IMPACT OF OLIVE GROVE MANAGEMENT ON SPIDER (ARANEAE) COMMUNITY IN NORTHWESTERN TURKEY

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ABSTRACT

IMPACT OF OLIVE GROVE MANAGEMENT ON SPIDER (ARANEAE) COMMUNITY IN NORTHWESTERN TURKEY

Elverici, Mert
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Olive (*Olea europea*) plantations cover vast areas on the Aegean coast of Turkey, including the southern foothills of Kaz Mountains. In such environments, olive trees dominate the terrestrial vegetation in such a way that they might form a novel ‘ecosystem’. The ‘olive grove agro-ecosystem’ is shaped by agricultural management while impacts on biodiversity are largely unknown. As representatives of local biodiversity, spider communities were sampled in soil and vegetation microhabitats, alongside with various environmental variables linked to management or habitat features in organic and conventional olive groves, and in relict patches of natural habitats. 278 spider species from 20,167 specimens and >300 sampling units were collected and standardized by effort. Data obtained were analyzed by using multivariate statistics to interpret associations within the community (NMDS, CCA, RDA models) and by diversity statistics (ANOVA, ANOSİM and permutation tests). Community composition and diversity differed by habitat type (moist or dry grove, maquis, pine and mixed forests) and microhabitat (soil, herb and canopy). Olive groves and mixed forests had richer communities compared to maquis and pine patches. Soil microhabitats were more species rich and diverse than herb or canopy layers. Difference between “organic” or “conventional” management types could not be detected at the community level, whereas impact of soil management (tillage and herbicides) and vegetation cover (tree, shrub or herb cover) were significant on the spider community. Our results may serve as evidence for community composition -

environment relationship for spider communities, and help towards more nature-friendly olive grove management in the region.

Keywords: Arachnology, Community Ecology, Agri-environments, Biodiversity

ÖZ

TÜRKİYENİN KUZEYBATISINDA YAPILAN ZEYTİNCİLİK UYGULAMALARININ ÖRÜMCEK(ARANEAE)KOMÜNİTESİ ÜZERİNDEKİ ETKİSİ

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Zeytin (*Olea europaea*) Akdeniz ekosisteminin doğal bir elemanıdır ve zeytinlikler yerel biyolojik çeşitliliği ve ekosistem servislerini kararlı bir şekilde muhafaza eden yaşam alanları olarak kabul edilmektedir. Zeytin tarımsal ekosistemi tarımsal aktivitelerle şekillenmiş ve şekillemektedir, ancak bu durumun biyolojik çeşitlilik üzerindeki etkileri iyi bilinmemektedir. Toprak ve vejetasyon mikrohabitatları (ağaç tacı ve zeytin altı bitki örtüsü) örümcek komüniteleri ile yerel veya tarım uygulamalarına bağlı habitat özellikleri ve çevresel değişkenler, organik ve geleneksel uygulamalar altındaki zeytinliklerde ve kalıntı doğal habitat parçalarında örneklenmiştir. Veriler çok değişkenli analizlerle incelenmiş, komünite kompozisyonu ordinasyonla (NMDS, CCA, RDA modelleri); biyoçeşitlilik değişkenleri ise regresyon ve varyans analizleriyle (ANOVA, ANOSİM ve permütasyon testleri) değerlendirilmiştir. 20,167 birey ve 278 türe ait ergin örümcek çalışmaya dahil olmuş >300 örnekleme biriminde incelenmiş, örneklem büyüklüğüne göre standardize edilmiştir. Komünite kompozisyonu habitat tipi ve microhabitatlara göre değişiklik göstermiştir. Zeytinlik ve karışık ormanların, maki ve çam ormanlarına göre daha zengin örümcek komüniteleri olduğu bulunmuştur. Toprak mikrohabitatı vejetasyona göre daha zengin ve çeşitlidir. Organik ve geleneksel tarım arasında, çalışma alanında komünite düzeyinde fark bulunamamış, ancak toprak yönetimi ve otsu bitki örtüşünün komünite üzerinde anlamlı etkileri ölçülmüştür. Sonuçlar zeytinliklerde örümcek kompozisonu – çevresel değişkenler arasındaki etkileşiminin anlamlı olduğunu göstermektedir ve biyoçeşitlilik barışık zeytin tarımı uygulamalarının geliştirilmesine katkıda bulunacaktır

Anahtar Kelimeler: Araknoloji, Komünite Ekolojisi, Tarım-Çevre, Biyoçeşitlilik

For all contributors of ecological science

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INTRODUCTION

1.1.Anatolian Coast of the Mediterranean – An Antropogenic Ecosystem

Mediterranean Basin is one of the ’Biodiversity Hotspots’ as human interference has been prominent in this ecosystem historically for thousands of years, beside the edaphic, climatic, ecological or bio-geographical ‘factors’ which have all been contributed to an elaborate biological diversity. These natural or anthropogenic ‘factors’ show striking mosaic patterns in the Mediterranean landscape together with a species rich ecological community and there is a tendency to explain this coexistence by a presumed adaptation settled between two (human influence & biological diversity) over the evolutionary past (Blondel et al. 2010).

Although this explanation is strong, it is barely the picture today. Land use patterns have been dramatically changed-intensified within the last several decades and the scale of threats are now huge, threatening the persistence of Mediterranean biodiversity. Exploitive trends in land use policies are the main drivers of the supposed change and the consequences on biodiversity are still not yet well documented.

Agriculture has a major share among other sources of anthropogenic impacts in the region. It has left its mark in the Mediterranean over thousands years of history; and olive has been the single most widespread and dominating agricultural product in the region. Olive groves represent a biotope in the Mediterranean, for being very widespread and potentially rich in native biodiversity; thus potentially constitute a rarely found opportunity for conservation. Current knowledge suggests that low intensity farming systems are particularly better for harboring native biodiversity compared to high intensity farming systems (Siebert, 2004). However, farming intensity can be quite variable and neither its magnitude or variability nor the impacts on biodiversity are unknown at the Anatolian coast, which was a focal point in this study.

Agricultural policies directed on enhanced productivity have transformed the formerly low-input traditional farming systems into high intensity conventional agriculture, which have lead spreading of monoculture landscapes with intensive chemical use (Piorr, 2003). Farmland abandonment has been another consequence, when some particular traditional farmland systems are not compatible with modern trends and thus not profitable anymore (e.g. olive groves on the sloping land, Stroosnijder et al. 2008). In both cases, farmland habitats are subject to dramatic changes in structure and habitat quality, which can be relatable to loss of native biodiversity.

Loss of natural habitats due to anthropogenic land exploitation is another actual threat, especially due to its coupling with lack of conservation concerns. This is currently true for natural habitats as well as olive groves at the Anatolian coast. Pine forests of *Pinus brutia* constitute one of the dominant Mediterranean-type habitats beside the olive groves; others can be classified under tall and low shrubs (maquis and phrygana respectively) or mixed forests. Urbanization, tourism related construction, energy or mining industries are very prominent threats, as there is no valid protection on a significant portion of these habitats. Forests are protected by the Turkish Forest Law (6831), but this law does not provide protection in practice for majority of the ecosystem since 1950's, as the Mediterranean characteristic shrub vegetation is not categorized as forest in the same law, but instead, as 'degraded' forest which are prone to 'misguided forestation' activities or available for exploitative use (2B-lands) (Şekercioğlu et al., 2011). Protected areas represent more or less reliable conservation alternatives for now, but their extent is currently too small (Konukçu, 2001, Yurdakul et al. 2011).

With its diverse but largely unrecognized habitats, the Mediterranean coast of Turkey corresponds to one of the least known Pleistocene glacial refuge in west Palearctic (Médail and Diadema, 2009), with endemic or un-described species representing a significant portion of the community among invertebrates. This unique fauna is consequently exposed to a potential risk of extinction (with reference to a brief review by Baletto & Casale, 1991 on the overall Mediterranean). Understanding the effects of different land use types on such a unique biodiversity is crucial for developing case specific strategies of biodiversity conservation and sustainable land use.

1.2. Natural Value of Olive Groves

Olive is a widespread cultivar in the Mediterranean as already mentioned, as well as it is in Turkey. Its role in economy and social development is crucial as it provides the needs for nutrition, raw material as well as employment for the population. According to the records from 2015 Turkish Statistical Institute, olive groves represent a quarter of the total orchard area and according to Boyraz et al. (2010) olive farming provide direct mainstay for around 400.000 families in Turkey.

Apart from its economic relevance, olive tree (Olea europapaea) is also important as a native component in the Mediterranean ecosystem. Over its long and largely forgotten history, olive groves spread in the Mediterranean by replacing natural habitats and causing dramatic changes in the landscape (Loumou and Giourga, 2003; Zaferatos, 2011; also see Kocadağlı, 2009 for history of olive groves in Turkey). Today, on the other hand, olive groves under traditional agriculture are regarded as resilient habitats, which potentially harbour native biodiversity and ecosystem services (Loumou and

Figure 1. 1. A typical scenery from the Aegean coast of Turkey, Muğla, Milas

Giourga, 2003; Stroosnijder et al., 2008; Simon et al., 2009; Resica et al., 2013). Some notions for explaining the natural value & biological diversity of traditional olive groves are as follows:

- olive tree is a natural component of the Mediterranean native vegetation maquis;
- olive farming has a thousand years long history in the Mediterranean;
- olive farming has been inherited with sustainable and low-input farming practices over the history;
- groves represent a permanent habitat (unlike most annual crops which are being destroyed with harvest);
- groves are often found in mosaic patterns with natural habitats, enabling migration & colonization in between.

Recently, however, traditionally low-input olive farming systems have been converted into intensive forms of conventional farming by the influence of European Union (EU) agricultural policies aiming to increase agricultural productivity (Zaferatos, 2011; Resica et al., 2013). Philosophy of intensive farming systems is based on acquiring production yields at maximum and keeping a fixed production rate between harvests. Such systems are often implemented in monoculture fields with closely planted and short olive cultivars which are under intensive tillage, pruning, irrigation or chemical use (pesticides, herbicides, fungicides and synthetic fertilizers) practices. It is known that such systems can harbour only a small portion of the native biodiversity (Biaggini et al., 2007; Zaferatos, 2011).

1.3. High Nature Value Farming and Agri-Environment Schemes

Soon after realization of the variability in agriculture – biodiversity interactions (as exemplified above with the olive example), it was also understood that high intensity farming practices have unacceptably high costs and negative consequences on the ecosystems. 'High Nature Value Farming' (HNVF) concept has been defined against high intensity farming and became more of an issue as it has also understood that biodiversity conservation is possible with certain farming practices (Baldock et al., 1993).

Today, HNVF is integrated in the European Union (EU) Common Agricultural Policy and there are supports through the Agri-Environment schemes. Turkey has started to realize the agri-environment schemes under the framework of IPA (Instrument for Pre-Accession Assistance) Rural Development Program (IPARD) in a limited context; but as there have been no criteria identified on biodiversity-agriculture interaction, currently there are no support schemes which may target this subject. From a conservationist perspective, it is important to clear the way for such promotions. For this purpose, revealing the effects of different agricultural practices on the biodiversity in local scale is necessary for planning agriculture aiming biodiversity conservation.

Organic farming is a similar approach and a modern alternative as a low input system against the high intensity conventional farms, and also supported with agri-environment schemes (Tuck et al. 2014). Due to the restrictions on chemical use in organic farms, economic feasibility largely relies on ecosystem services (pest control; matter cycle / soil fertilization) provided by biodiversity (Vandermeer, 1995, Barbosa., 1998; Zehnder et al., 2007). Indeed, organic farms usually constitute a better alternative on hosting biodiversity compared to high intensity conventional farming (Bengtsson et al., 2005; Hole et al., 2005; Birkhofer et al., 2008a; Rader et al., 2014). However, interaction between organic farming and biodiversity depends on the organism group (Birkhofer et al., 2014) or the type of individual practices. For example, monoculture farms being managed without taking landscape complexity into account tend to be poor in native biodiversity (Fahrig et al., 2011). On the other extreme, it is also possible to form a biodiversity friendly system in high intensity farming as well, with proper decisions in management (Gomiero et al., 2011).

In order to define criteria for regulation of agricultural practices aiming conservation of biodiversity, it is necessary to monitor and compare local biodiversity in farmlands under different farming practices. In Turkey, there have been no studies regarding interaction of olive farming practices and biodiversity until recently. Aiming to fulfil such a shortcoming, a three years long project (TOVAG 213O147) with the title "Identifying Biodiversity Friendly Agriculture in Olive Groves at Kaz Mountains" has been started in 2014. This project was focused on determination of high nature value farming by monitoring and comparing biodiversity in olive groves under organic and conventional farming practices at the southern foothills of Kaz (Ida) Mountains. In this respect, biodiversity was intended to be summarized as community composition and diversity of popular indicator groups: birds, butterflies (Lepidoptera: Rhopalocera) and understory plants. Ideas and efforts spend for this project were used as a basis for studies performed under this thesis.

1.4. Spider’s in Agricultural Biotopes

Spiders are one of the most abundant and diverse organisms in the terrestrial biotopes, therefore being able to observe them in agricultural environments is not surprising. However, their presence has been consistent enough in such biotopes in a way that their use as subject organisms in agro-ecology is getting widespread (Birkhofer et al. 2013). If we call this “agro-arachnology”, most popular topics in this field are about predator nature of spiders, emphasizing their biological control potentials. Enhancing their abundance in agricultural biotopes where they naturally, occur is one of the major concerns of so-called conservation biological control (Barbosa, 1998). Spiders however, haven’t become number one biological control agents, due to their generalist (unselective) predatory nature (Pekar et al. 2012), which is not appreciated by agriculturists. On the other hand, it has been demonstrated that they have established with persistent species and communities in agricultural landscapes. These species are called agrobionts (Samu & Szinetar 2002). Certain traits were associated to agrobiont spiders, such as being habitat generalists (Duffey, 1978 and Luczak,1979). Wissinger (1997) proposed that agrobiont adaptations are for predictably ephemeral habitats. Life history characteristics (Duffey, 1978), competitive ability (Marshall & Rypstra 1999), tendency for intraguild predation (Wagner & Wise 1996) and colonization power (Marshall et al. 2000) have been counted among these traits (Samu & Szinetar 2002).

1.5. Spiders for Bio-Indication

The ecological indicator concept is a practical way of reasoning over biological communities or natural value of an environment, by monitoring a portion of the organisms in sympatry with the rest of the community (McGeoch, 1998; Caro, 2010). Birds, butterflies (Lepidoptera: Rhopalocera) and plants are among popular taxa as surrogates, which have been used for a variety of purposes including ecological indication (Padoa-Schioppa et al. 2006; Bonelli et al., 2012; Rader et al., 2014). Their popularity is mainly based on the following reasons:

- availability of specialists: as these are visual organisms, many people bear affection among the academy or amateurs and specialists are easy to find from both sources;

- ease of monitoring: these organisms are usually easy to observe & identify in the field and there are well established practical methods for their monitoring in the field.

On the other hand, it is controversial that whether their use is appropriate in every situation or not (Landres et al., 1988; Andelman & Fagan 2000). Besides, the representativeness of indicators can be increased by use of a variety of different organisms together (Allen et al. 1999; Fleishman et al. 2005). For these reasons, there is a need for identifying alternative organisms for their potential role as indicators.

Terrestrial arthropods are prominent candidates among the alternatives. Some of their advantageous characteristics can be summarized as follows:

I. Rigorous sampling protocols present. Arthropod sampling methods are repeatable and eligible for standardization; thus it is easy to provide comparable results. Also they are cheaper in both time and money and the quality of data is usually better compared to vertebrate sampling (Coddington et al.,1991; Cardoso et al., 2008; Cardoso, 2009).

II. Their ubiquity and sensitivity. Arthropods are usually species rich and abundant in communities and their response to environmental variation is fast, resulting dramatic changes in communities following impacts (Kremen et al., 1993).

III. They are easy to sort into morphospecies. Despite of difficulties on their taxon identifications, morphological characters are diverse and variable among arthropods, enabling para-taxonomy as an eligible approach. Recognizable taxonomic units can be identified in high accuracy as an analogous approach to actual taxonomy (Oliver & Beattie, 1996; Derraik et al. 2002), enabling comparisons between community-ecological samples without getting limited by the rarely found taxonomists.

Certain arthropod groups are getting prominent as indicators of environmental change and their use in this context is getting increasingly prevalent (Mc Geoch et al. 2011; Gerlach et al. 2013). Among these, spiders represent one of the most abundant and species rich predatory group in natural and agricultural habitats (Wise, 1993; Marc et al. 1999; Morris et al., 1999). Their use as indicators of anthropogenic land use patterns or habitat types has been pronounced many times in the literature (e.g.: Gunnarsson, 1990; Uetz, 1991; Wise, 1993; Marc et al.,1999).

In addition to the characteristics counted for arthropods above in general, spiders are prominent as an indicator group for the following specifications:

IV. Taxonomic diversity is followed by ecological diversity. When considered as a guild, spiders do not represent a homogeneous functional group in terms of their ecology, unlike many other invertebrate taxa. Instead, they are very diverse in terms of body size, predation or foraging strategies and habitat choice (Marc et al., 1999); which correspond to many functional groups in a given community (Cardoso et al. 2011).

V. Spiders are everywhere in high richness and abundance; being able to provide data that is appropriate for statistical analyses. They are exceptionally well dispersed through all possible micro-habitats, from soil surface to the highest canopy in a typical terrestrial ecosystem. As this is combined with their high species richness and abundance, differences on their composition between communities (from different habitats or land use patterns) are easy to measure (Prieto-Mendez & Mendez, 2011).

VI. Despite their ubiquity, many spiders have a limited dispersal capacity (except for ballooning species) and they have high local and regional endemism levels; therefore, they present high spatial resolution data (New, 1999; Feest & Cardoso, 2012).

