species level. Underpinning such approach is the need to understand the trade-offs, in terms of environmental impacts, among different local authorities to propose a unified planning of the functional urban area. The HII, based on the distance from “reference” values of landscape-based and species-based measures, could help to interpret and characterize the degree of impact in relation to this reference, but it could also be used to monitor and characterize changes in time. Landscape and species-based measures combined into one surrogate measure (i.e., a HII reference value at MCC level), could help managers to define an acceptable and feasible threshold, locally defined, to which local authorities should tend to mitigate the negative effects produced by fragmentation and land-use change at landscape and species level. Decision-makers or planners could take their decision considering the multiple levels of organization (landscape and species) (Noss 1990; Poiani et al. 2000; Carignan and Villard 2002; Siddig et al. 2016), proposing achievable targets for a more sustainable planning (Acosta-Alba and Van der Werf 2011), and addressing additional efforts and corrective actions to achieve a long-term conservation of habitats and species (Haddad et al. 2015; Wilson et al. 2016).

Conclusion

To fulfill sustainable management and nature-based solutions’ objectives, planners and decision-makers must consider conservation of natural resources, and human wants and needs. The strengthened collaboration and joint planning between ecologists and managers can support this task: HII turned out to be easily measurable for a specific location and able to capture the ecosystem complexity, considering multiple levels of organization. This confirms the advantages to follow a flexible and multidisciplinary approach which is able to weight the variables composing the index (landscape/species) according to local needs.

Individual tables

Table 1 List of landscape-level metrics calculated for each municipality (see McGarigal et al., 2002 for a complete description of each metric)

Acronym	Name of metric	Type of metric
NumP	Patch number	Dimension
MPS	Mean patch size	Dimension
PSCoV	Patch Size Coefficient of Variance	Dimension
TE	Total edge	Edge
MPE	Mean Patch Edge	Edge
MPAR	Mean Perimeter-area ratio	Shape
AWMSI	Area-weighted mean shape index	Shape
AWMPFD	Area-weighted mean patch fractal dimension	Shape

Table 2 List of collected beetles species divided into four groups based on the specific autoecology, chorology, importance for conservation and management

Family	Species	Author	Group of species
Carabidae	*Licinus* (*Licinus*) *punctatulus*	(Fabricius, 1792)	Anthropogenic or alien species
Cerambycidae	*Phoracantha recurva*	Newmann, 1840	Anthropogenic or alien species
Cerambycidae	*Phoracantha semipunctata*	(Fabricius, 1775)	Anthropogenic or alien species
Tenebrionidae	*Akis trilineata barbara*	Solier, 1837	Anthropogenic or alien species
Tenebrionidae	*Blaps gigas*	(Linné, 1767)	Anthropogenic or alien species
Tenebrionidae	*Scaurus atratus*	Fabricius, 1775	Anthropogenic or alien species
Carabidae	*Cicindela* (*Cicindela*) *campestris nigrita*	Dejean, 1825	Endemic species
Carabidae	*Percus* (*Percus*) *strictus oberleitneri*	(Dejean, 1831)	Endemic species
Carabidae	*Percus* (*Percus*) *strictus ellipticus*	(Porta, 1901)	Endemic species
Melolonthidae	*Hoplia* (*Hoplia*) *pubicollis*	Küster, 1849	Endemic species
Tenebrionidae	*Blaps nitens mercatii*	Canzoneri, 1969	Endemic species
Tenebrionidae	*Tentyria grossa sardiniensis*	Ardoin, 1973	Endemic species
Bostrichidae	*Apate monachus*	Fabricius, 1775	Threatened species
Buprestidae	*Capnodis tenebrionis*	(Linnaeus, 1761)	Threatened species
Buprestidae	*Latipalpis* (*Latipalpis*) *plana plana*	(A.G.Olivier, 1790)	Threatened species
Cerambycidae	*Aegosoma scabricorne*	(Scopoli, 1763)	Threatened species
Cerambycidae	*Arhopalus ferus*	(Mulsant, 1839)	Threatened species
Cerambycidae	*Cerambyx cerdo cerdo*	Linnaeus, 1758	Threatened species
Cerambycidae	*Hesperophanes sericeus*	(Fabricius, 1787)	Threatened species

Cerambycidae	*Stictoleptura cordigera*	(Fuessly, 1775)	Threatened species
Cetoniidae	*Cetonia carthami carthami*	Gory & Percheron, 1833	Threatened species
Cetoniidae	*Protaetia* (*Netocia*) *sardea*	(Gory & Percheron, 1833)	Threatened species
Cetoniidae	*Protaetia* (*Potosia*) *cuprea cuprea*	(Fabricius, 1775)	Threatened species
Cetoniidae	*Protaetia* (*Potosia*) *opaca*	(Fabricius, 1787)	Threatened species
Dynastidae	*Oryctes nasicornis corniculatus*	Villa & Villa, 1833	Threatened species
Elateridae	*Ampedus melonii*	Platia, 2011	Threatened species
Elateridae	*Lacon punctatus*	(Herbst, 1779)	Threatened species
Lucanidae	*Dorcus musimon*	Gené, 1836	Threatened species
Scarabeidae	*Trichius gallicus zonatus*	Germar, 1831	Threatened species
Tenebrionidae	*Allardius sardiniensis*	Allard, 1877	Threatened species
Tenebrionidae	*Probaticus ebeninus cassolai*	(Ardoin, 1973)	Threatened species
Tenebrionidae	*Probaticus ebeninus ebeninus*	(Villa, 1838)	Threatened species
Carabidae	*Calosoma* (*Calosoma*) *sycophanta*	(Linnaeus, 1758)	Widespread species
Carabidae	*Carabus* (*Macrothorax*) *morbillosus constantinus*	Lapouge, 1899	Widespread species
Carabidae	*Chlaenius* (*Chlaenius*) *velutinus auricollis*	Gené, 1839	Widespread species
Cetoniidae	*Protaetia* (*Netocia*) *morio morio*	(Fabricius 1781)	Widespread species
Dynastidae	*Pentodon algerinus algerinus*	(Fuesslin, 1778)	Widespread species
Dynastidae	*Phyllognathus excavatus*	(Forster, 1771)	Widespread species
Histeridae	*Pactolinus major*	(Linnaeus, 1767)	Widespread species
Hydrophilidae	*Hydrophilus pistaceus*	(Laporte, 1840)	Widespread species

Table 3 Species richness and total number of records by municipality

Municipality	**Species richness**	**Number of records**
Assemini	22	81
Cagliari	28	579
Capoterra	18	71
Decimomannu	1	3
Elmas	10	21
Maracalagonis	33	487
Monserrato	4	24
Pula	25	95
Quartu Sant'Elena	32	403
Quartucciu	15	95
Sarroch	14	41
Selargius	4	10
Sestu	14	28
Settimo San Pietro	1	2
Sinnai	33	1110
Uta	20	146
Villa San Pietro	6	9

List of figures

Fig. 1 Flow chart of the methodological framework

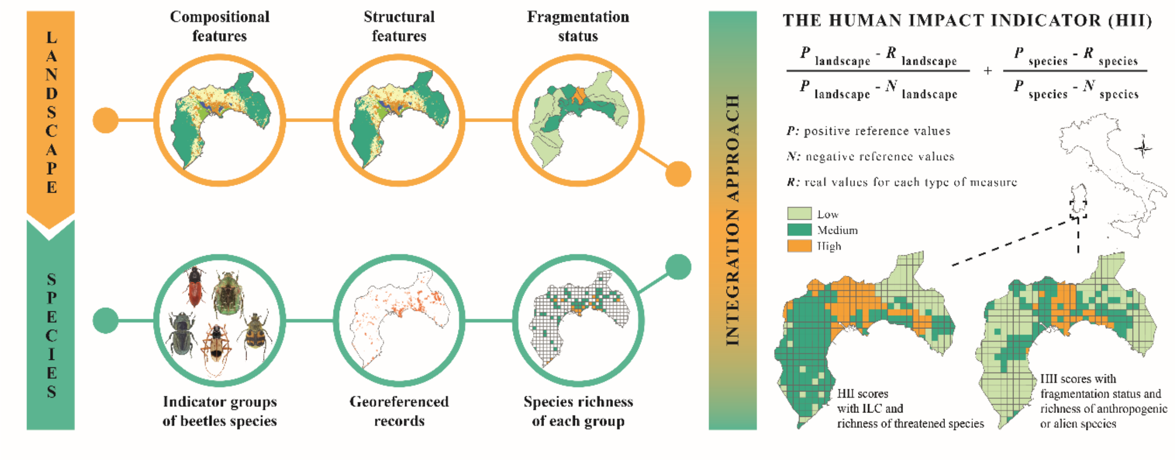


Fig. 2 On the left: dendrogram of the agglomerative hierarchical clustering of all landscape-based measures (ILC and metrics at landscape level) computed for 17 municipalities. On the right: clusters based on their compositional and configurational features describing the conservation status, size, edge and shape of polygons, projected on the PCA scatterplot

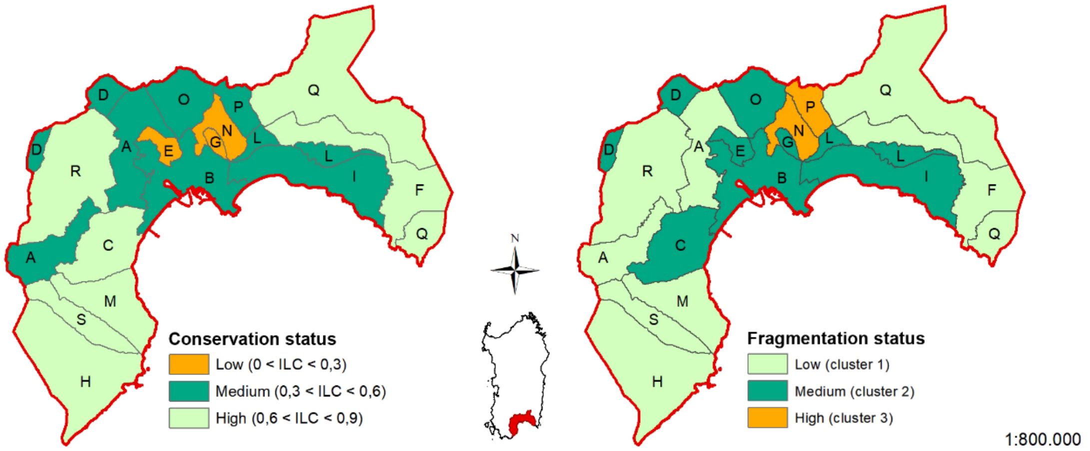


Fig. 3 On the left: heterogeneous degree of naturalness and conservation status on the municipalities, ranging from 0,20 to 0,83. On the right: fragmentation status based on compositional and configurational features of each municipality: C, Capoterra; D, Decimomannu; A, Assemini; B, Cagliari; I, Quartu Sant'Elena; S, Villa San Pietro; L, Quartucciu; G, Monserrato; M, Sarroch; N, Selargius; Q, Sinnai; O, Sestu; P, Settimo San Pietro; F, Maracalagonis; H, Pula; E, Elmas; R, Uta

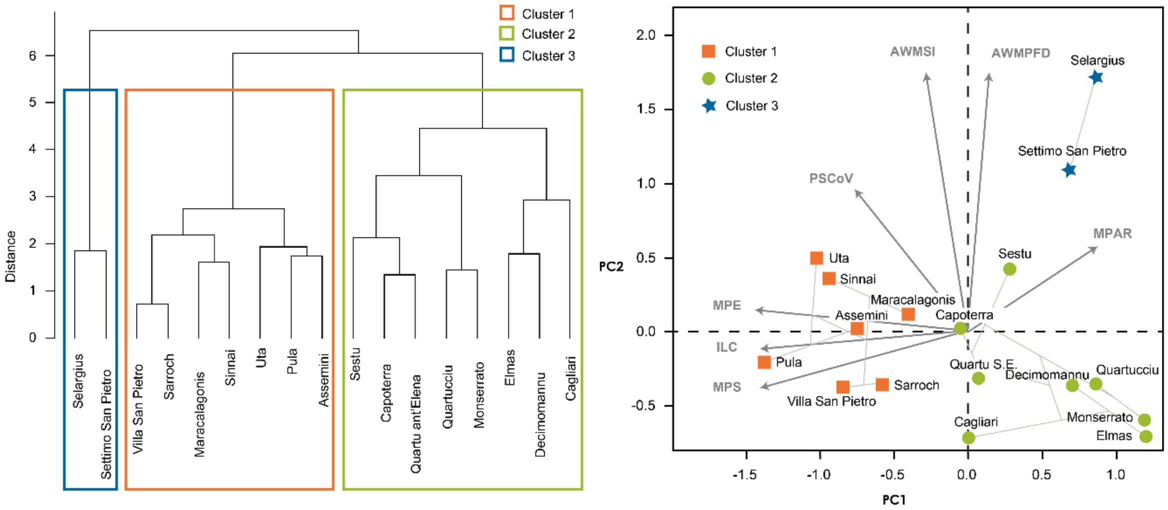


Fig 4 Species richness classes of each group of beetles: threatened species, endemic species, widespread species, and anthropogenic and alien species

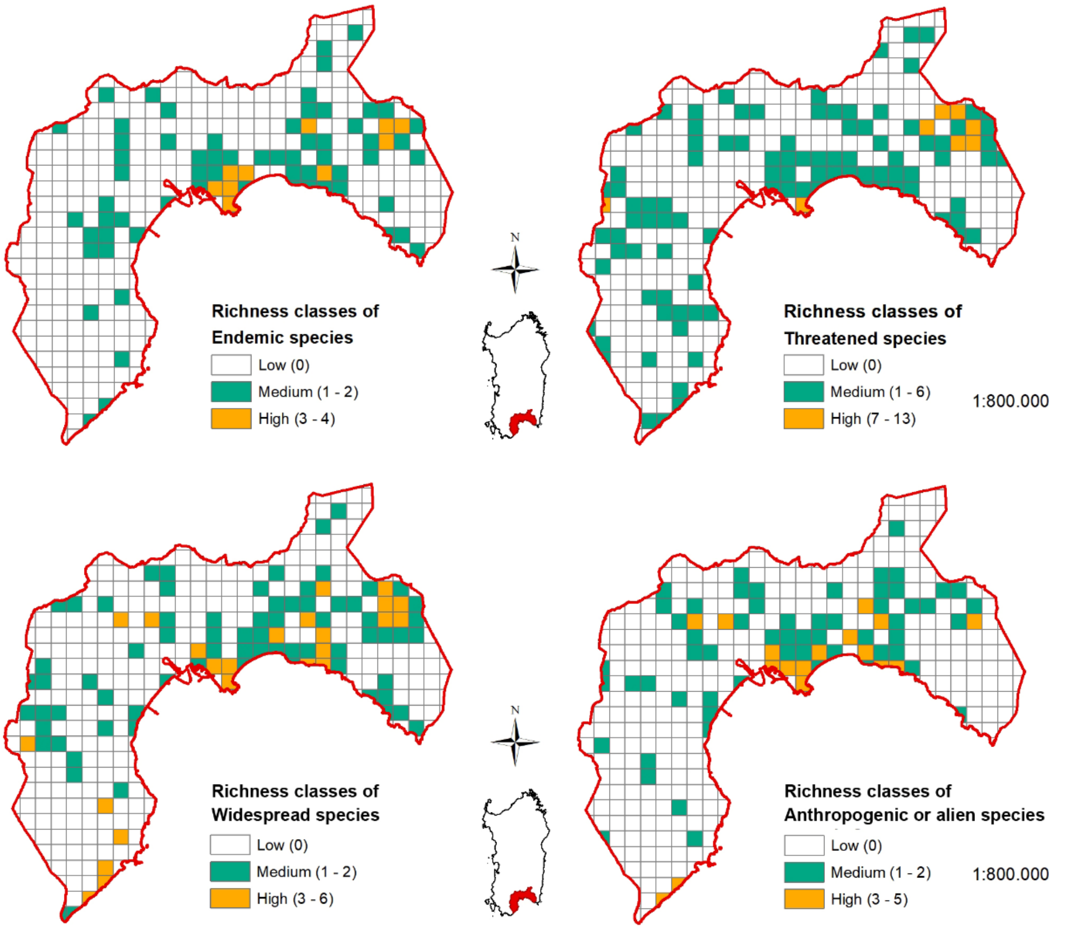


Fig. 5 Spatial distribution of cell-by-cell indicator score (Human Impact Index, HII) on the entire planning area using different reference values

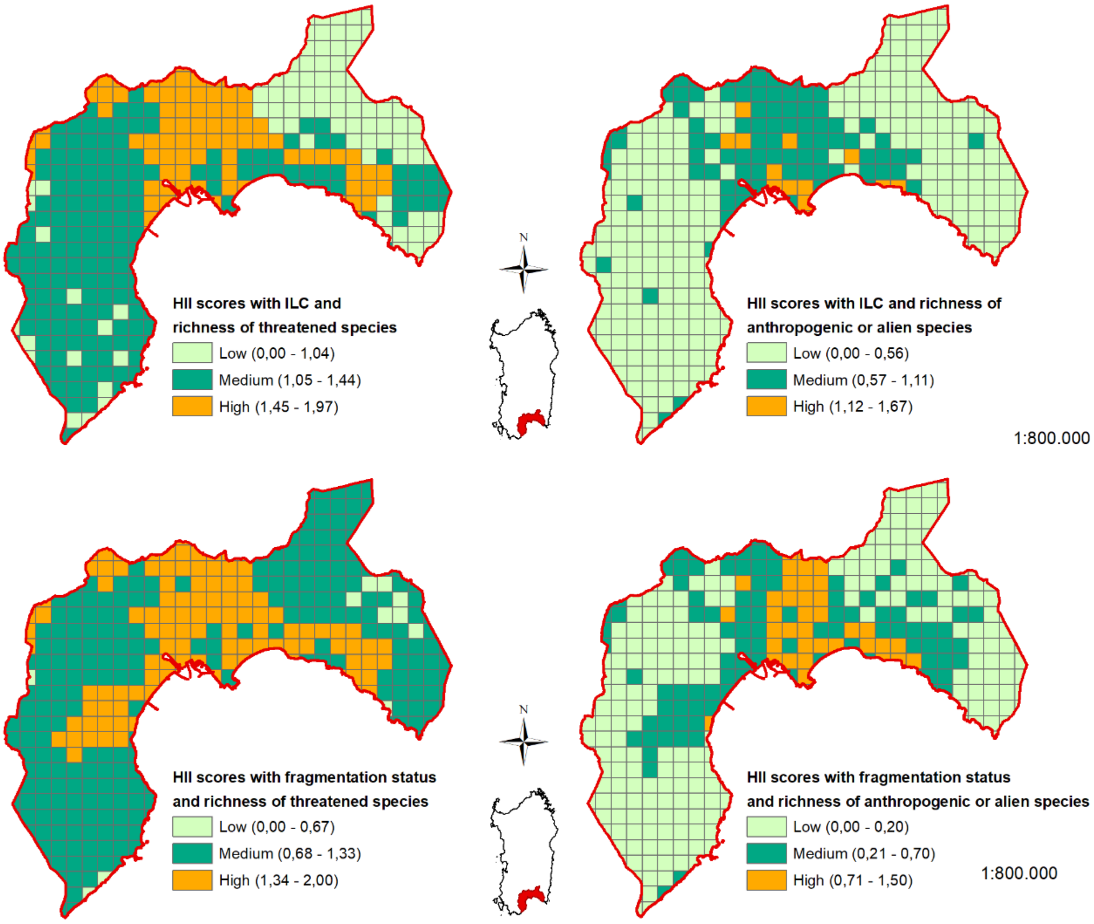
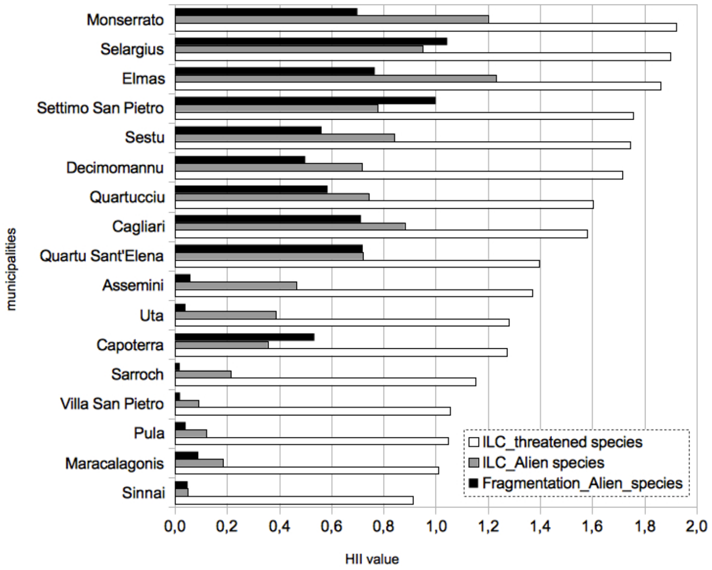


Fig. 6 The level of Human Impact (HII) in the municipalities using as reference value the richness of threatened species was always higher if compared to the one using the richness of anthropogenic and alien species, either using conservation value or fragmentation



Acknowledgments

Special thanks are extended to private collectors (Alamanni F., Ancona C., Atzori M.G., Cabitta N., Lecis A., Leo P., Fancello L., Fois F., Rattu R.) for supplying the beetle data which increased the number of records available on the personal collection of the authors (E.B., D.C.) and public collections (University of Cagliari and University of Sassari).

Author contributions

Material preparation and data collection were carried out by Erika Bazzato and Davide Cillo. Erika Bazzato performed the analysis and wrote the first draft of the manuscript. All authors contributed to the study conception and design, commented on previous versions of the manuscript, read and approved the final manuscript.

Subchapter 5.2: strategic plan of Metropolitan City of Cagliari

Environmental dimension into strategic planning. The case of Metropolitan city of Cagliari

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Abstract

Global changes in the Anthropocene are unprecedented in history. They are closely linked to the use of the soil, the sea and the exploitation of natural resources and in turn determine important changes in the values and socio-cultural behavior of entire populations. In this context, the focus on the environmental dimension is the main way to govern the city and territory. In this sense, the environmental assets through the criterion of participation in decision-making processes, the identification and assessment of reasonable plan / program alternatives through the construction of forecast scenarios related to the evolution of the state of the environment constitutes the spatial planning paradigm, from the municipal level implementation strategy and the metropolitan level strategic one. Although in fact all Italian metropolitan cities are oriented towards adopting strategic and sustainable development models, capable of fighting the consumption of soil and natural resources in general, these have not always correspondence in an approach that specific environmental assessments part of the plan process and therefore functional for future governance choices. In this context, the objective of this work is to describe the case of the metropolitan city of Cagliari highlighting how the environmental dynamic and assets should be considered into its (actually in defining phase) strategic plan.

Keywords: Environment, Strategic planning, Metropolitan cities, Climate change

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1. A new season for the Strategic Planning in Italy

The "strategic" adjective has become commonly used in the language of territorial planning, even if, as often happens, it is not always used with the appropriate level of precision and / or awareness and is now applied in a variety of experiences in an easy way [1].

The origin of the term must be sought in military science and is often linked to the term tactics; a strategy is a long-term action plan used to set up and coordinate actions aimed at achieving a predetermined goal or objective, while by tactics we mean a targeted action aimed at the short term, at a specific and specific episode, a segment of that wider goal which is the field of strategy; in short, the strategy is war, the tactic is the single battle. To win a war (strategy) you can also order a retreat or lose a battle (tactic).

Strategic planning then entered predominantly within the private sector to define the competitive strategies of companies in the markets, aimed at achieving precise business objectives through short-term measures and actions. Strategic business planning is now a consolidated practice and is a basic technique taught in business administration and business administration schools and in recent years it has gradually spread also in the non-profit sector and in the public sector.

In many European cities, between the 80s and 90s, strategic planning was included among the tools for territorial and urban planning to experiment with new methods and procedures that would go beyond traditional urban planning tools. A little later, with the new millennium, the territorial strategic planning tools also made their appearance in Italy.

Starting from the early 2000s, in fact, we witness the first Italian experiences of Strategic Planning, both in the urban / administrative sphere and in the disciplinary and transdisciplinary scientific sphere [2]. In those years, the Italian network of strategic cities (which included the pioneering cities of Turin, Florence, Pesaro, Trento, Piacenza and Verona, was quickly established, which was later joined by Venice, Perugia, La Spezia, Naples and the Province of Trento) with the aim of exchanging experiences and good practices, examining the main unresolved political and organizational issues and connecting with the most important experiences realized in the European panorama.

Apart from some pioneering experience, it was practice that imposed a new and relevant reflective approach on theory, especially in the urban field, which viewed the new tool with skepticism, tending to give it a minimalist, pejorative and misleading interpretation. In response to the traditional "plan crisis", strategic planning provides a rational and viable response, allowing to get out of the contrast between cognitive limits and implementation rigidities of regulatory-totalising planning and irresponsibility in terms of interest collective of purely derogatory practices.

However, if at the beginning the strategic territorial planning had a mere voluntaristic nature, it is only recently that in Italy it has become mandatory, even if only for metropolitan cities with the Law 56/2014 (Delrio Law). This Law defined also the metropolitan cities as a new governance level between regions and municipalities replacing, de facto, the Province level. By the way not all the Province has been replaced by metropolitan cities but only 14[1], and 13 of them (with the exception of Cagliari) are constituted by the same municipalities of the old Province.

2. Resilience, sustainability and smartness as the fulcrum of strategic planning action

In defining their strategic plans, the Italian metropolitan cities have stepped up and started setting their own agendas on the base on sustainability, resilience or smartness concepts focusing in different ways on them.

It's to note that in the last years these concepts have been coupled by researchers and institutions generating crossing paradigms, i.e. incorporating sustainability in smart city approaches for developing a more complex smart sustainable urban model.

The increasing awareness about environmental and sustainability issues related to urban growth and technological transformation is at the basis of the Smart Sustainable Cities concept [3]. The cities which has to face climate change as well as other challenges as concentration of population within an urban area, have become to use this concept widely since mid-2010s [4, 5]. With smart sustainable city, it is described a city "that is supported by a pervasive presence and massive use of advanced ICT, which, in connection with various urban domains and systems and how these intricately interrelate, enables cities to become more sustainable and to provide citizens with a better quality of life" [5]. The new technology, based on the Internet of Things (IoT) [6], allows citizens to be always connected through several devices. The real-time data may provide the opportunity of real-time feedback which may support real-time citizens' decisions in light of sustainable choices. The smart sustainable city allows decoupling high quality of life and economic growth from resource consumption and environmental impact [7].

Moreover, sustainability has been closely associated with the concept of resilience [8], since this last term "is often used to describe characteristic features of a system that are related to sustainability" [9].

[10] distinguishing among three aspects, economic, social and environmental, underline how these have resulted in the development of Sustainable Development Goals [11]. These

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[1] The Italian metropolitan cities are: Bari, Bologna, Catania, Cagliari, Firenze, Genova, Messina, Milano, Napoli, Palermo, Reggio di Calabria, Roma, Torino, Venezia.

goals allow both developing and developed Nations to reach sustainable development through a holistic approach. In particular, Sustainable Development Goal 11 vows to "Make cities and human settlements inclusive, safe, resilient and sustainable".

However, there are some authors [12] which disapprove this connection considering resilience as just a label. To be sustainable, cities and urban areas must be ready to face shocks and stresses which undoubtedly sooner or later will occur and will modify the state and the operating ways. In other words, they must be resilient [13]. Coherently with this approach, [14] propose the term of Biophilic City. The idea is that to make cities greener, more natural or, in their words, more biophilic, it is important to make them more resilient. This target can be reached in a direct way when investments in green infrastructure – i.e. a strategically planned network of natural and semi- natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services' in both rural and urban settings [15] – achieve resilience outcomes; or in an indirect way when actions or projects stimulate green and healthy behaviors that in turn serves to enhance the resilience of a city and of individuals.

Over the past decade and from a political point of view, urban resilience concept has emerged as one of the core principles of sustainable urban development widely acknowledged among various agreements such as the 2030 Agenda for Sustainable Development with its dedicated goal on cities—SDG 11, the Paris Agreement on climate change and the Sendai Framework for Disaster Risk Reduction.

It is worth to note that the urban resilience issue has also been associated with the smart city concept [16]. In fact, both concepts "are operationalized on the basis of similar or even the same systems, having similar trajectories of development and similar dilemmas to be solved" [17]. Moreover, these notions aim at improving sustainability and increase the quality of life, although follow different paths. Even if some international organizations or networks as well as a wide number of cities are fostering integrated projects and strategies for building up smarter and more resilient cities, a theoretical framework is still missing.

3. **The Metropolitan City of Cagliari (MCC)**

In this framework the Metropolitan city of Cagliari started the definition process of its strategic plan in the 2019. The process is articulated in three main phases:

- *Collecting*: data collection according to an objective approach (desk analysis) and a subjective one (participatory diagnosis);
- *Frameworking*: identification of the vision, formulation of the objectives and identification of the actions necessary to achieve the objectives;
- *Assessment and monitoring*: construction of indicators that allow to verify the implementation of the plan and the achievement of the objectives.

Actually, the first phase is concluded and the second one is ongoing. The results of the first phase highlight the central role of the environmental aspect in order to define the strategic pathway.

3.1 Environmental factors shaping the MCC

The areas occupied by the Metropolitan City of Cagliari is characterized by a high heterogeneity of the environmental mosaic, as a consequence of a wide variability of the physical, geomorphological, pedological-vegetational and historical-cultural elements.

Geology, geomorphology and hydrography

From a geological point of view, the territory of the Metropolitan City falls into three large geological areas:

- Campidano, an area whose geological structure consists of a series of geological formations from the Oligocene up to the recent Quaternary, such as terraced ancient floods, clay soils and recent soils of reclaimed marsh areas. From a geomorphological point of view, this area is characterized by landscape with “conoids”, typical in the western sectors of Campidano, or “plains” modified by agricultural activities

- Linas-Sulcis, consisting of three large homogeneous units: the valley area of Cixerri and the foothills. The sedimentation phases can be distinguished in: a pre-Pliocene sedimentation related to the opening of the great tectonic structure known as “Fossa Sarda” and a Plio-Quaternary sedimentation related to the opening of the Campidano graben; the volcanic reliefs; metamorphites and Paleozoic intrusions;

- Sulcis and gulf coasts, with a geo-structural conformation derived from a series of ancient orogenesis, extensional or compressional tectonic phases, volcanic activity and erosion and sedimentation phases over time. Characterized by hills and predominantly rounded forms, this area represents a small portion of the southern sector of the large Oligo-Miocene tectonic structure known as “Fossa Sarda”.

The urban area of Cagliari shows a hilly morphology connected to coastal morphological systems by a complex hydrography. The coastal system shows an articulated system of lagoons, ponds, marshes and salt marshes separated from the sea by coastal cords. Dynamics are strongly influenced by intense anthropization, which, by reducing its runoff, has strongly compromised the drainage network of the coastal areal, fundamental for the maintenance of the coastal ponds system.

From a hydrographic point of view, the Metropolitan City of Cagliari falls into the Flumendosa-Campidano-Cixerri sub-basin, which extends for almost 6000 km^2. The main rivers are “Flumini Mannu”, major tributary of the pond of Santa Gilla, and “Rio Cixerri”, once a tributary of the Flumini Mannu, then artificially separated near the S. Gilla lagoon. The intense urbanization has drastically reduced the recharge potential of the aquifers. The strong contamination also prevents their use for drinking purposes. Further damage derives from the excessive drainage activity near the coasts, which caused the rise of waters with a high salinity.

Climate and natural and semi-natural vegetation

The territory of the metropolitan area is characterised by the Mediterranean macrobioclimate, falling within a upper or lower mesomediterranean and thermo-mediterranean phytoclimatic belt [18]. Potential vegetation ranges from areas of scrublands and coastal scrub to areas of thermo-xerophilous woods and thermophilic holm oaks, especially in areas belonging to the districts of “Sette Fratelli” and “Monti del Sulcis”.

The vegetation is mainly composed of matorral of evergreen oaks, *Olea europaea* and *Pistacia lentiscus* formations, garrigues and silicic mesomediterranean scrub vegetation [19]. The territory has a high heterogeneity, with 40 different land use coverages [20]. More than half of the territory (52.3%) is represented by wooded areas and semi-natural environments; 31.9% of the area is occupied by agricultural areas, while 10.2% is occupied by artificial surfaces, mainly residential urban areas or industrial, commercial and infrastructural areas. The remaining territory is occupied by an important system of wetlands (3.3%) and water bodies (2.3%).

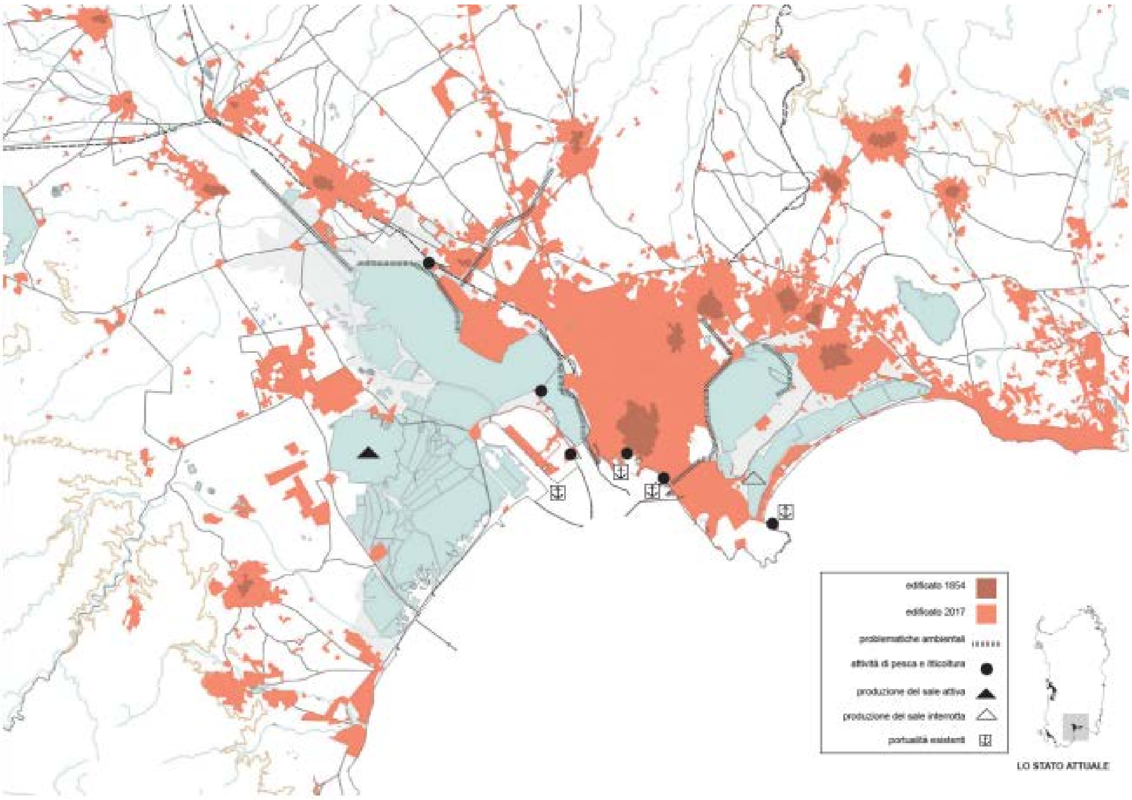
Coastal wetland ecosystem

Coastal wetlands are characterized by a delicate balance linked to the supply of solid materials from water courses: the deposition of sediments shapes the mouth of the rivers and constitute a determining agent in the drainage of the hydrographic-lagoon-sea basin. The functional role of hydraulic regulation of the territory depends on this dynamic equilibrium, that appears particularly relevant during the flood waves following the extreme meteoric events.

The resilience of coastal wetlands is therefore strictly connected to continental contributions which, due to morphological alterations and/or pollution of the areas further upstream of the river basin, may present poor water quality, or be unavailable during the summer period.

The consequences of these phenomena can have a negative impact on the ecology of these environments, specifically on the components of biodiversity of ecosystems (flora and fauna), but also on lagoon production, resulting in fluctuating returns, negatively influencing the ecosystem services provided by the wetlands. In particular, the wetlands of Cagliari are subject to a condition of "urban encirclement" or the tendency to weld of the urban centers of the MCC (Fig.1), which progressively leads to reducing the residual physical and functional corridors of communication between the wetlands and their feeding basins [21]. These vulnerabilities, added to the future instability caused by climate change, represent a great challenge for the management of coastal wetlands.

Fig. 1. Current state of the building in the wetland system of the city of Cagliari. Source: DICAAR-DISVA-CRENOS Interdepartmental research University of Cagliari



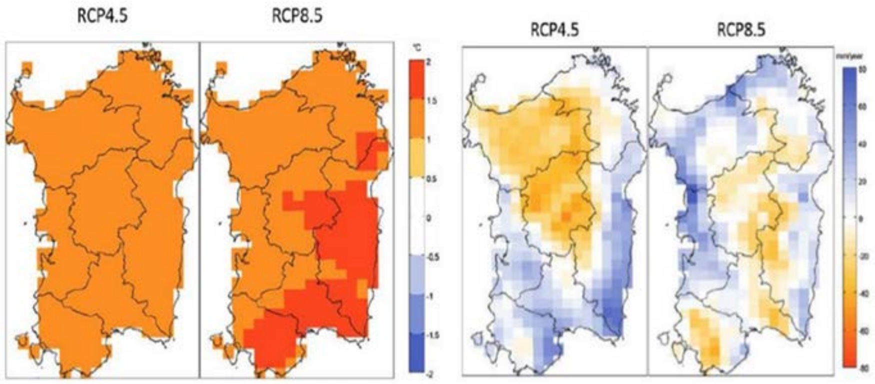
3.2 Threats and vulnerability

Climate and land use change

According to the National Climate Change Adaptation Strategy, in the coming decades the impacts resulting from climate change in the European Mediterranean region will be particularly negative and, combined with the effects of anthropogenic pressures on natural resources, it will make this area one of the most vulnerable in Europe. The future climate projections, included in the Regional Strategy of Adaptation to Climate Change of the Sardinia Region (reference period 1981-2010) and performed according to two scenarios, show for MCC an expected increase in the average temperature which varies between + 1 ° C and + 2 ° C in the period 2021-2050 (Fig. 2).

As regards rainfall, the projections show an increase in the annual values for the municipalities of the Metropolitan City in the first scenario, and a significant reduction in the second scenario, particularly marked in the municipalities of the eastern arch (Fig. 2). A general slight increase in the number of days with more intense rainfall is also expected, which suggests a future scenario in which rainfall could be concentrated in a limited number of intense events.

Fig. 2. Anomalies [° C] of the average temperature (left) and anomalies [mm / year] of the AP indicator for the period 2021-2050 (right). Source: Regional Strategy for Adaptation to Climate Change.



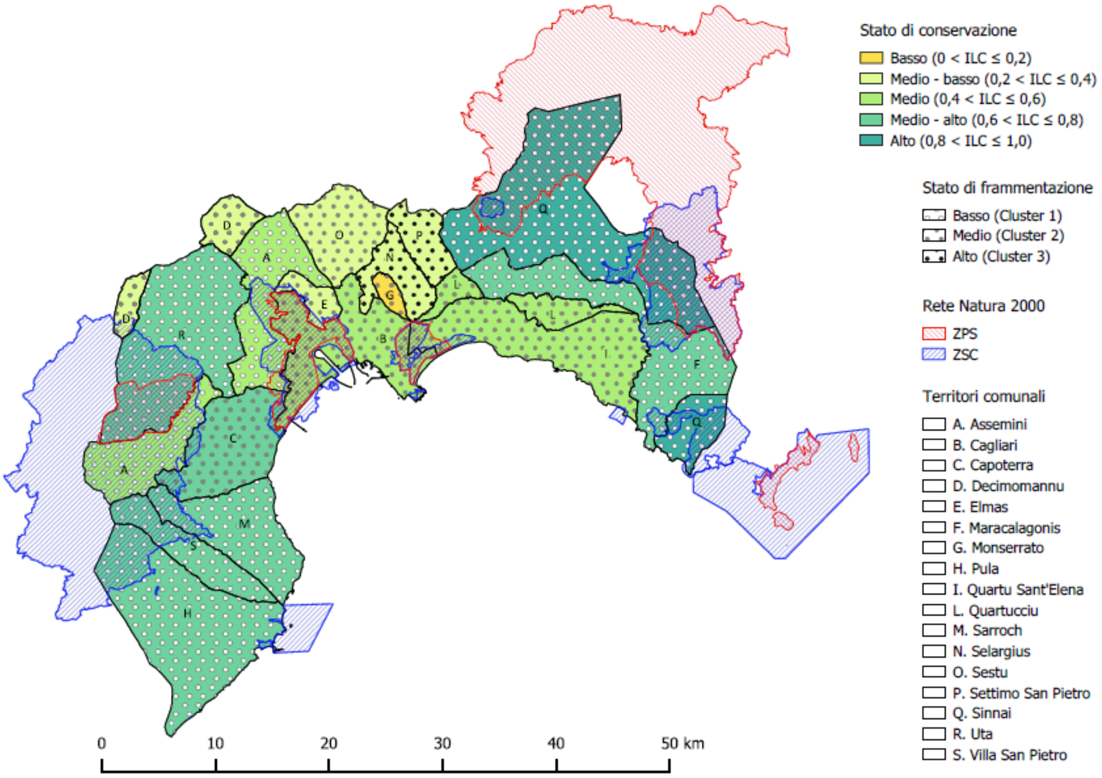
The main vulnerabilities related to the natural landscape are directly or indirectly related to anthropic activities, influencing hydrogeological processes and altering the ecological connections of the territory, through alterations and changes in land use which lead to habitat loss and environmental fragmentation, with a special intensification on coastal areas [22].

Fragmentation and conservation status of the landscape

Landscape level metrics and specific metrics show an overall medium-low degree of environmental fragmentation in the metropolitan area [23]; only 2 municipal territories, out of a total of 17, have a high degree of fragmentation (Fig. 4). The ILC Landscape Conservation Status Index [23, 24] calculated for the metropolitan area shows an overall conservation status with a high value (ILC = 0.63), except for only one municipality with a low conservation status ($0 < ILC \leq 0.2$) [23].

The analysis of the degree of fragmentation and the state of conservation of the administrative units of the MCC [23] allows to identify the municipal territories that present the most critical conditions and, at the same time, to highlight the territories that would need the implementation of strategies aimed at the protection and/or restoration of natural and semi-natural habitats. By framing the state of conservation and fragmentation of the municipal territories belonging to the MCC within the system of protected natural areas, it is possible to highlight and locate the inconsistencies existing between ecological emergencies and current distribution of the areas subject to conservation actions (Fig. 4).

Fig. 3. Protected areas and the Natura 2000 network in relation to the state of conservation and degree of fragmentation of the Metropolitan City of Cagliari.



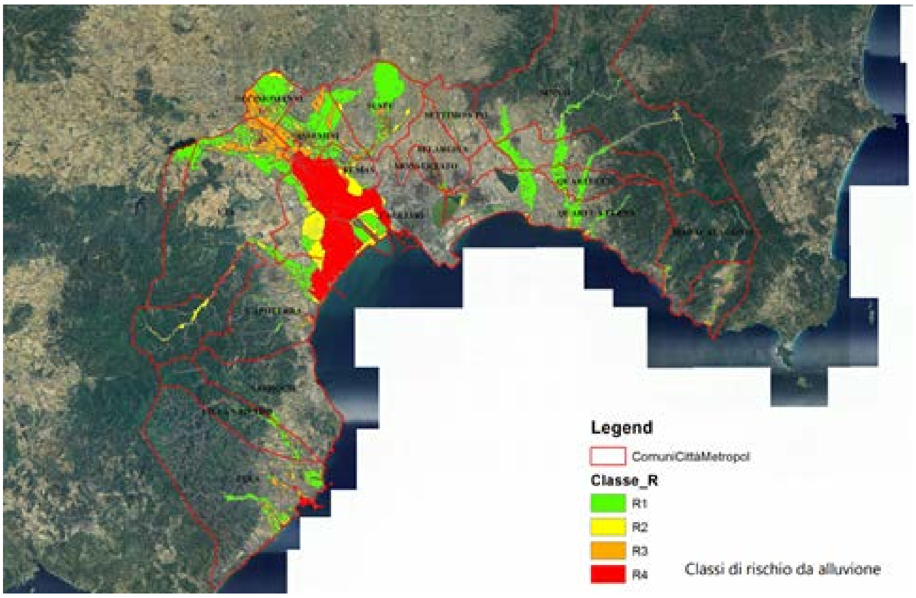
Hydrogeological risk

The concept of flood risk pursuant to art. 6 of Legislative Decree n.49 / 2010 is linked to the contextual analysis of Flood Hazard (H) and Potential Damage (D). Flood hazard is based on modeling referring to flood events, floods, linked to different return times. The potential damage is based on the analysis of the elements at risk present in the territory and their respective vulnerability.

The planning tools adopted or approved by the Sardinia Region (PAI, PSFF, studies pursuant to art. 8 paragraph 2 of the NA of the PAI) identify 3 hazard classes and increasing probability of occurrence and four classes of potential damage to people, to the socio-economic system and to non-monetizable assets. The Flood Risk map is the results of the overlaying of the Hydraulic Hazard map and the Potential Damage map. In accordance with the operational guidelines prepared by the Italian Ministry of the Environment (MATTM), the Flood Risk identifies four classes of increasing risk degree, ranging from R1 no risk (yellow) to R4 very high risk (red).

Only 4% of the entire territory is subject to very high hydrogeological risk (Fig. 3); this area is mainly concentrated in the territory of the municipalities of Elmas and Cagliari. A complete study of the network is missing: in this context, a lack of coordination between the different levels of constraint and study of the individual branches of the basin represent a major threat.

Fig. 4. Flood risk map of MCC



<u>Drought and wild fires</u>

The following indicators can be used to map the vulnerability to fire and drought risk as developed in the project Life "Master Adapt" (https://masteradapt.eu/?lang=en):

- <u>exposure indicators</u>, used to identify the main categories of activities and services exposed to fires and droughts, including the percentage of industrial and residential areas that could be mainly affected by fires and droughts;

- <u>sensitivity indicators</u>, which indicate how much the potential impact of climate change will be greater for each category of environmental typology involved;

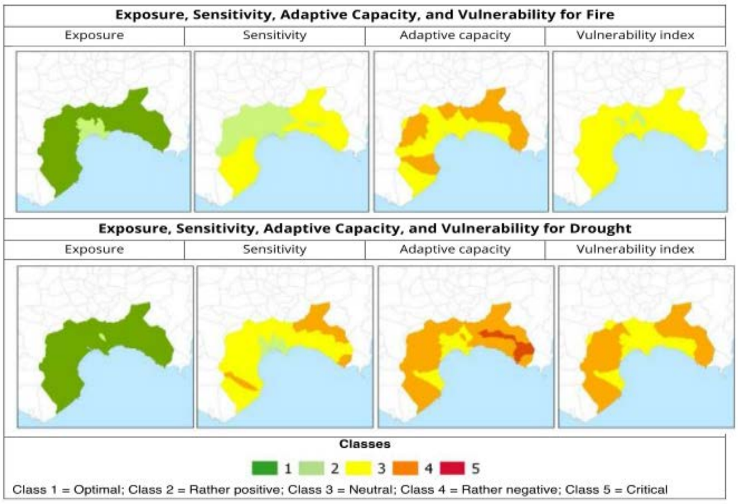
- <u>indicators of adaptive capacity</u>, calculated considering the level of education, the economic resources available per capita, the people employed in the agricultural and forestry sectors, the people employed to manage the risk of fires and the presence of fire risk plans, as well as projects o plans relating to adaptation to climate change for each municipality;

• global vulnerability indicators, derived from the aggregation of the normalized values of the global sensitivity index and the global adaptive capacity index.

As regards fires, there is a general low level of sensitivity in almost all the metropolitan area, because of the presence of vast irrigated lands and green urban areas. The adaptation capacity for the area is medium, therefore the global vulnerability index is classified at medium level (class 3).

As regards the drought, an average sensitivity to drought is reported for the area, apart from the municipalities of Sinnai and Villa San Pietro with a medium-high sensitivity level. The municipality of Cagliari reported a sensitivity class of 2. The global vulnerability index is therefore classified at a medium and medium-high level with classes 3 and 4.

Fig. 5. Global exposure, sensitivity, adaptive capacity and vulnerability for fires and droughts in the MCC. Source: Life Project "Master Adapt"



Climate projections indicate a marked future heating for the MCC, with an increase in the minimum, maximum and average temperature (from + 1.3 ° C to - 3.6 ° C, depending on the CPR scenario and the future period considered). It is also expected a sharp increase in hot extremes (summer days, consecutive dry days, etc.) and a decrease in cold extremes. A slight general reduction in total rainfall is also expected, which could exacerbate fires and drought.

Heat waves

To map the vulnerability related to heat wave risk as developed in the project Life "Master Adapt" (https://masteradapt.eu/?lang=en) the following indicators can be used:

• exposure indicators, considering the population density, which determines the “Urban Heat Island” effect;

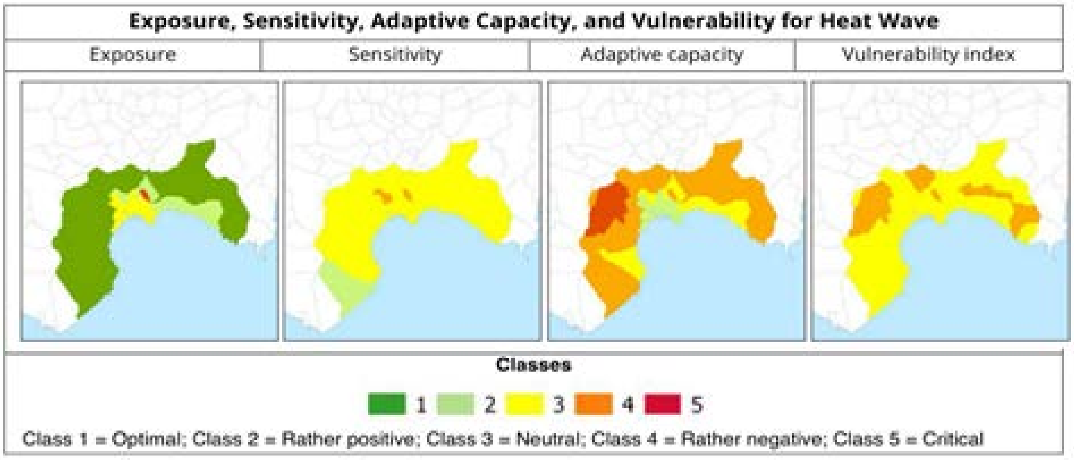
- sensitivity indicators, considering heat related diseases and deaths, two categories in direct relation with the intensity peaks of the urban heat islands (UHI), therefore representative for studying the sensitivity of the heat waves;

- indicators of adaptive capacity to cope with heat waves, considering the level of education, the economic resources available per capita, the unemployment percentage and the number of medical points and projects related to climate change.

- global vulnerability indicators, derived from the aggregation of the normalized values of the global sensitivity index and the global adaptive capacity index.

The global heat wave vulnerability index, however, reports a higher class for the hinterland of Cagliari. Climate projections indicate an increase in extreme temperatures, especially on tropical nights (21 - 61 days) and on summer days (22 - 53 days). This could lead to a greater vulnerability for heat waves, in particular for the municipality of Cagliari.

Fig. 6. Global exposure, sensitivity, adaptive capacity and vulnerability for heat waves in the MCC. Source: Life Project "Master Adapt"



3.3 The main environmental assets to build the Strategic plan on

The strong characterization of the Metropolitan City of Cagliari highlights how the environment should be at the center of the targets of the strategic plan. In particular, there are some assets to be considered as main reference: coastal wetlands, protected areas and hills system.

Coastal wetlands

The wetlands of Cagliari constitute a single environmental macrosystem consisting on the western side of the Santa Gilla lagoon, Macchiareddu salt pans and Capoterra pond, which, together with the Molentargius-Saline system, located symmetrically east of the city, complete the belt of wetlands of the city. In the overall view of the Metropolitan City, the pond of Nora or Sant’Efisio, in the municipality of Pula, is also included.

The wetland system of Cagliari is one of the most important wetlands in Italy and in the Mediterranean basin: it represents a highly complex system, affected by strong alterations and multifaceted environmental dynamics given by the overlaps between natural habitats, production systems, infrastructure and ecological systems.

The wetlands of the MCC, with the traditional fishing, salt cultivation, combined with activities for recreational, tourist and cultural purposes (visits to the park and salt marshes, birdwatching, fish tourism, sport fishing etc...), represent a social, cultural and economic wealth, closely linked to the MCC natural capital. This natural capital (e.g., the landscape and the biodiversity therein) structured in communities rich in highly specialized species, with functions related to resilience and resistance to drastic environmental variations, represent a reservoir for ecosystem goods and services [25].

The coastal wetlands system of Cagliari (Santa Gilla lagoon, Macchiareddu salt marshes and Capoterra pond) are characterized by different levels of exploitation for production, settlement and infrastructure. In addition to traditional fishing and water-culture activities, salt extraction and agricultural activities, industrial structures and plants coexist with some areas of urban functions (service areas, infrastructures, purification and green public areas). In the same area we find concentrated the strategic large infrastructures of Porto Canale, the International Airport, railway and road network and the industrial area of Macchiareddu and Elmas, for which the lagoon represents both a point connection and separation (Fig. 1).

Terrestrial protected areas and green infrastructures

The MCC territory hosts several types of protected areas s.l., such as permanent oases of fauna protection and capture, IPA areas (Important Plant Areas), IBA areas (Important Bird Areas), Ramsar areas ("Stagno di Santa Gilla" and "Stagno di Molentargius”), Regional Natural Parks (Molentargius-Saline Regional Nature Park and Gutturu Mannu Regional Nature Park) and Natura 2000 Network sites. The latter is represented by 12 Special Conservation Zones (SACs) and 4 Special Protection Zones (SPAs), falling totally or at least in part within the MCC. The area belonging to the Natura 2000 network amounts approximately to 52,000 ha of SAC areas, of which more than 31,000 ha fall entirely within the territory of the MCC and approximately 49,000 ha of SPA areas, of which about 18,000 ha fall entirely within the MCC.

Protected areas can be considered as core areas of the green infrastructure, showing a great potential to create a network for the protection of the natural capital of MCC [26, 27].

The calcareous hills of Cagliari

According to some scholars, the name of Cagliari (*Krly*) derives from a "*particular geophysical condition: the imposing masses of bare and craggy limestone rocks of the current castle and of Mount S. Elia, bleached ... are the most characteristic and suggestive,* Emidio De Felice”. Cagliari, therefore, is simply the place of the white hills.

These biotopes therefore represent a characterizing and identifying element of the territory, but they also have a naturalistic and environmental relevance. In fact, Capo S.Elia Promontory and the calcareous hills rise in the southern part of the Campidano plain, the only limestones emergencies of south-eastern Sardinia, places where, in some cases, high levels of biodiversity are preserved. In the last 50 years, the development of the city has profoundly changed these sites and in the near future a further alteration of the environments is expected with a strong compromise of naturalistic values.

The promontory of Capo Sant'Elia consists of reliefs aligned according to the SE-NO direction. Cala Mosca divides the promontory into two parts, one higher in the east (136 m) and one in the west, called Sant'Ignazio (94 m). The promontory of Sant’Elia to the south is joined to the other hills by flood lands of the Pleistocene. The hills of Cagliari are emergencies set on the terminal part of the Campidano plain, partly eroded by the quarries and compromised by the building development, they are distributed along two alignments with direction NNO-SSE and include the hill of S. Michele, the hill of mount Claro, the hill of Tuvixeddu - Tuvumannu, mount Is Mirrionis, the hill of Castello, the hill of Monte Urpino, the hill of Mount Mixi, the hill of Bonaria and the hill of San Bartolomeo.

Where natural vegetation is still present, the hills retain high values of naturalness in the urban area, representing unique reservoirs for biodiversity. For example, the garrigues present in these territories are recognized as of particular botanical interest: in addition to the presence of Sardinian endemics (*Genista corsica* (Loisel.) DC., *Helichrysum italicum* (Roth) G.Don subsp. *tyrrhenicum* (Bacch., Brullo & Giusso) Herrando, JMBlanco, L.Sáez & Galbany), these plant communities are characterized by having plant species that have a Mediterranean-Eastern gravitation (*Satureja thymbra* L., *Thymbra capitata* (L.) Cav., *Poterium spinosum* L.) which differentiates them from other plant communities in Sardinia that generally show a Western-Mediterranean floristic contingent. In particular, the Promontory of Sant'Elia (Cagliari) is the only place where *Poterium spinosum* is present in Sardinia and represents the western limit of distribution of the species at a global level (Natura 2000, habitat 5420: *Sarcopoterium spinosum* phryganas), while on Colle San Michele we can find a widespread population of *Satureja thymbra*.

On the other hills of Cagliari, in the remaining fragments of natural vegetation, the garrigue vegetation appears floristically depleted compared to that of Capo Sant'Elia. For these reasons, the Promontory of Capo Sant'Elia and the system of the hills of Cagliari still characterized by the presence of spontaneous vegetation are of strategic importance for the preservation of urban diversity. These elements, closely correlated with the human settlement, could identify some of the nodes of the green infrastructure of the MCC, to be developed for the conservation of biodiversity and its sustainable use in the urban area.

4. Conclusions

The Metropolitan City of Cagliari is characterized by an important amount of environmental assets but at the same time presents a high degree of vulnerability due to internal characteristics but also to external pressures, due primarily to climate change, to which it is subjected. The definition of a medium-long term development perspective must necessarily deal with this situation and must include within its development vision the integration of the principles of sustainability, resilience and smartness. The smartness pursues sustainability through creating a digitally- enabled environment which promotes a more efficient use of energy consumptions and a more effective management of networks. The more a city is innovative, the more information and communication technologies is used improving the quality of life and the sustainable development. Uncertain events such as weather and climate negative events at urban level, together with a growing population which increases the urban sprawl phenomenon, feature the need of creating and maintaining prosperous social, economic and ecological systems through sustainable urban systems [28]. Moreover, the capability of a city planner to develop a strategic approach that adopts a wide and long-term vision may contribute to make a city more resilient and less vulnerable. Climate resilience as well as a digital environment may contribute to support strategies for reducing vulnerability and achieving sustainability. In fact, the more information and data are available from multiple sources in a smart city context the more it may facilitate the knowledge of potential climate- related risks and damages. This may increase urban resilience due to a more conscious planning and decision-making process in reducing urban vulnerability. Finally, technology may contribute to better planning and managing a resilient city through the improvement of city’s adaptive capacity and the implement of city's mitigation strategies [29]. As a consequence, these three definitions provide a common paradigm of future urban development and structure. The city’s evolution aims at increasing the quality of life and reducing vulnerability following a sustainable path of development in the near future as well as guaranteeing further progress in the future. This new paradigm for a sustainable, digital, and less vulnerable city may be defined as “bright city” [30], where combined actions are implemented in order to maximize city’s efficiency and management efficacy.

Author contributions

This paper is the result of the joint work of the authors. For Italian evaluation purposes MEP takes responsibility for section 4.1, LM for sections 3 and 5, GB for section 2, EB for section 4.2 and MM for section 4.3.

GENERAL DISCUSSIONS AND CONCLUSIONS

Trees outside forest (i.e., scattered trees, small woodlots - SWOFs, trees lines) represent important multifunctional resources, providing different ecosystem services at both global and local scales (Manning et al. 2006; Prevedello et al. 2018). Considering that policymakers, land planners, and conservation organizations generally focus their efforts on large, intact and well-connected areas by underestimating the importance of these small but wide-spread resources (Wintle et al. 2019), more studies are urgently needed to support and promote their conservation (Lindenmayer 2019). This need is further reinforced by the limited number of studies conducted at the local scale on TOF distribution and coverage in different land-use types (Schnell et al. 2015; Price et al. 2017), especially considering the different TOF types and the wide range of possible attributes measurable at multiple scales.

This lack underlines the need to implement the monitoring of these important resources, especially in Mediterranean areas recognized as one of the most important hotspots for biodiversity conservation (Marignani et al. 2017a, b; Médail 2017), but strongly affected by human activities (Marignani et al. 2017b) and considered particularly sensitive to the growing threats of climate change (Giorgi and Lionello 2008) and biological invasions.

The broad goal of this thesis was to investigate the role of Mediterranean SWOFs for biodiversity conservation, along a gradient of land-use intensification from natural areas to urbanized zones (natural and semi-natural, agriculture and artificial areas), by using a multi-scale and multi-taxa approach based on different statistical modelling tools.

Different approaches combining statistical modelling tools and biodiversity monitoring of multiple taxonomic groups allow relating taxonomical and functional diversity patterns with environmental conditions (Guisan and Thuiller 2005; Elith et al. 2006; Lomba et al. 2010), contributing to better understand the effect of human-disturbance with important theoretical and applied implications for small woodlots outside forest management (Wintle et al. 2019; Lindenmayer 2019).

Several studies indicate that climate can be considered the most apparent factor driving species distributions at macro-scales (Pearson et al. 2004; Vicente et al. 2011), whereas microclimate, topography, land cover and land use, disturbance, resources, and biotic interactions become increasingly important at the local and more detailed spatial scale (Vicente et al. 2014).

The development of high spatial resolution bioclimatic data conducted in this research emphasized the climate and environmental heterogeneity present in this Mediterranean region (Chapter 1). The newly developed data will support a new generation of research studies in a broad array of ecological applications at a much finer scale than previously possible (Bazzato et al. 2021) and played a key role in the achievement of the general goals of this thesis (Chapter 1, 3-4).

One of the main achieved conclusions relies on the fact that small woodlots do not represent a negligible component of natural and human-impacted land-uses in the considered Mediterranean area, supporting a good level of naturality also in the human-altered matrix and particularly in agricultural landscapes (Chapter 1). These level of naturality, consistently with the evident dynamics of Mediterranean forests (Falcucci et al. 2007; Carranza et al. 2020) and agricultural areas (San Roman Sanz et al. 2013), highlight the chance to consider these patches as an opportunity for natural conversion to forest land and consequent rewilding ecosystems with important benefits for biodiversity and ecosystem services.