VII. Spiders are also prominent due to ease of their taxon identifications (Jocque et al., 2013) among the megadiverse arthropod taxa. One specialist can identify almost all spider genera in a typical spider community, which is unlikely for many invertebrate groups with such a high diversity. E.g.: An equally intensive workload on insects from different orders would require collective work of many entomologists.

VIII. Predatory roles. As a group unexceptionally composed of predators, spiders are placed at the highest trophic level among terrestrial invertebrates (Main, 1987) and insects represent their main prey item (Turnbull, 1973; Riechert & Luczak, 1982). As one of the major insect predators, spiders take role in controlling insect pest populations as an ecosystem service (Riechert & Lockley, 1984; Nyffeler, 2000). Spiders are also known with their ability to track resource abundance and multiple species can simultaneously increase their abundances in a community, by assembling in response to prey populations (Marshall and Rypstra, 1999; Marshall et al. 2000; Harwood et al. 2003). Such a predatory response refers to an indication value on the most diverse terrestrial invertebrate group, insects.

1.6. Spider Biodiversity in Turkey

Understanding the nature of determinants on community structure and composition is the first step for evaluating spider’s potential as ecological indicators. However, it is obvious that there is almost no information on spider communities in Turkey at all, regardless of the extensively mosaic conformation of land use patterns and habitat diversity in the country. A review of the available literature indicates that our knowledge on spider biodiversity in Turkey is mainly based on scattered faunistic records, lacking comprehensive catalogues, community level surveys or discussions on effects of habitat type or land use patterns on spider communities. Published literature on spider biodiversity have been mostly in the form of incomplete faunistic catalogues, with contents limited to municipalities or smaller governmental borders and sometimes delimited natural areas, but unexceptionally lacking any reports on sampling efforts or inventory completeness and uninformative on the variability between habitat types or landuse patterns. See Bayram et al. (2016) for a complete bibliography. Below, I have tried to summarize the most relevant studies with a community level perspective, among the available literature:

- The most extensive studies were carried out by one of the first Turksih arachnologists, Abdullah Bayram. Bayram et al. published reports providing species lists from a variety of farmlands such as clover, cotton, tobacco and wheat fields (Bayram ve Allahverdi, 1994; Bayram vd., 1998; 1999; 2000; Efil vd., 2012) and also published about activity and seasonal dynamics of some species (Bayram 1994; 1996a;b; Bayram ve Varol, 1996; Bayram ve Varol 2001). He has also published faunistic lists for some delimited areas (Bayram ve Varol 1996; Bayram 1996c; Bayram vd., 2005).

- Various other authors similarly published faunistic lists for delimited territories (Topçu vd. 2005; Özdemir vd. 2006; Kaya ve Uğurtaş 2007; 2011; Lecigne 2011; Helsdingen 2013; Türkeş ve Karabulut 2013).

- There are only three publications found in the available literature, reporting faunistic lists from perennial orchards in Turkey; Bolu vd. (2008) studied in almond orchards from South-Eastern Anatolia; Öztürk vd. (2013) studied in olive groves and pomegranate orchards from Adana, Mersin and Osmaniye regions; Kaçar (2015) in persimmon, almond, avocado, locust, citrus, fig, loquat and pomegranate orchards from Çukurova and Amik basins. All three publications provided faunistic lists for taxa included in their samples, but these lists are far from being complete and uninformative on spider communities for the sampled habitats.

- Master thesis by Elverici (2012) is currently the only available report on olive grove spider communities from Turkey. Elverici (2012) studied spider communities of various microhabitats in traditional olive groves found in a mosaic landscape with shrub lands at the Aegean coast of Turkey, in Muğla, Milas, Kıyıkışlacık. Some traits described for olive grove spider communities and relevant to the current proposal are as follows:
    - Spider species richness was measured and estimated as 221 and 250-300 respectively, indicating a comparable richness with the tropics. Identification as fine as species level was not possible for 13% among the observed taxa, despite of extensive literature research or consulting

efforts with senior colleagues. This implies a species rich community and high ratio of undiscovered or new species for the study region.
- Bulk of the spider community is composed of annual stenochronous species and community composition varies greatly between seasons. Species richness is most measurable in late spring and early summer as most of the species show activity as adults (the only life form identifiable to species level) during this period; while winter and early spring are prominent with higher proportions of endemics.
- The spider fauna apparently belongs to East Mediterranean with high similarity between checklists from Aegean islands, Greek, Bulgarian and Israel main lands; while northern, central and western European fauna appears to be more distant. This implies the necessity for special interest focused on the impacts of land use patterns on biodiversity at the East Mediterranean, which is the least studied region on this manner.

1.7. Impacts of Farming Practices on Spiders

1.7.1. Community Level

In general speaking, the impact of agriculture on spiders is dramatic at the community level (Prieto-Benitez & Mendez, 2011). However, variability of input intensity between different farming systems (e.g.: organic or high-intensity conventional) induce differential responses on spider communities in return. Abundance and diversity tend to remain higher with organic farming, in comparison to high-intensity conventional farms (Glück & Ingrisch, 1990; Basedow, 1998; Feber et al., 1998; Pfiffner & Luka, 2003; Birkhofer et al., 2008a; Tahir & Butt, 2009; Eyre & Leifert, 2011; Batáry et al., 2012). On the other hand, there have been some reports without any measured differences (Clough et al., 2005, 2007; Schmidt et al., 2005; Diekötter et al., 2010) but mainly in cereal farms. Responses also vary among taxa; e.g.: Linyphiidae as one of the richest spider families with many agrobiont species (agricultural habitat specialists), tend to decline with organic farming (Öberg, 2007; Birkhofer et al., 2014).

Beside the holistic assessment of farming systems as discussed above, it is also important to identify community responses on each particular farming practice separately from one another:

- Pesticide use: Response to pesticides depends on type of pesticide, although it is usually negative (Birkhofer et al. 2008a; Oraze & Grigarick 1989; Riechert 1999). Spiders are shown to be more vulnerable to broad-spectrum pesticides compared to pest prey (Yardim & Edwards, 1998; Maloney et al., 2003); while they are usually tolerant to specific pesticides, herbicides and fungicides (Pekar, 2002). Indirect effects have also been depicted by decrease of spiders associated with decrease on non-target arthropod prey following the pesticide use (Marko et al., 2009).

- Herbicide use and weed control: There is a positive interaction between vegetation structural diversity and spider biodiversity (Diehl et al. 2013). As spiders are usually tolerant to herbicides, responses are usually indirect: spiders tend to decline with loss of vegetation structural diversity (Chiverton & Sotherton 1991). Some explanations in this respect are as follows: 1- habitat loss; 2- bottom up cascade following decrease on herbivore prey abundance; 3- loss of refuge against predation or cannibalism (Sunderland & Samu, 2000; Birkhofer et al., 2008b).
- Tillage: Impact of direct physical interference on soil is usually negative on both spiders and other soil invertebrates including spider prey (Heimbach & Garbe, 1996; Holland & Reynolds, 2003; Butt & Sherawatt, 2012). Thorbek & Bilde (2004) emphasized that spiders are also vulnerable to mechanic effects which occur during execution of tillage. Reducing the extent of tilled area has been suggested to increase spider biodiversity in farmlands (Samu et al.,1999). Furthermore, Rodriguez et al. (2006) suggested spiders as an indicator group on identifying intensity of interference on the soil.
- Fertilizer Use: Organic fertilizers such as animal manure have been associated to increase in numbers of soil invertebrates (Pimentel & Warneke, 1989; Moreby et al., 1994; Eaton et al., 2004). Increase in abundance of spider prey has been related to numerical response by spiders in the literature (Birkhofer et al., 2008b). However, unlike organic fertilizers, extensive use of synthetic fertilizers has shown to have negative impact on spider biodiversity (Batáry et al., 2008; Birkhofer et al., 2008a).

1.7.2. Population Level

Unlike community level approaches as exemplified above, research on population level responses of spiders or similar taxa against agricultural impact have been scarce in the literature and present studies have focused on a few parameters, such as fitness related measures (body condition & fecundity):

- Some species take advantage from farmland intensification. Oberg (2009) reported that *Pardosa* wolf spiders (Lycosidae) were in better condition in structurally simpler monoculture cereal farms compared to heterogeneous farmlands, regardless of the management type (organic & conventional). Drapela et al. (2011) reported similar results in canola fields: Pardosa spiders were in better condition in monoculture fields compared to heterogeneous fields in mosaic patterns with natural vegetation.
- Contrarily, beneficial effects of low intensity farming have also been measured for spiders or similar predatory organisms. Östman (2001) found evidence for beneficial effects of organic farming on body condition of another predatory

ground dwelling arthropod group (Carabidae). Similarly, Corrales and Campos (2004) calculated lower mortality and higher fecundity rates for another predatory arthropod group (Neuroptera) in organic olive groves, compared to high input conventional farms. Jakob et al. (1996) measured both body condition indices and body size measures of Pardosa spiders in tilled and non-tilled soybean fields and found that individuals were significantly larger in no tillage systems.

There are several studies known from habitats other than farmlands or based on laboratory experiments, but still interesting enough to mention here in order to point out spiders response to environmental variables, mostly related to variations on prey availability and feeding history. A better understanding on the relationships between such variables and traits of spiders may contribute to our understanding on indication value of spiders and there is a gap on the topic in the literature regarding impacts of agriculture (Winqvist, 2012; Birkhoefer et al. 2013).

- Spiders are commonly food limited in nature (Anderson, 1974; Miyashita, 1968,1992; Wise, 1993; Chen and Wise, 1999; Moya-Larano et al. 2003).
- Spiders modify their foraging behaviour in response to prey availability (Riechert & Luczak, 1982). For example: Lycosa wolf spiders were observed to perform seasonal movements from areas of low prey availability to high (Kronk & Riechert, 1979). Similarly, Miyashita (1992) reported orb weaver spiders to increase their population density with increasing prey abundance by foraging. Marshall et al. (2000) measured increases on population densities in two species of wolf spiders (Lycosidae) in response to experimental prey addition. Hardwood et al. (2003) showed that availability of preferred prey type is a determinant on web site selection in web-building spiders.
- Feeding history is another determinant on spider foraging. Pasquet et al. (1999) showed that experimentally food limited web-builders (Eresidae) tend to build larger webs at maturity compared to food supplemented individuals, which corresponds to increased foraging activity. Moya-Larano et al. (2003) measured similarly higher foraging activity in hungry cursorial Lycosa wolf spiders compared to supplemented ones, while this effect turned into the opposite at maturity as historically well fed individuals were more active at the adult-hood.
- Spider body condition is relatable to prey availability or feeding history. Bucher and Entling (2011) recorded an orb-weaver spider species in better body condition in higher prey abundances. Walker et al. (1999) observed better body condition measurements in experimentally well fed spiders compared to hungry ones.

- Fecundity and quality of offspring can increase with higher prey abundances or feeding success (Miyashita 1986,1992, Moya-Larano, 2002). Miyashita (1986) further reported ‘numerical response’ by recording higher juvenile abundance in prey-rich sites at the following year (in an annual orb-weaver). Body size in female spiders (related to foraging success) tend to be positively correlated with the number or biomass of eggs produced (Vollrath, 1987).

1.8. Impacts of Olive Farming Practices on Spider Communities

Olive groves are unique among agricultural habitats particularly as they are potentially rich in native biodiversity. In such an orchard environment, it is possible to identify several micro-habitats: soil surface, understory herbaceous vegetation and tree canopy can be classified as the most conspicuous micro-habitats, which are also rich in spiders (Elverici, 2012).

On the other hand, the information reviewed above on the impacts of agricultural practices on spiders and related taxa are largely based on studies performed in agricultural environments other than olive groves and from distant locations in the world. The literature is currently poor on agriculture - biodiversity interactions in olive groves and it is not yet possible to make general interpretations on impacts of individual agricultural practices.

A review of the available literature can be summarized as follows:

- Spider communities are characterized with higher species diversity under low input olive farming practices compared to high intensity farming. Ruano et al. (2004) compared soil and canopy micro-habitats in olive groves under organic, integrated and conventional farming systems in Spain and found that invertebrate orders are more abundant in organic and integrated farms. Jerez-Valle et al. (2013) compared soil micro-habitat in organic, tilled and non-tilled conventional olive farms in Spain and measured higher diversity of arthropod orders in organic farms compared to conventional farms independent from the soil management type. Gkisakis et al. (2015) similarly compared soil micro-habitat in organic, integrated and high intensity conventional olive groves in Crete and again found high diversity of arthropods in organic and integrated farms.
- Pesticides commonly used in olive groves have negative impacts on spiders. Scalercio et al. (2009) reported negative effects of pesticides used in olive fruit fly (*Bactrocera oleae*) control (rotenon, azadiraktin and copper oksiklorid) on non-target arthropods with decreases on spider abundances following the application. Rodriguez et al. (2003) similarly reported negative effects of deltametrin use on invertebrates, which is particularly used against olive bark beetle (*Phloeotribus scarabaeoides*). In a similar study Santos et al. (2007)

showed dramatic declines in predatory arthropods including spiders, following dimethoat use against olive moth (*Prays oleae*).

- Olive tree canopy is the most sensitive micro-habitat in olive groves. Ruano et al. (2004) showed that olive tree canopy community in Spain is more dramatically affected from management type, compared to soil community. Cardénas et al. (2006), again in Spain, compared canopy spiders in groves under different management intensities and found that abundance and family level diversity are significantly low under intensive farming and family composition is different. Pascual et al. (2010) measured declines on diversity of canopy arthropods including spiders, following use of kaolin against olive fruit fly in Spain.
- The nature of interaction between the understory vegetation and soil spider biodiversity is not clear; although spider abundance seems to increase by the presence of vegetation. Cardenas et al. (2012) reported a decrease on soil spider abundance following vegetation clearing, although species composition was unchanged. Paredes et al. (2013) examined the interaction between understory vegetation and natural vegetation adjacent to olive groves; maximum spider abundance was measured with co-occurrence of both vegetation types.
- Similarly, impacts of direct physical interference on soil spiders or other soil arthropods are not clear. Rodriguez et al. (2012) compared predatory arthropods in olive groves with cover crops or tillage systems, but did not find significant differences in family and order level abundances. However, Rodriguez et al. (2006) in their comparison between tilled and non-tilled olive groves, found that spiders are more abundant both on tree canopy and soil microhabitats in non-tilled farms. Both studies were performed in Spain.

In the generality of studies reviewed above, taxonomic resolution was left at the order level (Araneae) and the compared variable was abundance. There are only a small number of studies in which taxonomic resolution was kept at generic or specific level. Thaler and Zapparoli (1993) defined community composition of soil spiders in Italian olive groves. Morris et al. (1999) reported composition of olive canopy spider communities from Spain and also defined spiders as the most species rich predatory arthropod group at the canopy. By also considering that all of the comparative studies were performed in Spain or Italy, it is not yet possible to characterize olive grove spider communities and their responses to olive farming practices solely from literature for the Mediterranean coast of Turkey.

1.9. Spider Fauna of the Study Area and Its Environs

Our knowledge on spiders of Çanakkale province and Kaz Mountains range is severely limited. To my knowledge, there have been one master thesis (Ayva, 2002) and one discrete poster presentation (Varlı et al., 2012) reporting preliminary catalogues for spiders of Kaz Mountains independently from each other. Both studies are narrowscoped by the effort spend for sampling and number of specimens examined. Bosmans et al. (2014) described Zodarion bigaense as a new species for science from Biga district (Çanakkale). There is no study regarding spiders from olive groves from the region.

Although the literature is poor for the Kaz Mountain range and north western Anatolia, the region has an advantageous geographical position for spider community ecology studies, for being in close proximity to better studied Aegean islands and Balkans, due to availability of taxonomic literature which would be helpful for taxon identifications. Bosmans et al. gathered catalogues for spiders of Lesbos (Bosmans et al., 2009) and Crete (Bosmans et al. 2013) islands. Russell-Smith et al. (2011) catalogued spiders of Chios island. Fauna of Greek (e.g.: Deltshev, 1999; Bosmans & Chatzaki, 2005) and Bulgarian (e.g.: Deltshev 1999; Deltshev 2005) mainlands were studied by numerous authors and catalogued by Van Helsdingen (2013).

1.10. Standardized Sampling of Spiders - Summary of Methods

It is usually not feasible to measure biodiversity of spiders completely due to their "mega-diversity" and also due to common limits on monitoring opportunities (limited time, money, availability of specialists etc.; Cardoso, 2009). Rapid biodiversity assessment (RBAS) programs have been developed in order to meet this problem. This approach did not originally target megadiverse taxa such as invertebrates, but it is applicable for a wide range of taxa by keeping costs at reasonable levels, at the same time aiming to get: 1) reasonably representative, rigorous measures for sampling sites, 2) comparable results between sites, on the organism group under examination (Oliver ve Beattie, 1996; Duelli, 1997). Thanks to adaptation of RBAS into arachnological inventory sampling protocols, feasible alternative protocols for the Mediterranean region could be developed (Cardoso et al. 2008).

Here, it is important to note that neither sampling nor identification of spiders can be performed "so rapidly". Rather than that, arachnologists approach focus on delimited spatial and temporal aspects of sampling. This can be practiced by quadrat (or transect) based sampling at a certain time, for a delimited period. Best sampling period for spiders in the Mediterranean have been repeatedly shown as late spring / early summer (Cardoso et al., 2007; Chatzaki et al. 2005; Elverici, 2012), as the identifiable adult activity is known to be highest in the community.

There are commonly held sampling protocols already established for sampling spiders. Use of a combination of methods and repetitive sampling for collecting community level data was first offered by Coddington et al. (1991) and then repeated, developed and standardized by many authors (Silva and Coddington, 1996; Coddington et al. 1996; Dobyns, 1997; Toti et al. 2000; Sørensen et al. 2002; Scharff et al. 2003; Cardoso et al. 2008; Coddington et al. 2009; Cardoso et al. 2009). Sampling methods used in these studies can be classified under semi-quantitative protocols, in which sampling efforts are standardized.

Further approaches for standardization of semi-quantitative samples require evaluation of equal number of samples (Simpson, 1964), sampling effort (Colwell et al. 2004), individuals (Gotelli & Colwell, 2001) or equal levels of sample completeness (in terms of coverage; Chao & Jost, 2012) between samples. Such standardizations may produce statistics that differ in nature however, as Magurran (2004) pointed out. Number of species per specified sample size or sampling effort (in terms of sampling units) gives the species density, which is different from number of species per specified number of individuals, which gives the numerical species richness. Sample based standardization require sampling units composed of multiple, smaller samples; whereas in individual based standardization, each sampling unit must be composed of numerous individuals.

Main methods for dealing with the aforementioned standardizations are rarefaction or extrapolation (Colwell et al. 2012). These two methods work complementarily on a species sampling curve based on an empirical reference sample (one like produced in this study), as more crowded datasets can be rarified into sparser ones, and sparser communities can be extrapolated to denser ones, this time guided by an asymptotic species richness estimator (Chao, 1984; Colwell and Coddington 1994). Such a framework is essential and becoming a standard tool today for comparison of richness (Chao & Chiu 2014) or diversity (Chao et al. 2014) analyses in megadiverse communities (also see Gotelli and Ellison, 2012).

Note that major concern behind these standardizations is enabling comparability between sample diversity measurements, as fundamental variables for the expression of biological diversity. However, there is a plethora of versions of biodiversity measurement in the literature (Washington, 1984), and it can be difficult to choose which statistic to use. A consensus has been achieved after a forum (Ellison, 2010), for using number equivalents (or effective numbers of species) of diversity indices, meaning that "*number of equally abundant species that would be needed to give the same value of a diversity measure*". Hill numbers met this requirement and have been increasingly in use recently, which were developed by Hill (1973) and reintroduced to ecologists by Jost (2006, 2007) and praised for many other advantages compared to rest of the diversity indices (Chao et al. 2014).

Hill numbers are statistics parameterized by a diversity order (q), which determines the measures sensitivity to species relative abundances in a sample (Hsieh et al. 2016) and gives the number equivalents. Species richness is a Hill number on its own (q=0) and the most widely used diversity indices Shannon (q=1) and Simpson (q=2) can be transformed into Hill numbers by taking the exponential or inverse of the original indices respectively. Hill's parameter value q=1 "can be interpreted as the effective number of common species", whereas q=2 "can be interpreted as the effective number of dominant species" in samples (Hsieh et al. 2016).

1.11. The Chorotype Concept in Biogeography and Its Use in Ecology

Faunistic checklists provide knowledge of local biodiversity, which represent a base for conservation or bio-indication purposes. Recently, many comprehensive checklists on terrestrial arthropods (particularly on spiders) from west Palearctic fauna have been accompanied by biogeographical assessments (Demir and Seyyar, 2017; Isaia et al. 2015; Komnenov et al. 2016; Paschetta et al. 2016). Such assessments are based on publications of Vigna Taglianti et al. (1993, 1999), who delimited geographical ranges of biogeographical significance for Anatolia and West Palearctic terrestrial arthropod fauna, based on known worldwide distributions of various arthropod taxa. Those efforts actually resulted by a classification of geographical ranges in a hierarchical or discrete fashion; all of them are shared by multiple taxa and they are called chorotypes. Later, Fattorini (2015) revised the chorotype concept and suggested its correct use as “global chorotype”, in order to discriminate the term from different meanings found in the literature. Chorotype concept primarily arose from biogeography, but it’s has also been proposed as a tool for assessing conservation value of insect communities Fattorini and Taglianti (2015).

1.12. Aims and Our Perspective on the Topic

During our efforts of designing studies in this thesis, we tried to focus on three main problems: a) exploring and documenting the spider fauna for the first time in an agricultural landscape; b) exploring community composition and diversity of spiders in major biotopes, native relicts or anthropogenic, in the olive agroecosystem; c) impact of agricultural management in olive groves on spider communities. By having the afore-mentioned perspective and with concurrent circumstances we had while performing the studies, we believe that direct measurements on little known taxonomic groups in the community ecological level are essential to understand the impact of agriculture, to describe sustainable alternatives to agricultural activities or to gain better understanding on conservation of biodiversity. Therefore we tried to solve these problems, as briefly described below.

Exploring the spider fauna was a challenge and a tough claim to achieve, since there wasn’t any previous study that we would make use of, and the necessary workload was huge for sustaining the assumptions of elaborate statistical methods. Standardized sampling was the only alternative approach to fulfill these requirements, which was quite labor intensive in the field. And then these efforts were accompanied by taxonomic analyses in the laboratory, to extract the needed data for the further analyses and forming the community datasets, similarly quite labor intensive.

Community composition and diversity of major Mediterranean biotopes have been rarely studied comparatively, therefore, many aforementioned info about diversity in olive groves has to be tested. We were especially curious about biodiversity of the maquis shrubs dominated biotopes, since these are threatened and unprotected in Turkey.

And finally, impact of agricultural management in olive groves on spider communities have been little known, with almost no empirical data available. Most of the speculations above were based on agri-environments other than olive groves. Turkish olive groves might be represented in the lower input management types, compared to rest of Europe. Understanding the drivers of biodiversity in such an environment is important to describe and compare with other parts of the world. In order to achieve this, it was necessary to monitor the management practices, together with other environmental variables alike in the agroecosystem, besides the spider communities. This was another challenge, which we could came through, hopefully in the best possible way.

CHAPTER 2

MATERIALS AND METHODS

2.1. Study Area

Study area is located at the west of Southern Marmara Region, on the northern shores of gulf of Edremit and southern foothills of Kaz Mountains; covers lands of Ayvacık and Edremit districts from Çanakkale and Balıkesir provinces respectively. Olive represents the dominating component of the vegetation, establishing an “olive grove belt” with a continuous cover up to an average of 350 m altitude, which is followed by pine forests at the higher altitudes (Kocadağlı, 2009). There are few remaining maquis, brutian pine or mixed forest patches in the “olive belt” (Cürebal, 2012). Plant diversity is high in the area, as it is located at the intersection point of the Mediterranean and Euro-Siberian bio-geographical regions (Eken vd., 2006).

Conventional olive farming represents the bulk of agricultural activity, while farming intensity varies from low input (almost no management) to high input practices with intensive chemical use. Moreover, “organic farming” and “good agriculture” practices are also known in the area. Organic olive groves are managed by TARIŞ Olive Oil Cooperative and it corresponds to 30% of total organic olive oil production in Turkey.

2.1.1. Biotopes

Olive groves represent the dominant biotope in the study area, as natural biotopes were replaced in the “olive belt” almost entirely. Hypothetically, maquis represented the majority of the transformed land as there has been no valid protection since 1950’s. However, there are reports indicating considerable losses of pine forests as well during the replacement process (Efe, 2000; Efe et al. 2008; Cürebal, 2012; personal communication with farmers). Today, there are few pine or mixed forest fragments left in the landscape, and maquis is almost extinct. Temporary surface water can be observed in the olive groves occasionally, creating a moist type of habitat.

We can summarize the most conspicuous biotopes in the study area and therefore evaluated in this study as follows:

a) olive groves – overly dominated by planted cultivars;
b) maquis shrub lands – short shrubs as typical in the Mediterranean;
c) pine forests – *Pinus brutia* pine forests;
d) mixed forests – dominated by oaks and brutian pine;

e) moist olive groves - with temporary surface water

2.1.2. Micro-habitats

Three types of micro-habitats are the most conspicuous in such an orchard environment:

a) soil surface
b) understory herbaceous vegetation
c) tree canopy (can be in the form of tree or shrub canopy)

These micro-habitats are known to differ in spider communities and sampling methods (Cardoso et al. 2008; Elverici, 2012), therefore treated separately in all analyses.

2.1.3. Sampling Periods

All sampling was designed to be performed in spring – early summer period, as this time of the year was suggested as the optimum time to sample the adult individuals of spiders most efficiently. Vegetation and soil surface sampling were performed at different periods, both within and between methods.

Soil surface was sampled for four times which corresponded to two different periods, the mid May and the late May – early June; as well as three different years: 2015, 2016 and 2017. Mid May sampling was performed once in 2015, whereas late May – early June sampling was repeated for three times: once in each year. We called these samples as periodic replicates. There were three reasons for collecting in such a design: 1) to be able to compare naturally occurring periodic shifts in species composition between sampling periods; 2) to be able to detect management impact on communities, in cases of management shifts between periods; 3) if there is no change detectable, then simply for increasing sample sizes for community or management impact analyses.

Vegetation sampling was performed earlier in the season, in order to avoid any damage to olive flowers and young fruits, as the olive blooming starts to occur in May. Sampling was repeated in 2016 and 2017 for herb. vegetation sampling, whereas tree canopy sapling was performed once in 2017.

You can find further details under each method below.

2.2.Sample Collection

Sampling was performed under various schemes aiming to produce datasets for data analyses in further chapters. Two different sampling methods were used: 1) pitfall trapping; 2) vegetation vacuuming. Sampling units were designed as amalgamations of multiple smaller replicates in order to satisfy requirements of statistical approaches. Sampling locations were determined in GIS software by using data from various sources in order to standardize environmental variables, then used after further approval in the field.

2.2.1. Sampling Site Selection

Sampling sites were partly common with those from TOVAG 213O147 project and chosen as the result of cooperative work by taxon specialists from universities (METU, Gazi and Erciyes Universities) and GIS specialists from Nature Conservation Center.

Step by step methodology, data sources and criteria for site selection can be summarized as follows:

- Spatial data from various sources were digitized in Geographic Information System (GIS) environment.
- Sites were selected by parcel ownership or land cover data.
- Cadastral information on olive groves were extracted from Parcel Inquiry Application (https://parselsorgu.tkgm.gov.tr) of the General Directorate of Land Registry and Cadastre, Ministry of Environment and Urbanization of the Turkish Republic.
- Lot and block number information of the ECOCERT certificated organic olive groves provided by TARIS Olive and Olive Oil Sales Cooperatives Union, Küçükkuyu Olive and Olive Oil Sales Cooperative and Çanakkale Provincial Directorate of Food, Agriculture and Livestock.
- Land cover info was extracted from STATIP land cover data based on satellite imaging; Corine land cover maps and Google Earth images.
- Sites were aimed to have minimum 6 hectares of homogenous biotope coverage in order to avoid edge effects. Most of the selected sites had single owner, but parcels larger than 6 hectares were rare in the study area. Inevitably we had to include smaller parcels, by combining adjacent lots (distance between lots <10m) under similar management, so that 6 hectare criteria could be met.
- Distances between sites were 500 m at minimum to be able to differentiate agricultural practices and microhabitat effects (spatial autocorrelation).
- Distances to urban lands, main roads or other biotopes were 500 m at minimum in order to avoid edge effects.
- Distances to permanent watercourses was 300 m at minimum.
- Altitude was restricted below 300 m.
- North bound aspects were excluded.
- Candidate sites were selected on the GIS environment and visited in the field at 17-21 April 2015 and 17 May 2015 in order to finalize decisions.

Site selection resulted in six organic and ten conventional olive groves, two mixed forests, three maquis shrublands and four pine forests.

2.2.2. Sampling Units

Transects

Sampling units were transects, which were designed as 100 m long lines systematically designated in the sites. One to six transects (mostly three) were designated depending on the total area in each site. Transects were located considering homogeneity in terms of land cover or management type.

Quadrats

Quadrats of 3x10 m^2 size were used as sub-sampling units on the transects. Two quadrats were systematically located towards the ends of transects. Sampling of vegetation dwelling spider communities (vacuum) and measurements of environmental variables were performed on the quadrats. These measurements were then transformed to transect level by amalgamation and by averaging respectively, and then used in the analyses.

Traps

Traps were unique to pitfall trap sampling, which was used to sample ground dwelling spider communities. Ten traps were deployed systematically per transect, equally distant to each other in 10 m intervals. Trap samples were transformed to transect level by amalgamation before the analyses.

2.2.3. Sampling Methods

This study follows a semi-quantitative approach for sampling spiders. Community level data have been collected in sampling units (transects) aiming to represent composition (=presence of taxa) and structure (=abundance of taxa) in the spider community. Different methods were used in order to sample different microhabitats in sampling sites: soil surface, understory vegetation and tree (or shrub) canopy:

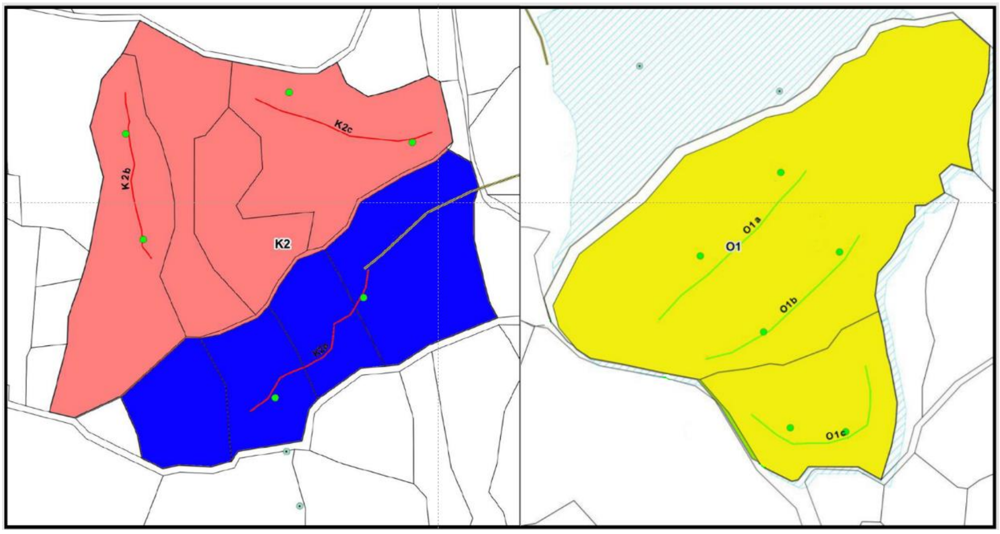


Figure 2. 1 Examples to sampling sites, transects and quadrats- black lines indicate parcels, color filled polygons indicate ownership, red and green lines indicate transects and green dots in the colored polygons are quadrats

Pitfall Trap Sampling

This method has been commonly applied for sampling edaphic spider fauna and became traditional (Uetz and Unsicker, 1976); while it is also eligible to use for comparative purposes on biodiversity (Cardoso et al. 2009). This method was used in olive groves in Muğla province (Elverici, 2012) and found to be very cost- effective while providing best results for comparisons between sampling units. At the same time, this method sampled almost half of the total richness measured in the groves (106 species of 221).

Pitfall trap methodology was performed by using plastic beakers of 9cm diameter and 12cm depth as traps; a hand type auger with 9 cm tip diameter for drilling holes for trap placement. It was important to place traps in these holes for keeping the beaker mouth at the same level with soil surface. Pure propylene glycol was used as fixative, which is a commonly used nonvolatile, antimicrobial preservative and the safest alternative among the available chemicals as it is tasteless, odor-free and non-lethal upon drinking by wildlife (Thomas, 2008). 1/5 of the beakers were filled by the fixative.

Traps were placed under tree shades, repeated for 10 times per transect, with 10 meters of distance to each other and kept in the field for 14 days of duration.

Pitfall trap sampling was performed four times in three consecutive years from 2015 to 2017. In 2015 sampling was performed for two times, first set of trapping sampled mid. to late May, and the second set of sampling covered late May to early June period. 2016 and 2017 sampling occasions were equivalent to second set of samples from 2015 by dates. In summary, all sampling occasions were performed during late spring – early summer period (following Cardoso et al., 2007; Chatzaki et al. 2005; Elverici, 2012).

Table 2. 1. Sampling Methods and Their Periods

Method	Vacuum Sampling periods			Pitfall Trap Sampling Periods			
Sample	Herb.16	Herb.17	Tree17	2015–1st	2015–2nd	2016	2017
Start	14Apr16	3Apr17	3Apr17	9May15	23May15	24May16	26May17
End	18Apr16	9Apr17	9Apr17	23May15	5June15	6June16	8June17

Vacuum Sampling

Vacuum samplers have been suggested as the best method for standardizing sampling on the herbaceous vegetation (Stewart et al., 1995; Samu et al., 1997) and since then have been a popular method for sampling spiders. A gasoline powered suction device (Oleo-Mac BV 300 vacuum cleaner) was used, modified by placing a net (mesh size = 1 mm) at the mouth of the inflow pipe for collection and security of samples. Both understory herbaceous vegetation and tree or shrub canopy was sampled by using this method with slightly different protocols:

Herbaceous vegetation cover

Herbaceous vegetation did not always remain in the olive groves due to ‘weed control’ practices, but there have been small patches of herb. remaining under tree shades. Therefore, sampled microhabitat is chosen as the tree shades.

Vacuum sampler was hold keeping the mouth of its inflow pipe 10-50 cm above the soil surface, and an approximately $1m^2$ area of herb. cover was vacuumed under tree shade; this procedure was repeated for 5 consecutive trees around a quadrat to form a single subsample of a transect. In 2016, three subsamples (two quadrats + their mid-point) were collected per transect and amalgamated to form a sampling unit. In 2017, subsamples were restricted to quadrats, therefore each sampling unit was composed of two.

This sampling method was performed at day time, between 9:00 – 17:00. during mid-April in 2016 and early-April in 2017, when the herbaceous vegetation was established and ‘green’ and adult spider activity was known to be high in vegetation (Elverici, 2012).

Tree canopy

Height of olive trees were strictly controlled by farmers by regular pruning in the study area. Therefore, olive tree branches were approachable and easy to sample from the ground in all of the studied olive groves. In the beginning we were planning to sample tree canopy by flight interception traps, which could be effective (Simon and Linsenmair, 2001; Niedobova et al. 2015); but due to problems regarding fund raising, we converted to vacuum sampler.