Thanks to the intensive field sampling conducted for almost two years, this research provides a remarkable contribution to the knowledge of floristic and faunistic species distribution at regional, but also national level (Chapter 2). The wealth of novel taxa of vascular plants, spiders, beetles, and ants here documented for the second largest Mediterranean island (Rosati et al. 2020; Schifani et al. 2021; Bazzato et al. submitted; Caria et al. accepted), but also for the Italian territory (Caria et al. accepted) underlines the importance to implement our knowledge of species diversity in human-shaped environments and sites most threatened by e.g. invasive alien species (Benedetti and Morelli 2017).

Although this research permitted to report a recent established alien beetle species for the investigated area (Chapter 2, subchapter 2.2), the patterns observed across different land-use matrices also revealed that the contribution of alien plants to the total diversity resulted particularly low in comparison with the native ones (Chapter 1, subchapter 1.1). The contingent of alien was concentrated in a few dominant species in the agricultural and urban land-use matrices, whereas the other alien species showed a negligible occurrence.

These findings may be especially relevant for these small patches. Small patches located in highly human-impacted land-uses are usually linked to the activities of different stakeholders, such as farmers, pastoralists, institutions linked to agriculture and rural development, people living in settlements and cities and institutions linked to urban management and development (Pauleit et al. 2005; de Foresta et al. 2013).

However, quantitative evidence also showed a stronger effect of dominant matrices surrounding SWOFs on species composition rather than on species richness of almost all considered taxonomic groups: plants (both native and alien contingent) and all ground-dwelling invertebrates except rove beetles (Chapter 1 and 3). Moreover, it was observed a general homogenization of diversity patterns across SWOFs surrounded by matrices at high or intermediate disturbance (urban and agricultural areas) (i) for the richness of plants (both for native and alien contingent), and (ii) for the richness, composition and beta diversity (except pseudoscorpions) of all ground-dwelling invertebrates (Chapter 1 and 3). Overall, this research highlighted that only overall vascular plants (but not the alien contingent) shifted their compositional features (except beta diversity pattern) between urban and agricultural SWOFs (Chapter 1 and 3).

In this framework, reasons for the stronger effect in higher trophic levels were linked to the increase of human-disturbance, the strong dependence on the lower trophic level (Lindenmayer et al. 2005) and the consequent multiple interacting factors (Scherber 2015) (Chapter 3).

The general homogenization on taxonomic diversity patterns was also observed on a key plant trait - seed mass - of two congeneric species (*Asparagus albus* L. and *Asparagus acutifolius* L., Chapter 4, subchapter 4.2). Landscape matrix influenced one of the most important seed traits, leading to a reduction of variability in human-altered contexts.

The homogenization along the land-use intensification gradient, here observed, corroborate results of previous studies on species richness (McKinney 2006), beta diversity patterns (Buhk et al. 2017) and other species traits (Gámez-Virués et al. 2015; Carmona et al. 2020).

On the other hand, the high number of key native plants found in each land-use matrix (Chapter 1) and the high level of richness and abundance of almost all ground-dwelling invertebrates (except rove beetles), recorded in urban and agricultural SWOFs (Chapter 3), underline the need to preserve these remaining patches to avoid that homogenization drives a generalized biodiversity loss (McKinney 2006; Buhk et al. 2017) and extinction of entire communities in the long-term (Gámez-Virués et al. 2015).

This research also showed evidence that each group of predictors (spatial-topographic, bioclimatic and landscape-level variables) contributed to explaining the total variation in the composition and cover of native species, whereas only climatic factors and spatial-topographic filters were implied in explaining the compositional variation for alien species (Chapter 1).

Taking into account the environmental heterogeneity in terms of climate, topography, land use, disturbance, resources, and biotic interactions, this study contributes to enhancing the knowledge on the highly complex and variable interplay of these drivers in explaining cross-taxon congruence in species composition along the considered gradient (Chapter 3).

In addition, this study draws attention to the importance of higher trophic-level organisms, positively related to the lower trophic level, highlighting as they can contribute to ecosystem functions and multifunctionality (Schuldt et al. 2018), representing a promise for the future studies and application of surrogates in the presence of strong stressor environmental gradient.

The knowledge provided is a baseline to comprehend the role of small woodlots outside forest supporting high levels of biodiversity and give useful indications for planning and management conservation efforts (Gerlach et al. 2013).

Notwithstanding that the impacts of human disturbance are neither temporary nor fully avoidable (Araia et al. 2020), solutions to preserve species and communities conservation with the inclusion of wildlife in agricultural (Simons and Weisser 2017) and urban areas (Apfelbeck et al. 2019, 2020) are possible and needed (Capotorti et al. 2020).

In this regard, the multidisciplinary approach for the definition of indicators may represent a pragmatic solution to informing managers on the degree of impact on a real planning jurisdiction and its local authorities (Chapter 5, subchapter 5.1).

Although all Italian metropolitan cities are oriented towards adopting strategic and sustainable development models, specific environmental assessments not always are part of the planning process and therefore are functional for future governance choices (Palumbo et al. 2020).

An important exception is represented by the metropolitan city of Cagliari (Chapter 5m subchapter 5.2), which is considering the environmental dynamic and assets into its strategic plan, actually in defining phase, thanks to the collaboration between ecologists, planners and managers.

Research outlook and future perspectives

Although this framework provides a valuable baseline for monitoring and preserving small woodlots outside forest in a Mediterranean fragmented landscape context, future outlooks can be considered to produce more exhaustive research.

Explored multidisciplinary aspects open possible future developments that include four main sub-types: (i) SWOF spatial distribution map, (ii) species diversity patterns, (ii) integration of taxonomic and functional diversity patterns, (iv) detection of key-small patches.

The method for the spatial delimitation of SWOFs, here implemented, was used efficiently to the census of these resources at the local scale. The monitoring of TOF coverage on a large planning jurisdiction (e.g., metropolitan cities, region, country) require alternative technics. For example, remote sensing methods based on high-resolution data, and terrestrial laser, capable of describing both the horizontal and the vertical distribution of the vegetation, appear to be promising techniques for the development of detailed maps and monitoring applications in combination with the field surveys (Schnell et al. 2015).

Different diversity measures can reveal more complex patterns (Chapter 1 and 3), emphasizing various ecological processes (Aggemyr et al. 2018). The implementation of measures like β diversity, which provides a direct link between biodiversity at the local (α diversity) and broader scales (i.e., regional species pool; γ diversity), can be meaningfully utilized to the development of robust frameworks for assessing ecological processes (Anderson et al. 2011). In this sense, simulations and models based on multiple approaches combining spatial or temporal species data and con-joint analyses of abundance, taxonomic, and functional diversity may be useful to narrow down potential mechanisms driving β diversity (Anderson et al. 2011), with implication on studies focusing in small patches in human-modified landscapes.

To better understand patterns of distribution of taxa inhabiting these small patches, the occurrence of TOF also could be included as a base datum in the species distribution models (Prevedello et al. 2018).

Ensemble modelling of species distribution, based on high-resolution climate or other spatial data (e.g., land-use maps, TOF maps), could usefully contribute to identifying key individual patches that should be prioritized for conservation and restoration efforts (Cadavid-Florez et al. 2020).

Accounting for these aspects will significantly promote fundamental research on different spatially-structured ecological patterns, enhancing future valuation studies on small woodlots outside forest. These will allow more complete conclusions, particularly for Mediterranean fragmented landscape and their ecosystems, with application in such fields as global change ecology, conservation planning, and management of these important resources.

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SUPPLEMENTARY MATERIALS

Chapter 1 - Subchapter 1.2: Annex S1

Table 1. Summary statistics of the number of total species richness and abundance classified as native or aliens. Land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

Category	NAT		AGR		URB		Total species richness	Total Coverage
	Richness	Coverage	Richness	Coverage	Richness	Coverage		
Native	195	7706	187	3226.3	88	2356.6	302	13288.9
Alien	10	921.3	13	3486.1	16	1869.8	22	6277.2

Table 2. Summary statistics of number of SWOF, mean species richness and abundance (mean ± standard error. minimum and maximum) of the two groups: native and alien plants. Land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). Statistical note: mean species richness (μ); standard error (SE); minimum (min) and maximum (max) values.

	Group	NAT	AGR	URB
Richness/SWOF	Native	43.45 ± 4.17 (28 - 71)	37.45 ± 6.8 (8 - 78)	21 ± 2.2 (13 - 32)
μ ± SE (min-max)	Alien	1.55 ± 0.62 (0 - 6)	3.64 ± 0.56 (1 - 8)	4.25 ± 0.94 (1 - 9)
Cover/SWOF	Native	2.32 ± 0.34 (0 - 422.6)	0.97 ± 0.16 (0 - 349.8)	0.98 ± 0.21 (0 - 320)
μ ± SE (min-max)	Alien	3.81 ± 1.39 (0 - 195)	14.41 ± 3.57 (0 - 420.8)	10.62 ± 2.84 (0 - 280.5)
No. SWOFs		11	11	8

Table 3. PERMANOVA results based on the Euclidean distance of species richness data of the two groups (GR) classified as native or aliens, sampled from 30 SWOFs at the three land-use matrices (MA). Significance codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

Source of variation	df	SS	MS	Pseudo-F and p-values	Variance components (%)
MA	2	970.95	485.47	3.55*	9.24
GR	1	13937	13937	101.83***	47.76
MAxGR	2	1487.7	743.83	5.44**	17.24
Res	54	7390.2	136.86	-	25.76
Total	59	25434	-	-	100.00

Table 4. PERMANOVA t statistic and significance values of pair-wise tests for the interaction term “MA x GR” for pairs of levels of factor “land-use matrix” within each taxonomic group (native or alien plants) based on species richness data. Land-use matrices (MA): natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). Significance codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

Category	**Pairs of levels of factor “land-use matrix"**		
	AGR versus NAT	**AGR versus URB**	**NAT versus URB**
Native	0.75	2	4.26***
Alien	2.49*	0.59	2.5*

Table 5. PERMANOVA results based on zero-adjusted Bray-Curtis dissimilarity on square-root transformed abundance data analyzed separately for native and alien classified groups sampled from 30 SWOFs at the three land-use matrices (MA). Significance codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

Source of variation	**df**	**NATIVE**		**ALIENS**	
		Pseudo-F	**Variance components (%)**	**Pseudo-F**	**Variance components (%)**
MA	2	3.55***	20.46	3.32***	19.00
Res	27	-	79.54	-	81.00
Total	29	-	100	-	100

Table 6. PERMANOVA t statistic and significance values of pair-wise tests for the main effect of land use matrices on species abundance analysed separately for native and aliens at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). Significance codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

	Pairs of levels of factor “land-use matrix"		
Category	**AGR versus NAT**	**AGR versus URB**	**NAT versus URB**
Native	1.97***	1.51*	2.1***
Alien	2.17**	1.45	1.75*

Chapter 1 - Subchapter 1.2: Annex S2

Supplementary Data of subchapter 1.2 (S2): plot of the ten principal component analysis (PCA) run for each predictor set of environmental variables

Fig. 1. Plot of the first two principal component analysis (PCA) axes for spatial-topographic (scaled to zero mean and unit variance) recorded from the 30 SWOFs along the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

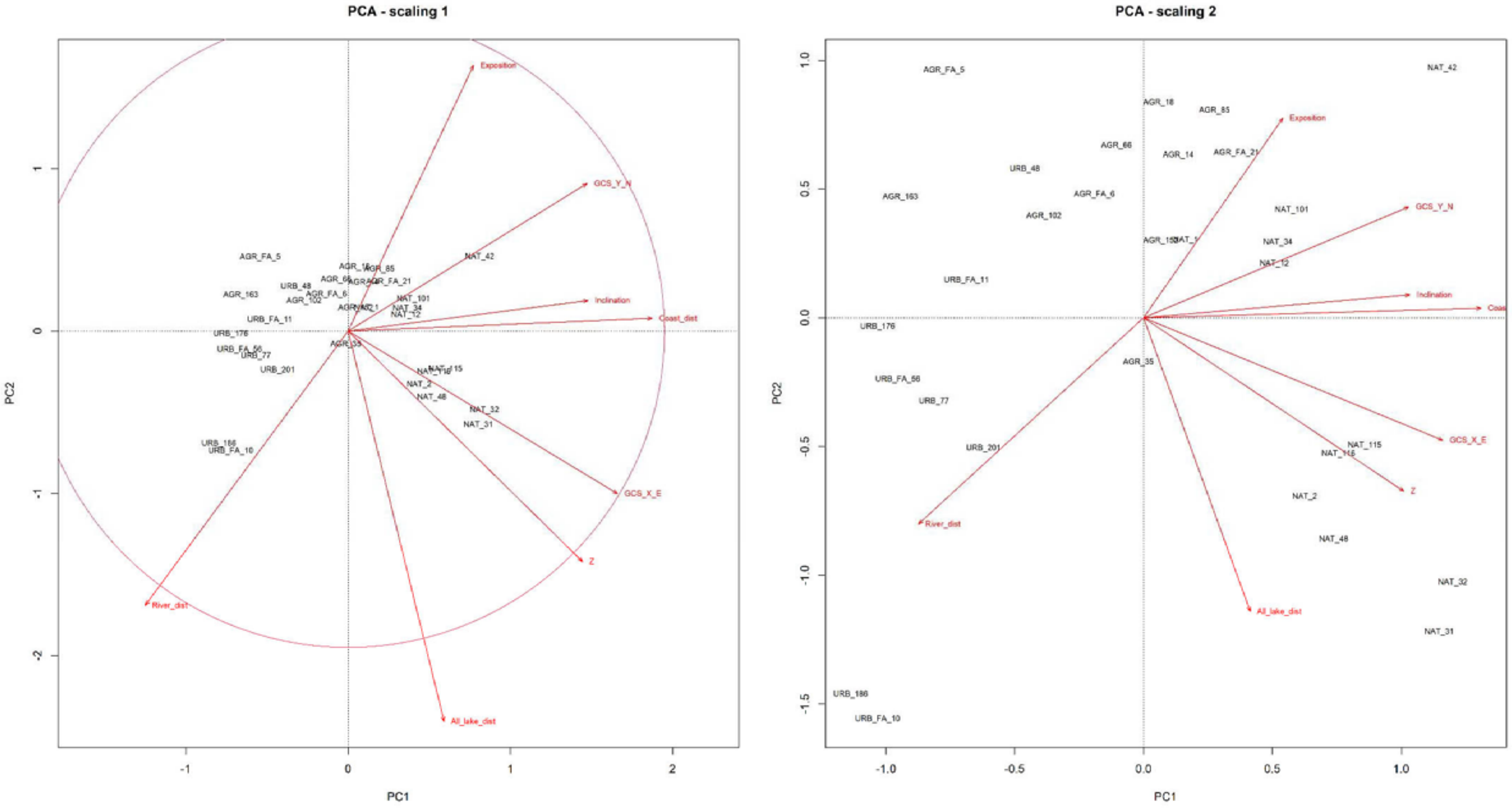


Fig. 2. Spearman's rank correlation between selected axes and spatial-topographic variables recorded from the 30 SWOFs along the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

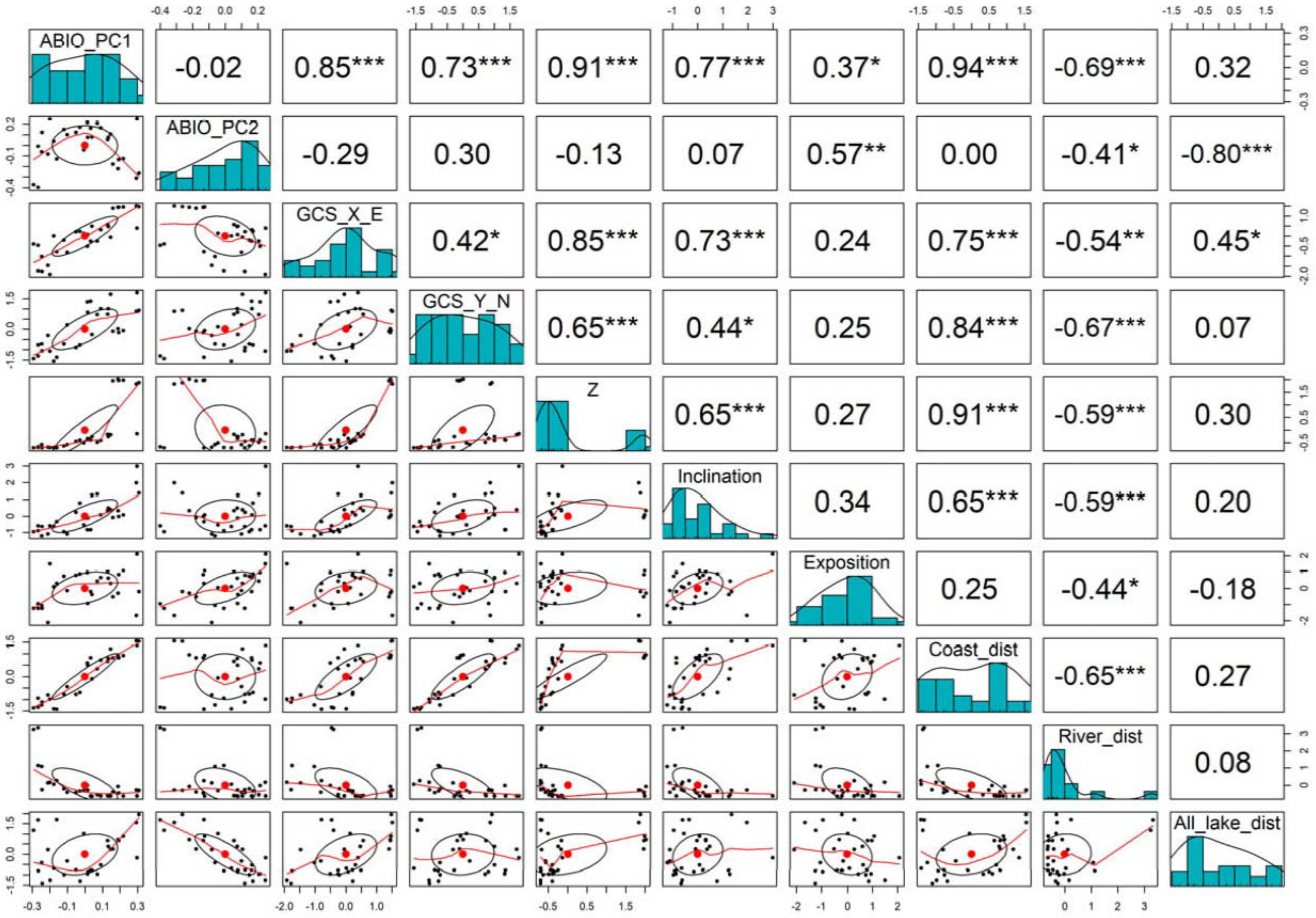


Fig. 3. Plot of the first two principal component analysis (PCA) axes for climate variables (scaled to zero mean and unit variance) recorded from the 30 SWOFs along the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

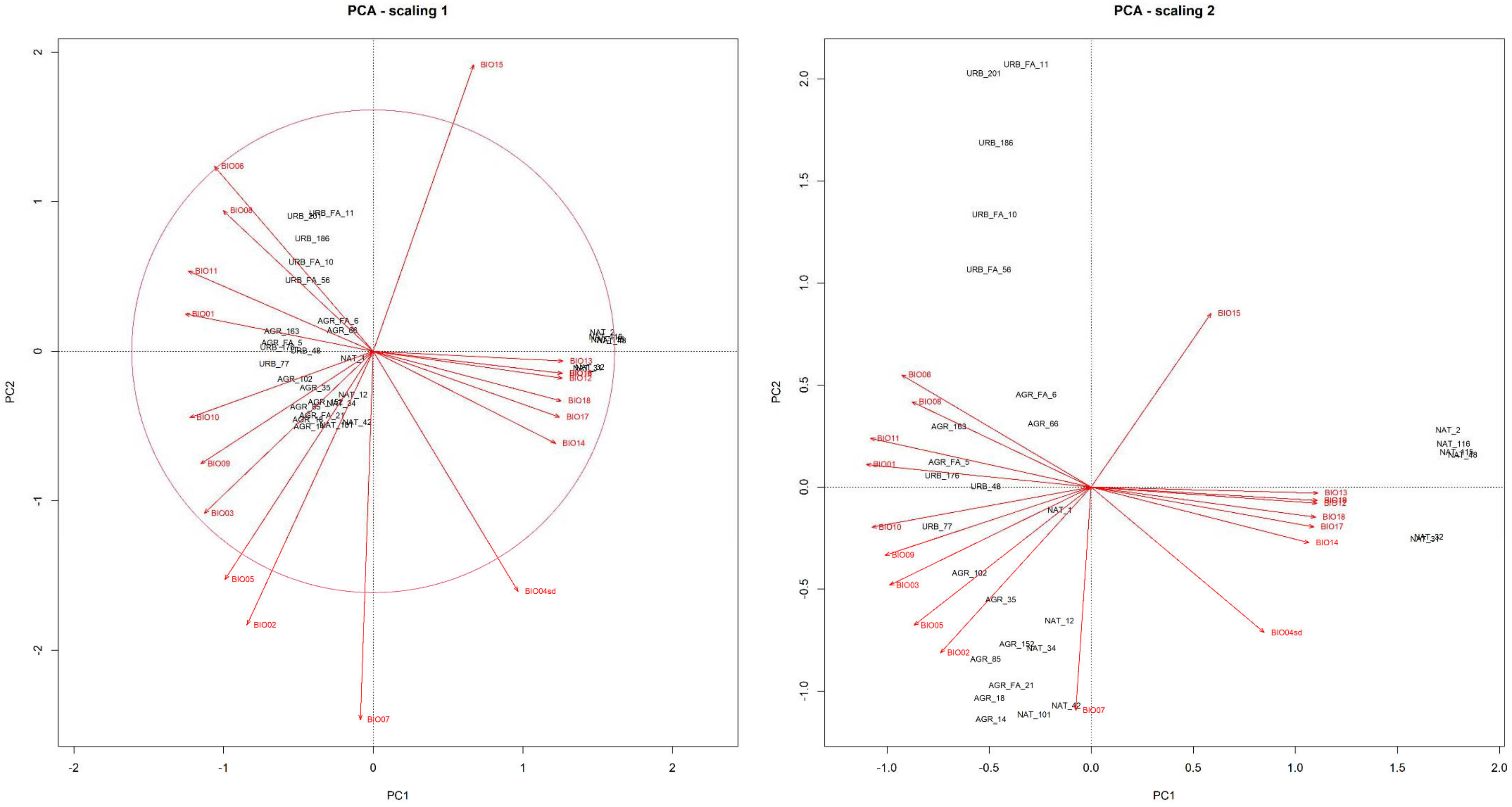


Fig. 4. Spearman's rank correlation between selected axes and climate variables recorded from the 30 SWOFs along the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

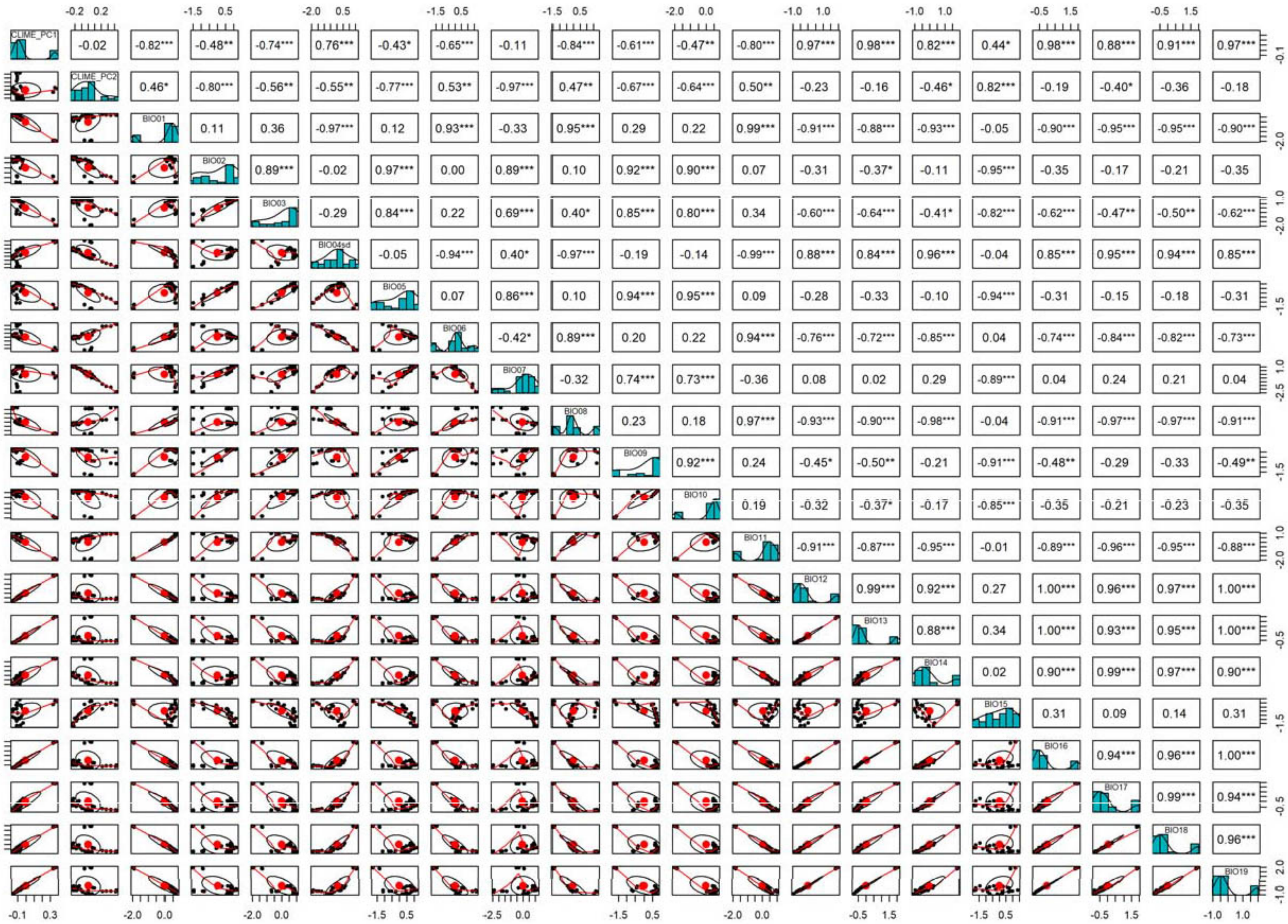


Fig. 5. Plot of the first two principal component analysis (PCA) axes for landscape metrics (scaled to zero mean and unit variance) recorded around the 30 SWOFs along the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

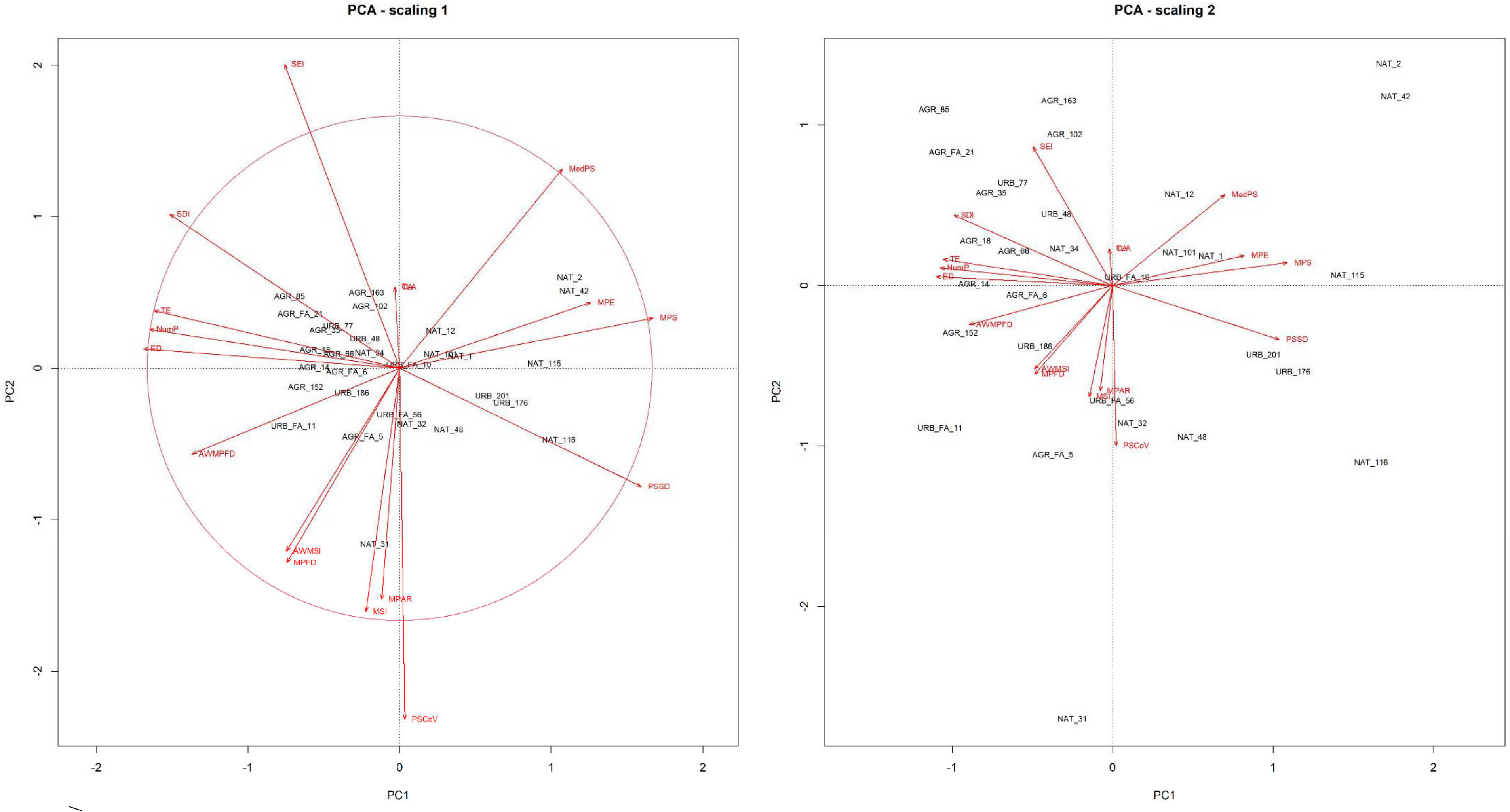
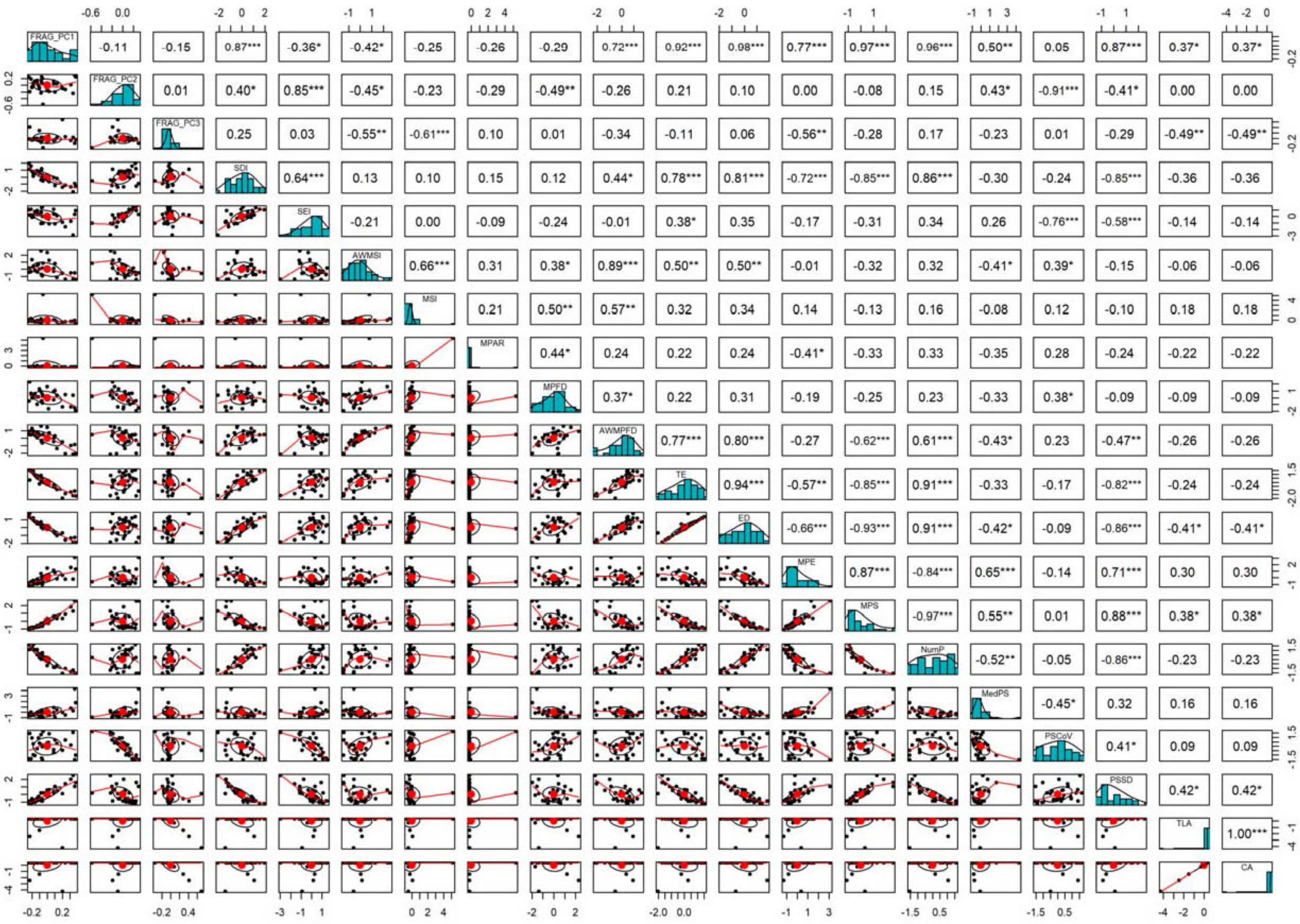


Fig. 6. Spearman's rank correlation between selected axes and landscape metrics recorded around the 30 SWOFs along the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).