Vacuum sampler was hold keeping the mouth of its inflow pipe directed to a branch, vacuuming the entire branch by regular movements; this procedure was repeated for 5 consecutive trees (a single branch per tree) around a quadrat to form a single subsample in a transect. Subsamples were restricted to quadrats, therefore each sampling unit was composed of two. Sampling was performed during early-April in 2017 (alongside with herb. vacuum sampling), before the blooming season of olive trees.

2.2.4. Quantifying Farming Practices and Environmental Variability

There wasn’t any report available on the olive grove management practices in the study area, therefore one of the original outputs of this thesis and the accompanied project has been planned as monitoring and documenting the extent of agricultural activities. However, the idea behind documenting agricultural practices is not limited to their discovery, but it also includes treating them as another set of variables besides the biodiversity & its composition, as well as to study their impacts or interaction between each other.

Monitoring have been performed by field studies and questionnaires since the start of the joint project TOVAG 213O147 in 2015. Field studies are covered below in detail, but questionnaires have been mentioned briefly, which can be found in the final report of the project in detail.

Following practices were identified or measured as detailed below:

- Pesticide use: Questionnaires indicated that pesticide use was very rare during the studied period. Our second approach was to analyze pesticide residuals from fruit samples. Field studies were performed to collect fruit samples of approximately 100gr from quadrats before the harvest (October) in 2016. 101 samples were collected from olive groves and natural biotopes (if olive fruit was available) from two to four trees around the quadrat. Residual analyses were performed by two independent laboratories: one from TÜBİTAK MAM Gebze Food Institude and the other in Erciyes University TEKMER Research and Development Center. Pesticides from a wide spectrum were targeted including Cyflutrin, Cypermethrin, Deltametrhrini Dimethoate and Lamda cyhalothrin.

- Herbicide spraying: Herbicides were used as an alternative to tillage or hay cutting for weed control and impacted the herbaceous vegetation dramatically. Spraying occurred in March or early April and easily identified in the study area by herb. vegetation structure. It was measured as percent sprayed area in quadrats.

- Soil Tillage: It was one of the most commonly held practices by farmers in the study area. It was mainly used for weed control, but reasons behind this were variable: for preventing water competition with olive; as a precaution against fire; for providing easier harvest (personal communication with farmers). Also, there were beliefs that it would increase water retention capacity of the soil. Soil tillage was performed in mid to late May, traditionally by horse power using ploughs, but usually by tractors. Tilled area was measured as a percentage in the quadrats.

- Hay cutting & Pasturage: Stock farming was rare in the study area and pasturage significantly impacted herb. cover in the olive groves on rare occasions. Mowing on the other hand is a common alternative to tillage or herbicide spraying; it was applied in autumn before harvest. Both of these activities were rare in the studied olive groves, recorded as present–absent due to difficulties related to quantifying and associating with biodiversity, as their impact on vegetation structure appear in summer or autumn (community samples collected in spring).

- Fertilizer use: Various types of animal manure or synthetic fertilizers were used in the groves. Type of fertilizer was monitored both in the field and by questionnaires, but it was a difficult variable to assess in the field especially for synthetic fertilizers due to difficulties related to observation.

Table 2. 2. Environmental Parameters Measured in the Olive Groves and Native Habitats

Type of Parameter	Parameters
Local Habitat Parameters	
	Altitude
	Slope
	Tree Circumference (cm)
	Tree Cover %
	Tree Height (m)
	Herb Height (cm)
Management Based Parameters	
	Herb Covered Area %
	Shrub Covered Area %
	Herbicide Sprayed Area %
	Tillage Applied Area %

Environmental variables were measured at 6 different periods during the study; late May 2015; late April 2016; late May 2016; November 2016, early April 2017; mid. August 2017. Datasets composed of various variables were produced for each sampling period, and each one of them corresponds to a certain community dataset for spiders.

Figure 2. 2. . Examples to alternative types of weed control practices and resulting habitat structure: A. no weed control; B. herbicide spraying; C,D. soil tillage

Alternatives by management types: organic & conventional; by weed control methods: herbicide spraying, tillage, hay cutting or quiet enjoyment (all contributing in variability on herb. covered area); by traits of olive trees or other vegetation: tree cover, tree circumference, herb. height and shrub cover; by topography: slope and altitude.

Table 2. 3. Community or environmental variables sampled in each period

Periods	**SpiderComm.**	**Herbicide**	**Tillage**	**H.Cover**	**S.Cover**	**T.Cover**	**T.Circ**	**H.Height**	**Slope**	**Altitude**
2015 May	Soil'15.2	X	X	✓	✓	✓	✓	✓	✓	✓
2016 April	Herb'16	✓	✓	✓	✓	✓	✓	✓	✓	✓
2016 May	Soil'16	✓	✓	✓	✓	✓	✓	✓	✓	✓
2016 Oct.	NA	✓	✓	✓	✓	✓	✓	✓	✓	✓
2017 April	Herb&Tree'17	✓	✓	✓	✓	✓	✓	✓	✓	✓
2017 May	Soil'17	✓	✓	✓	✓	✓	✓	X	✓	✓

2.2.5. Sorting, Preparation and Identification of Materials

Specimens collected by various methods in the field were transferred into leak proof plastic containers or bags with 70% ethanol addition and were carried to laboratory; then sorted and prepared as museum materials in 70% ethanol and glass vials with rubber lids. Specimens then examined under a Leica S8AP0 stereomicroscope by using fine tip forceps, petri dishes filled with 70 % ethanol and quartz sand for ease of positioning and better view. Digital images were taken when necessary by a Leica DFC280 digital camera equipped on the stereomicroscope. For a higher focal depth, 2-20 photographs were taken in different focal planes and combined using "Combine ZP-image stacking software".

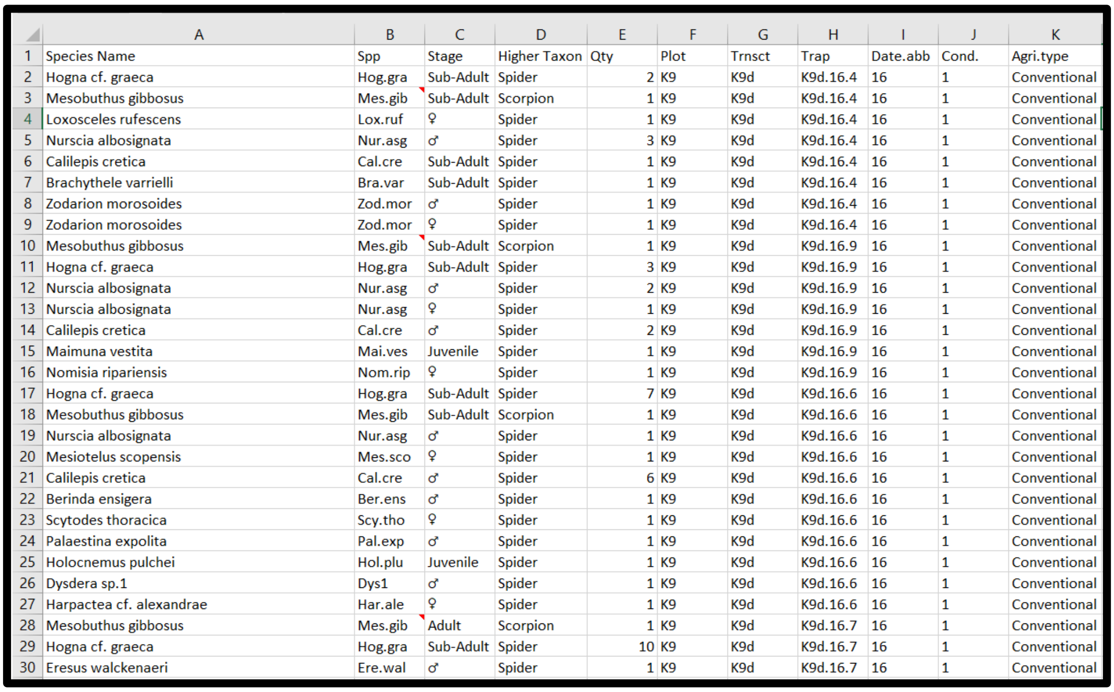
Adult specimens were identified based on morphology of copulation organs (which carry traits valuable as taxonomic characters) to the finest taxon level applicable (species) by using keys provided by Nentwig et al. (2018) or references from World Spider Catalog (2018). Final step for concluding about identity of a doubtful adult specimen was to consult a senior arachnologist. If the identification efforts were not enough or the specimens were immature, a para-taxonomic approach was used: morpho-types were designated for each unique type of habitus or copulation organ morphology combinations and denominated by using the name of the identified taxon and a number (using a generic or family name; e.g.: *Ozyptila* sp.1, Gnaphosidae sp. 1).

Special efforts were given for identification of immature specimens, as it has been shown that their inclusion could contribute to sample size (Sackett et al. 2008). Immature specimens were identified at the species level by using unique traits of somatic characters, which can be productive for certain taxa (traits other than copulation organs; e.g.: by abdomen colors and patterns, *Araneus diadematus*; by unique shape of abdomen, *Glyptogona sextuberculata*). Immature individuals were included in the datasets if they could possibly get associated to known taxa. If it was not possible, they were designated into a morpho-type based on short descriptions and photographs, but as “speculative” and discarded from the taxon based analyses in order to avoid duplicates, but planned to be used in higher taxa or functional group evaluations.

2.2.6. Enumeration and Formation of the Database

Taxonomic analyses, identifications and enumeration of the samples were performed simultaneously to form a database. Every single individual arachnid was recorded in a Microsoft Excel 2016 spreadsheet file in the biological recording format.

Every species or morpho-type was recorded in order with its name, abbreviated name, stage or gender identity, higher taxa identity (order), number of individuals per sample, site, transect and trap (or quadrat) names, abbreviated period of collection, sample condition for traps and microhabitat info for vegetation samples alongside with land use type in the site where the sample was collected. Names were abbreviated with the first three letters of generic and specific names or morpho-type numbers.



	A	B	C	D	E	F	G	H	I	J	K
1	Species Name	Spp	Stage	Higher Taxon	Qty	Plot	Trnsct	Trap	Date.abb	Cond.	Agri.type
2	Hogna cf. graeca	Hog.gra	Sub-Adult	Spider	2	K9	K9d	K9d.16.4	16	1	Conventional
3	Mesobuthus gibbosus	Mes.gib	Sub-Adult	Scorpion	1	K9	K9d	K9d.16.4	16	1	Conventional
4	Loxosceles rufescens	Lox.ruf	♀	Spider	1	K9	K9d	K9d.16.4	16	1	Conventional
5	Nurscia albosignata	Nur.asg	♂	Spider	3	K9	K9d	K9d.16.4	16	1	Conventional
6	Calilepis cretica	Cal.cre	Sub-Adult	Spider	1	K9	K9d	K9d.16.4	16	1	Conventional
7	Brachythele varrielli	Bra.var	Sub-Adult	Spider	1	K9	K9d	K9d.16.4	16	1	Conventional
8	Zodarion morosoides	Zod.mor	♂	Spider	1	K9	K9d	K9d.16.4	16	1	Conventional
9	Zodarion morosoides	Zod.mor	♀	Spider	1	K9	K9d	K9d.16.4	16	1	Conventional
10	Mesobuthus gibbosus	Mes.gib	Sub-Adult	Scorpion	1	K9	K9d	K9d.16.9	16	1	Conventional
11	Hogna cf. graeca	Hog.gra	Sub-Adult	Spider	3	K9	K9d	K9d.16.9	16	1	Conventional
12	Nurscia albosignata	Nur.asg	♂	Spider	2	K9	K9d	K9d.16.9	16	1	Conventional
13	Nurscia albosignata	Nur.asg	♀	Spider	1	K9	K9d	K9d.16.9	16	1	Conventional
14	Calilepis cretica	Cal.cre	♂	Spider	2	K9	K9d	K9d.16.9	16	1	Conventional
15	Maimuna vestita	Mai.ves	Juvenile	Spider	1	K9	K9d	K9d.16.9	16	1	Conventional
16	Nomisia ripariensis	Nom.rip	♀	Spider	1	K9	K9d	K9d.16.9	16	1	Conventional
17	Hogna cf. graeca	Hog.gra	Sub-Adult	Spider	7	K9	K9d	K9d.16.6	16	1	Conventional
18	Mesobuthus gibbosus	Mes.gib	Sub-Adult	Scorpion	1	K9	K9d	K9d.16.6	16	1	Conventional
19	Nurscia albosignata	Nur.asg	♂	Spider	1	K9	K9d	K9d.16.6	16	1	Conventional
20	Mesiotelus scopensis	Mes.sco	♀	Spider	1	K9	K9d	K9d.16.6	16	1	Conventional
21	Calilepis cretica	Cal.cre	♂	Spider	6	K9	K9d	K9d.16.6	16	1	Conventional
22	Berinda ensigera	Ber.ens	♂	Spider	1	K9	K9d	K9d.16.6	16	1	Conventional
23	Scytodes thoracica	Scy.tho	♀	Spider	1	K9	K9d	K9d.16.6	16	1	Conventional
24	Palaestina expolita	Pal.exp	♂	Spider	1	K9	K9d	K9d.16.6	16	1	Conventional
25	Holocnemus pulchei	Hol.plu	Juvenile	Spider	1	K9	K9d	K9d.16.6	16	1	Conventional
26	Dysdera sp.1	Dys1	♂	Spider	1	K9	K9d	K9d.16.6	16	1	Conventional
27	Harpactea cf. alexandrae	Har.ale	♀	Spider	1	K9	K9d	K9d.16.6	16	1	Conventional
28	Mesobuthus gibbosus	Mes.gib	Adult	Scorpion	1	K9	K9d	K9d.16.7	16	1	Conventional
29	Hogna cf. graeca	Hog.gra	Sub-Adult	Spider	10	K9	K9d	K9d.16.7	16	1	Conventional
30	Eresus walckenaeri	Ere.wal	♂	Spider	1	K9	K9d	K9d.16.7	16	1	Conventional

Figure 2. 3 Sample data in the database produced

Community matrices were produced to form ecologically meaningful datasets by using pivot tables and then extracted and maintained in .csv format, before being analyzed in R software. The database in currently in preparation for publishing, but can be available upon request.

2.3.Chorotype Classification of the Fauna

Known global distributions of the recorded species were extracted from World Spider Catalog (WSC) (2018). Nentwig et al. (2018) visualized this info on a map display which was used as a practical way for assessments. In case of endemic taxa with narrow distributions (East Mediterranean endemics), relevant publications were reviewed for chorotype assessments instead of WSC (2018), as the catalogue transform localities into political territories, which is usually inappropriate.

Corresponding chorotypes were assigned based on Vigna Taglianti et al. (1999), who described global chorotypes for Anatolia and West Palearctic terrestrial arthropod fauna based on their review on distribution patterns of numerous taxa. Assigned chorotypes were categorized under hierarchically inclusive ones. Introduced populations were not considered. Unidentified or questionable taxa were left out of assessment except new taxon candidates (assumed local endemics) or species complexes (by referring distribution range of the complex). Unidentified Linyphiidae and Therididae were not considered.

2.4.Statistical Approaches

Statistical part of this chapter is mostly about production of biodiversity statistics, therefore could be regarded as descriptive on this manner. Further data analyses were spared for the following two chapters, mainly analyzing statistics or community datasets produced or described here. R statistical software version 3.5.1. (R Development Core Team, 2018) was the main tool for calculations and plotting, with occasional use of Microsoft Excel for the arrangement of datasets and EstimateS (Colwell, 2016) for production of particular statistics.

Statistics regarding richness and diversity were calculated in multiple levels, following various methods for different purposes as described below.

a) Complete Inventory – All sampling units (transects) collected during the study were used in order to evaluate the complete spider inventory.
b) Microhabitat Inventory – Different microhabitats were expected to differ in the spider communities they possess, so spider inventories of microhabitats were evaluated separately as well.
c) Sampling Unit Inventory – Transects were designated as the fundamental sampling units in this study as described above, therefore diversify measurements in transects represent the main variables that will be used in the hypothesis tests in the following chapters. Both production of the variables and evaluation of their value as inventory was important.

2.4.1. Diversity

Measuring and reporting diversity can be tricky. Different approaches for its measurement often result in completely different variables in nature, therefore we used a variety of different indices and analyzed them separately. Primary measure of richness and diversity were Hill numbers (effective numbers of species for orders q: 1,2,3). Alternative standardization methods were used for the sake of comparability between sampling units, resulting in different indices. Reference samples that were created by sampling and enumeration were rarified or extrapolated to a common threshold before any comparison, resulting in the following statistics:

a) Species Density – an index giving the number of species per sampling unit, standardized by the number of replicated subsamples. Species density estimation was unique to pitfall trap sampling, in which 10 traps were deployed per transect, but some eventually lost due to vandalism or natural causes. Remaining subsamples (#traps) were extrapolated to 10 by sample based rarefaction or extrapolation following Colwell et al. (2012). Transects fallen below 5 subsamples or 20 individuals were discarded from calculations.
b) Computed or Observed Abundances – this statistic accompanies species density and calculated in sample based rarefaction or extrapolation to 10 subsamples, by the formula [t/T]*N where t is the number of sampling units accumulated, T is number of sampling units in the reference sample and N is the total number of individuals in all T samples (Gotelli and Colwell (2011). Calculations were unique to pitfall trapping, whereas the index was reported as observed abundance in vegetation samples.
c) Species Richness (Numerical) – calculated by individual based rarefaction for each transect and gives the number of species per 40 individuals, either by rarefaction or extrapolation.
d) Species Richness Estimator – Chao's abundance based estimator (Chao1) was used for richness estimation, both as a guide to extrapolation or as a statistic on its own.
e) Hill Number q=1 (Numerical) - calculated by individual based rarefaction for each transect and gives the number of common species per 40 individuals (exponential of Shannon), either by rarefaction or extrapolation.
f) Hill Number q=2 (Numerical) - calculated by individual based rarefaction for each transect and gives the number of abundant species per 40 individuals (exponential of Shannon), either by rarefaction or extrapolation.

2.4.2. Inventory Completeness

Inventory completeness was assessed by comparing rarefaction & extrapolation curves together with singleton and doubleton sampling curves 1) for transects; 2) for the entire dataset; 3) for microhabitats. Other approaches were to measure sample coverage as described in Chao & Jost (2012) or sample completeness as calculated by the ratio between observed and estimated richness values based on previous authors (Cardoso et al. 2008).

Sampling units (transects) were amalgamated under the hierarchically higher levels of sample classes for their assessment. Calculations and plotting were performed by using R basic commands and also by package INEXT (Hsieh et al. 2016).

2.4.3. Indirect Gradient Analyses

In this study, we use community composition and biodiversity as the fundamental variables, measure and document them in sampling units, then make comparisons for assessing the magnitude of differences between different categories, or perform analyses how these variables respond to environmental gradients as outcomes of anthropogenic land use. But what are these variables really?

Community Composition – This variable was analyzed in terms of pairwise dissimilarity between sampling units; dissimilarity measures (beta diversity) between all sampling units were gathered in a matrix and used to infer relatedness by relevant ordination procedures, we used Non-Metric multi-Dimensional Scaling (NMDS). Composition data is strongly left skewed due to very abundant or rare species, Hellinger transformation was used. Dissimilarity indices of choice were Bray-Curtis, Hellinger and Morisita-Horn dissimilarities. Indirect gradient analyses were accompanied by the analysis of similarity (ANOSIM) for testing the significance of groupings by making use of simulations and comparing within group and between group dissimilarities

2.4.4. Direct Gradient Analyses

Impacts of environmental variables were studied in direct gradient analyses. Principal Components Analyses (PCA) was used to study variability of environmental variables between sampling units. Redundancy Analyses (RDA) and Constrained Correspondence Analyses (CCA) models were used to study interference between environmental variables and spider community datasets. Environmental variables were non-normally distributed in general and transformed into parametric distributions by using relevant transformations methods (log, square-root, for continuous variables; logit for percent coverage measures). Direct gradient analyses were accompanied with ANOVA like permutation tests to assess significance of the constraints (environmental variables) as well as to assess significance of the models.

CHAPTER 3

RESULTS AND DISCUSSION

3.1. Discovered Spider Fauna

386 taxa or morpho-types were recorded by 27.432 individuals in total. 278 of these from 20.167 individuals were either represented with adults or had distinct characters for species level identification or morpho-type level designation. Among these 14 were represented by immature specimens only, whereas 79 were recorded both by adults or immatures. See Appendix 1 for the taxonomic list.

108 morpho-types from 7.265 specimens were based on immatures without any association to known taxa, or without strong evidence for morpho-type designation, therefore difficult to justify and excluded from lists or analyses provided below.

Table 3. 1 Species richness (abundance in parenthesis) and numbers of sampling units

	Total Richness	Sub-adult Taxa	Discarded Morphotypes	Sampling Units
Total	278 (20.167)	14 (5.920)	108 (7.265)	NA
Pitfall	256 (15.374)	5 (3.310)	96 (6.426)	210 (1.791)
Herb. Veg.	123 (3.717)	31 (1.905)	20 (593)	99 (248)
Canopy	44 (1.052)	28 (685)	7 (237)	49 (98)

Among the 278 taxa or morpho-types recorded, 264 species were represented by adult specimens. Among these, 46 could not be identified at the species level due to various problems. I have classified these problems as below:

a) Immatures: some species were represented only by immature species and identified at the generic level, it was not possible to decide about the specific identity due to availability of various alternative candidate taxa (e.g.: *Pistius* cf. *truncatus*; *Phlegra* cf. *lineata*)

b) Species Complexes: some species were identified at the generic level and associated to a species complex, but could not be identified at the species level due to lack of relevant revisions or material for comparisons between similar species for proper identifications (e.g.: *Pellenes* cf. *allegrii*; *Euophrys* cf. *gambosa*; *Zelotes* cf. *segrex*)

c) Inconclusive Literature: some taxa were described poorly and there aren't any voucher specimens available for comparing; therefore, it is currently impossible to decide about specific identity (e.g.: *Hogna* cf. *graeca*; *Theridion* cf. *cyprusense*);

d) New Taxon Candidates: Some species identified at the generic level may belong to possible new taxa, which must be re-assessed after further taxonomic evaluations (e.g. *Dysdera* sp.; *Dasumia* sp.; *Lathys* sp.; *Phrurolithus* cf. *thracia*)

e) Difficult Taxa: Some species could be identified at the family level if possible, without any further resolution. Most of these were from family Linyphiidae.

f) Lack of Specimens: Some species were represented with single specimens, sex or damaged, therefore difficult to identify (rare taxa, e.g. *Tegenaria* sp.).

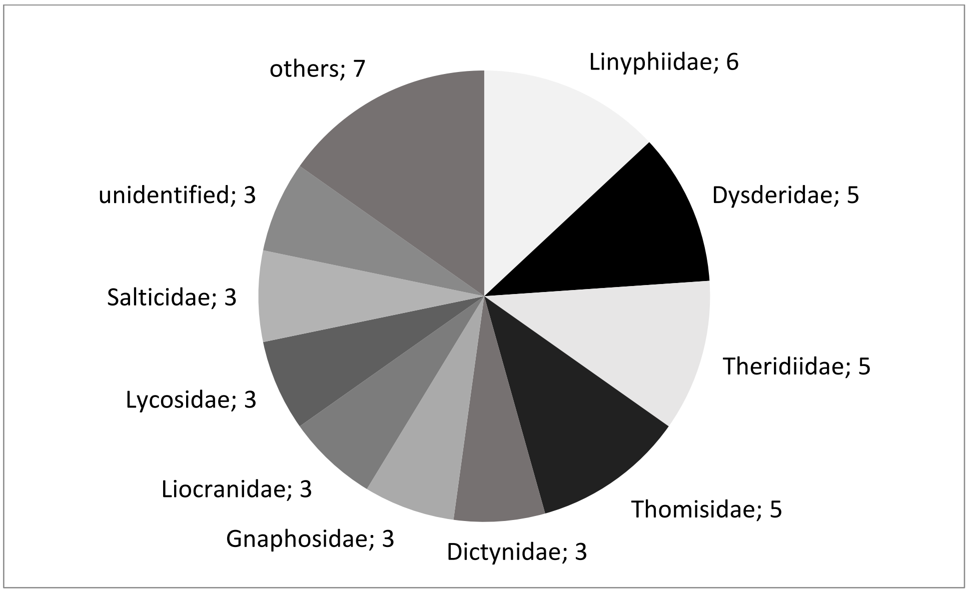


Figure 3. 1. Representation of unidentified species per spider family

Among the unidentified taxa or morpho-types, 25 were classified as candidate new taxa which needed further taxonomic research in order to make sure their species identity.

Apart from these, all species recorded from the region were new records for the study area, except one (*Zodarion bigaense*), while 42 species were recorded for the first time from the Turkish territory. New taxon candidates and Turkish new records represent 24% of the complete inventory. Discovery of such an apparently novel spider community with high amount of new records in an agricultural ecosystem under huge anthropogenic impact, can serve as an indicator of lack of knowledge that exist even for the most intensively used landscapes.

Table 3. 2. Spider families found and their species richness

Family	# species	Family	# species	Family	# species
Gnaphosidae	45	Oonopidae	2	Amaurobiidae	1
Theridiidae	35	Eutichuridae	2	Miturgidae	1
Linyphiidae	32	Filistatidae	2	Palpimanidae	1
Salticidae	31	Titanoecidae	2	Scytodidae	1
Thomisidae	22	Mimetidae	2	Zoropsidae	1
Araneidae	13	Sparassidae	2	Sicariidae	1
Philodromidae	11	Oecobiidae	2	Nemesiidae	1
Dysderidae	11	Clubiobidae	1		
Lycosidae	10	Uloboridae	1	Unidentified	3
Dictynidae	8	Hahniidae	1		
Liocranidae	7	Phrurolithidae	1		
Zodariidae	5	Eresidae	1		
Oxyopidae	5	Pisauridae	1	TOTAL FAMILIES	37
Agelenidae	5	Anyphaenidae	1		
Pholcidae	3	Ctenizidae	1		

Majority of the unidentified taxa were Linyphiidae, Theridiidae, Thomisidae or Dysderidae, which was not a surprise in a Mediterranean environment. Theridiidae and Linyphiidae were the most problematic families, with identifications limited at the family level. Three morpho-types could not be identified even at the family level. Finding such a high number of unidentified species indicated persistence of many taxonomic problems regarding spiders of the region.

37 spider families were identified among all spider specimens examined. Gnaphosidae was the most species rich family as usual for the Mediterranean region, followed by Theridiidae, Linyphiidae, Salticidae and Thomisidae. From these, 164 spider genera were identified.

By-catch arachnids other than spiders were also enumerated by means of taxonomy or para-taxonomy; 2 scorpion species among 482 specimens, 28 pseudoscorpion morpho-types among 196 specimens and 8 opilionid morpho-types among 10,259 specimens were represented in the collection.

The most speculative counts were for Acari with 50 morpho-types amon 3.502 specimens, which needs specialist examination obviously. Among these, only the scorpions were identified at the species level, which were *Mesobuthus gibbosus* (Brullé, 1832) and *Euscorpius idaeus* Yagmur & Tropea, 2017. These two species were included in the pitfall trap based community datasets.

Table 3. 3. List of unidentified species

Morpho-type Name	Family	Rationale
Tegenaria sp.1	Agelenidae	single damaged female
Cyrtocarenum cf. *cunicularium*	Ctenizidae	inconclusive literature
Archaeodictyna cf. *consecuta*	Dictynidae	inconclusive literature
Dictynidae sp	Dictynidae	single female
Lathys sp.1	Dictynidae	new taxon candidate
Dasumia sp.	Dysderidae	new taxon candidate
Dysdera cf. *lata*	Dysderidae	inconclusive literature
Dysdera sp.1	Dysderidae	new taxon candidate
Dysdera sp.2	Dysderidae	new taxon candidate
Harpactea sp.1	Dysderidae	new taxon candidate
Drassodes cf. *lapidosus*	Gnaphosidae	species complex
Phaeocedus cf. *hebraeus*	Gnaphosidae	inconclusive literature
Zelotes cf. *segrex*	Gnaphosidae	species complex
Erigonolophus sp1	Linyphiidae	new taxon candidate
Linyphiidae sp. 1	Linyphiidae	difficult taxon
Linyphiidae sp. 2	Linyphiidae	difficult taxon
Linyphiidae sp. 3	Linyphiidae	difficult taxon
Linyphiidae sp. 4	Linyphiidae	difficult taxon
Palliduphantes cf. *byzanthinus*	Linyphiidae	inconclusive literature
Liocranidae sp.1	Liocranidae	new taxon candidate
Liocranidae sp.2	Liocranidae	new taxon candidate
Liocranidae sp.3	Liocranidae	new taxon candidate
Hogna cf.*graeca*	Lycosidae	inconclusive literature
Pardosa cf. *roscai*	Lycosidae	inconclusive literature
Trabea cf. *paradoxa*	Lycosidae	inconclusive literature
Zora cf. *prespaensis*	Miturgidae	inconclusive literature
Orchestina sp.1	Oonopidae	single female
Philodromus cf. *lunatus*	Philodromidae	species complex
Pholcidae sp.1	Pholcidae	new taxon candidate
Phrurolithus cf. *thracia*	Phrurolithidae	new taxon candidate
Euophrys cf. *gambosa*	Salticidae	species complex
Pellenes cf. *allegrii*	Salticidae	species complex
Phlegra cf. *lineata*	Salticidae	immature specimen
Theridion cf. *cyprusense*	Theridiidae	inconclusive literature
Theridiidae sp.1	Theridiidae	new taxon candidate
Theridiidae sp.2	Theridiidae	new taxon candidate
Theridiidae sp.3	Theridiidae	new taxon candidate
Theridiidae sp.4	Theridiidae	new taxon candidate
Ozyptila sp.1	Thomisidae	new taxon candidate
Ozyptila sp.2	Thomisidae	new taxon candidate
Ozyptila sp.3	Thomisidae	new taxon candidate
Pistius cf. *truncates*	Thomisidae	immature specimen
Xysticus sp.1	Thomisidae	single female
Unidentified sp.1	Unidentified	new taxon candidate
Unidentified sp.2	Unidentified	new taxon candidate
Unidentified sp.3	Unidentified	new taxon candidate

3.2. Inventory Completeness and Biodiversity

The complete inventory covering all studied transects appeared as one of the most comprehensive ever produced, with more than 360 sampling units, 20,000 individuals and 278 species included. Sample coverage and completeness estimations indicated high values of representation of the actual spider community in the samples, with estimates of 99% and 83% respectively. Rarefaction curve also confirmed this finding with low final slope. Representation of the singletons was one of the lowest ever reported with 16%. Singletons and doubletons curves were quite stable and continued their existence in the samples however, with a slight drop in the doubletons curve and departure from singletons in the rarefaction curves. This was due to consistent appearance of new rare species in the samples. This can be explained by the insufficiency of sampling to detect rare species due to high detection limits, but it is difficult to judge sampling strategy, as the main cause of such a signal it is difficult to track.

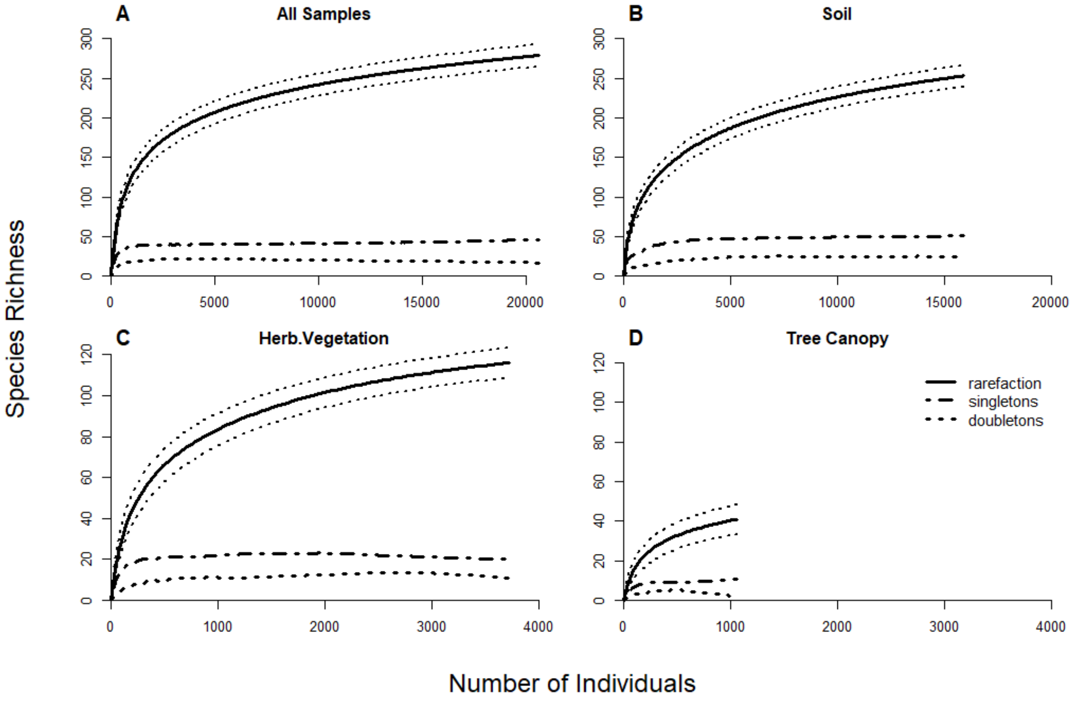


Figure 3. 2. Rarefaction, singleton and doubleton curves for the given samples

Rarity is a natural phenomenon in megadiverse taxa and also in sampling attempts covering spatial scales wider than certain limits, with inevitable persistence of rare species (Chao et al. 2009; Longino et al. 2002). Other potential drivers of rarity persistence might be the presence of continuous disturbance in the study plots, in the form of agricultural management in olive groves, and other types of land use (e.g.:domestic grazing) or consequences of biotope fragmentation resulting in ecological drift (Vellend, 2010).

Microhabitat inventories similarly resulted in high sample coverage which was measured higher than 98% for all classes. Completeness measures slightly differed however, with 88% for herbaceous vegetation spiders inventory, 83% for soil spider inventory and 69% for tree canopy inventory, which were all at the higher side of the reported values in the literature. Soil was the most intensively sampled microhabitat with more than 15,000 individuals, and its main difference from the complete inventory appeared as the representation of the rare groups, singletons and doubletons. Proportion of singletons was higher than the complete inventory with 20%. Main reason of such a difference might be the contamination of pitfall traps from microhabitats other than soil, which result in species apparently rare in the traps, but actually abundant in other microhabitats. Regardless of this, behavior of the rarefaction curves did not differ significantly from those of complete inventory. Herb. vegetation inventory was similarly exhaustive, with singletons and doubletons curves approaching to each other, but not crossing, with 17% representation of singletons. Tree canopy was the least exhaustively sampled microhabitat, with singletons and doubletons curves departing from each other and highest proportion of singletons measured with 27%. Both numbers of singletons and doubletons were lowest however, with eleven and two species respectively. These results might be explained by the "not so exhaustive" sampling of the tree canopy, which received the least amount of effort both in terms of number of samples and sampling periods.