Chapter 2 - Subchapter 2.1: Appendix A

***Alyssum diffusum* Ten. subsp. *garganicum* Španiel, Marhold, N.G.Passal. & Lihová**

Brassicaceae Hemicryptophyte scapose Italian endemic

First record for Basilicata

Specimen: 8-4-2011, Difesa S.Biagio (Montescaglioso, Matera province), rocky outcrop within dry grassland, 155 m a.s.l., 40.5044°N-16.6876E°, *leg. et det* S. Fascetti (HLUC).

Note. This endemic taxon was until now recorded only for Puglia region [56]. It is noteworthy to underline the presence in Basilicata of two infraspecific taxa of *Alyssum diffusum*: the subspecies *calabricum* Španiel, Marhold, N.G.Passal. & Lihová in the Campano-Lucano Apennine, along the border with Campania and Calabria regions [16] and the subsp. *garganicum* in the easter part, along the border with Puglia region.

***Anredera cordifolia* (Ten.) Steenis** [≡ *Boussingaultia cordifolia* Ten.]

Basellaceae Phanerophyte climbing Neophyte, S-America

First record for Basilicata (casual)

Specimen: 15-6-2014, Lido di Policoro (Policoro, Matera province), roadside, 40.17361°N-16.70099°E, 3 m a.s.l., *leg*. G. Salerno *et* M.R. Lapenna, *det*. G. Salerno (Herb. Salerno).

Note. Casual or naturalized alien species in the most of the peninsular and insular regions except for Umbria [17]. It is considered an invasive taxon in Toscana [17].

***Asclepias physocarpa* (E.Mey.) Schltr.** [≡ *Gomphocarpus physocarpus* E.Mey.]

Apocynaceae Phanerophyte caespitose Neophyte, S-Africa

First record for Calabria (casual)

Specimen: 1-7-2019, Campo Calabro (Reggio Calabria province), roadside, 38.20633°N-15.64897°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface *et* G. Spampinato (REGGIO).

Note. Five plants were recorded along the road, mixed with *Hyparrhenia hirta* (L.) Stapf. subsp. *hirta* and *Rubus ulmifolius* Schott. It was previously recorded as casual or naturalized alien species only in Toscana, Campania and Sicilia regions [17].

***Asplenium viride* Huds.**

Aspleniaceae Hemicryptophyte rosulate Circumboreal

First record for Sardegna

Specimen: 14-5-2013, Monte Albo (Siniscola, Nuoro province), 40.54141°N-9.62394°E, sinkhole shaft in montane karstfield, 880 m a.s.l., *leg. et det*. P. J. de Lange (AK, SS).

Note. Recorded from all regions of Italy except for Puglia and doubtfully for Basilicata, but until now absent in the insular regions (Sardegna and Sicilia) [16]. However, *Asplenium viride* had been reported from Sardegna from a locality close to Fluminimaggiore (Iglesiente, South-western Sardegna) from where it has not since been seen [57]. Recently Ballero et al. [57] suggested its exclusion from the flora of Sardegna.

Image available at: https://inaturalist.nz/observations/4046536

***Aubrieta columnae* Guss. subsp. *italica* (Boiss.) Mattf.** [≡ *A. italica* Boiss.]

Brassicaceae Chamaephyte subshrubs Italian endemic

First record for Lazio (naturalized)

Specimen: 5-8-2019, Old town (Anticoli Corrado, Roma province), old walls, 500 m a.s.l., 42.00991°N-12.99103°E, *leg. et det.* L. Rosati (HLUC).

Note. A small population of this Italian endemic taxon, which grows in nature only in Puglia (Gargano peninsula) and doubtfully in Basilicata, was observed for over 10 years in anthropogenic habitat at Anticoli Corrado (Rosati, pers. observ.). These individuals have spread by seeds from a nearby cultivated plant in pots and urban gardens. Our specimens were identified using the key by [58].

***Bellevalia boissieri* Freyn** [≡ *Bellevalia dubia* (Guss.) Kunth subsp. *boissieri* (Freyn) Feinbrun]

Asparagaceae Geophyte bulbouse E-Mediterranean

First record for Basilicata

Specimen: 28-3-2018, Iazzo di Tucci (Rotondella, Matera province), olive groves, 40.17157°N-16.53832°E, 298 m a.s.l., *leg*. G. Salerno *et* G. Misano, *det*. G. Salerno (Herb. Salerno).

Note. Until these discoveries this species was known from Italy only from the Calabria region [16].

***Bupleurum rollii* (Montel.) Moraldo** [≡ *B. gracile* d’Urv. var. *rollii* Montel.]

Apiaceae Therophyte scapose Italian endemic

First record for Campania

Specimen: 8-9-2013, Monte Vesole (Roccadaspide, Salerno province), chestnuts, 580 m a.s.l., 40.41156°N-15.16761°E, *leg. et det.* G. Salerno (Herb. Salerno).

Note. This Italian endemic had previously been recorded from Lazio to Sicilia, but not from Campania [16].

***Campanula poscharskyana* Degen**

Campanulaceae Hemicryptophyte scap Neophyte, SE-European (Dinarides)

First record for Lazio (casual)

Specimen: 5-8-2019, Old town (Roviano, Roma province), calcareous rock crevices, 495 m a.s.l., 42.02565°N-12.99403°E, *leg. et det.* L. Rosati (HLUC).

Note. This alien species was reported as casual or naturalized only in Lombardia, Veneto and Toscana [17].

***Cenchrus longisetus* M.C.Johnst.** [= *Pennisetum villosum* R.Br. ex Fresen.]

Poaceae Hemicryptophyte caespitose Neophyte, Paleotropical

First record for Calabria (casual)

Specimen: 16-6-2019, San Roberto (Reggio Calabria province), sidewalks, 280 m a.s.l., 38.21074°N-15.736°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface, C.M. Musarella *et* G. Spampinato (REGGIO).

Note. Collected from the margins of sidewalks where it was probably derived from a nearby flowerbed. This species has been recorded from Italian regions as casual or naturalized [17]; only in Sardegna is it considered invasive [17].

Chamaerops humilis **L.**

Arecaceae Nano-phanerophyte Neophyte, W-Mediterranean

First record for Basilicata (casual)

Specimen: 14-12-2017, S.Basilio, Marina di Pisticci (Pisticci, Matera province), clearings in mediterranean maquis, 3 m a.s.l., 40.2998°N-16.78265°E, *leg. et det.* S. Fascetti (HLUC).

Note. Although this palm is indigenous to the neighbouring regions of Campania and Calabria [16], it has not been recorded in the previous flora of Basilicata [51]. Therefore, as for the neighbouring Puglia region [16], we consider it as introduced to this region. Plants probably spread from cultivation as the species was present in nearby gardens, and only juvenile plants and seedlings were observed.

Commelina erecta **L.**

Commelinaceae Terophyte crawling Archeophyte, C-America

First records for Calabria and peninsular Italy (casual)

Specimens: 23-6-2019, Lazzaro (Motta San Giovanni, Reggio Calabria province), crevices between road and sidewalks, 16 m a.s.l., 37.97394°N-15.66507°E, *leg. et det.* C.M.Musarella (REGGIO); 25-6-2019, Catona (Reggio Calabria, Reggio Calabria province), sidewalks, 22 m a.s.l., 38.17987°N-15.64811°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface, C.M. Musarella *et* G. Spampinato (REGGIO).

Note. Observed growing in crevices and on the edge of the roadside. Probably escaped from plants cultivated in pots placed on sidewalks. It was recently reported for the first time in Italy for Sicilia in [38].

Conringia orientalis **(L.) Andrz. ex DC.** [≡ *Brassica orientalis* L.;≡ *Gorinkia orientalis* (L.) J.Presl & C.Presl]

Brassicaceae Therophyte scapose Neophyte, W-Asia

First record for Campania (casual)

Specimen: 15-5-2018, Campitello, Monti della Maddalena (Padula, Salerno province), forest edges, along a track, 1270 m a.s.l., 40.3452°N-15.68886°E, *leg*. S. Fascetti, *det* S. Fascetti *et* L. Rosati (HLUC).

Note. It is considered an archaeophyte associated with cereal crops, as stated for example in a previous record for Basilicata [59]. However, our finding of *C. orientalis* was in a silvo-pastoral habitat, quite far from cultivated areas.

Convolvulus siculus **L. subsp.** ***siculus***

Convolvulaceae Therophyte scapose S-Mediterranean

First record for Basilicata

Specimen: 12-3-2012, Torre Caina (Maratea, Potenza province), clearings in mediterranean maquis, 39.94319°N-15.73534°E, 880 m a.s.l., *leg. et det*. G. Salerno (Herb. Salerno).

Note. It was previously recorded in Liguria, Toscana, Lazio, Campania, Puglia, Sicilia and Sardegna regions [16].

***Coprosma repens* A.Rich.**

Rubiaceae Nano-phanerophyte Neophyte, New Zealand

First record for Italy (casual)

Specimen: 10-03-2013, Torre Nuova, Porto Conte (Alghero, Sassari Province), occasional shrubs growing by old Spanish tower near beach, 2 m a.s.l., 40.593647°N-8.20430°E, *leg. et det*. P.J. de Lange (AK).

Note. Growing amongst *Myoporum laetum* G.Forst., *M. aff. insulare* R.Br., *M. laetum* × *M. aff. insulare*. Plants with flower buds, seedlings present. In Europe, although it has been recorded only from Great Britain [60], this species was commonly seen by PdL along the coastline of southern France (1999, 2013), and northern Spain (Catalonia) (2013). It seems surprising it has not naturalized further as under the vernacular ‘mirror bush’ it is recorded as an aggressive invasive species throughout much of the world [61].

***Cordyline australis* (G.Forst.) Endl.** [≡ *Dracaena australis* G.Forst.]

Asparagaceae Phanerophyte scapose Neophyte, New Zealand

First record for Sardegna (casual)

Specimen: 12-3-2012, Via Tarragona (Alghero, Sassari province), courtyard garden 30 m a.s.l., 40.55422°N-8.32475°E, *leg. et det*. P.J. de Lange (AK, SS).

Note. Seedling collected from rank grassland within inner courtyard garden. Growing close to planted adult fruiting trees. Although a very commonly cultivated tree in Sardegna this species is, so far, only sparingly naturalizing. This may be because mostly only the one clone is grown and the species is self-incompatible [62]. Outside Sardegna this species has been reported as a casual alien only in Marche, Abruzzo and Campania [17].

***Cyclamen balearicum* Willk.** [≡ *C. repandum* Sm. subsp. *balearicum* (Willk.) O. Schwarz]

Primulaceae Geophyte bulbouse NW-Mediterranean

First record for Italy (Sardegna)

Specimen: 20-4-2008, Monte Doglia (Alghero, Sassari province), evergreen woodlands of *Prasio majoris-Quercetum ilicis chamaeropetosum humilis*, 40.60742°N-8.24683°E, 190 m a.s.l., *leg. et det.* E. Farris (SS).

Note. Cyclamen balearicum differs from *C. repandum* in having leaves usually adaxially speckled with white and flowers that are either fully white or with a white-veined pale pink corolla [42]. Recent research [63,64] highlighted the presence of a hybrid zone between *Cyclamen repandum* and *C. balearicum* in NW Sardegna, where hybrid plants displaying bi-coloured flowers prevail, and pure *C. balearicum* plants were scarce. However, based on original observations, at Mt. Doglia plants with small white flowers and non-exerted stigma are dominant, though only occasional individuals have the diagnostic leaf ornamentation of *C. balearicum s.s.* The nearest known localities of this species according to Flora Gallica and Flora Iberica are in Corsica (France), Provence (S-France) and Balearic Islands (Spain).

***Daucus aureus* Desf.**

Apiaceae Therophyte scapose S-Mediterranean

First record for Basilicata

Specimen: 1-5-2019, Serra delle Grotte (Rotondella, Matera province), fallows on clay soils, 120 m a.s.l., 40.19691°N-16.49405°E, *leg*. G. Salerno *et* G. Misano, *det*. G. Salerno (Herb. Salerno).

Note. At the present known only from Calabria and Sicilia[16]. There are historical doubtful records of this species from Liguria and Campania, respectively [16].

***Dichoropetalum carvifolium-chabraei* (Crantz) Soldano, Galasso & Banfi** [≡ *Selinum carvifolium-chabraei* Crantz; – *Holandrea carvifolia* Reduron, Charpin & Pimenov; = *Peucedanum carvifolium-chabraei* (Crantz) Soldano]

Apiaceae Hemicryptophyte scapose Europeo-Caucasian

First record for Lazio

Specimen: 16-7-2015, Monte Tilia, Fondo del Laghetto (Leonessa, Rieti province), secondary grassland with *Brachypodium genuense*, marly-calcareous slopes, 1400 m a.s.l., 42.55842°N-12.95441°E, *leg. et det.* E. Del Vico (Herb. Del Vico).

Note. In Italy it is reported in many northern and central regions and in Sicilia [16].

***Epilobium nummularifolium* R.Cunn. ex A.Cunn.**

Onagraceae Hemicryptophyte crawling Neophyte, New Zealand

First record for Italy (casual)

Specimen: 24-04 2013, via Castelsardo (Alghero, Sassari province), damp ground on side of road adjacent pasture used for hay making, 6 m a.s.l., 40.57221°N-8.32010°E, *leg. et det.* P.J. de Lange (AK, SS).

Note.

Image from a very close locality to the above indicated available at: https://inaturalist.nz/observations/4437892.

Epilobium nummularifolium is apparently not known wild elsewhere in greater Europe [65]. However, this species has a superficial similarity to *E. brunnescens* (Cockayne) Raven et Engelhorn subsp. *brunnescens*, which is abundantly naturalized in the British Isles, and reported from the Guernesy Islands, so it is possible that it has been confused with that species, and so overlooked [65]. *E.*

nummularifolium, a New Zealand endemic, is well known for its aggressive, weedy nature. Raven & Raven [65] argue that this species was once a narrow range endemic of northern New Zealand, which has spread throughout that archipelago as a garden plant pot contaminant. We suspect that this is how it has reached Sardegna, i.e. as a possible soil/garden plant contaminant, introduced alongside anyone of the many New Zealand indigenous/endemic plants seen there in cultivation. This species should be looked for elsewhere in Europe. *Epilobium nummularifolium* is distinguished from both *E. brunnescens* and *E. komarovianum* H. Lev, which is also naturalized in Europe [65] by the yellow-green, serrulate leaves, purple-blue stems and grey-strigulose capsules [65-68] (Table 1, Figure 3). These species are keyed out as follows:

1. Adaxial leaf surface deeply rugose-impressed.. *E. komarovianum*

1. Adaxial leaf surface smooth, not rugose-impressed...................................... *2*

2. Leaves yellow-green, with red margins, serrulate bearing 2-11 pairs of teeth either side of midrib; stems purple-blue, capsules grey-strigulose........... *E. nummularifolium*

2. Leaves dark green, tinged red or brown, entire or with 1-4 pairs of indistinct teeth either side of midrib, capsules glabrate, rarely very sparsely hairy... *E. brunnescens*

Table 1. Distinguishing characters between three New Zealand creeping *Epilobium* species naturalized in Europe/British Isles

	E. brunnescens subsp. *brunnescens*	*E. komarovianum*	*E. nummularifolium*
Growth Habit	Densely matted, creeping herb forming patches up to 1 m diameter	Matted, creeping herb forming patches of tightly appressed stems and leaves up to 1 m diameter (usually less)	Loosely matted, creeping herb patches up to 1 m diameter
Stems	Light brown	Yellow-green	Purple-blue

Leaves	Dull green often with a red or brown-tinge, bearing 0-1(-2) obscure laterial veins either side of midrib; lamina 1.5-13.0 × 1.5-12.0 mm ovate to broadly ovate, apex rounded to obtuse, base obtuse to truncate, entire, rarely with 1-2(-4) weakly developed teeth on each margin	Dull reddish-green to coppery, adaxially rugose-impressed, bearing 1-4 lateral veins on each side of the midrib; lamina 2.0-12.0 × 1.5-9.0 mm, usually orbicular, but occasionally oblong or ovate (sometimes with all forms on the same plant), apices subacute to obtuse, base attenuate to obtuse, entire or occasionally with 1-3 remote, weak teeth on each side of leaf	Yellowish-green, usually with red margins and 1-3(-4) inconspicuous lateral veins on either side of the midrib; lamina 3.0-13.0 × 3-11.0 mm, broadly ovate to oblate, obtuse or rounded at apex, rounded to truncate at base, margins remotely and shallowly serrulate with 2-11 teeth on either side
Flowers	Flowers nodding, falling before full pedicel elongation is achieved	Flowers falling before full pedicel elongation is achieved	flowers falling when pedicel elongation is complete
Capsules	Capsule glabrate or sparsely hairy, indumentum of appressed eglandular or erect glandular or eglandular hairs running along lines of dehiscence, 12-60 mm long, borne on a glabrous pedicel (16-)35-80(-120) mm long	Capsule subglabrous or sparsely furnished with hairs, 4-30 mm long, borne on a pedicel 3-93(-135) mm long	Capsule densely grey-strigulose, (10-)15-40 mm long, borne on a strigulose pedicel 23-130 mm long

***Epipactis schubertiorum* Bartolo, Pulv. & Robatsch** [≡ *Epipactis helleborine* (L.) Crantz subsp. *schubertiorum* (Bartolo, Pulv. & Robatsch) Kreutz]

Orchidaceae Geophyte rhizomatous Italian endemic

First record for Basilicata

Observatum: 25-7-2017, Abetina di Laurenzana (Viggiano, Potenza province), *Abies alba* and *Quercus cerris* forest, 1130 m a.s.l., 40.40708°N-15.95478°E, *leg. et det.* V.A. Romano (HLUC).

Note. This Italian endemic species had previously been recorded only from Puglia and Calabria [16].

Eragrostis barrelieri* Daveau subsp. *barrelieri

Poaceae Therophyte scapose S-Mediterranean

First record for Basilicata

Specimen: 16-6-2014, Concio, Sinni river (Policoro, Matera province), roadside, 40.17126°N-16.64227°E, 10 m a.s.l., *leg. et det.* G. Salerno (Herb. Salerno).

Note. In Italy it is recorded for some northern and central regions, Sicilia and Sardegna [16].

***Euphorbia nutans* Lag.** [≡ *Chamaesyce nutans* (Lag.) Small]
Euphorbiaceae Therophyte scapose. Neophyte, N-America
First record for Basilicata (casual) *and Sardegna* (casual)
Specimens: 3-8-2017, Porto (Maratea, Potenza province), roadside, 15 m a.s.l., 39.98978°N-15.70888°E, *leg.* A. Stinca *et* M. Eviano, *det.* A. Stinca (PORUN-Herb. Stinca); 20-8-2017, along SS18 road, close by loc. La Torre (Maratea, Potenza province), roadside, 60 m a.s.l., 40.03127°N-15.67405°E, *leg.* A. Stinca *et* M. Ravo, *det.* A. Stinca (PORUN-Herb. Stinca); 29-9-2019, Tuerredda (Teulada, Cagliari province), pots and paths along the dune, 2 m a.s.l., 38.89686°N-8.81394°E, *leg. et det.* L. Rosati *et* M. Marignani (HLUC).
Note. Euphorbia nutans was recorded as casual or naturalized alien in most of the Italian regions [17]. It is considered invasive in Lombardia and Veneto [17].

***Euphorbia stricta* L.** [= *Euphorbia serrulata* Thuill.]
Euphorbiaceae Therophyte scapose Europeo-Caucasian
First record for Basilicata and confirmation for peninsular Italy
Specimen: 12-6-2018, Vascarano, Pergola (Marsico Nuovo, Potenza province), traditional arable land, 915 m a.s.l., 40.42385°N-15.68713°E, *leg. et det.* L. Rosati (HLUC).
Note. Previously, and reliably reported only from the northern Italian regions [16]; this record provides the first verified evidence of its presence in peninsular Italy. Previously it was considered as doubtfully present in Lazio [16], while it has not been recently recorded from Abruzzo and Calabria [16], and was apparently recorded by mistake from Sicilia [16].

***Fagopyrum esculentum* Moench**
Polygonaceae Therophyte scapose Neophyte, Asia
First record for Basilicata (casual) *and confirmation for Calabria* (casual)
Specimens: 7-9-2011, Diga Masseria Nicodemo, Lago di Cogliandrino (Lauria, Potenza province), fallows, mesic grasslands, 670 m a.s.l., 40.08151°N-15.93680°E, *leg. et det.* S. Fascetti (HLUC); 30-8-2017, Contrada Schiena La Torre (Verbicaro, Cosenza province), vegetable garden, 575 m a.s.l., 39.76166°N-15.89780°E, *leg. et det.* S. Fascetti (HLUC).
Note. This species is known with certainty only from the northern Italian regions [17]; otherwise considered doubtfully present in Lazio, and not recently confirmed from Abruzzo and Calabria (after 1950) [17]. In Basilicata it was observed escaping from small cultivated fields of buckwheat. In the Calabria region it was observed for some years as a weed of vegetable gardens and along arable land edges.

***Ficus microcarpa* L.f.** [– *Ficus retusa* auct. p.p., non L.]
Moraceae Phanerophyte scapose Neophyte, E-Asia

First record for Basilicata (casual)

Specimen: 3-8-2017, Porto (Maratea, Potenza province), walls and crevices in the road pavements, 5 m a.s.l., 39.98949°N-15.70868°E, *leg.* A. Stinca *et* M. Eviano, *det.* A. Stinca (PORUN-Herb. Stinca).

Note. Treated as a casual alien in Sardegna, Campania and Puglia; naturalized in Sicilia [17]. Recently recorded in southern Italy for Calabria [14].

***Filago asterisciflora* (Lam.) Sweet** [≡ *Gnaphalium asterisciflorum* Lam.; ≡ *Evax asterisciflora* (Lam.) Pers.]

Asteraceae Terophyte crawling Steno-Mediterranean

First record for Basilicata

Specimen: 2-5-2019, Bosco Finocchio (Rotondella, Matera province), roadside, 40.15282°N-16.47588°E, 600 m a.s.l., *leg.* G. Salerno *et* G. Misano *det.* G. Salerno (Herb. Salerno).

Note. Previously *Filago asterisciflora* has been recorded from Toscana, Lazio, Campania, Puglia, Calabria, Sicilia and Sardegna [16].

***Galanthus reginae-olgae* Orph. subsp. *vernalis* Kamari**

Amaryllidaceae Geophyte bulbouse Europeo-Caucasian

First record for Basilicata

Specimen: 20-2-2014, Sellata (Pignola, Potenza province), beech forest, 1270 m a.s.l., 40.53127°N-15.78871°E, *leg. et det*. S. Fascetti (HLUC).

Note. Several plants observed in a large population extending for approximately 200 m^2. Previously reported from the Calabria and Sicilia regions [16].

***Geranium pusillum* L.**

Geraniaceae Therophyte scapose Eurasian

First record for Lazio

Specimen: 4-6-2015, Colle Pietrolone (Poggio Bustone, Rieti province), secondary grassland with *Bromus hordeaceus, Poa trivialis* and *Poa sylvicola,* gently marly slopes, 1070 m a.s.l., 42.50015°N-12.90332°E, *leg.* E. Del Vico *et* L. Facioni, *det.* E. Del Vico (Herb. Del Vico).

Note. It was previously recorded in all Italian regions except for Lazio and Sicilia [16].

***Glandularia tenera* (Spreng.) Cabrera** [≡ *Verbena tenera* Spreng.; ≡ *Shuttleworthia tenera* (Spreng.) Walp.]

Verbenaceae Chamaephyte subshrubs Neophyte, S-Amer*ica*

First record for Sardegna (casual)

Specimen: 27-4-2019, Colle S. Michele (Cagliari, Cagliari province), grassland, 95 m a.s.l., 39.24345°N-9.11095°E, *leg. et det.* L. Rosati *et* M. Marignani (HLUC).

Note. It was previously recorded only for Lazio as casual [17]. Plant samples were identified using the diagnostic characters and keys of [69-71].

Gleditsia triacanthos **L.**

Fabaceae Phanerophyte scapose, Neophyte, N-America

First record for Basilicata (casual)

Specimen: 25-5-2017, M. Vulture, Foggianello (Rionero in Vulture, Potenza province), forest edges and road side, 570 m a.s.l., 40.96473°N-15.59563°E, *leg. et det.* E. Fanfarillo *et* L. Rosati (HLUC)

Note. Previously reported from all Italian regions except for Molise and Basilicata [17].

Hieracium pallescens **Waldst. & Kit.** ***s.l.***

Asteraceae Hemicryptophyte scapose SE-Europe

First record for Lazio

Specimen: 29-7-2015, Rifugio Sebastiani, Monte Terminillo (Rieti province), pastured grassland dominated by *Bromopsis erecta* with *Helictochloa praetutiana*, *Brachypodium genuense*, *Carex kitaibeliana* and *Festuca violacea* subsp. *italica*, limestone, 1870 m a.s.l., 13.01343°N-42.46902°E, *leg*. E. Del Vico *et* L. Facioni, *det*. G. Gottsclich (Herb. Del Vico).

Note. This species is present in all the northern regions of Italy except for Liguria [16]. In peninsular Italy it was recorded only from Abruzzo, Campania, Basilicata and Calabria [16]. About twenty subspecies of *H. pallescens* are present in Italy; however our specimen does not allowed to ascertained the subspecific level. Further on filed investigations should be performed to fix this gap.

Himantoglossum hircinum **(L.) Spreng.** [≡ *Satyrium hircinum* L.]

Orchidaceae Geophyte bulbouse Mediterranean-Atlantic

First record for Sardegna

Specimen: 22-5-2013, Illorai forest (Illorai, Sassari province), within a small clearing in otherwise dense *Quercus ilex* and *Q. pubescens* forest, 40.40494°N-8.92588°E, photo *et det*. P.J. de Lange.

Note. This species is present in Italy in the north-western (Piemonte, Liguria, Toscana and Emilia-Romagna) and in the southern regions with a clear gap in central Italy [16]. Although it is a very showy orchid it was never seen before in Sardegna, probably due to its rarity.

Images available at: https://inaturalist.nz/observations/4060025

Impatiens noli-tangere **L.**

Distribution clarification for Basilicata

Balsaminaceae Therophyte scapose Eurasian

Specimen: 31-7-2015, Fossa Cupa (Sasso di Castalda, Potenza province), *Fagus sylvatica* forest, 40.50972°N-15.73002°E, *leg*. V.A. Romano, *det*. L. Rosati (HLUC).

Note. This species was reported first from Basilicata by Bartolucci et al. [16] based on our unpublished record above. Here we provide full details of the discovery.

Isoëtes durieui **Bory**

Isoëtaceae Geophyte bulbouse W-Steno-Mediterranean

First record for Basilicata

Specimen: 2-5-2019, Bosco Finocchio (Rotondella, Matera province), ponds within *Quercus cerris* forest, 600 m a.s.l., 40.15282°N-16.47588°E, *leg*. G. Salerno *et* G. Misano, *det*. G. Salerno (Herb. Salerno).

Note. It was previously recorded for all western Italian regions though considered doubtfully in Molise [16].

***Isopyrum thalictroides* L.**

Ranunculaceae Geophyte rhizomatous Eurasian

First record for Sardegna

Specimen: 15-6-2014, Fontana Is Bidileddos (Desulo, Nuoro province), riparian forest of *Glechomo sardoae-Alnetum glutinosae*, 1660 m a.s.l., 40.01029°N-9.309394°E, *leg. et det*. E. Farris (SS).

Note. Locally common on the Fonni mountain side, exclusively in *Alnus glutinosa* forests above 1500 m a.s.l. (see https://inaturalist.nz/observations/4112222). This taxon is present throughout Italy, in most of the central-northern regions of Italy and though absent south of Toscana [16]. Our finding is the first for Mediterranean islands. This is noteworthy as it is believed to be absent from Corsica [72].

***Kalanchoë × houghtonii* D.B.Ward** [≡ *Bryophyllum ×houghtonii* (D.B.Ward) P.I.Forst.; *Kalanchoë daigremontiana* Raym.-Hamet & H.Perrier × *Kalanchoë delagoensis* Eckl. & Zeyh.]

Crassulaceae Ch succ Neophyte, horticultural origin

First record for Basilicata (naturalized)

Specimen: 3-8-2017, Porto (Maratea, Potenza province), calcareous sea cliffs, 6 m a.s.l., 39.98835°N-15.70995°E, *leg*. A. Stinca *et* M. Eviano. *det*. A. Stinca (PORUN-Herb. Stinca).