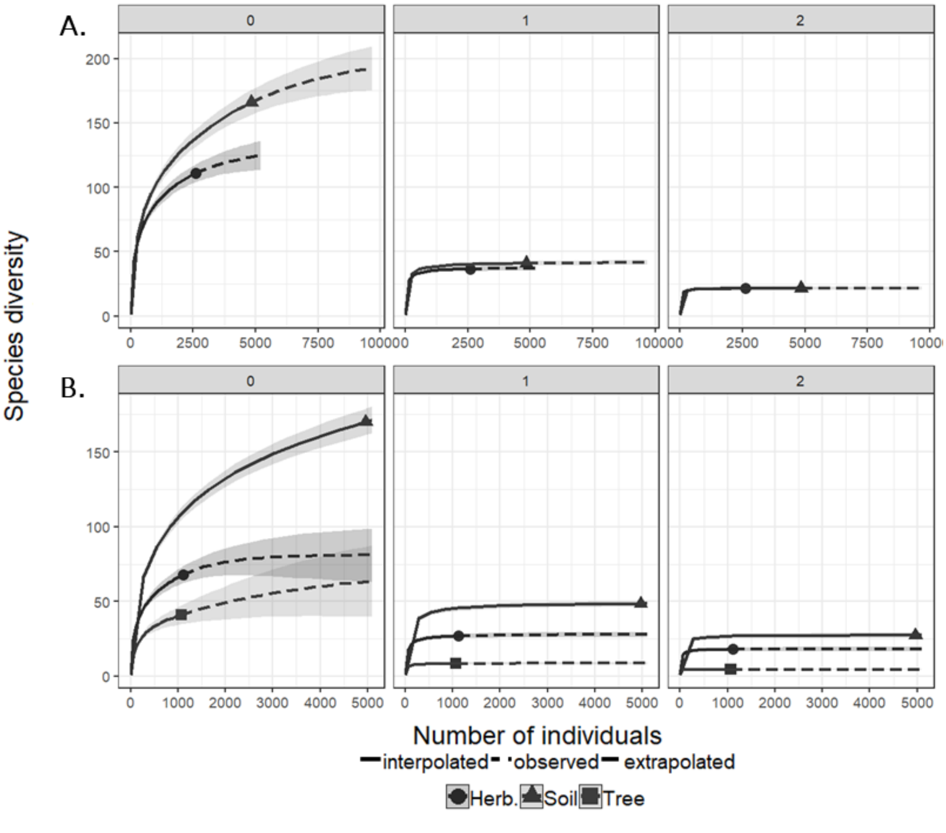


Figure 3. 3. Microhabitat rarefaction curves for richness (q=0) and diversity (q=1,q=2): A. 2016 period, herbaceous vegetation and soil microhabitat spider communities. B. 2017 period, herb. vegetation, soil and tree canopy microhabitats.

Biodiversity assessments and comparison of microhabitats per each sampling period (year) was also important to understand the extend of variability between microhabitats and periods, by excluding the effect of sampling intensity, which was apparent between studied microhabitats and acts as a source of variation. Alternative microhabitats were sampled only in 2016 and 2017, with two sampled in 2016 and all three sampled in 2017.

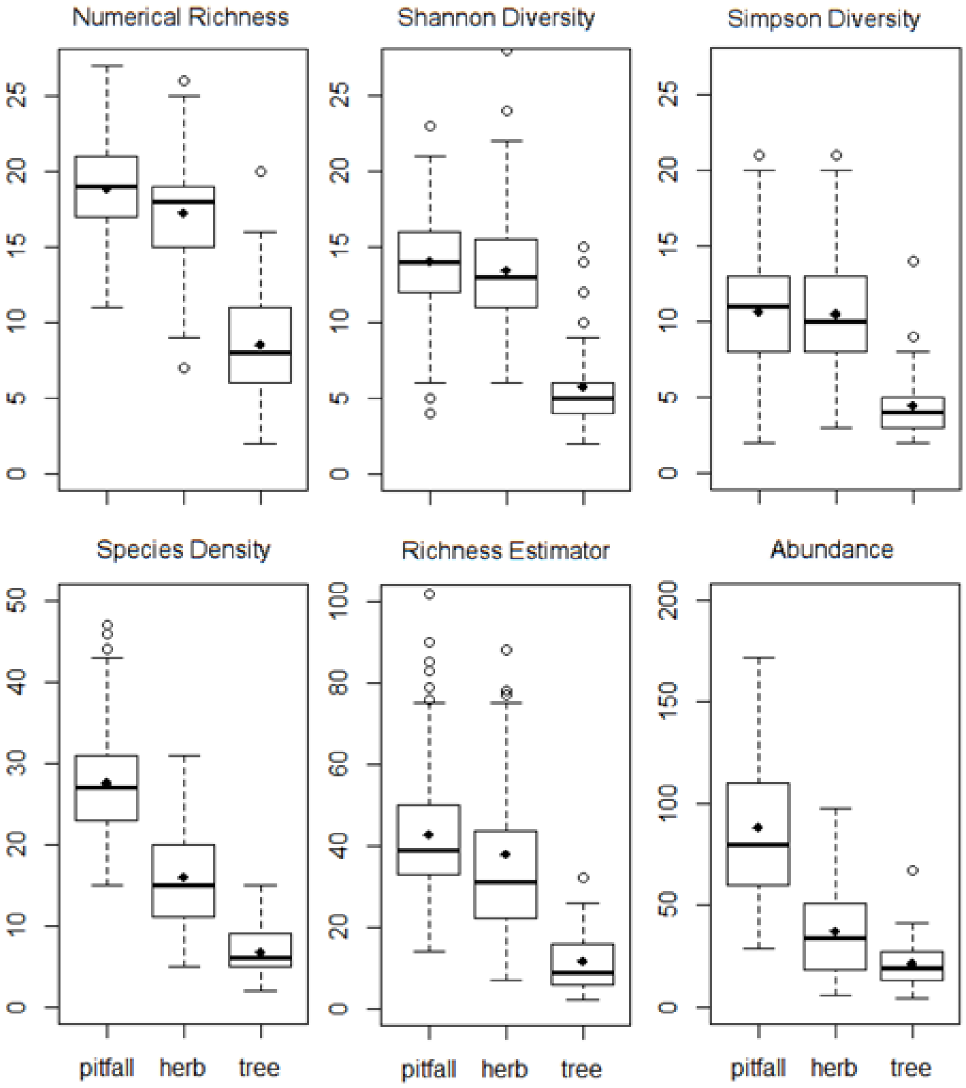


Figure 3. 4. Transect based diversity statistic summaries

In 2016, there was a two-fold difference between soil and herb. inventories on the number of individuals included, however, it was possible to deal with such a difference by rarefaction and extrapolation. Both these methods indicated a richer inventory on the soil surface compared to herb. vegetation, but differences on diversity measures were not significant.

In 2017 however, both richness and diversity was much higher in soil spider inventory, and herb. vegetation inventory was significantly richer and more diverse than the tree canopy. Difference between soil and herb. was indistinct in 2016 but became more pronounced in 2017, probably as an effect of sampling periods, which differ almost two weeks for herb. vegetation sampling between two years. Soil inventories were similarly rich and diverse between two years, but for herb. vegetation, both richness and diversity differed, with a significantly rich and diverse inventory in 2016, compared to 2017.

3.3. Transect Based Diversity Statistics

Spiders were significantly more abundant in pitfall trap samples compared to vegetation, but transect counts were satisfactory in terms of number of individuals (abundance) for both pitfall trap and herb. vegetation vacuum sampling, as almost all of the transects were resulted in observed abundances of more than 20, which has been recommended as a threshold for biodiversity evaluations (Gotelli and Colwell, 2011). A significant portion of tree canopy vacuum samples (49%) fall below this threshold however. Scarcity of spiders in tree canopy samples was perceived in all of the statistics.

Differences between pitfall trap and herb. vegetation samples were obvious in abundance and species density, whereas rest of the statistics did not differ significantly in between, indicating presence of equally rich and diverse communities in the soil and herb. vegetation microhabitats at the transect or local scale. This is contrasting with the diversity comparisons between two microhabitats after amalgamation of transects at the microhabitat level (regional scale), which indicated one and a half to three times richer, and up to two times more diverse communities for soil microhabitat. This was due to higher species turn over in the soil microhabitat, which can be explained by narrower habitat choice and limited dispersal abilities of soil dwelling spiders.

10 transects were lost from alpha diversity analyses due to dropping less than 5 traps due to trap loss (K1c.15.2, K3b.15.2, K4c.15.1, K5C.16, K7d.15.2, K9b.15.1, K9b.15.2, O5a.16, O5b.16, O5c.15.1)

3.4. Spider Community Datasets Produced and Analyzed

Ecologically meaningful datasets were extracted from the main database in order to analyze the community composition descriptively or prior to analyses. Following datasets were the most pronounced and meaningful ones, described, analyzed and discussed in the following pages. These datasets and their purposes are briefly described.

3.4.1. Soil Surface Spider Community Datasets

a) **Datasets of Each Sampling Period** – There are four different datasets representing four different sampling periods, 2015.1; 2015.2; 2016 and 2017. Last three of these were accompanied with environmental measurements, which were elaborately quantified for each sampling unit at each sampling period (see chapter4 for details).

b) **Periodic Replicates of Soil Surface Spider Communities**– Spider communities of sampling units with replications at four different periods were used to compare community composition and biodiversity parameters between different periods, in order to assess the magnitude of species turnover and diversity dynamics within season or between years. There are 30 sampling units with replicated measurements from four different periods: 1) 2015 mid.-late May; 2) 2015 early June; 3) 2016 early June; 4) 2017 early June. This dataset includes sampling units from both olive groves and natural biotopes.

c) **Biotopes Dataset** – Spider communities of sampling units representing different biotopes available in the study area: olive groves with or without surface water, maquis, mixed woodlands of low or high canopy cover and pine forests. Sampling units from early June period were used in this dataset, primarily from 2016 period; units from other periods (2015.2 or 2017) were included if they represent biotopes or sites absent in the 2016 samples. Olive groves with intensive management were discarded from this dataset in order to concentrate solely on biotope effect on the communities.

d) **Management Dataset** – Spider communities of sampling units representing different olive farming practices or habitat features. Alternatives by management types: organic & conventional; by weed control methods: herbicide spraying, tillage, hay cutting or quiet enjoyment (all contributing in variability on herb. covered area); by traits of olive trees or other vegetation: tree cover, tree circumference, herb. height and shrub cover; by topography: slope and altitude. Community dataset is accompanied with an environmental dataset, which was elaborately quantified for each sampling unit at each sampling period. Biotopes other than olive groves were discarded from this dataset.

3.4.2. Vegetation Spider Community Datasets

e) **Herbaceous Vegetation Datasets –** Spider communities sampled by vacuum sampling of understory herb. vegetation in olive groves or natural biotopes. There are two different datasets from two periods: 1) 2016 late April; 2) 2017 early April. These datasets are accompanied with environmental measurements, which were elaborately quantified for each sampling unit at each sampling period.

f) **Tree Canopy Dataset -** Spider communities sampled by vacuum sampling of tree or shrub canopies in olive groves or natural biotopes. Sampling was performed for once in 2017 early April. This dataset is accompanied with environmental measurements, which were elaborately quantified for each sampling unit at each sampling period.

3.4.3. Mixed Assessments

g) The Complete Dataset for Assessment of Regional Species Pool – All specimens and taxa (or morpho-types). This dataset was used for assessing species richness in the entire dataset.

h) Microhabitat Community Compositions Dataset – Spider communities sampled by various methods from all biotopes or microhabitats from 2017 period were involved in this dataset, as 2017 is the only period that all microhabitats (canopy, understory veg. and soil) were sampled. Aim of this dataset is to compare biodiversity statistics and community composition between different microhabitats or biotopes.

Table 3. 4. Chorotypes, number of species represented and their categories

Chorotype	Letter Code	N	Category
Sub-Cosmopolitan	SCO	14	Cosmopolitan
Asiatic-European	ASE	30	Palearctic
CentralAsiatic-European-Mediterranean	CEM	26	CentralAsiatic-European-Mediterranean
CentralAsiatic-European	CAE	10	
CentralAsiatic-Mediterranean	CAM	12	
Turano-European-Mediterranean	TEM	6	Turano-European-Mediterranean
Turano-European	TUE	8	
Turano-Mediterranean	TUM	1	
Turanian	TUR	3	
SW-Asiatic	SWA	3	
European	EUR	8	European
S-European	SEU	4	
E-European	EEU	2	
European-Mediterranean	EME	11	
Mediterranean	MED	34	Mediterranean
W-Mediterranean	WME	1	
E-Mediterranean	EME	43	
Peloponnesian Endemic	PELO	21	Endemics
Macedonian Endemic	MACE	3	
Anatolian & Levantian Endemic	ANAT	1	
West Anatolian Endemic	ANAW	4	
North West Anatolian Endemic	ANNW	7	

3.5. Chorotype Classification

252 taxa were assessed for chorotype classifications and assigned into 22 chorotypes from 7 categories (see table). Classification resulted in dominance of the

Mediterranean chorotype categories, whereas rest of the categories were composed of widely distributed taxa mainly from Palearctic or Central Asia. Chorotypes widespread in the Europe were represented with surprisingly low number of species.

Representation of the cosmopolitan species was low among the studied taxa, even the "cosmopolitan" chorotype which was found in the original descriptions of Taglianti et al. (1996) was absent in our study, excluding the introduced populations. (introduced populations were not considered). Sub-cosmopolitan species were also rare and their distribution ranges might not be significantly different from those of represented in the Palearctic category.

East Mediterranean was the most species rich among the represented chorotypes, followed by the Mediterranean-widespread chorotype. Narrowly distributed endemics also had a high representation in the Mediterranean category, mainly composed of Peloponnesian endemics, but North West Anatolian or West Anatolian endemicity were also common among the studied species.

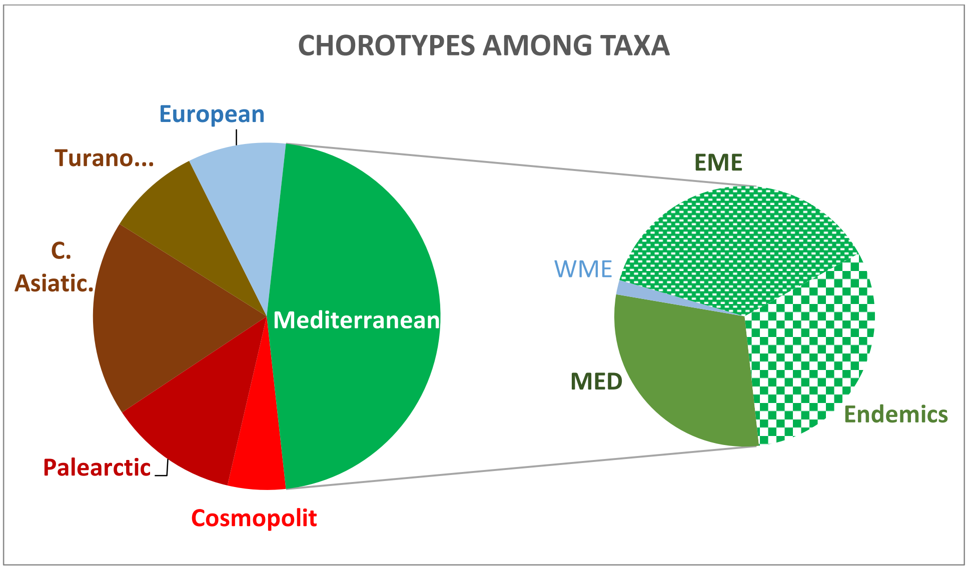


Figure 3. 5. Representation of chorotypes identified in this study

Rest of the chorotypes or categories were represented with few number of taxa. A further classification of the most widely represented (dominant) chorotypes identified three distributional groups: the east Mediterranean endemics; widely distributed species within the Mediterranean; and widely distributed species in the Palearctic.

Evaluations regarding chorotype compositions have been quite common in spider inventory reports from Europe as stated before. It was surprising to find out about the dominance by the Mediterranean chorotype categories, which have not been replicated in other inventories from the Mediterranean or Europe. Vrenozi (2012) reported chorotype assessments for 82 species from Albanian coast, without including any Mediterranean or endemic taxa. Komnenov (2013) and Deltshev et al (2013) reported diverse spider inventories from Macedonia (Osogova and Galichitsa Mountain ranges respectively), both with high representation of endemics, but chorotypes dominated with European or widely distributed taxa. Schröder et al. (2010) and Komnenov et al. (2016) reported chorotype assessments from Greece (Aladjagiola wetland complex and Dadia Forest, respectively) which were also dominated by European or widespread chorotypes as well.

Dominance of the Mediterranean chorotypes in our study might be due to study site features, sampling periods or sampling methodology. Our study area was delimited in an agro-ecosystem – namely in the olive grove belt – which only covered altitudes below 300 m, and natural biotopes included were severely fragmented and surrounded by olive groves. Such consequences indicate huge alternation on the native ecosystem in the study area, but in the end resulting in a typically Mediterranean type landscape. This might be an explanation for the dominance of Mediterranean-widespread taxa. Agro-biont species might have strong representation in this group of spiders, due to old agricultural history in the Mediterranean. It is also important to note that, the inventory evaluated in this study is actually biased by the sampling periods and sampling methodology, which must be considered as important determinants on chorotype compositions, but their effect seems to be more difficult to explain.

Important Note: In our analyses of chorotype identifications, we relied on chorotype identities described by Vigna Taglianti et al. (1993, 1999) and known (reported) distributions of taxa without any further spatial analyses. The reader must be aware of that there is a high risk of subjectivity bound with such data.

3.6. Indirect Gradient Analyses

We tried to summarize the results of beta diversity analyses (pairwise dissimilarity) in NMDS ordination and plotted the results of the models in two dimensional space. In these graphs plotting the NMDS models, each point represents a sampling unit, and distance between every two points is a correlate of the actual dissimilarity measure (the beta diversity, in terms of differentiation) between two sampling units. Therefore, it is easier to visualize the dissimilarity between sampling units.