Note. At Maratea *Kalanchoë* × *houghtonii* seems to form populations capable of self-sustaining reproduction. Indeed, because of the vivaporous nature, whereby the leaf lamina produces myriad plantlets this species has the potential to be highly invasive. It is recorded as naturalized from Liguria, Toscana, Lazio and Calabria; it is present as casual in Campania, Puglia, Sardegna and Sicilia [17].

Kickxia spuria* (L.) Dumort. subsp. *spuria [≡ *Antirrhinum spurium* L. subsp. *spurium*]

Plantaginaceae Therophyte scapose Eurasia*n*

First record for Campania

Specimen: 29-7-2014, near Vallone Serra Melito (Capaccio Paestum, Salerno province), arable land, 40.41596°N-15.08801°E, *leg. et det*. Stinca (PORUN-Herb. Stinca).

Note. This is the first record for the Southern Italian peninsula [16].

***Lantana montevidensis* (Spreng.) Briq.** [≡ *Lippia montevidensis* Spreng.]

Verbenaceae Nano-phanerophyte Neophyte, S-American

First record for Sardegna (casual)

Specimen: 30-8-2018, Su Giudeu (Domus de Maria, Cagliari province), crevices in the walls, 5 m a.s.l., 38.88749°N-8.85886°E, *leg. et det*. L. Rosati *et* M. Marignani (HLUC).

Note. Previously reported as a casual alien only from Liguria and Calabria regions [17]. This species is widely grown in urban green areas, pots and Mediterranean gardens (pers. observ.) so further naturalizations are likely.

***Lathyrus cirrhosus* Ser.**

Fabaceae Hemicryptophyte NW-Steno-Mediterranean

Exclusion for Italy (Sardegna)

Note. This taxon was reported only once at Monte Arviganu and Monte S. Giuliano, near Alghero, Sardegna by Schmid [49]. Pignatti [19,47] and Arrigoni. [73] already considered the presence of *L. cirrhosus* in Sardegna doubtful, due to probable confusion with *L. heterophyllus* L. Nevertheless, it was considered present in Sardegna in the Italian Flora checklists of vascular plants [74,16]. We have unsuccessfully searched for this species in the two sites during the last decade, and in the absence of verified herbarium evidence of its historic presence we recommend that *L. cirrhosus* be excluded from the Flora of Sardegna and Italy.

***Lathyrus inconspicuus* L.**

Fabaceae Therophyte scapose Euri-Mediterranean

First records for Lazio

Specimens: 11-6-2015, Monte Porillo (Micigliano, Rieti province), secondary grassland with *Bromopsis erecta* and *Brachypodium rupestre*, gently marly slopes, 1000 m a.s.l., 42.45552°N-13.06439°E, *leg*. E. Del Vico *et* L. Facioni, *det*. E. Del Vico (Herb. Del Vico); 28-6-2016, Micigliano (Rieti province), secondary grassland with *Bromopsis erecta* and *Brachypodium rupestre*, gently marly slopes, 1130 m a.s.l., 42.46011°N-13.06226°E, *leg. et det*. E. Del Vico (Herb. Del Vico); 8-6-2016, Monte Porillo (Micigliano, Rieti province), secondary grassland with *Bromopsis erecta* and *Brachypodium rupestre*, gently marly slopes, 1130 m a.s.l., 42.46301°N-13.05907°E, *leg*. E. Del Vico *et* S. Bonacquisti, *det*. E. Del Vico (Herb. Del Vico); 20-7-2016, Monte Porillo (Micigliano, Rieti province), secondary grassland with *Cynosurus cristatus*, 1110 m a.s.l., 42.46596°N-13.06396°E, *leg. et det*. E. Del Vico (Herb. Del Vico); 20-7-2016, Erta (Micigliano, Rieti province), secondary grassland with *Brachypodium rupestre*, marly substratum, 1110 m a.s.l., 42.46428°N-42.46428°E, *leg. et det*. E. Del Vico (Herb. Del Vico).

Note. In peninsular Italy *L. incospicuus* was previously recorded from Toscana, Marche, Abruzzo and Puglia; indeed in Umbria and Campania is considered doubtful present [16]. It is absent from southern and insular regions [16].

***Linum tenuifolium* L.**

Linaceae Chamaephyte subshrubs Steno-Mediterranean-Euxinian

Exclusion for Sardegna

Note. The presence of this taxon in Sardegna was reported by Mossa et al. [50] and so accepted in the Italian checklist of vascular flora [74,16]. Nevertheless, the species is excluded by [75] who considers its presence a mistake, due to the confusion with *L. bienne*. Our surveys in the site from where *Linum tenuifolium* was reported by Mossa et al. [50] resulted only *L. bienne.* Therefore, *L. tenuifolium* has to be excluded from Sardegna. It was also reported by mistake in Sicilia region [16].

Lonicera japonica **Thunb.**
Caprifoliaceae Phanerophyte climbing Neophyte, E-Asiatic
First record for Basilicata (casual)
Specimen: 14-6-2019, Macchia Romana, Campus of University of Basilicata (Potenza), crevices in road pavements close to the walls, 40.68829°N-15.80848°E, 735 m a.s.l, *leg. et det*. L. Rosati (HLUC).
Note. Lonicera japonica has been recorded from all Italian regions except for Molise and Basilicata [17]. It is considered invasive in several regions of Northern Italy [17].

Lupinus albus **L. subsp.** ***graecus*** **(Boiss. & Spruner) Franco & P.Silva** [≡ *L. graecus* Boiss. & Spruner]
Fabaceae Therophyte scapose E-Steno-Mediterranean
First record for Campania
Specimen: 22-4-2011, M. Vesole (Roccadaspide, Salerno province), chestnuts, 40.41538°N-15.17548°E, 500 m a.s.l., *leg. et det*. G. Salerno (Herb. Salerno).
Note. Previously reported from the Italian Peninsula for all of the regions, except Abruzzo, Molise and Campania [16].

Melampyrum barbatum **Waldst. & Kit. subsp.** ***carstiense*** **Ronniger**
Orobanchaceae Therophyte scapose SE-European
First record for Basilicata and confirmation for Puglia
Specimens: 9-6-1981, Trivigno (Potenza), oak forests, 500 m a.s.l., 40.58074°N-15.98965°E, *legit et det.* F. Cremonini (HLUC); 6-6-2018, V.ne Mariano (Laurenzana, Potenza province), edge of *Quercus pubescens* woods, siliceous substratum, 40.48337°N-15.96811°E, *leg*. L. Rosati *et* G. Potenza, *det*. L. Rosati (HLUC)*;* 20-7-2014, Foresta Umbra (Vico del Gargano, Foggia province), roadside of forest road and edge of forest clearing, 560 m a.s.l, 41.854115°N-16.001818°E , *leg. et det.* G. Mei (ANC, Herb. Mei).
Note. In Southern Italy previously known only from doubtful historic records [16].

Metrosideros excelsa **Sol. ex Gaertn.** [= *M. tomentosa* A.Rich.; = *Nania tomentosa* (A.Rich.) Kuntze]
Myrtaceae Phanerophyte caespitose Neophyte, New Zealand
First record for Europe (Azores and Madeira excluded) (casual)
Specimen: 31-5-2013, Costa Rei, Monte Nai (Muravera, Cagliari province), urban areas, 20 m a.s.l., 39.2565°N-9.575611°E, *leg. et det*. P.J. de Lange (AK, SS).

Note. A few seedlings, and saplings growing in vicinity of planted adult trees; this species is common in many coastal towns and resorts around Sardegna. *Metrosideros excelsa*, commonly known as "New Zealand Christmas tree", is a coastal evergreen tree that produces a brilliant display of red flowers. It has been recently added to the list of host plants referred to in Article 1(b) of Commission Implementing Decision (EU) 2015 2417 of 17 December 2015, which have been found to be susceptible to *Xylella fastidiosa* in the European Union territory.

Until this record *Metrosideros excelsa* had been recorded in Europe as a naturalized plant only from the Azores and Madeira (Portugal) by [76]. Thus, our finding represents the first record for Europe (Azores and Madeira excluded).

Muscari parviflorum **Desf.**

Asparagaceae Geophyte bulbouse Central-E-Mediterranean

Confirmation for Calabria

Specimen: 10-10-2011, Orsomarso (Cosenza), fallows, 488 m a.s.l., 39.78545°N-15.90759°E, *leg. et det* S. Fascetti (HLUC).

Note. Previous records of this species from Calabria had been considered doubtful [16].

Nicandra physalodes **(L.) Gaertn.** [≡ *Atropa physalodes* L.]

Solanaceae Therophyte scapose Neophyte, S-American

First record for Calabria (casual)

Specimen: 3-6-2019, S. Litterata (Belvedere Marittimo, Cosenza province), fallows, 35 m a.s.l., 39.65594°N-15.83856°E, *leg. et det.* A. Capano (HLUC).

Note. This species was recorded in Italy from several regions: it is present in almost all of the central and northern regions as a casual alien; in Piemonte, Trentino-Alto Adige and Friuli-Venezia Giulia it is considered naturalized [17]. It is absent in the most of the southern regions, except for Sicilia [17].

Nigella arvensis **L. subsp.** ***glaucescens*** **(Guss.) Greuter & Burdet** [≡ *N. arvensis* L. var. *glaucescens* Guss.]

Ranunculaceae Therophyte scapose Euri-Mediterranean

First record for Basilicata and confirmation for Puglia

Specimens: 15-4-2019, Padula (Ferrandina, Matera province), fallows, 40,5237°N-16,39444°E, *leg*. G. Marsilio, *det*. L. Rosati (HLUC)*;* 15-4-2019, Accetta Grande (Statte, Taranto province), herbaceous fallows, 500 m a.s.l., 40.55389°N-17.17667°E, *leg*. G. Santoro, *det*. G. Santoro *et* L. Rosati (HLUC).

Note. Previously it was only recorded from Calabria, Sicilia and Sardegna; from Puglia there are not recent records [16]. The presence of this taxon in Basilicata (Matera and Potenza Province) was also reported in the Forum Acta Plantarum ("*Nigella arvensis* subsp. *glaucescens* (Guss.) Greuter & Burdet - Scheda IPFI, Acta Plantarum". Available on line (access date: 02-10-2019): http:\\www.actaplantarum.org\flora\flora_info.php?id=5297).

***Nothoscordum gracile* (Aiton) Stearn** [≡ *Allium gracile* Aiton]

Amaryllidaceae Geophyte bulbouse Neophyte, S-America

First record for Calabria (casual)

Specimen: 3-5-2019, Lazzaro (Motta San Giovanni, Reggio Calabria province), crevices in the sidewalks, 17 m a.s.l., 37.97408°N-15.66485°E, *leg. et det.* C.M. Musarella (REGGIO).

Note. Probably escaped from cultivated plants growing in pots placed on the sidewalks from where it had spread into crevices and the edge of the roadside. It was recorded as naturalized in Toscana, Campania and Puglia [17] and as casual alien in Piemonte, Liguria and Sardegna [17].

***Oeosporangium tinaei* (Tod.) Fraser-Jenk.** [≡ *Cheilanthes tinaei* Tod.; = *C. corsica* Reichst. & Vida]

Pteridaceae Hemicryptophyte rosulate Steno-Mediterranean-Macaronesian

First record for Lazio

Specimen: 23-5-2012, Cerveteri (Roma province), trachytes steep slope, 60 m a.s.l., 42.00000°N-12.10000°E, *leg. et det.* E. Del Vico *et* L. Facioni (HRO).

Note. Although new to the Lazio region this West steno-Mediterranean species has been recorded in all Italian peri-Thyrrhenian regions, except for Campania [16].

***Oloptum miliaceum* (L.) Röser & H.R.Hamasha** [≡ *Agrostis miliaceum* L.; ≡ *Oryzopsis miliacea* (L.) Asch. & Schweinf.; ≡ *Piptatherum miliaceum* (L.) Coss.]

oaceae Hemicryptophyte caespitose Mediterranean-Turanian

First records for Sardegna

Specimens: 2-6-2018, Sirigragiu, Corongiu (Maracalagonis, Cagliari province), reforestation with *Eucaliptus*, 84 m a.s.l., 9.2791°E-39.28434°N, *leg. et det*. E. Lallai (Herb. E. Lallai); 7-6-2018, C. D'Aquila, Stagno di Quartu (Quartu Sant'Elena, Cagliari province), olive grove, 3 m a.s.l., 9.17736°E-39.22613°N, *leg. et det*. E. Lallai (Herb. E. Lallai); 14-6-2018, Corti de Perda (Quartucciu, Cagliari province), reforestation with *Eucaliptus* and *Pinus* spp., 135 m a.s.l., 9.32868°E-39.24517°N, *leg. et det.* E. Lallai (Herb. E. Lallai).

Note. In the recent checklist of Italian vascular flora only *O. thomasii* and not *O. miliaceum* has been reported from Sardegna [16]. However, *O. miliaceum* is the only species that has ever been considered present in the flora treatments of Sardegna e.g. [47,74-75]. In addition, several authors consider questionable the distinction between these two taxa and include *O. thomasii* in *O. milaceum*. Our observations confirm the presence of *O. miliaceum,* accordingly this name has to be re-added to the checklist of vascular flora of Sardegna. *O. miliaceum* is present in almost all Italian regions except Valle d'Aosta, Piemonte e Molise [16].

***Ophrys crabronifera* Mauri** [≡ *O. argolica* H.Fleischm. ex Vierh. subsp. *crabronifera* (Mauri) Faurh.; = *O. exaltata* Ten. subsp. *sundermannii* Soó; = *O. fuciflora* (F.W.Schmidt) Moench subsp. *sundermannii* Soó]

Orchidaceae Geophyte bulbouse Italian endemic

First record for Basilicata

Observation: 15-4-2015; 30-4-2019, along the road S.P. 12 Vietrese (Vietri di Potenza, Potenza province), roadside on calcareous substratum, 371 m a.s.l., 40.57086°N-15.52102°E, photo *et det.* VA Romano (Figure 2).

Note. This species was previously recorded in almost all the peninsular regions except for Puglia, Calabria and Basilicata. This observation extends southward the range of this endemic taxon in Italy [16]. It is absent form insular regions [16].

Ophrys marmorata **G.Foelsche & W.Foelsche** [≡ *Ophrys fusca* Link subsp. *marmorata* (G.Foelsche & W.Foelsche) Kreutz]

Orchidaceae Geophyte bulbouse Italian endemic

First record for Basilicata

Observation: 14-2-2015, Lido di Scanzano Jonico (Scanzano Jonico, Matera province), *Pinus halepensis* forest, 10 m a.s.l., 40.26114°N-16.75518°E, photo *et det.* S. Fascetti, V.A.Romano (Figure 2).

Note. Since 2005, a few isolated flowering plants have been observed at this site flowering between February and March. The new find extends southward the range of this endemic taxon in Italy; previously it had only been recorded from Toscana [16].

Ophrys sipontensis **(Gumpr.) O.Danesch & E.Danesch** [≡ *O. sphegodes* Mill. subsp. *sipontensis* Gumpr.; ≡ *O. garganica* O.Danesch & E.Danesch subsp. *sipontensis* (Gumpr.) Del Prete]

Orchidaceae G bulbouse Italian endemic

First record for Basilicata

Observation: 25-4-2015, Balvano (Potenza province), shrublands with *Spartium junceum*, 726 m a.s.l., 40.67352°N-15.52065°E, photo *et det.* S. Fascetti S. *et* V.A. Romano (Figure 2).

Note. Since 2002, at this location only a few isolated flowering plants have been observed flowering between April and May. Hitherto this endemic orchid has only been recorded from Campania and Puglia regions[16].

Ophrys speculum **Link** [= *O. ciliata* Biv.]

Orchidaceae G bulbouse W-Steno-Mediterranean

Confirmation for Calabria

Observation: 15-4-2010, Frascineto (Cosenza province), roadside, 39.801091°N-16.241693°E, 420 m a.s.l., photo H.Presser, *det.* V.A. Romano (Figure 2).

Note. Previously reported from southern Italy only from Campania and Molise, it had otherwise been regarded as doubtfully present in Calabria[16].

Oxalis debilis **Kunth** [= *O. corymbosa* DC.]

Oxalidaceae G. bulbouse Neophyte, S-American

First record for Calabria (casual)

Specimens: 3-5-2019, Lazzaro (Motta San Giovanni, Reggio Calabria province), crevices in the road pavements, 17 m a.s.l., 37.97405°N-15.66492°E, *leg. et det.* C.M. Musarella (REGGIO); 25-09-2019, Catona (Reggio Calabria, Reggio Calabria province), along the roadside, 15 m a.s.l., 38,189334°N-15,643462°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface *et* C.M. Musarella (REGGIO).

Note. At these locations only a few plants were recorded growing in the crevices and at the edge of the roadside; these probably escaped from plants cultivated in pots on the sidewalks. It is reported for several regions of Italy as a naturalized or casual alien [17]; it is considered doubtfully present in Lazio, Basilicata and Puglia[17].

Oxalis latifolia **Kunth**

Oxalidaceae Geophyte bulbouse Neophyte, Americ.

First record for Calabria (casual)

Specimens: 10-5-2018, Tortora Marina (Tortora, Cosenza province), fallows close to a plant nursery, 12 m a.s.l., 39.92276°N-15.76723°E, *leg. et det.* S. Fascetti (HLUC); 29-7-2019, Catona (Reggio Calabria, Reggio Calabria province), sidewalks, 15 m a.s.l., 38.187390°N-15.643697°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface, C.M. Musarella *et* G. Spampinato (REGGIO); 29-7-2019, Porelli di Bagnara SS18 (Bagnara Calabra, Reggio Calabria province), along the roadside, 142 m a.s.l., 38.286887°N-15.810304°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface, C.M. Musarella, G. Spampinato (REGGIO); 25-9-2019, Ferrito SS. 18 (Villa San Giovanni, Reggio Calabria province), along the roadside, 38 m a.s.l., 38.233512°N-15.660117°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface, C.M. Musarella, G. Spampinato (REGGIO); 27.07.2019, Catona (Reggio Calabria, Reggio Calabria province), sidewalks, 10 m a.s.l., 38.188338°N-15.641358°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface, C.M. Musarella, G. Spampinato (REGGIO); 25.09.2019, Melia di Scilla (Melia di Scilla, Reggio Calabria province), 653 m a.s.l., 38.228695°N-15.757547°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface, C.M. Musarella (REGGIO); 28.09.2019, Pietrastorta (Reggio Calabria, Reggio Calabria province), along the roadside, 317 m a.s.l., 38.113405°N-15.684027°E, *leg. et det*. V.L.A. Laface (REGGIO); 21.10.2019, Molochio (Molochio, Reggio Calabria province), along the sidewalks, 316 m a.s.l., 38.310638°N-16.029284°E, *leg. et det.* V.L.A. Laface (REGGIO).

Note. Abundantly naturalized. In some stations of Calabria, it probably escaped from plants grown in pots cultivated on the sidewalks as plants were seen in nearby crevices and on the nearby roadside. It was previously reported for almost all the Italian regions except for Molise, Basiicata and Calabria in southern Italy [17]. In Basilicata region it is reported in the forum Acta Plantarum (2007 on - "*Oxalis latifolia* Kunth - Scheda IPFI, Acta Plantarum". Available on line (access date: 16/10/2019): http://www.actaplantarum.org/flora/flora_info.php?id=502806).

Paspalum notatum **Flüggé** [= *P saurae* (Parodi) Parodi]

Poaceae Hemicryptophyte caespitose Neophyte, Meso and S-American

First record for Basilicata (casual)

Specimen: 3-8-2017, Porto (Maratea, Potenza province), shores, 1 m a.s.l., 39.98927°N-15.70905°E, *leg*. A. Stinca *et* M. Eviano. *det*. A. Stinca (PORUN-Herb. Stinca).

Note. Hitherto reported for Italy from the Calabria region as a casual alien at Scalea [77].

***Pedicularis petiolaris* Ten.**

Orobanchaceae Hemicryptophyte scapose E-Mediterranean

Confirmation for Campania

Specimen: 27-5-2014, Mandranello (Padula, Salerno province), carbonatic dry grasslands, 1100 m a.s.l., 40.38762°N-15.68548°E, *leg. et det*. L. Rosati (HLUC).

Note. In Italy it was previously reported only for Abruzzo, Basilicata and Calabria regions[16]. In Campania it had previously only been considered doubtful present[16].

***Pittosporum crassifolium* Banks & Sol. ex A.Cunn.**

Pittosporaceae Nano-phanerophyte Neophyte, New Zealand

First record for Italy (casual)

Specimen: 11-03-2013, Sardegna, Lungomare Valencia (Alghero, Sassari province), at back of disco bar near former castle, 9 m a.s.l., 40.55058°N-8.31974°E,. *leg. et det*. P.J. de Lange (AK)

Note. Associated with the superficially similar *Pittosporum tobira* (by far the most commonly grown and naturalized *Pittosporum* species in Sardegna). *Pittosporum crassifolium* was only seen here and at Torre Nuova (Alghero). Seedlings occasional, these appear to be pure *Pittosporum crassifolium*. In Europe it has previously recorded only from Great Britain [60].

***Phoenix canariensis* H.Wildpret**

Arecaceae Phanerophyte scapose Neophyte, Macaronesian

First record for Basilicata (casual)

Specimens: 14-12-2017, S.Basilio, Marina di Pisticci (Pisticci, Matera province), roadside along *Pinus halepensis* forests, 5 m a.s.l., 40.29815°N-16.77887°E, *leg. et det.* S. Fascetti (HLUC); 25-08-2019, Pisticci (Matera province), road escarpment of SS Jonica 106, 21 m a.s.l., 40.325447°N-16.746782°E, leg. et det. C.M. Musarella (REGGIO).

Note. In loc. S. Basilio a few juvenile plants were observed probably spread from nearby gardens. In loc. Pisticci two mature individuals, away from each other about 50 m, were recorded at the edge of a drainage channel along the SS Jonica 106, outside of a lay-by. *Phoenix canariensis* is reported as casual or naturalized for almost all coastal regions of peninsular Ital [17].

***Phyla nodiflora* (L.) Greene** [≡ *Lippia nodiflora* (L.) Michx.; ≡ *Verbena nodiflora* L.]

Verbenaceae Neophyte, Tropical America, Africa and Asia

Confirmation for Marche (casual)

Specimen: 3-8-2019, Osteria (Serra de' Conti, Ancona province), lawn borders and slopes of agricultural ditches for rainwater runoff, 135 m a.s.l., 43.541875°N-13.020947°E, *leg. et det.* G. Mei (ANC, Herb. Mei).

Note. Previously *P. nodiflora* was reported as naturalized from almost all the peninsular regions but it was considered as doubtfully present in Marche [17].

***Phyllostachys aurea* Carrière ex Rivière & C. Rivière**

Poaceae Phanerophyte caespitose Neophyte, E-Asian

First record for Calabria (casual)

Specimens: 2-7-2019, Pellaro, Torrente Fiumarella (Reggio Calabria, Reggio Calabria province), dry riverbed, 6 m a.s.l., 38.02138°N-15.64416°E, *leg. et det*. C.M. Musarella (REGGIO); 2-9-2019, C.da Monacelli, (Melicucco, Reggio Calabria province), roadside, 118 m a.s.l., 38.44528°N-16.04166°E, *leg. et det.* C.M. Musarella (REGGIO); 9-8-2019, C.da Sant'Angelo (Melia di San Roberto, Reggio Calabria province), road escarpment, 749 m a.s.l., 38.22281°N-15.77548°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface *et* G. Spampinato (REGGIO); 19.10.2019, Fiumara di S. Lorenzo (S. Lorenzo, Reggio Calabria province), bedriver, 6 m a.s.l., 37.920494°N-15.839534°E, *leg. et det*. V.L.A. Laface (REGGIO).

Note. In southern Italy it was previously recorded only for Campania (naturalized) and Sicilia (casual) [17].

***Physalis peruviana* L.**

Solanaceae Hemicryptophyte scapose Neophyte, S-American

First record for Calabria (casual)

Specimen: 23-8-2019, Thurio (Corigliano Calabro, Cosenza province), fallows, 8 m a.s.l., 39.69515°N-16.46777, *leg*. P. Nicoletti, *det*. L. Rosati (HLUC).

Note. In southern Italy recorded previously only from Puglia and Sicilia regions[17].

***Pinus nigra* J.F.Arnold subsp. *laricio* Palib. ex Maire** [= *P. laricio* Poir. subsp. *calabrica* (Loud.) Cesca & Peruzzi]

Pinaceae Phanerophyte scapose Italo-Corsican Endemic

First record for Basilicata (casual) *and Campania* (casual)

Specimens: 15-05-18, Mandrano, Monti della Maddalena (Paterno, Potenza province), pastures and roadsides, 1110 m a.s.l., 40.36917°N-15.71119°E, *leg. et det*. L. Rosati (HLUC); 15-05-18, Mandrano, Monti della Maddalena (Padula, Salerno), pastures and roadsides 1075 m a.s.l., 40.36901°N-15.70758°E, *leg. et det.* L. Rosati (HLUC).

Note. Several young trees deriving from nearby reforestation areas where this species had been deliberately planted were observed. It was recorded as alien taxa in others Italian regions (Sardegna, Fruli-Venezia Giulia, Toscana), whereas it is considered native only in Calabria and Sicilia [16].

Plantago argentea* Chaix subsp. *argentea

Plantaginaceae Hemicryptophyte rosulate S-Europe-S-Siberian

First record for Campania

Specimen: 20-7-2014, M. Alburno under M. Panormo (Sicignano degli Alburni, Salerno province), rocky slopes, 1402 m a.s.l., 40.53226°N-15.33386°E, *leg*. A. Stinca *et* M. Ravo, *det*. A. Stinca (PORUN-Heb. Stinca).

Note. Plantago argentea was previously indicated in sensu lato in Campania, thus, here we report the presence of the nominal subspecies[16].

***Plumbago auriculata* Lam.** [= *Plumbago capensis* Thunb.]

Plumbaginaceae Nano-phanerophyte Neophyte, S-African

First record for Calabria (casual)

Specimen: 02-09-2019, C.da Monacelli (Melicucco, Reggio Calabria), roadside, 117 m a.s.l., 38.445020°N-16.042060°E, *leg. et det*. C.M. Musarella (REGGIO).

Note. In Calabria, we observed a few individuals growing along the roadside; these had probably escaped from a nearby garden. It is reported as casual alien from almost all the regions of Italian peninsula, except for Umbria, Molise and Basilicata; in Sardegna it is considered naturalized [17].

***Polygala rupestris* Pourr.**

Polygalaceae Chamephyte subshrubs W-Mediterranean

First record for Sardegna and confirmation for Italy

Specimens: 02-11-2005; 14-05-2014; 08-04-2016, Canale Oma Molt (Alghero, Sassari province), garrigues with *Rosmarinus officinalis* and *Viola arborescens*, 33 m a.s.l., 40.52452°N-8.33371°E, *leg. et det.* E. Farris (SS).

Note. In the recent checklist of the Italian vascular flora the presence of this species was considered doubtful in Italy[16]. We found this species in Sardegna along the coast South of Alghero, where it was usually present, though at low densities in the calcareous coastal garrigues. The nearest known localities of this species, according to Flora Gallica and Flora Iberica are in Provence (S-France) and Balearic Islands (Spain).

***Populus × canescens* (Aiton) Sm.** [≡ *P. alba* L. var. *canescens* Aiton]

Salicaceae Phanerophyte scapose S-European

First record for Sardegna

Specimens: 21-6-2016, Settefratelli, along the road from Codoleddu to the giant's tombstone (Maracalagonis, Cagliari province), near the creek, 465 m a.s.l., 39.26022°N-9.37907°E, *leg. et det*. L. Rosati (HLUC); 13-5-1981, Mandas (Mandas, Cagliari province), uscita dal paese verso Sorgono, 30 m a.s.l., 39.65376N, 9.12843°E, *leg. et det*. I. Camarda (SS); 23-5-1975, Piana di Posada (Nuoro province), 5 m a.s.l., 40.635480°N-9.72371°E, *leg.* S. Vanelli *et* B. Corrias (SS); 01-06-2018, Sedda Brandanu, Corongiu (Maracalagonis, Cagliari provinca), remnants of riparian forest, 140 m a.s.l., 39.29282°N-9.29236°E, *leg. et det*. E. Lallai (Herb. E. Lallai).

Note. In North Sardegna this species was also reported in a vegetation survey from the Cala Regina (Isola dell'Asinara, Sassari province) by [78]. In Italy it is widespread and known from all regions except for Liguria. This hybrid was reported for Sardegna by Camarda and Valsecchi [79] and Arrigoni [80] but not subsequently considered (by mistake?) in the recent checklist of vascular flora [74,16]. In the last checklist of the Italian vascular flora [16] it is not indicated as a hybrid (i.e. *P. canescens* (Aiton) Sm).

Potamogeton pusillus **L.**

Potamogetonaceae Idrophyte rooted Subcosmopolitan

Confirmation for Campania

Specimen: 16-7-2016, Sele river, Oasi Persano (Serre, Salerno province), lotic freshwaters, 40.60488°N-15.13717°E, *leg. et det*. L. Rosati *et* G. Salerno, *rev*. L. Lastrucci (HLUC).

Note. Outside Campania this species is not known from the southern Italian Peninsula [16]; it was recently confirmed from Basilicata [39].

Prangos ferulacea **(L.) Lindl.** [≡ *Laserpitium ferulaceum* L.; ≡ *Cachrys ferulacea* (L.) Calest.]

Apiaceae Hemicryptophyte scapose Mediterranean-Turanian

Confirmation for Lazio

Specimen: 16-7-2015, Monte Tilia, Fondo del Laghetto (Leonessa, Rieti province), secondary grasslands with *Brachypodium genuense*, marly slopes, 1400 m a.s.l., 42.55842°N-12.95441°E, *leg. et det*. E. Del Vico (Herb. Del Vico).

Note. Doubtfully reported from Lazio on the basis of a historic record (1914) from Terminillo of Cavara and Grande which had not been confirmed by recent surveys [81]. On the basis of the record above *Prangos ferulacea* is now verified from the Lazio region. This species is, with the exception of Molise, otherwise known from the other regions of the Italian Peninsula [16].

Puccinellia festuciformis **(Host) Parl. subsp. *festuciformis*** [≡ *Poa festuciformis* Host]

Poaceae Hemicryptophyte caespitose Steno-Mediterranean

Confirmation for Campania

Specimen: 12-6-2014, Patria lake, close to the dewatering pump (Giugliano in Campania, Napoli province), temporary flooded grasslands, 0 m a.s.l. 40.94181°N-14.0235°E, *leg. et det*. A. Stinca *et* G. Salerno (PORUN-Herb. Stinca).

Note. Until this discovery the presence of this grass in Campania had been considered doubtful [16]. Outside Campania this *Puccinellia* had been reported from several regions of the Italian Peninsula and considered as extinct in Abruzzo [16].

Quercus petraea **(Matt.) Liebl. subsp. *austrotyrrhenica*** **Brullo, Guarino & Siracusa**

Fagaceae Phanerophyte scapose Italian endemic

First record for Basilicata

Specimens: 15-6-2015, M.te Serranetta (Abriola, Potenza province), oak forests, 1300 m a.s.l., 40.56351°N-15.81886°E, *leg. et det*. S. Fascetti (HLUC); 15-8-2002, Serra di Calvello (Calvello, Potenza province), oak forests, 1300 m a.s.l., 40,45683°N-15,77855°E, *leg. et det*. M.R. Lapenna (HLUC).

Note. This Italian endemic is uncommon in Basilicata, where it is generally found in mixed forests with *Quercus cerris*, in relatively xeric habitats on mountain ridges, exclusively on subacid substrata (e.g., Flysch Galestrino). Previously it was recorded only from Calabria and Sicilia regions[16].

***R.* × *bishopii* Wolley-Dod** [= *Rosa micrantha* Borrer ex Sm. × *R. agrestis* Savi]

Rosaceae Nano-phanerophyte

First record for Basilicata

Specimen: 3-6-2011, Fiumara di Laurenzana (Laurenzana , Potenza province), shurbland in dry riverbed, 577 m s.l.m., 40.49004°N-15.9583°E, *leg. et det* S. Fascetti, *rev*. E. Lattanzi (HLUC).

Note. Several hybrids into the wild dog roses group have been reported mainly in north-central Europe [82]; on the other hand, in southern Europe, knowledge concerning existence and distribution of these hybrids is very scarce. As a consequence chorology for this taxon is not defined.

***Rosa montana* Chaix × *R. villosa* L.**

Rosaceae Nano-phanerophyte

First record for Basilicata

Specimen: 25-6-2016, Piano Jannace (San Severino Lucano, Potenza province), pastures, 1679 m a.s.l., 39.93977°N-16.19668°E, *leg. et det* S. Fascetti, *rev*. E. Lattanzi (HLUC).

Note. Rosa villosa is one of the species of sect. *Caninae* DC. that can easily hybridize with other sympatric dog roses [83]. According to some authors, it is probably the most common hybridogenic species among the dog roses [84]. To the best of our knowledge, a binomial name for this hybrid does not exist yet. As a consequence chorology for this taxon is not defined.

***Rosa subcollina* (Christ) Vuk.** [≡ *R. coriifolia* Fr. f. *subcollina* Christ]

Rosaceae Nano-phanerophyte Europe-W-Asian

First record for Basilicata

Specimen: 16-6-2011, Fossa Cupa (Pignola, Potenza province), shrublands, 1000 m a.s.l., 40.55197°N-15.73003°E, *leg. et det*. S. Fascetti, *rev*. E. Lattanzi (HLUC).

Note. It is recorded from most of the Italian Peninsula regions.

***Rumex maritimus* L.**

Polygonaceae Therophyte scapose Eurasian

First records for Campania

Specimens: 23-5-2014, near the harbour (Pozzuoli, Napoli province), sea sands, 0 m a.s.l., 40.82935°N-14.11412°E, *leg. et det*. A. Stinca (PORUN-Herb. Stinca); 2-6-2014, Ponte del Diavolo on the banks of the Patria lake (Castel Volturno, Napoli province), artificial river bank, 1 m a.s.l., 40.92918°N-14.0287°E, *leg*. A. Stinca *et* G. Salerno, *det*. A. Stinca (PORUN-Herb. Stinca); 3-6-2014, along the Canale Vico Patri close by Masseria De Chiara (Villa Literno, Napoli province), artificial river bank, 3 m a.s.l, .40.95818°N-14.04141°E, *leg*. A. Stinca *et* G. Salerno, *det*. A. Stinca (PORUN-Herb. Stinca).
Note. Previously recorded only from Calabria in southern Italy [16].

***Salvinia minima* Baker**

Salviniaceae Idrophyte floating Neophyte Meso-S-American
First record for Europe (casual)
Specimen: 1-7-2019, Petile di Calanna (Calanna, Reggio Calabria province), impluvium, 38.19045°N-15.73117°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface *et* C.M. Musarella (REGGIO).
Note. Salvinia minima is a non-rooted aquatic fern native to South and Central America. Madeira et al. [85] report this species as locally introduced for Spain, citing Lawarlee in Flora Europaea [42]. However, *S. minima* is not considered in Flora Europaea, neither in Flora Iberica [86]. For this reason, it may be that this record for Calabria is not the first for Italy but also for Europe. In this context, a case of accidentally introduction of *Salvinia minima* in Europe has been only documented in the botanical garden greenhouses in Jibou, Romania [87]. Detailed images of plants collected in Calabria are showed in Figure 4. The plants collected in Calabria were identified using [88-91]; diacritical characters useful to distinguish *S. minima* respect to the most similar taxa are showed in Table 2.

Table 2. Distinguishing characters between *Salvinia minima* and the most similar species (data retrieved from [91] including measures taken on our samples collected in Calabria.

	Floating blade length (mm)	**Hairs on papillae**	**Length/ width ratio**	**Papillae on upper face (mm)**
S. adnata Desv. (= *S. molesta* D.S. Mitch.)	15-25	joined into dark knot at tips	< 1.5	≤ 1
S. oblongifolia Mart.	25-45	not joined at their tips	2-3	< 0.1
S. minima Baker	5-10(15)	not joined at their tips	< 1.5	≤ 1
S. minima Baker (our sample)	6-9	not joined at their tips	< 1.5	≤ 1

***Sempervivum tectorum* L.**

Crassulaceae Chamephyte S-European

First record for Basilicata

Specimen: 15-6-2019, M. Volturino (Marsico Vetere, Potenza province), siliceous rocks and cliffs, 1500 m a.s.l., 40.4026°N-15.81642°E, *leg*. L. Rosati *et* G. Potenza, *det*. L. Rosati (HLUC).

Note. Present in central and northern regions of Italy but in southern regions, records of it from Puglia, Basilicata and Calabria have been treated as mistakes[16]. Therefore, the discovery of Mt. Volturino reported here now confirms its presence in the southern Apennine.

***Solandra maxima* (Sessé & Moc.) P.S.Green** [≡ *Datura maxima* Sessé & Moc.]

Solanaceae Phanerophyte climbing Neophyte, Mesoamerican

First record for Calabria (casual) *and Sicilia* (casual)

Specimen: 9-8-2019, Ferrito (Villa San Giovanni, Reggio Calabria province), roadside, 34 m a.s.l., 38.23229°N-15.64493°E, *leg*. V.L.A. Laface, *det*. V.L.A. Laface, C.M. Musarella *et* G. Spampinato (REGGIO); 5-7-2019, Canalicchio (Catania), road escarpment, 91 m a.s.l., 37.53119°N-15.09410°E, *leg.* C.M. Musarella, *det*. C.M. Musarella *et* G. Spampinato (REGGIO).

Note. Solandra maxima was recorded for the first time in Europe by Salerno and Stinca [92] from the Campania region. Our two records are therefore the second for the Italian peninsula and the first for Calabria and Sicilia respectively.

***Tagetes erecta* L.** [= *T. patula* L.]

Asteraceae Therophyte scapose Neophyte, Mesoamerican

First record for Basilicata (casual)

Specimen: 3-8-2017, Porto (Maratea, Potenza province), crevices in road pavements, 16 m a.s.l., 39.98889°N-15.70958°E, *leg*. A. Stinca *et* M. Eviano. *det*. A. Stinca (PORUN-Herb. Stinca).

Note. This species has been recorded as casual alien from many Italian regions [17].

***Tetragonia tetragonoides* (Pall.) Kuntze** [≡ *Demidovia tetragonoides* Pall.]

Aizoaceae Therophyte scapose Neophyte, Australia and New Zealand

Confirmation for Sardegna (casual)

Specimen: 19-05-2013, Golfo di Orosei, Cala Gonone (Orosei, Nuoro province), cobble beach, 1 m a.s.l., 40.28270°N-9.63620°E, *leg. et det*. P.J. de Lange (AK).

Note. Flowering and fruiting plants have been observed (see https://inaturalist.nz/observations/1509928). In Italy it was previously reported in Friuli Venezia Giulia, Trentino-Alto Adige, Lombardia, Toscana, Lazio, Campania and Sicilia regions[17]. Prior to this record this species was known from Sardegna on the basis of old reports that required confirmation [17].

***Trifolium spumosum* L.**

Fabaceae Therophyte scapose E-Steno-Mediterranean

Confirmation for Basilicata

Specimen: 16-6-2014, Anglona Sanctuary (Tursi, Matera province), dry grassland, 262 m a.s.l., 40.2449°N-16.55653°E, *leg*. G. Salerno *et* M.R. Lapenna, *det*. G. Salerno (Herb. Salerno).

Note. The presence of this species in Italy was ascertained only from Sardegna, Sicilia, Calabria and Lazio[16]. In recent times this species has not been recorded from most of the Italian peninsular regions[16]. Our discovery from Basilicata confirms its continued presence from at least part of that region.

***Urginea fugax* (Moris) Steinh.** [≡ *Anthericum fugax* Moris]

Asparagaceae Geophyte bulbouse W-Mediterranean

Exclusion for continental and peninsular Italy (Basilicata)

Note. In Italy, we can only confirm this species as present in Sardegna [16], which is consistent with its W-Mediterranean distribution. Accordingly, we regard past records of the species from Basilicata and neighbouring regions as erroneous. Although Gavioli [51] reported the species from Basilicata in 1915 the samples he noted had been lost, so the record cannot be verified. In absence of further specimens of this distinctive plant from the Italian Peninsula we suggest that *U. fugax* had probably been reported by mistake, due to confusion with a similar species (likely small size individuals of *Urginea maritima* auct. Fl. Ital.). Therefore, we recommend that this species is excluded from the flora of Basilicata.

Veronica spicata* L. subsp. *fischeri (Trávn.) Albach [≡ *Pseudolysimachion spicatum* (L.) Opiz subsp. *fischeri* Trávn]

Plantaginaceae Hemicryptophyte crawling Eurasian

First records for Lazio

Specimens: 2-7-2015, Monte Rosato (Poggio Bustone, Rieti province), secondary grassland with *Bromopsis erecta*, *Carex humilis* and *Plantago argentea*, calcareous slopes, 1300 m a.s.l., 42.51662°N-12.88791°E, *leg. et det*. E. Del Vico (Herb. Del Vico); 10-7-2016, Monte Rosato (Poggio Bustone, Rieti province), secondary grassland with *Bromopsis erecta*, calcareous slopes, 1350 m a.s.l., 42.51387°N-12.88557°E, *leg*. E. Del Vico *et* L. Facioni *det*. E. Del Vico (Herb. Del Vico).

Note. In Italy this taxon has a Northern (Lombardia, Trentino-Alto Adige, Veneto) and Central (Marche, Umbria, Abruzzo) disjunct distribution [16]. In Piemonte it is recorded by mistake [16]. Subspecies identification of our samples was based on the key reported in [20]; however, in our view the subspecies delimitation of *V. spicata* as stated in [20] is unclear and needs further study.

***Viola kitaibeliana* Schult.**

Violaceae Therophyte scapose Europ-Caucasian

Confirmation for Campania

Specimen: 1-3-2016, Campitello, M. della Maddalena (Padula, Salerno province), pastures on siliceous schistous substrate, 1300 m a.s.l., 40.34573°N-15.69029°E, *leg. et det*. L. Rosati (HLUC).

Note. Viola kitaibeliana is recorded from most of the peninsular regions, except for Emilia-Romagna, Molise and Puglia; it was previously considered as doubtfully present in Campania [16]. Our discovery now fills this gap along the Apennine chain.

***Wisteria sinensis* (Sims) Sweet** [≡ *Glycine sinensis* Sims]

Fabaceae Phanerophyte climbing Neophyte, E-Asian

First record for Calabria (casual)

Specimen: 2-9-2019, C.da Monacelli (Melicucco, Reggio Calabria province), roadside, 118 m a.s.l., 38.44528°N-16.04166°E, *leg. et det.* C.M. Musarella (REGGIO).

Note. A few individuals noted growing together with *Phyllostachys aurea* and *Plumbago auriculata* along the roadside, probably escaped from a nearby garden. Previously *Wisteria sinensis* was recorded as casual alien from almost all of the Italian regions but only from Campania in the southern ones [17]. In Sardegna it is considered naturalized [17].

Figure 2. Species not documented by herbarium specimens in Appendix A: A) *Ophrys crabronifera*, B) *O. marmorata*, C) *O. speculum*, D) *O. sipontensis*.

Figure 3. - Comparison between three New Zealand creeping *Epilobium* species naturalized in Europe/British Isles: A) *Epilobium brunnescens* subsp. *brunnescens* fruiting plant showing growth habit, Mt Te Aroha, North Island, New Zealand, images: P.J. de Lange; B) *E. komarovianum*, flowering plant with immature capsules, showing distinctive rugose-impressed leaves, Long Point, Otago, South Island, New Zealand, image: J.W. Barkla; C) *E. nummularifolium*, vegetative material showing growth habit, note stem and leaf colour and toothed leaf margins; Alghero, Sardegna, images: P.J. de Lange; D) *E. nummularifolium* flowering and fruiting plant (note greyish colour of capsules); Centre Road, Otago Peninsula, South Island, New Zealand image: David Lyttle.

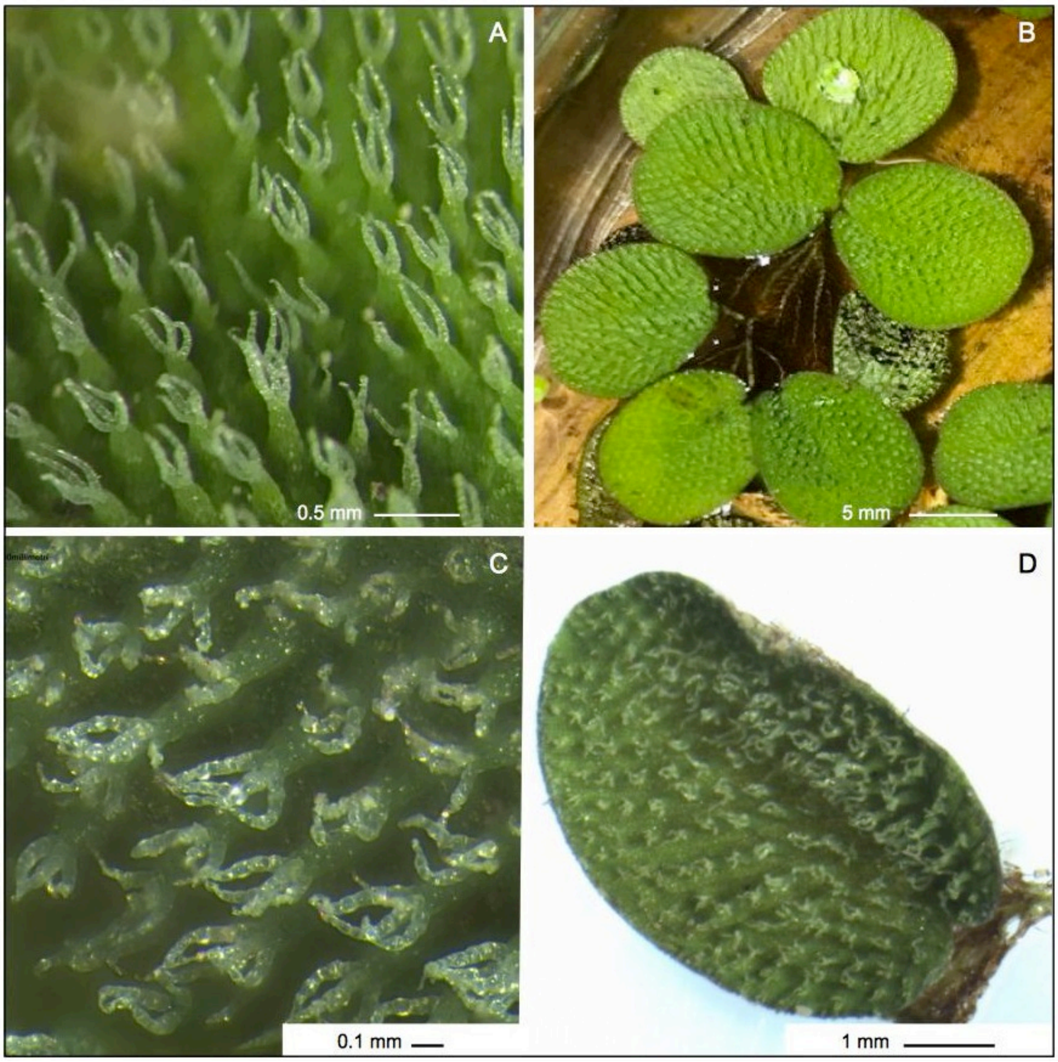


Figure 4. *Salvinia minima* plants collected in Calabria and photographed in laboratory at University of Reggio Calabria: A,C, D) upper side leaf hairs at different magnification, note distinctive hairs not joined at their tips B) entire plants with the typical rounded leaves shape;

Chapter 3 - Subchapter 3.1: Annex S1

Table 1. Summary statistics of the number of total species richness and abundance of each taxonomic group: pseudoscorpions (Pseudoscorpiones), spiders (Araneae), darkling beetles (Tenebrionidae), rove beetles (Staphylinidae), ground beetles (Carabidae), ants (Formicidae), and vascular plants. Land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

GROUP	**NAT**		**AGR**		**URB**		**Total species richness**	**Total abundance**
	Richness	**Abundance**	**Richness**	**Abundance**	**Richness**	**Abundance**		
Pseudoscorpiones	9	123	12	148	9	119	13	390
Araneae	59	355	75	1248	54	1218	106	2821
Tenebrionidae	12	102	14	672	12	310	22	1084
Staphylinidae	39	3195	36	2321	31	1699	55	7215
Carabidae	18	175	25	617	20	1985	38	2777
Formicidae	28	6090	30	16732	26	29303	35	52125
Vascular plants	208	8629.5	202	6712.9	106	4264.7	330	19607.1

Table 2. Summary statistics of number of SWOF, means species richness and abundance (mean ± standard error, minimum and maximum) of each taxonomic group: pseudoscorpions (Pseudoscorpiones), spiders (Araneae), darkling beetles (Tenebrionidae), rove beetles (Staphylinidae), ground beetles (Carabidae), ants (Formicidae), and vascular plants. Land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). Statistical note: mean species richness (μ); standard error (SE); minimum (min) and maximum (max) values.

	Taxonomic group	**NAT**	**AGR**	**URB**
Richness/SWOF μ ± SE (min-max)	Pseudoscorpiones	2.82 ± 0.44 (1 - 6)	3.27 ± 0.38 (1 - 5)	2.75 ± 0.45 (1 - 5)
	Araneae	13.91 ± 0.96 (10 - 19)	22.09 ± 1.98 (14 - 34)	20.12 ± 1.68 (14 - 27)
	Tenebrionidae	2.64 ± 0.58 (0 - 6)	4.36 ± 0.68 (2 - 9)	3.62 ± 0.68 (1 - 7)
	Staphylinidae	15.27 ± 1.48 (9 - 24)	12.27 ± 1.17 (5 - 19)	11 ± 1.89 (4 - 19)
	Carabidae	4.18 ± 0.69 (2 - 8)	5 ± 0.93 (2 - 11)	5.25 ± 1 (2 - 10)
	Formicidae	11.36 ± 0.64 (9 - 15)	13.64 ± 0.96 (7 - 18)	13 ± 0.8 (10 - 17)
	Vascular plants	45.36 ± 4.44 (30 - 73)	41.36 ± 7.01 (10 - 83)	25.75 ± 2.43 (19 - 38)
Abundance/SWOF μ ± SE (min-max)	Pseudoscorpiones	11.18 ± 1.72 (2 - 21)	13.45 ± 2.99 (1 - 37)	14.88 ± 4.64 (3 - 40)
	Araneae	32.27 ± 3.52 (19 - 63)	113.45 ± 19.41 (44 - 266)	152.25 ± 28.66 (35 - 316)
	Tenebrionidae	9.27 ± 4.99 (0 - 57)	61.09 ± 26.94 (2 - 289)	38.75 ± 12.69 (1 - 89)
	Staphylinidae	290.45 ± 33.73 (122 - 527)	211 ± 24.34 (110 - 351)	212.38 ± 32.71 (105 - 354)
	Carabidae	15.91 ± 4.38 (3 - 43)	56.09 ± 18.52 (2 - 197)	248.12 ± 145.62 (3 - 1224)
	Formicidae	553.64 ± 90.79 (129 - 1163)	1521.09 ± 166.59 (682 - 2413)	3662.88 ± 2047.95 (662 - 17772)
	Vascular plants	784.5 ± 65.25 (458.4 - 1112.8)	610.26 ± 32.96 (471.2 - 812.6)	533.09 ± 47.95 (288.7 - 679.8)
No. SWOFs		11	11	8

Chapter 3 - Subchapter 3.1: Annex S2

Table 1. PERMANOVA results based on the Euclidean distance of species richness data of all taxonomic groups (GR) sampled from 30 SWOFs (SW) at the three land-use matrices (MA). Significance codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

Source of variation	**df**	**SS**	**MS**	**Pseudo-F and p-values**	**Variance components (%)**
MA	2	282.78	141.39	2.74	4.74
GR	6	26095	4349.2	84.43***	50.36
MAxGR	12	2168.8	180.74	3.51***	15.03
Res	189	9736.2	51.514	-	29.86
Total	209	40584		-	100

Table 2. PERMANOVA t statistic and significance values of pair-wise tests for the interaction term “MA x GR” for pairs of levels of factor “land-use matrix” within each taxonomic group, based on species richness data. Land-use matrices (MA): natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). Taxonomic groups (GR): pseudoscorpions (Pseudoscorpiones), spiders (Araneae), darkling beetles (Tenebrionidae), rove beetles (Staphylinidae), ground beetles (Carabidae), ants (Formicidae), and vascular plants. Bold t statistic values indicate statistically significant p-values: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

	Pairs of levels of factor “land-use matrix"		
Taxonomic groups	AGR versus NAT	AGR versus URB	NAT versus URB
Pseudoscorpiones	0.78	0.88	0.11
Araneae	3.72***	0.72	3.42**
Tenebrionidae	1.94	0.75	1.11
Staphylinidae	1.59	0.6	1.81
Carabidae	0.71	0.18	0.92
Formicidae	1.98	0.48	1.62
Vascular plants	0.48	1.83	3.48**

Table 3. PERMANOVA results based on zero-adjusted Bray-Curtis dissimilarity on square-root transformed abundance data analyzed separately for each taxonomic group sampled from 30 SWOFs (SW) at the three land-use matrices (MA). Significance codes: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

		Pseudoscorpiones		Araneae		Tenebrionidae		Staphylinidae	
Source of variation	df	Pseudo-F	Variance components (%)	Pseudo-F	Variance components (%)	Pseudo-F	Variance components (%)	Pseudo-F	Variance components (%)
MA	2	4.98***	28.67	4.05***	23.57	2.42*	12.57	1.53	5.11
Res	27	-	71.33	-	76.43	-	87.43	-	94.89
Total	29	-	100	-	100	-	100	-	100

		Carabidae		Formicidae		Vascular plants	
Source of variation	df	Pseudo-F	Variance components (%)	Pseudo-F	Variance components (%)	Pseudo-F	Variance components (%)
MA	2	4.26***	24.75	4.63***	26.82	3.47***	19.99
Res	27	-	75.25	-	73.18	-	80.01
Total	29	-	100	-	100	-	100

Table 4. PERMANOVA t statistic and significance values of pair-wise tests for the main effect of land use matrices on species abundance analyzed separately for each taxonomic group: pseudoscorpions (Pseudoscorpiones), spiders (Araneae), darkling beetles (Tenebrionidae), rove beetles (Staphylinidae), ground beetles (Carabidae), ants (Formicidae), and vascular plants. Land-use matrices (MA): natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). Bold t statistic values indicate statistically significant p-values: (***) p ≤ 0.001, (**) p ≤ 0.01, (*) p ≤ 0.05.

	Pairs of levels of factor “land-use matrix"		
Taxonomic groups	AGR versus NAT	AGR versus URB	NAT versus URB
Pseudoscorpiones	2.17***	1.4	3.07***
Araneae	2.06***	1.36	2.39***
Tenebrionidae	1.69*	1.11	1.78*
Staphylinidae	1.37	0.98	1.29
Carabidae	2.19***	1.4	2.48***
Formicidae	2.51***	1.06	2.54***
Vascular plants	1.98***	1.52*	2.02***

Fig. 1. Nonmetric multidimensional scaling (nMDS) ordinations of community composition of each taxonomic group analyzed separately based on zero-adjusted Bray-Curtis dissimilarity on square-root transformed abundance data.

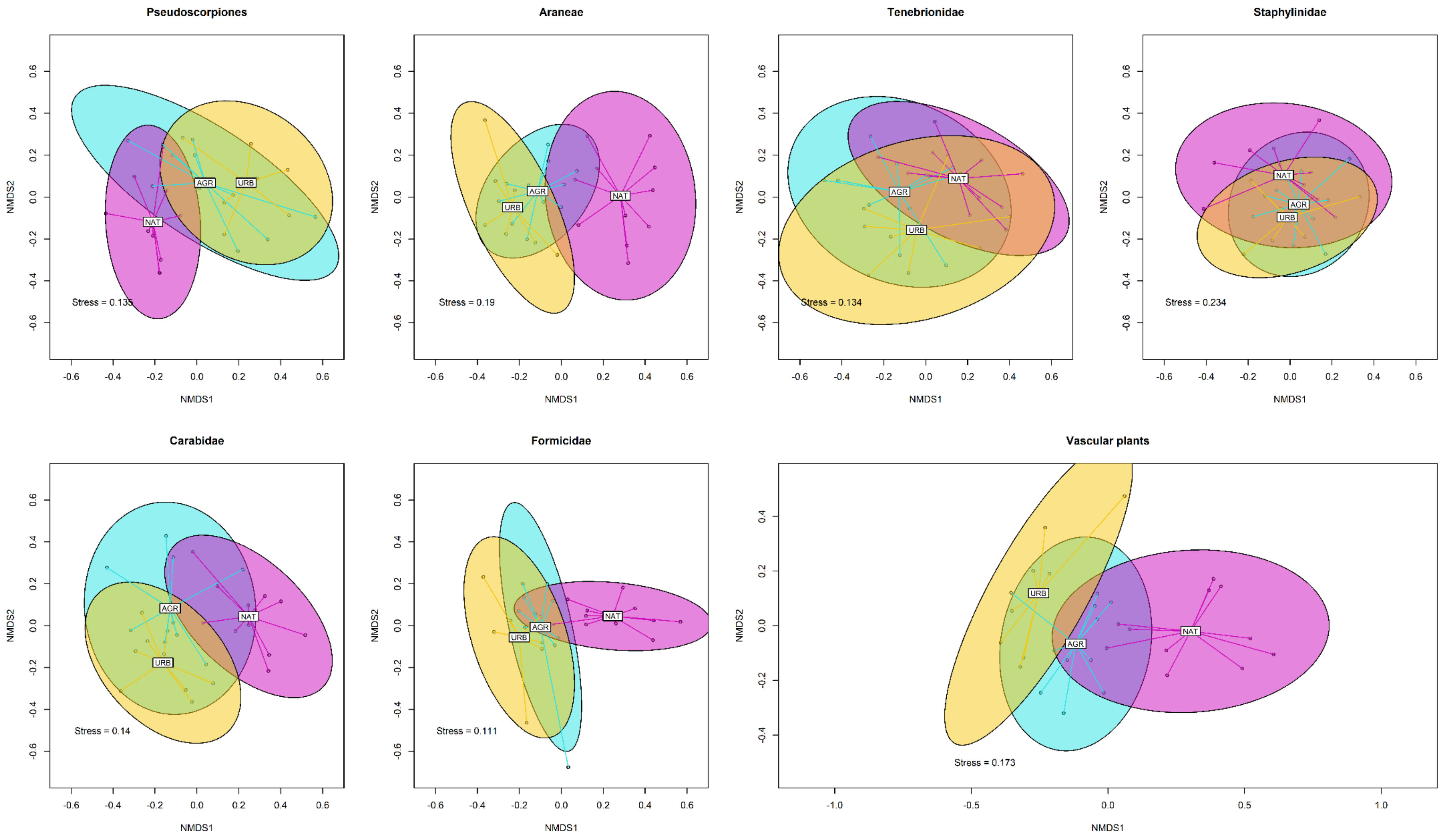
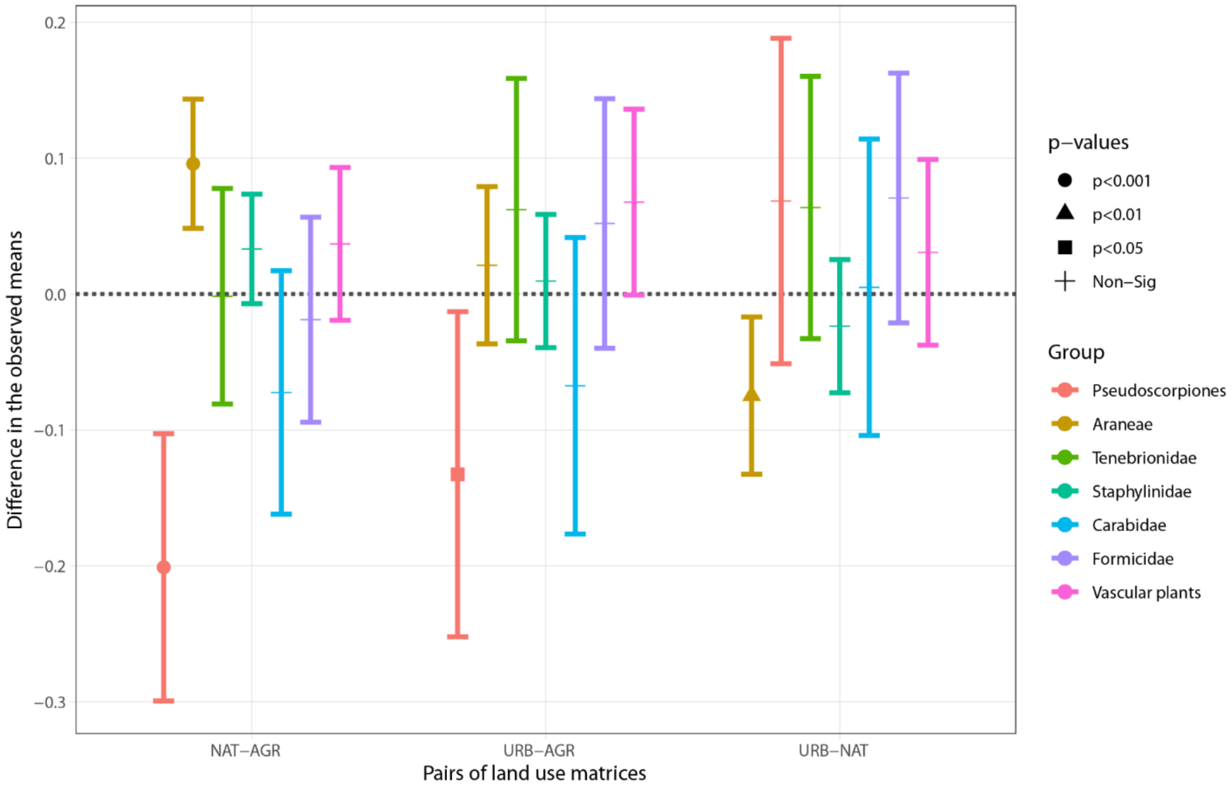


Fig. 2. Results of the Tukey HSD test on beta diversity analyses among land-use matrices calculated for each taxonomic group, separately, in considered SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB). Taxonomic groups: pseudoscorpions (Pseudoscorpiones), spiders (Araneae), darkling beetles (Tenebrionidae), rove beetles (Staphylinidae), ground beetles (Carabidae), ants (Formicidae), and vascular plants.