Grouping variables were used to categorize different treatments or groups of biotopes when necessary. Labels of these variables were provided in legends or marked at the centroids of the groups in the ordination plots. In other cases, ellipses were used to indicate standard deviation of the centroids or entire cluster of a group.

Dissimilarity calculated between spider communities by various indices did not differ from each other significantly, therefore we have used the Hellinger dissimilarity in the graphs, in order to have a standard. It has been indicated on the graph if a different measure was used.

3.6.1. Comparing Spider Community Composition Between Periods

Compositional differences between periods were studied between periodic replicates of soil surface and herbaceous vegetation samples, separately for natural and olive grove sampling units.

Soil surface samples were significantly different between mid-May and late May samples in natural sampling units. Different years clustered together with the corresponding periods however. These findings indicate a high dynamism due to species turnover in spider communities within a single season, which has never been shown before. At the same time, samples collected at similar dates between years reflect such similarity in community composition too, indicating the resilient nature of the soil surface spider communities.

The difference was still significant between olive grove periodic replicates too, but the overlap between different periods was much higher compared to natural sampling units. This might be due to higher variability in olive groves compared to other biotopes due to higher amount of management, which diminish the effect of periodicity. Such an effect of diminishing might be due to replacement of stenochronous specialist species by generalists, presumably which are also agrobionts, in olive groves.

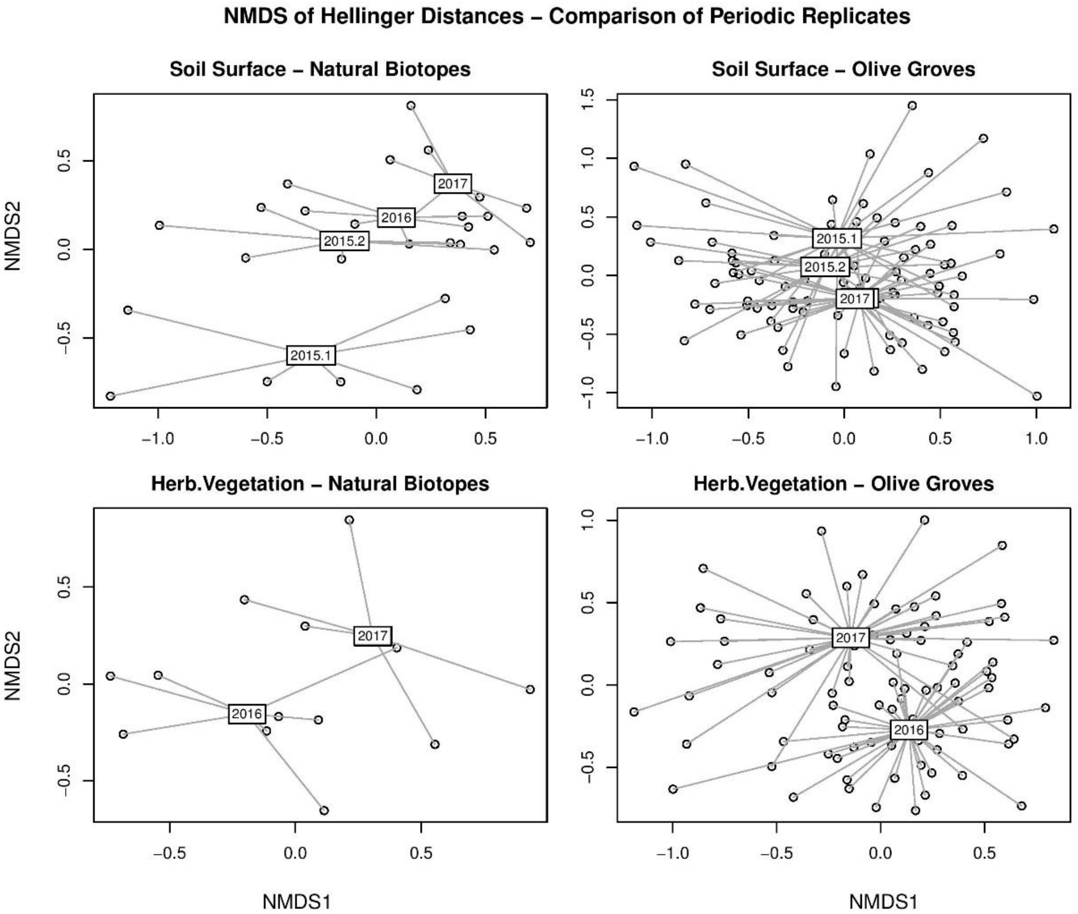


Figure 3. 6. NMDS of Hellinger distances for comparison of community composition between periodic replicates in soil surface and herbaceous vegetation spider communities

Nature of dissimilarity between herbaceous vegetation samples was similar to what we have observed in soil surface communities, with slightly different communities from different periods. However, this time periodic replicates differ in both sampling year and sampling period (early mid-April and early April in 2016 and 2017 respectively). Therefore, it has been more difficult to justify the source of variation in the herb. vegetation samples. By assuming that rate of turnover is effected from within year- within season differences, similar to what we observed in soil surface communities, then it is possible to speculate that within season variability has the larger share in this variability. Between year overlap was less than observed in oil surface samples, which can also be explained by assuming a stronger effect of within year variability.

3.6.2. Comparing Microhabitat Spider Communities

The most apparent differences ever measured in this study were for those of the microhabitat comparisons. Microhabitats differed significantly in both 2016 and 2017 sampling periods without any overlap between groups. Soil surface and vegetation spider communities were the most distinct; whereas herbaceous vegetation and tree canopy spider communities were also significantly different from each other.

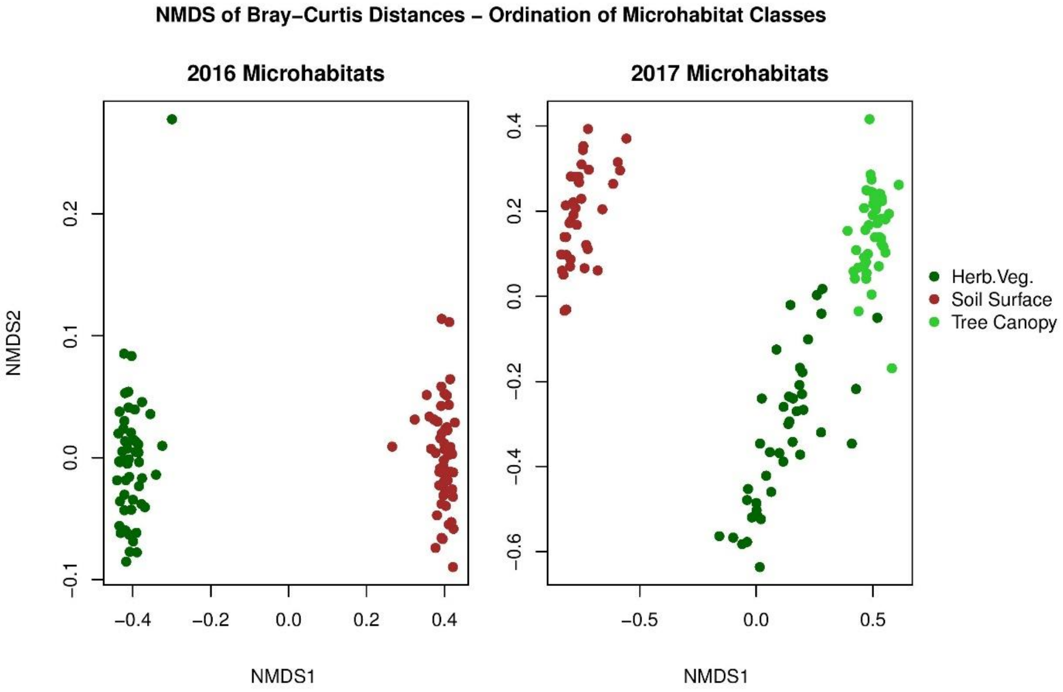


Figure 3. 7. NMDS of Bray-Curtis distances for comparison of community composition between micro-habitats in 2016 and 2017

Variability between sampling units were higher in soil surface and herbaceous vegetation spider communities, compared to tree canopy, which indicates higher higher species turnover in the former mentioned microhabitats.

3.6.3. Comparing Spider Communities Between Different Biotopes

Different biotopes were examined in higher detail for soil surface spider communities, with higher number of replicates from different biotopes, compared to vegetation communities. Biotopes included in our study differ from each other in terms of habitat structure, which can have direct effects on ecosystem processes such as primary productivity (e.g. canopy closeness). Olive grove management is another source of variation, which directly effects habitat structure (e.g. weed control by tillage or herbicide use, etc.). We wanted to study these different sources of variation between habitats, namely the natural and anthropogenic sources respectively. Therefore, we studied community composition first by excluding the olive groves with weed control practices, which leaves the natural biotope sampling units, as well as olive groves with no management at the studied periods.

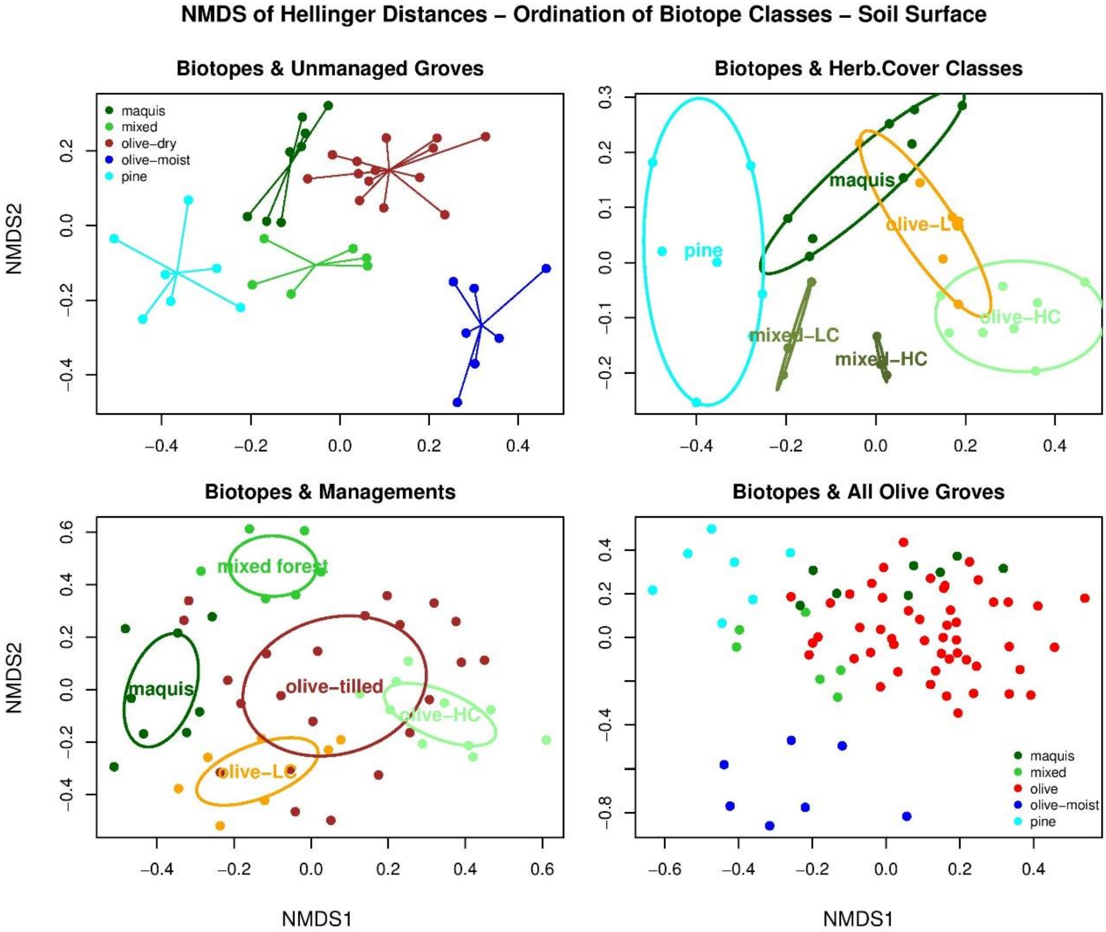
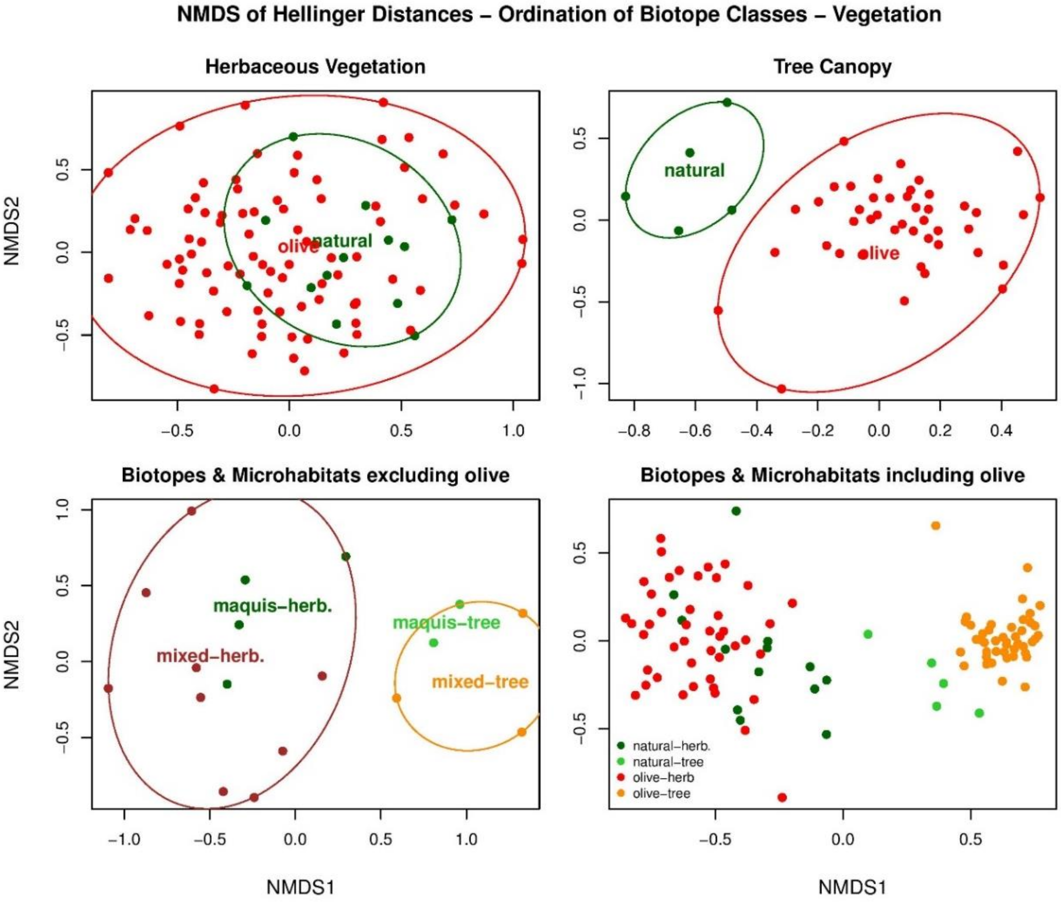


Figure 3. 8. NMDS of Hellinger distances to study soil surface spider community differences between different types of biotopes

Comparisons between biotope spider communities yielded significant differences between different groups for soil surface. Moist olive groves with temporary surface water had the most distinct type of communities, whereas maquis biotope remained as the most similar one with the olive groves. This was actually expected, since olive tree is a native component of the maquis, but still surprising as maquis and olive groves have different structures. Pine forest soil surface spider communities were also easily distinguished from the other biotopes, whereas mixed forests constituted transition type of communities between pine forests and olive groves.

This feature was further examined by studying herbaceous plant covered area, between sampling units. Olive groves with high herb. coverage were more similar to mixed forests with high herb. cover; whereas mixed forests with low herb. coverage were more similar to pine forests, which typically lack herbaceous plant cover at the understory. On the other hand, olive groves with naturally low plant cover had similar spider communities with those of the maquis, which also have naturally low herbaceous plant cover. Therefore, it is possible to speculate that, differences in spider communities between different biotopes might be a reflection of a gradient of herbaceous plant cover.

Including the olive groves with weed control practices and excluding the most marginal biotopes, pine forests and moist olive groves, laid emphasis on the differences on soil surface spider communities between olive groves and the natural biotopes. Tillage had an obvious impact on the soil surface spider communities, marked by a huge rate of species turnover between sampling units. Herbicide use had a similar effect on the spider communities, but discarded from the NMDS plots, in order to keep plots more readable.


NMDS of Hellinger Distances – Ordination of Biotope Classes – Vegetation

Herbaceous Vegetation

Tree Canopy

Biotopes & Microhabitats excluding olive

Biotopes & Microhabitats including olive

Figure 3. 9. NMDS of Hellinger distances to study herbaceous vegetation and tree canopy spider community differences between different types of biotopes

Herbaceous vegetation and tree canopy spider communities were studied in maquis and mixed forest biotopes, other than olive groves. Spider communities did not differ between olive groves and natural biotopes in herb. vegetation microhabitat; whereas they differ in tree canopy. This might be related to microhabitat structure, as herbaceous plant microhabitat did not differ between biotopes in terms of structure; whereas, a dramatic change in structure is expected for tree canopies, as the tree species differ between biotopes. The two microhabitats differed from each other by spider communities, regardless of the type of biotope.

3.6.4. Comparing Spider Communities Between Management Types

Management types were studied in olive groves, all microhabitats and in two different scales: the main management types (organic vs conventional) and environmental parameters that result from management activities (e.g. weed control practices).