Chapter 3 - Subchapter 3.1: Annex S3

Fig. 1. Plot of the first two principal component analysis (PCA) axes for spatial-topographic (scaled to zero mean and unit variance) recorded from the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

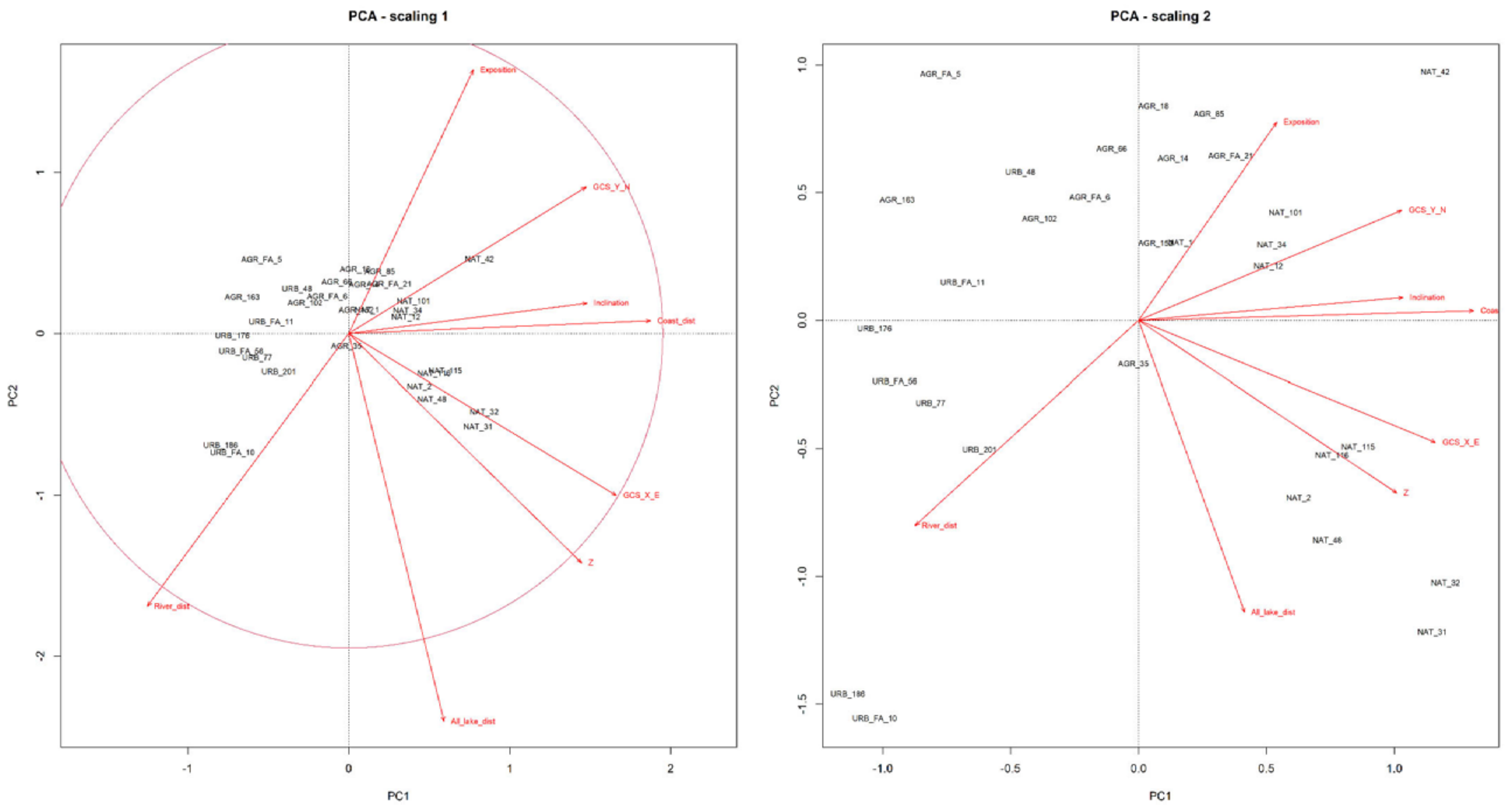


Fig. 2. Plot of the first two principal component analysis (PCA) axes for climate variables (scaled to zero mean and unit variance) recorded from the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

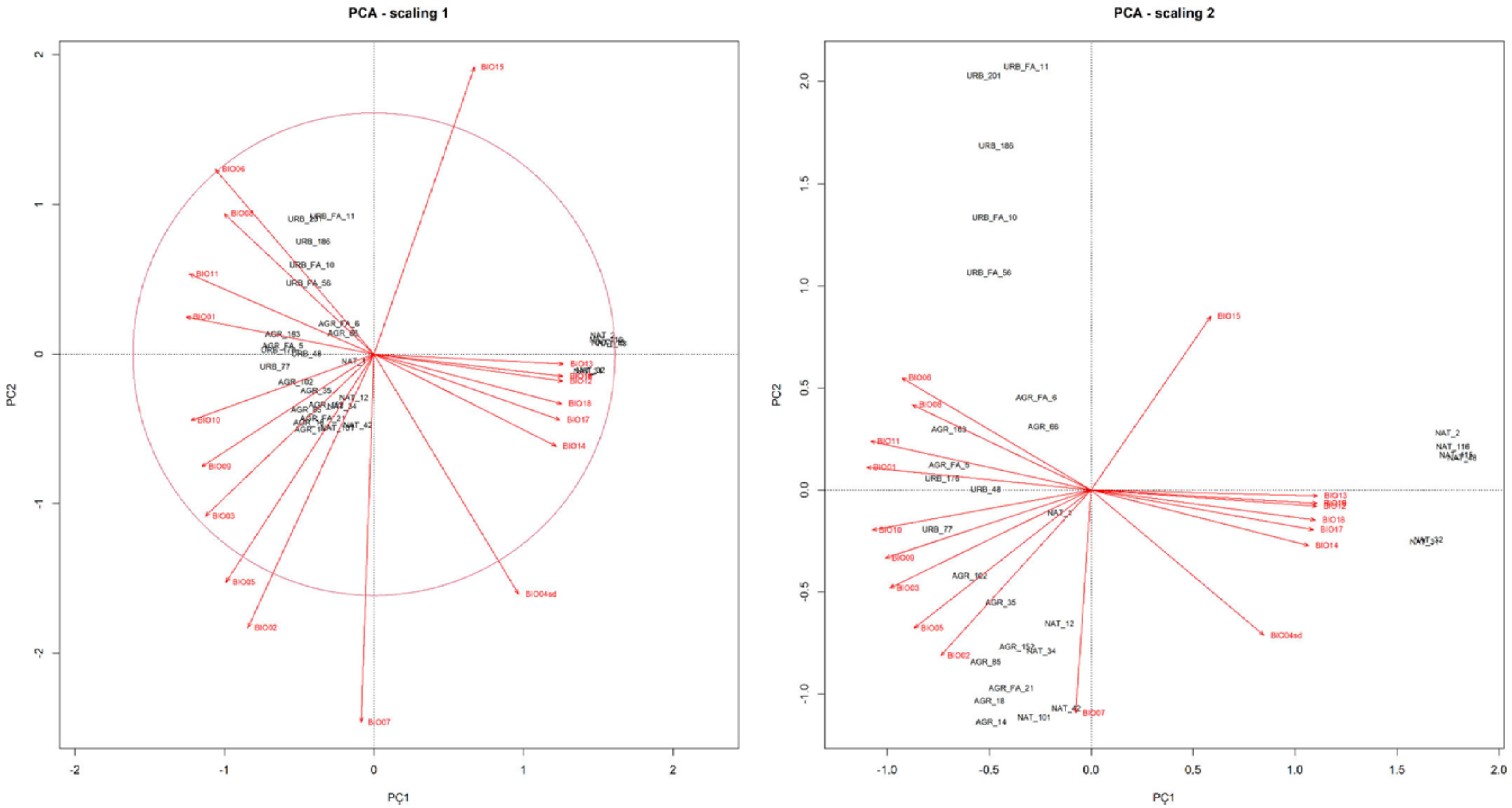


Fig. 3. Plot of the first two principal component analysis (PCA) axes for landscape metrics (scaled to zero mean and unit variance) recorded from the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

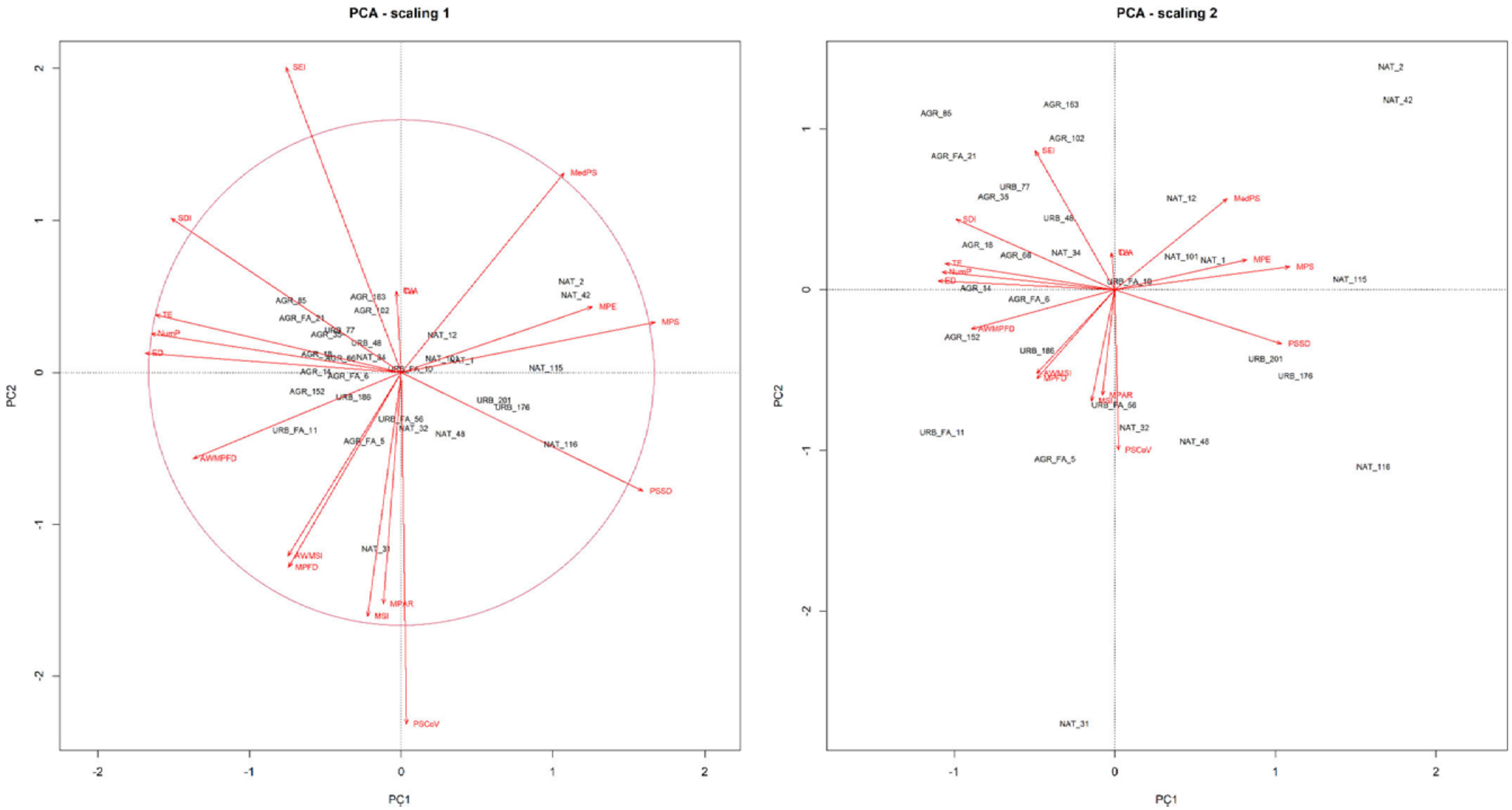


Fig. 4. Plot of the first two principal component analysis (PCA) axes for Hellinger transformed data of pseudoscorpions (Pseudoscorpiones) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

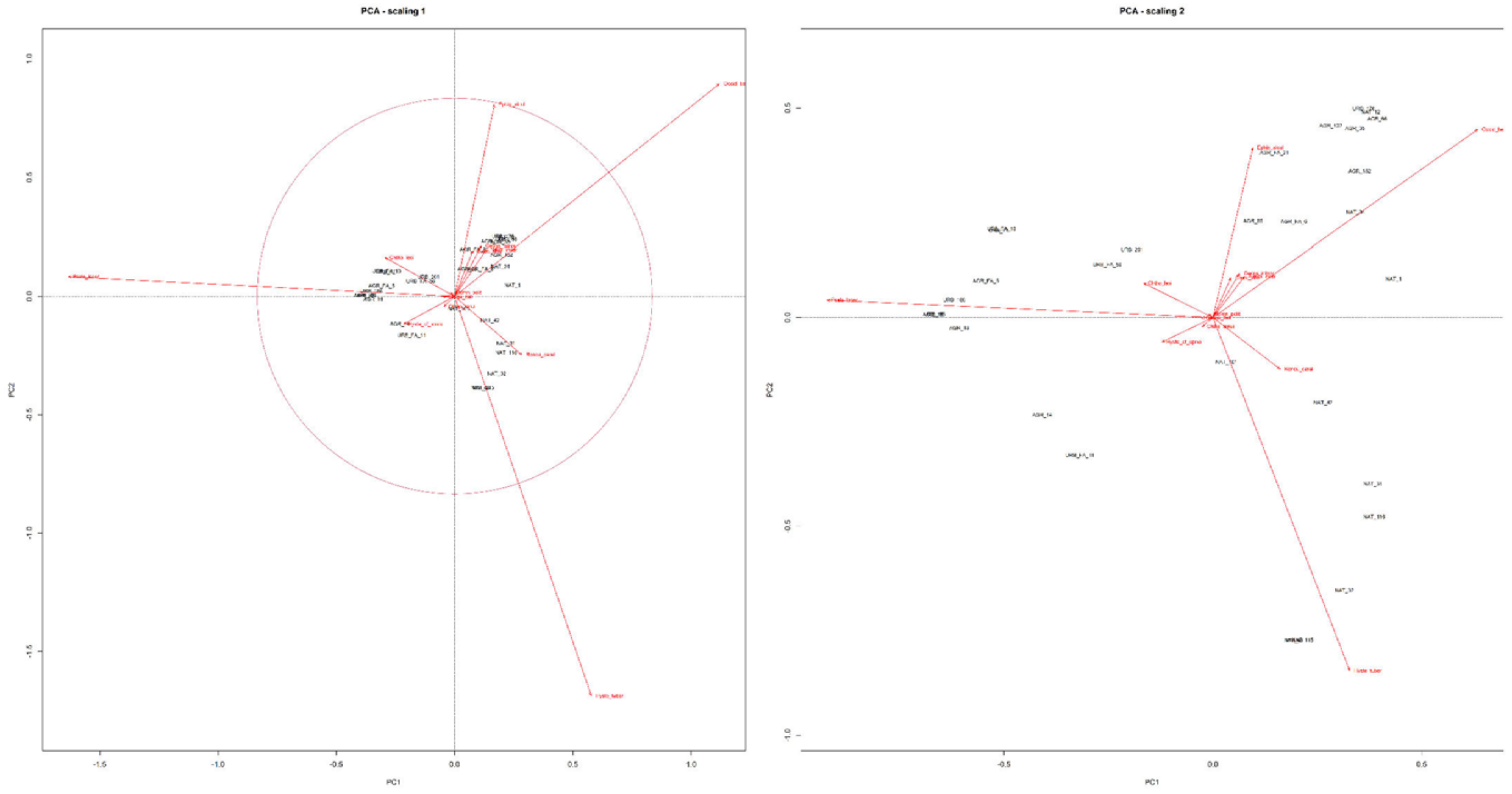


Fig. 5. Plot of the first two principal component analysis (PCA) axes for Hellinger transformed data of spiders (Araneae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

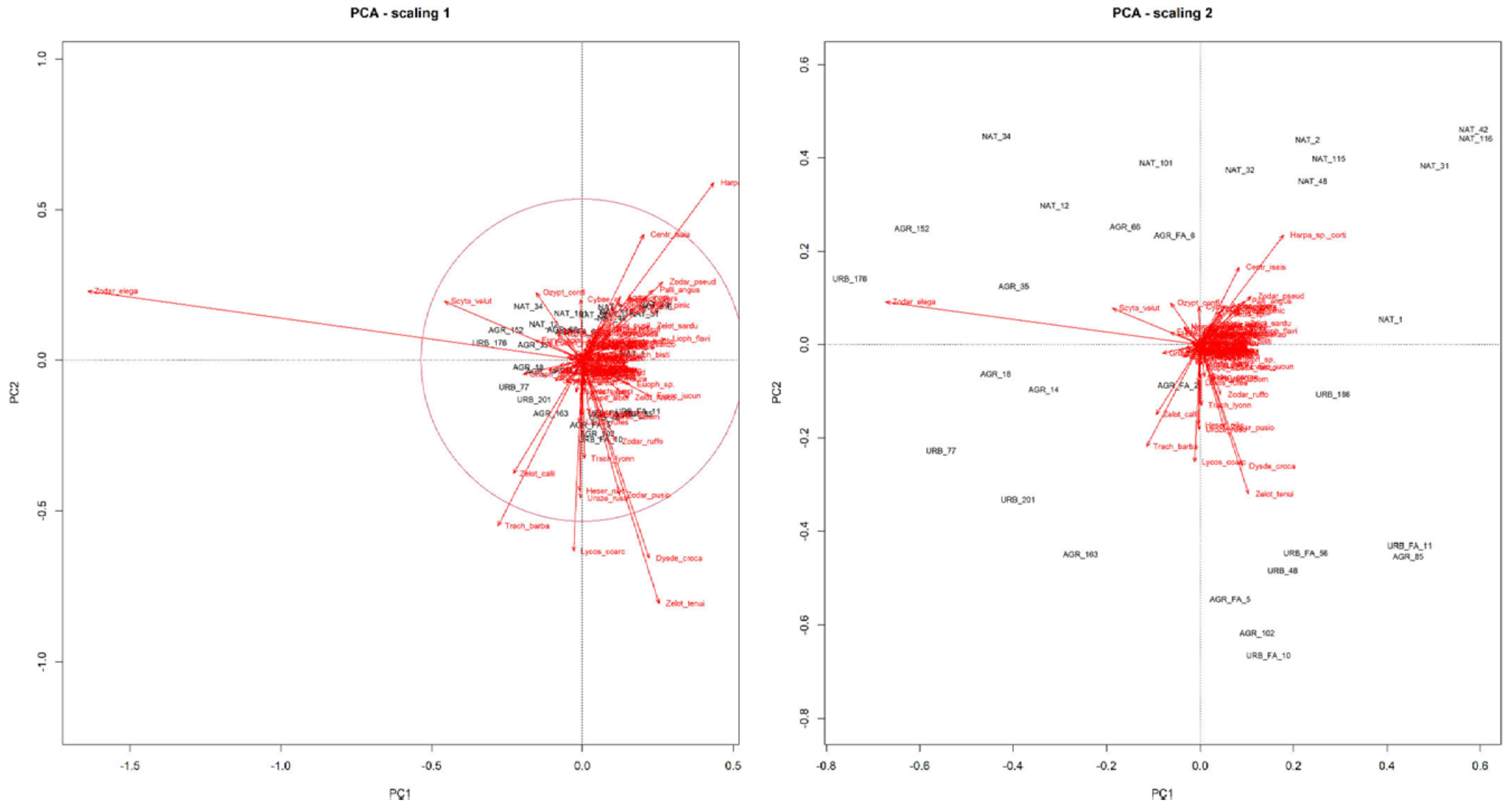


Fig. 6. Plot of the others principal component analysis (PCA) axes for Hellinger transformed data of spiders (Araneae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

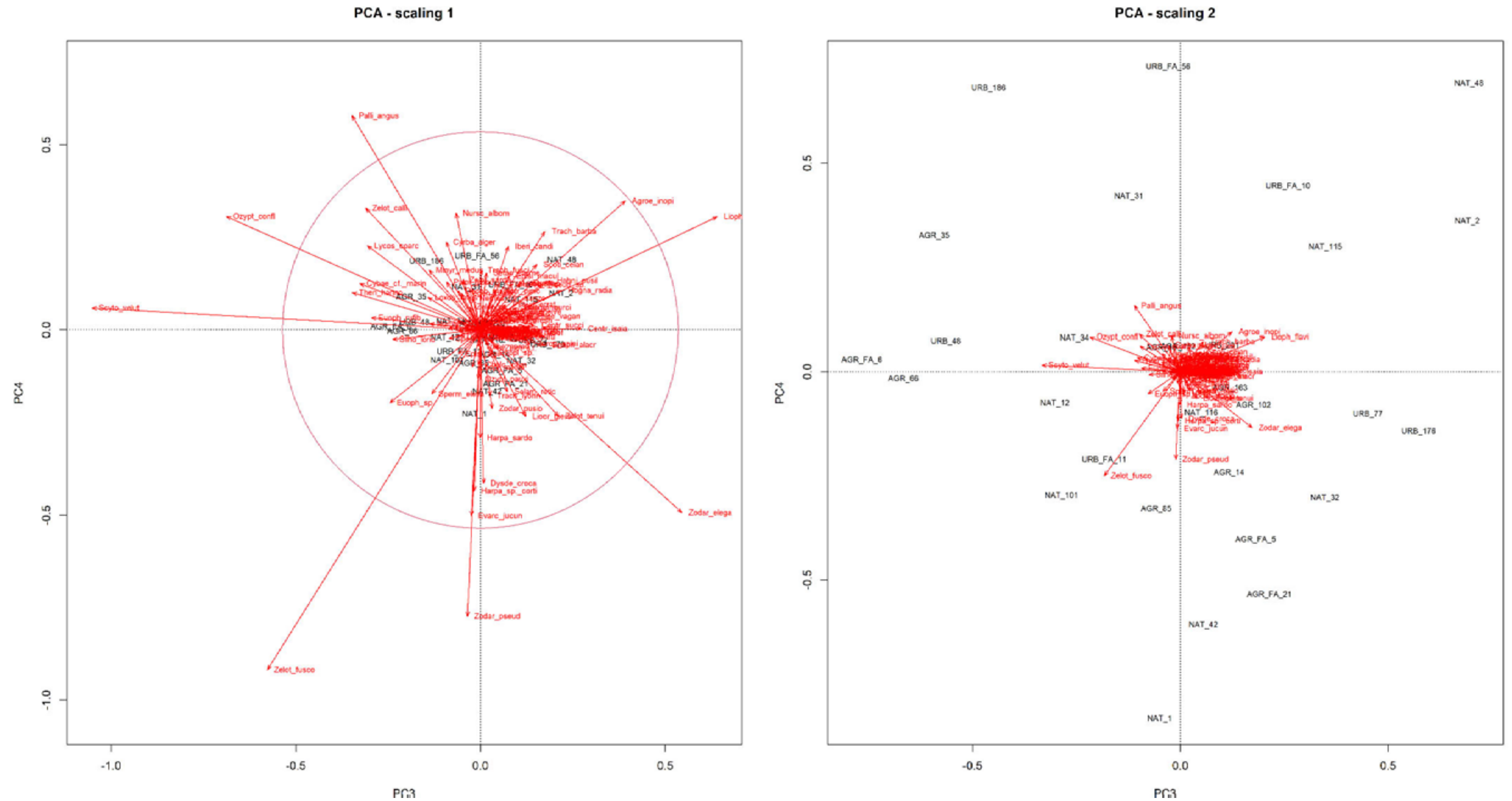


Fig. 7. Plot of the first two principal component analysis (PCA) axes for Hellinger transformed data of darkling beetles (Tenebrionidae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

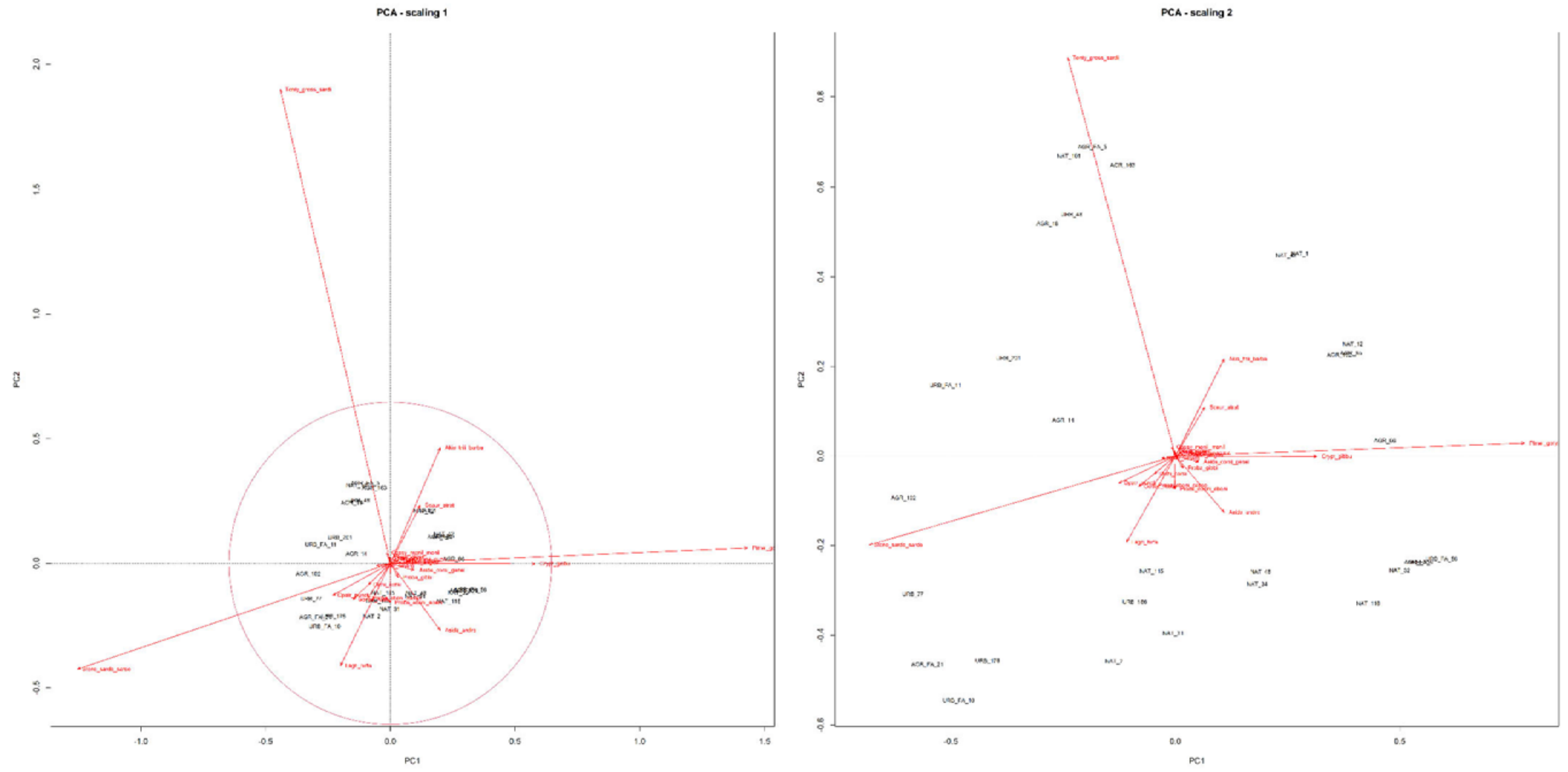


Fig. 8. Plot of the others principal component analysis (PCA) axes for Hellinger transformed data of darkling beetles (Tenebrionidae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

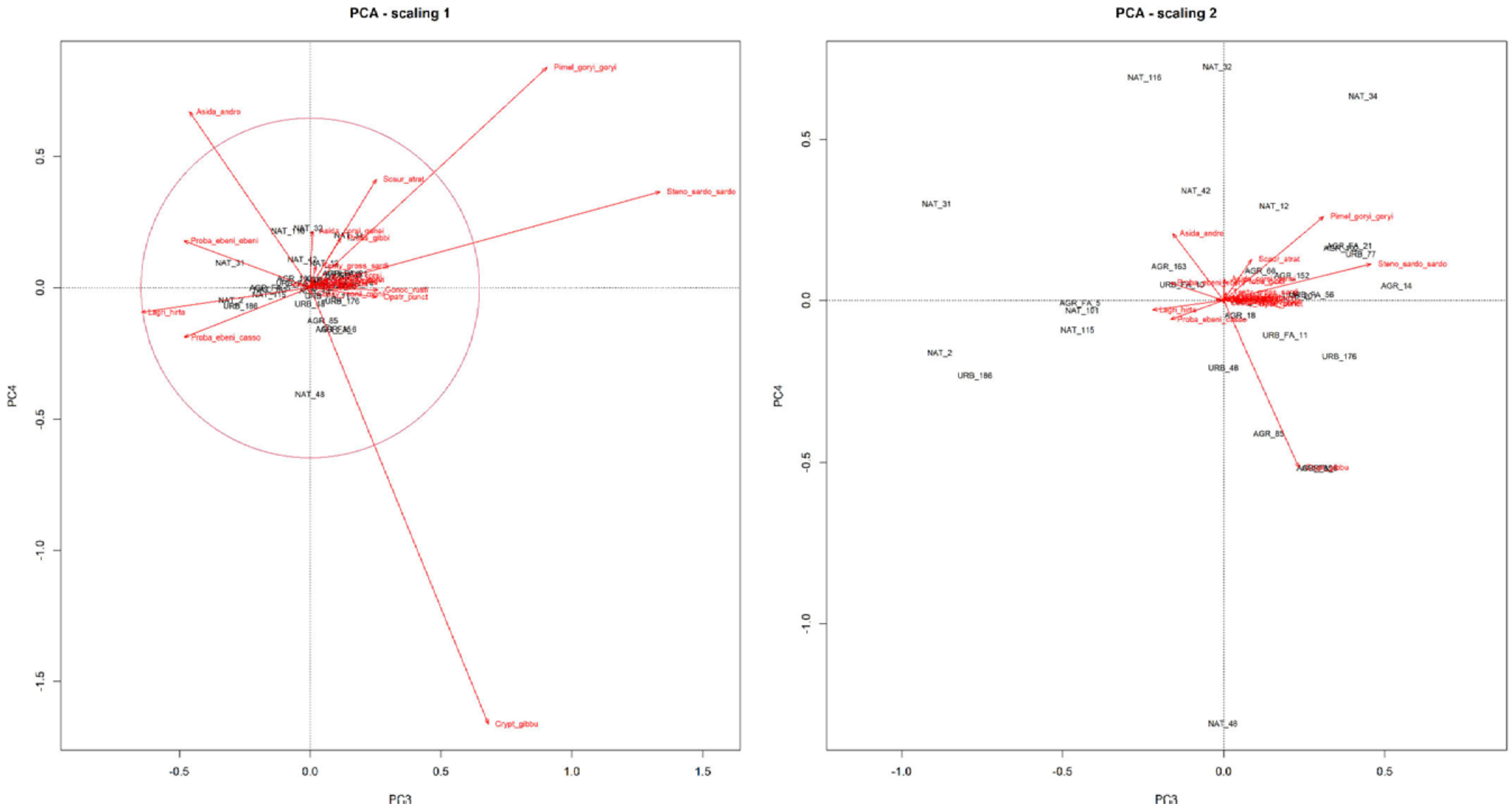


Fig. 9. Plot of the first two principal component analysis (PCA) axes for Hellinger transformed data of rove beetles (Staphylinidae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

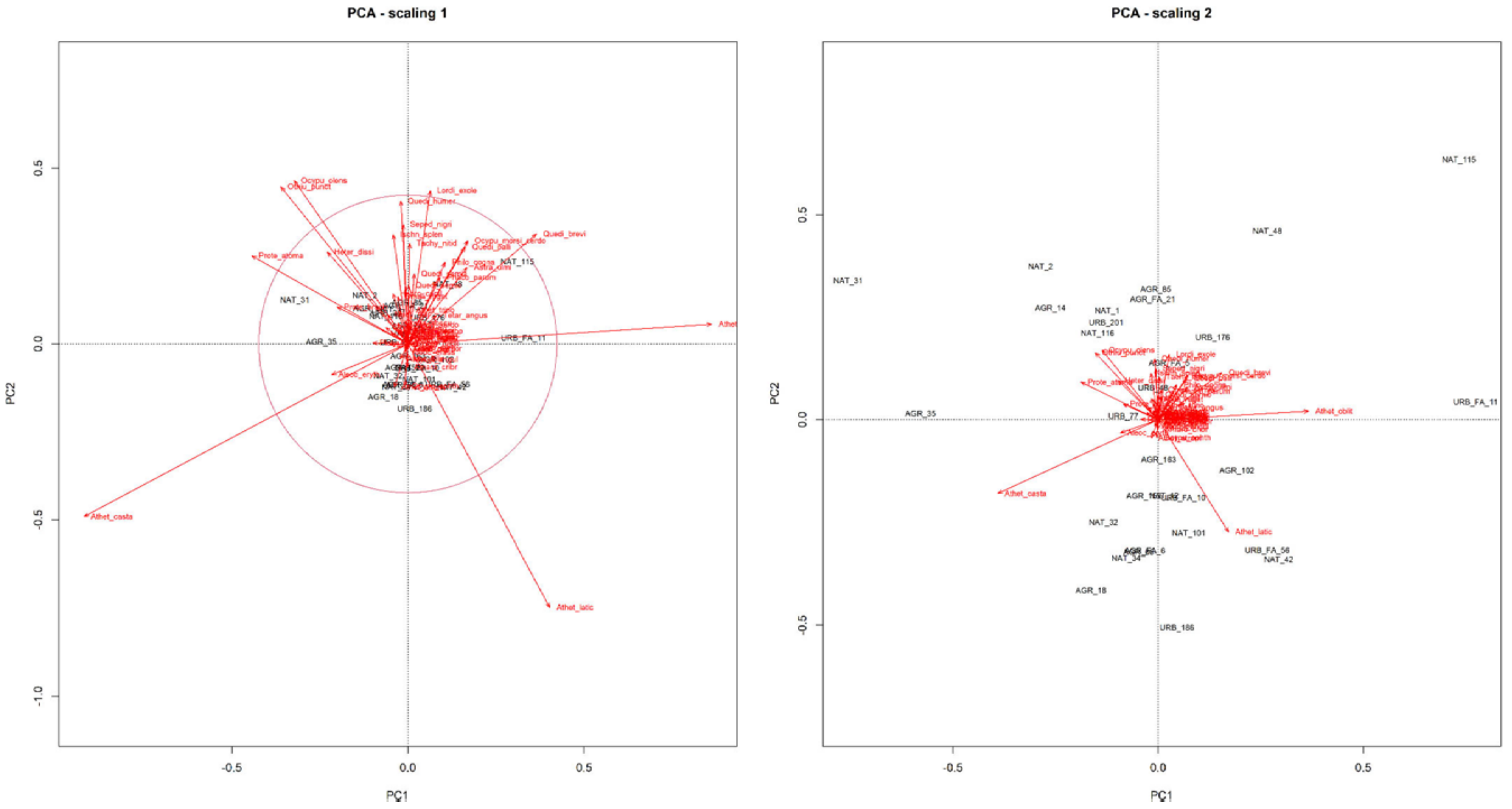


Fig. 10. Plot of the others principal component analysis (PCA) axes for Hellinger transformed data of rove beetles (Staphylinidae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

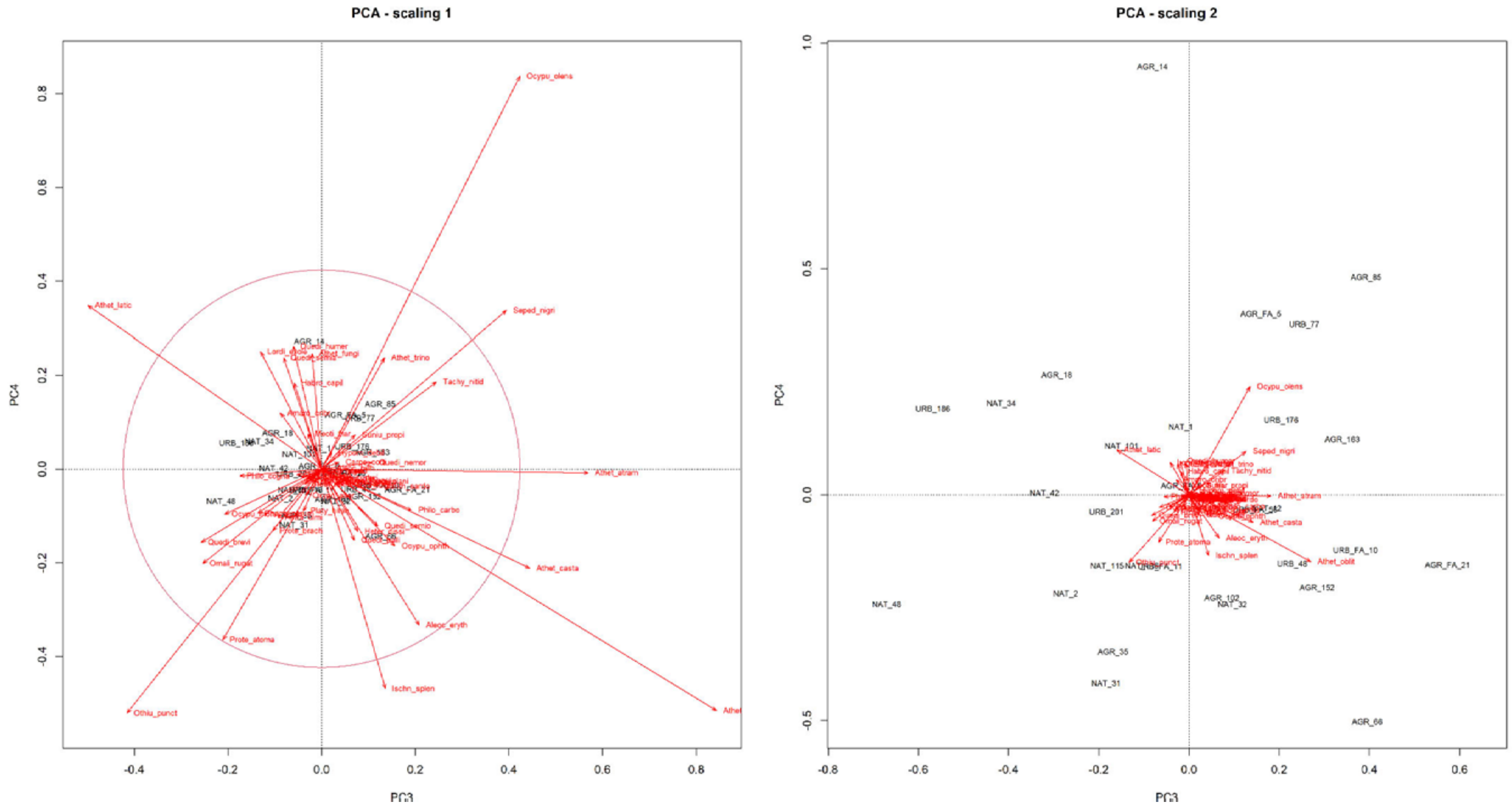


Fig. 11. Plot of the first two principal component analysis (PCA) axes for Hellinger transformed data of ground beetles (Carabidae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

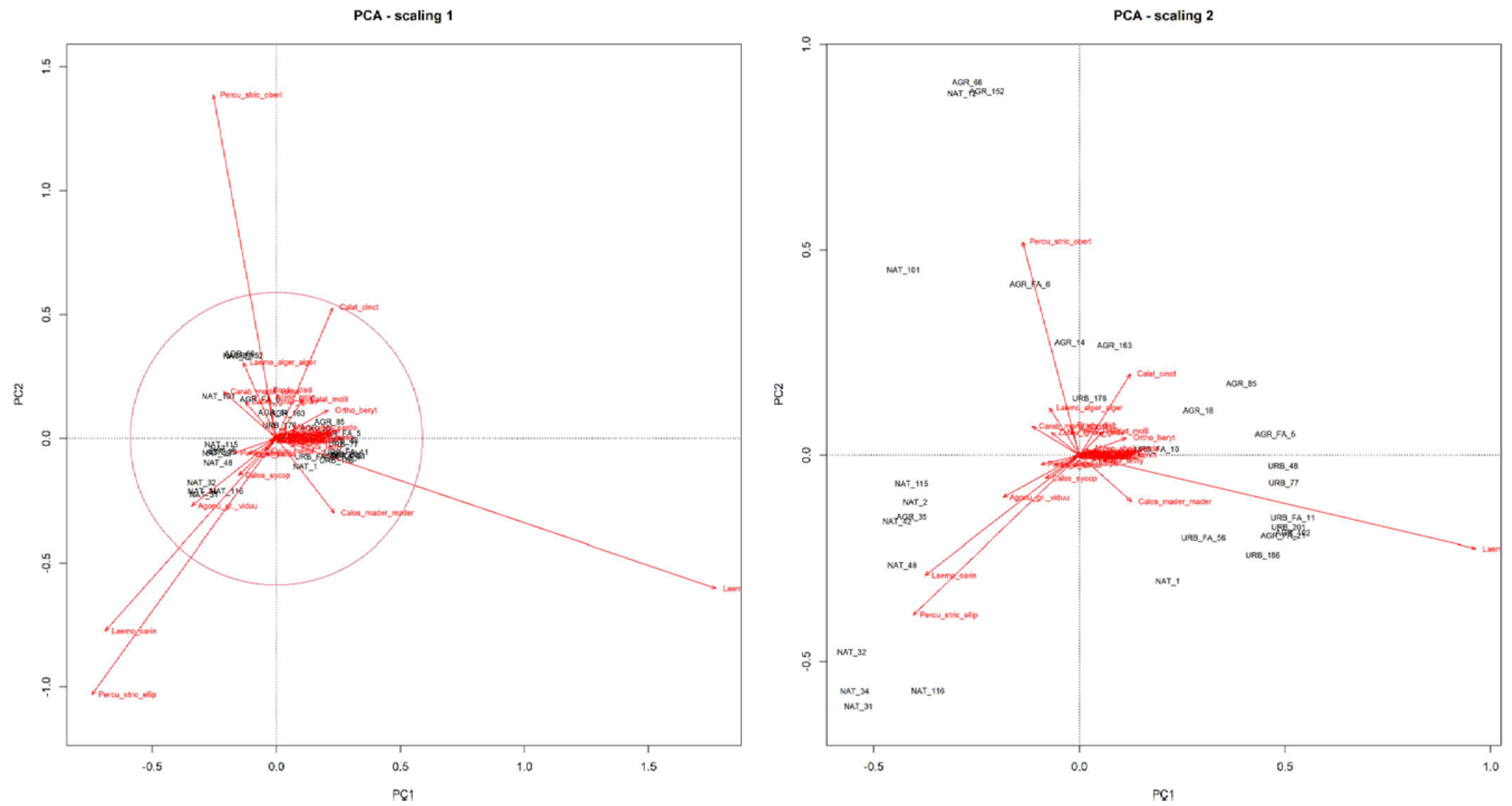


Fig. 12. Plot of the others principal component analysis (PCA) axes for Hellinger transformed data of ground beetles (Carabidae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

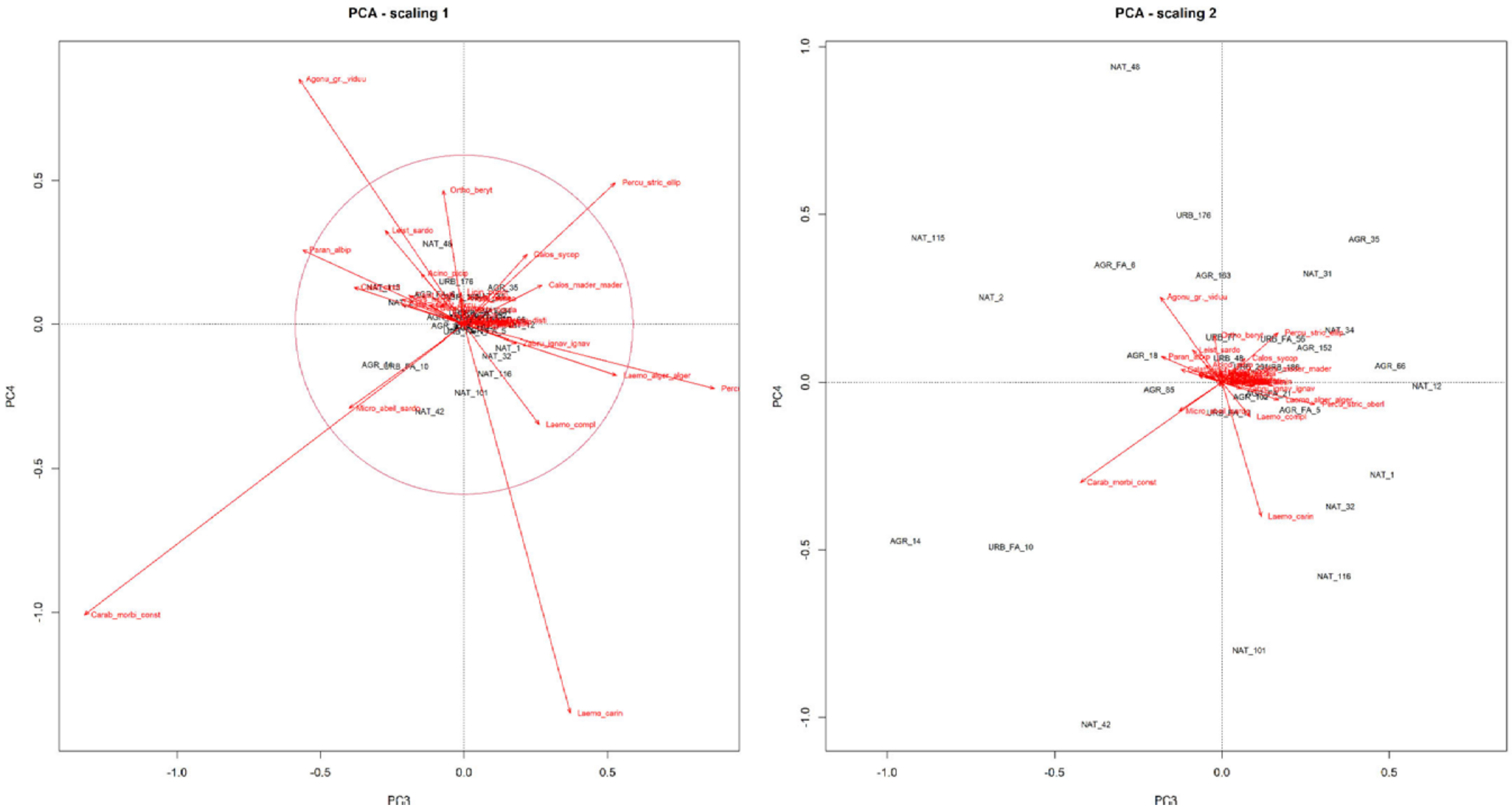


Fig. 13. Plot of the first two principal component analysis (PCA) axes for Hellinger transformed data of ants (Formicidae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

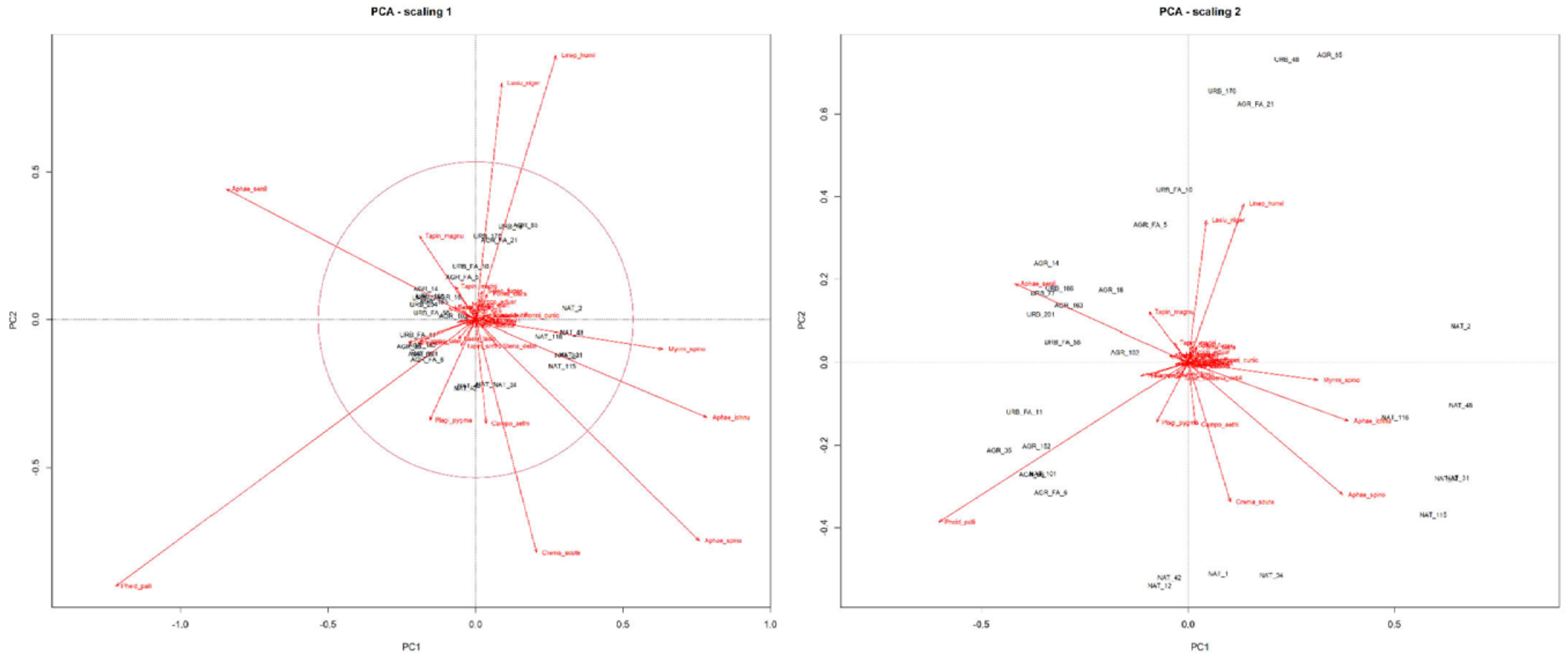


Fig. 14. Plot of the others principal component analysis (PCA) axes for Hellinger transformed data of ants (Formicidae) sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

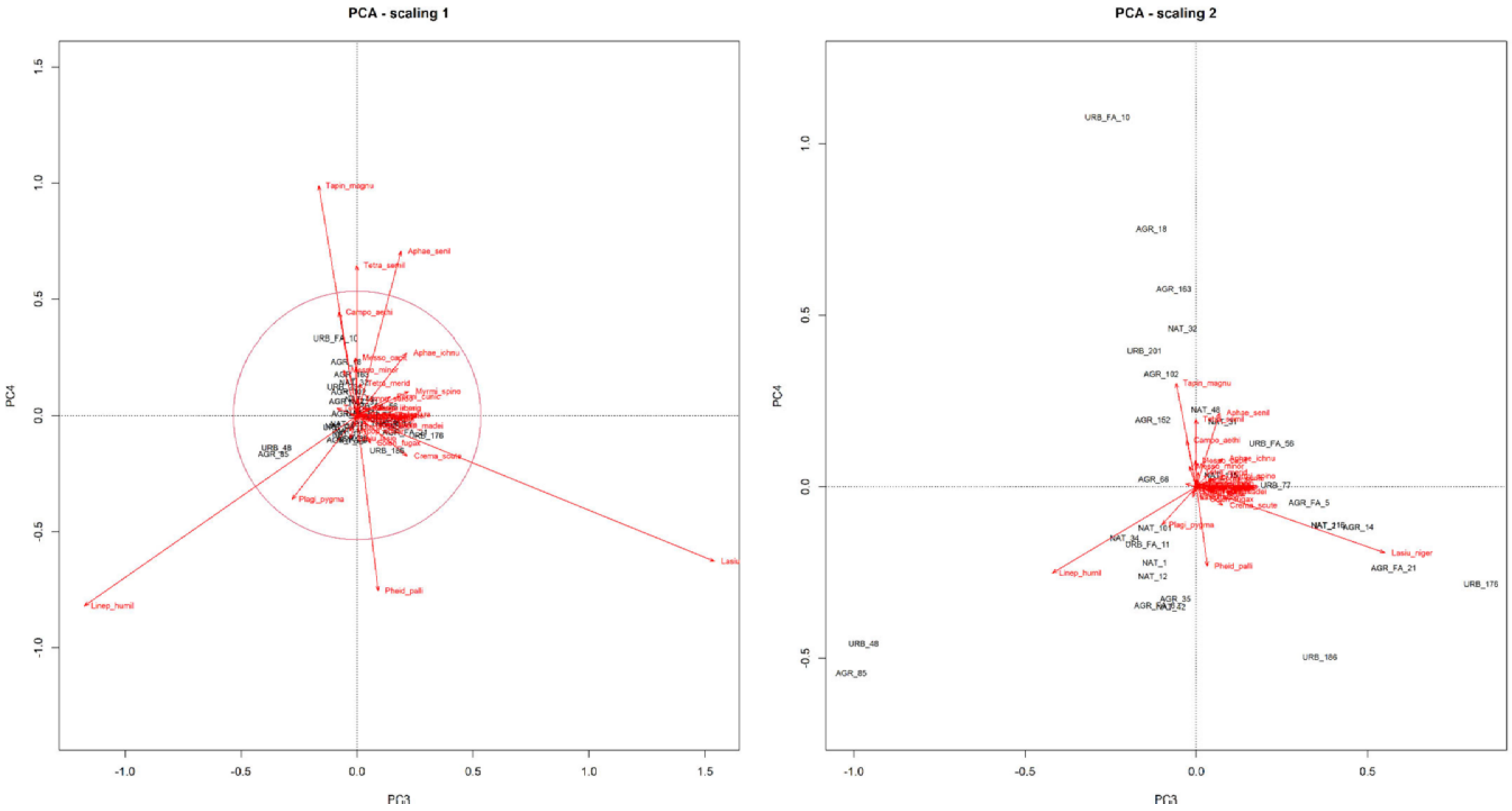


Fig. 15. Plot of the first two principal component analysis (PCA) axes for Hellinger transformed data of vascular plants sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).

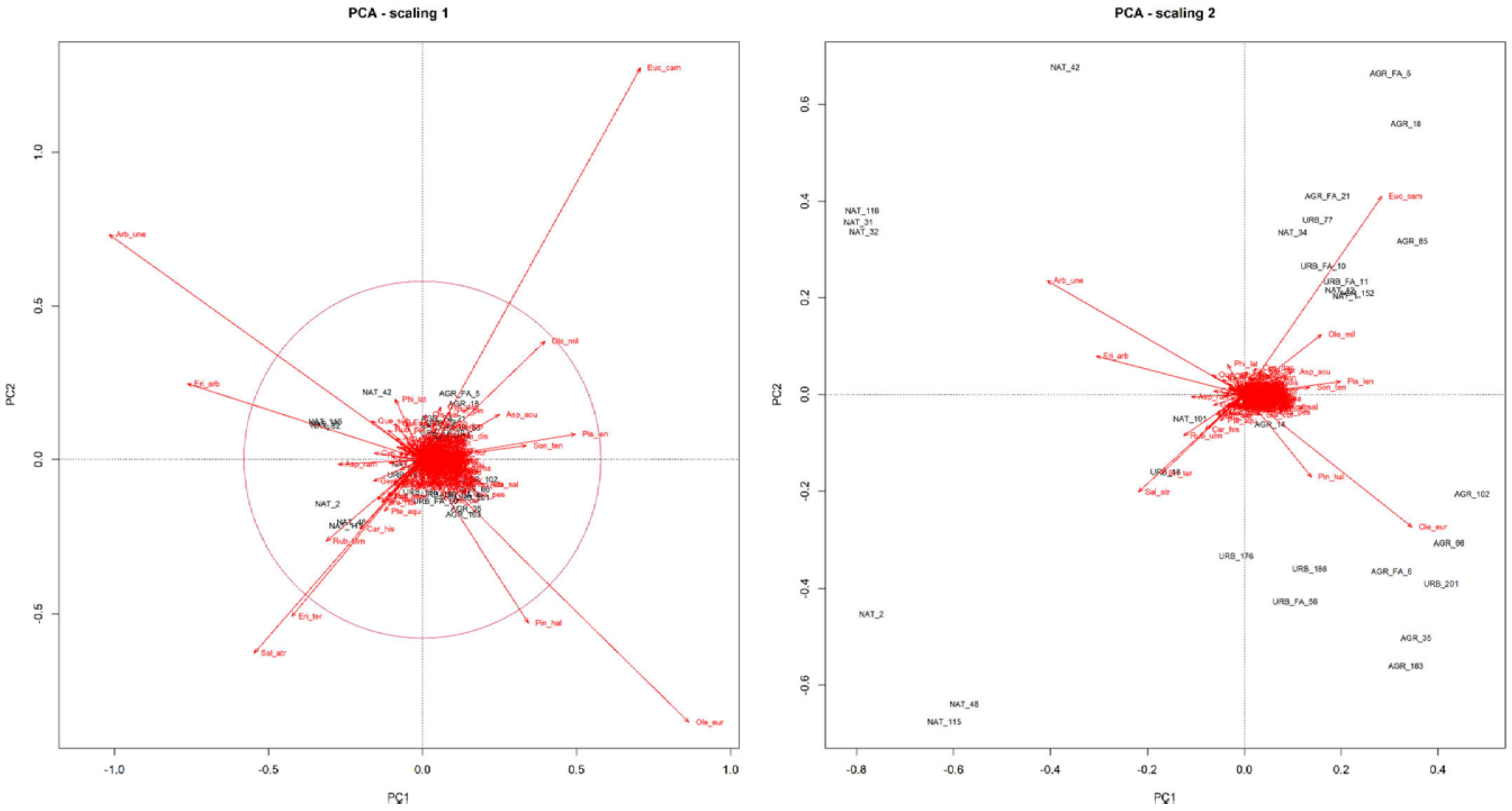
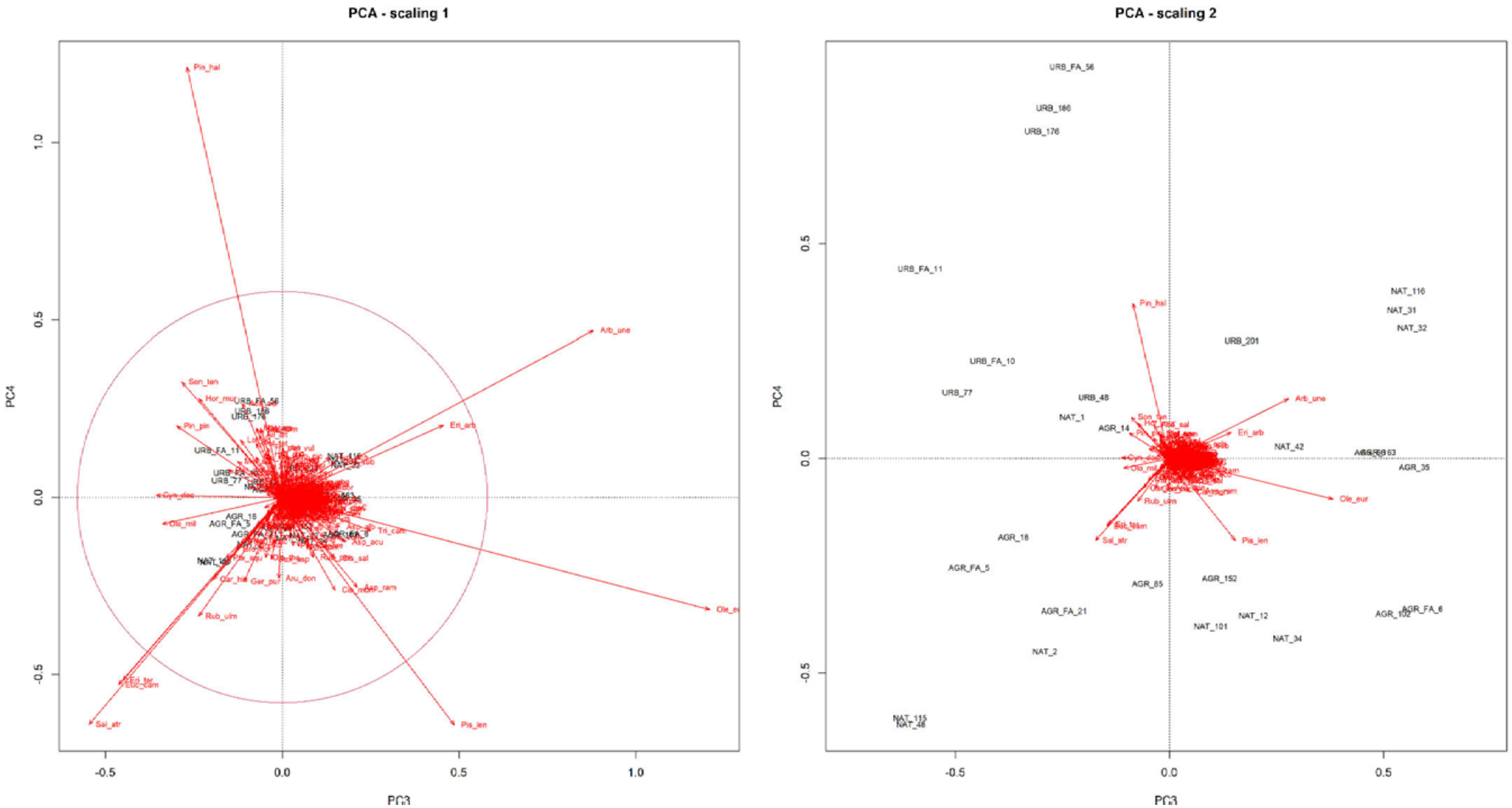


Fig. 16. Plot of the others principal component analysis (PCA) axes for Hellinger transformed data of vascular plants sampled in the 30 SWOFs at the three land-use matrices: natural and semi-natural areas (NAT), agricultural areas (AGR), urban and artificial surfaces (URB).



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Chapter 4 - Subchapter 4.1: References

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