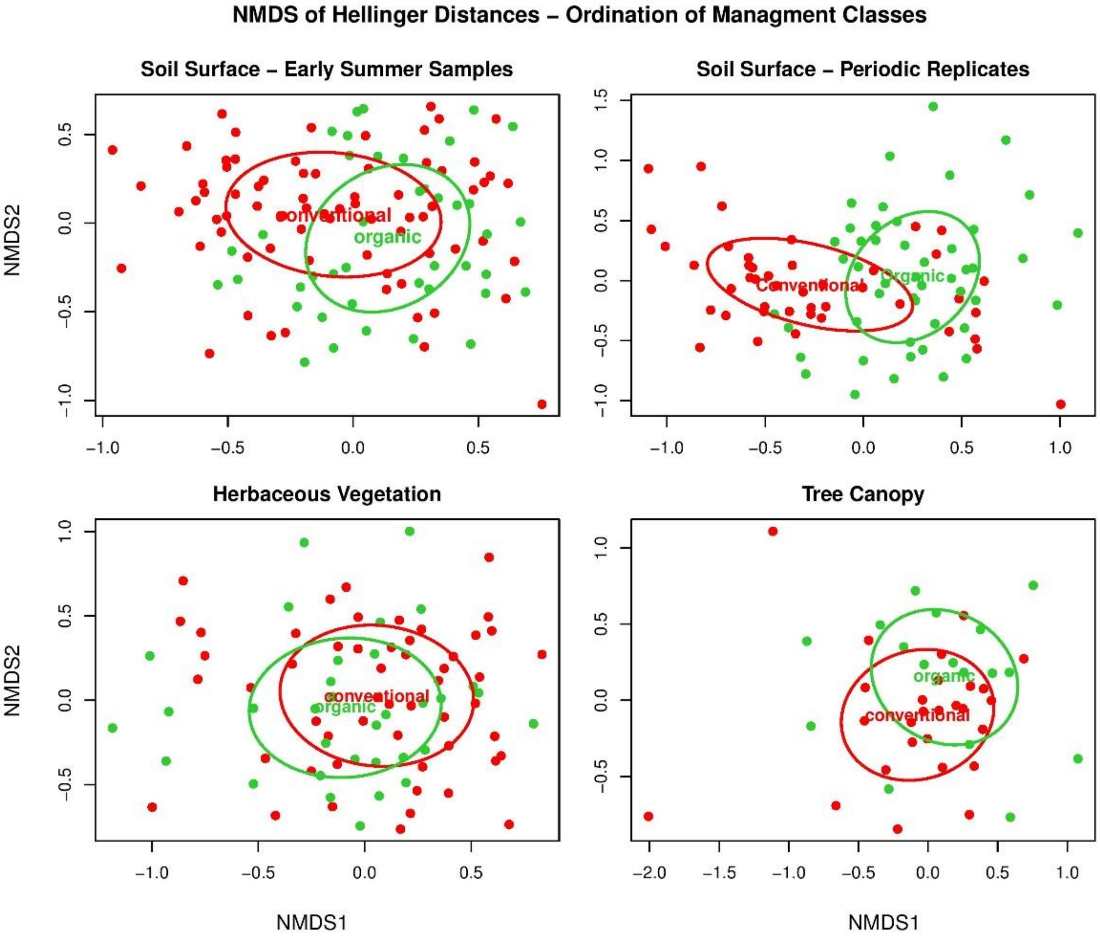


Figure 3. 10. NMDS of Hellinger distances between main management types

Olive groves under organic or conventional management could not be identified from each other by spider community composition in neither of the microhabitats in general. Only one of the evaluations yielded a siginificant difference, which was for the soil surface periodic replicates. However, this dataset was also characterized by huge overlap between two groups, therefore it was difficult to justify the difference between management types. This difference might be explained by chance as well, as the inclusion of sites or sampling units into periodic replicates was not an entirely random process. Analysis of the management dataset on the main management types yielded a similar result, with significant difference between organic and conventional management, but a huge amount of overlap.

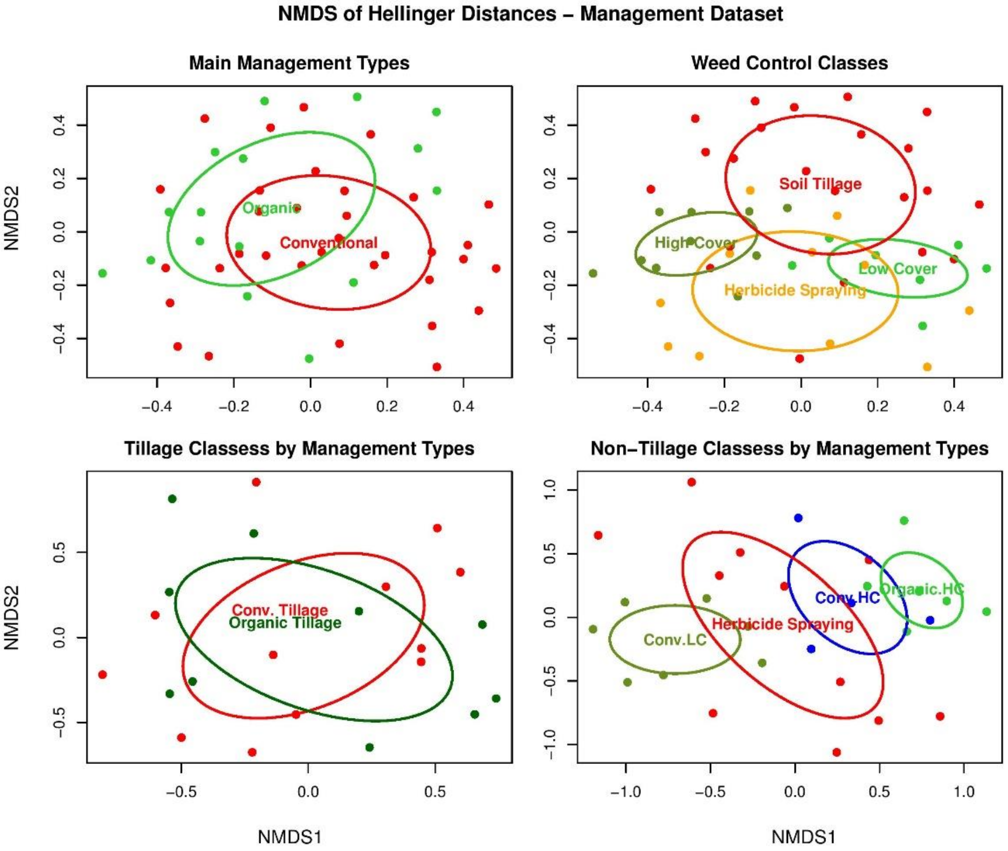


Figure 3. 11. NMDS of Hellinger distances between weed control & management types

Concentrating on the weed control practice classes resulted in significantly different clusters in the NMDS ordination, with unmanaged olive groves of high or low herb. cover classes differ from tilled and herbicide sprayed olive groves. Both tillage and herbicide spraying resulted in higher species turnover between sampling units. This might be explained by intermediate disturbance hypothesis, with weed control practices acting as a disturbance, and such disturbance increases the possibilities for species sorting.

Soil tillage was an abundant weed control practice in both organic and conventional management types, whereas no interaction effect could be observed on soil spider community composition. Non- tillage classes on the other hand, display signals of main management – weed control practice interaction effecting spider community composition, with differences between organic and conventional managed olive groves. Conventional un-management olive grove spider communities had higher overlap with herbicide sprayed olive groves, compared to organic unmanaged olive groves. This difference might be a reflection of the recent past of the conventional groves, if herbicide spraying was performed in these groves in the past. If this was the case, then it is possible to speculate that explored spider communities have been descended from the past communities from herbicide sprayed olive groves, and this explains the current overlap between these groups.

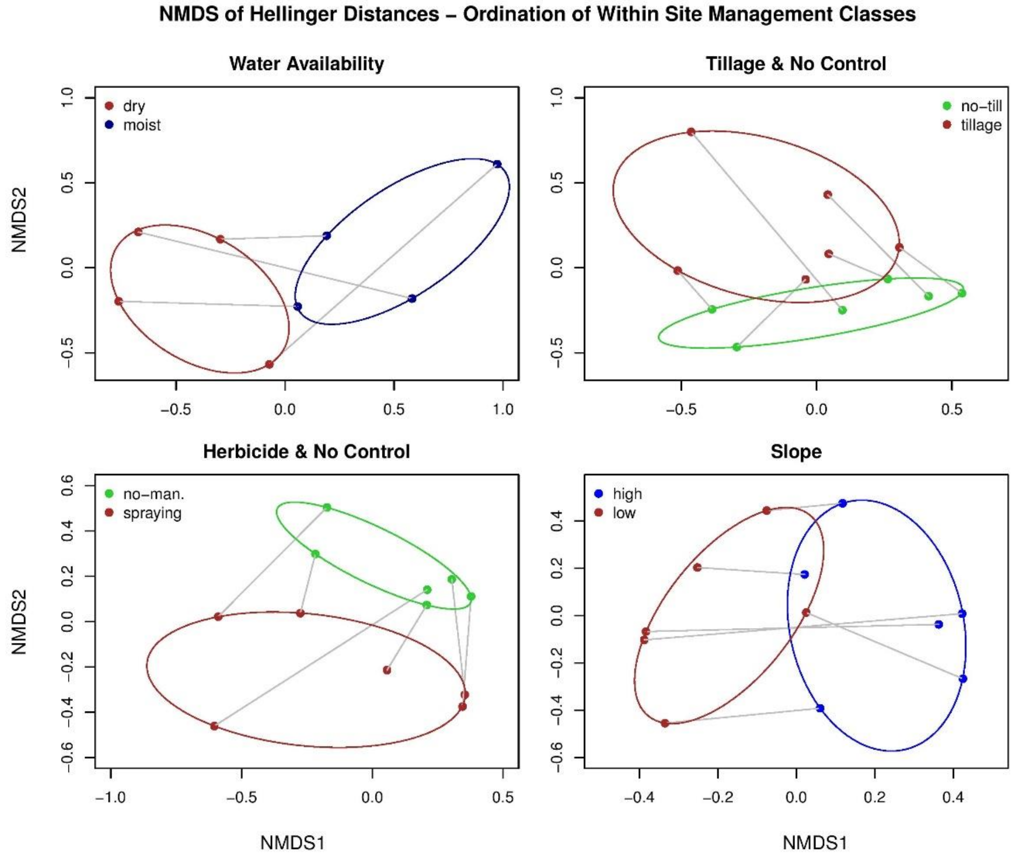


Figure 3. 12. NMDS of Hellinger distances – within site effect of management or habitat type classes on spider communities

Apart from the overall evaluations by covering all possible sampling units in the analyses, we also studied within site variability between sampling unit community compositions. We hypothesized that, there must be measurable directional change in community composition, due to management or habitat type differences between closely located sampling units in the same site. Such directional changes were detected for temporary surface water availability, tillage, herbicide spraying and alternating slopes within sites, resulting in significant differences between soil surface spider community composition.

A similar approach was to compare communities between cases of annual management shifts, which took place in particular sampling units, even though rarely.

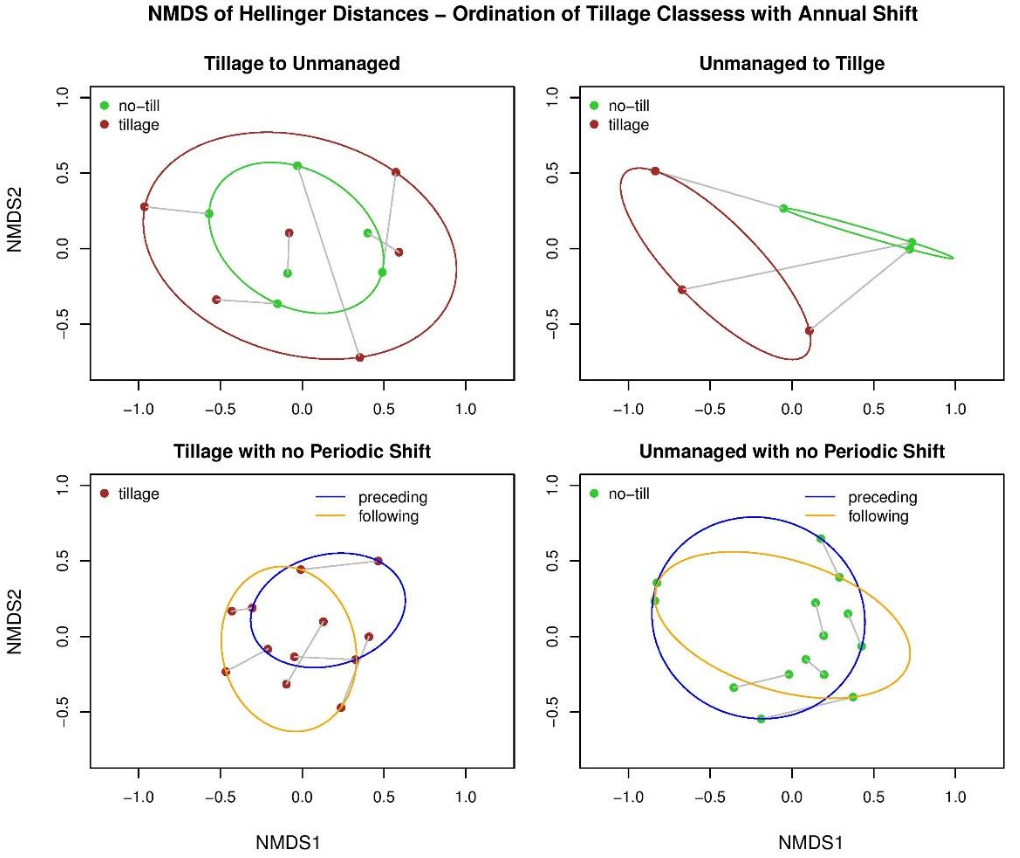


Figure 3. 13. NMDS of Hellinger distances – effects of management shifts on soil spider communities, in the same sampling units between periodic replicates

Directional change was not detected for tillage to unmanaged shifts as it was for the previous evaluations; but community dissimilarity decreased between sampling units after quitting tillage. Directional change was observed in unmanaged to tillage shifts however. Same type of analysis was also performed for olive groves with no periodic shift as control groups, resulting in no difference between sampling periods. Furthermore, dissimilarity between periodic replicates were higher in sampling units with management shifts, compared to no-shift sampling units.

Last two types of analyses indicated that directional, therefore predictable changes can occur due to agricultural practices in olive groves, however, these observations were severely limited in sample sizes, and further evaluations are required with controlled experiments.

3.7. Direct Gradient Analyses

This title includes two parts, and composed of assessments on environmental variability between different periods, and spider community – environment interactions. These topics were studied in PCA and RDA&CCA models s their results were summarized below.

3.7.1. Environmental Variability Between Sampling Periods

Field sampling of spider communities and environmental variables were performed at various periods. Spider community sampling was concentrated in spring in general, in April for vegetation samples (early season) and May-early June for soil surface samples (late season). Environmental variables associated to these community samples must have been representing the concurrent state of the environment when the community samples were collected, or the recent past. Environmental sampling was performed accordingly, either in early season or late season. Late season sampling was performed in May or August, assuming that environmental variability remains constant between these periods, as most of the management practices were performed in spring, between April – early May. One sampling occasion was performed in November, to be able to measure effects of mowing on habitat variables, as this activity was unique to late summer period.

Environmental variability in each sampling period was studied in PCA, indicating a major difference between early season and late season measurements of environmental variables.

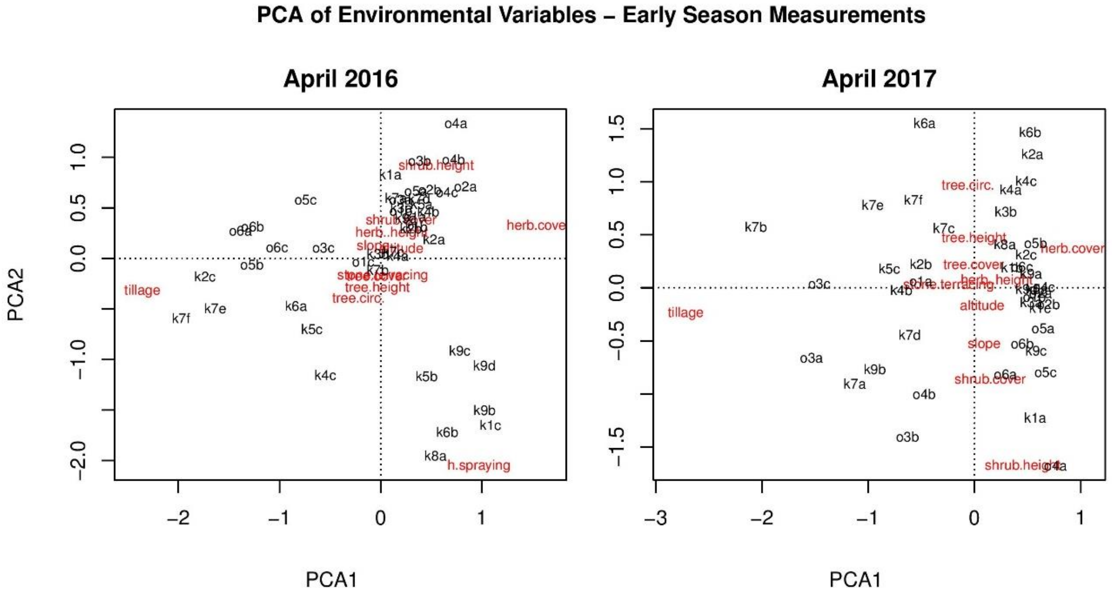


Figure 3. 14. PCA’s of environmental variables in early season

Environmental gradients resulting from management activities were not apparent in early season measurements, with only a few sites appeared as impacted by the management, while many other sites where still remaining unmanaged. Many of these latter type of sites or sampling units were managed later in the season, resulting in appearance of gradients only in the late season measurements.

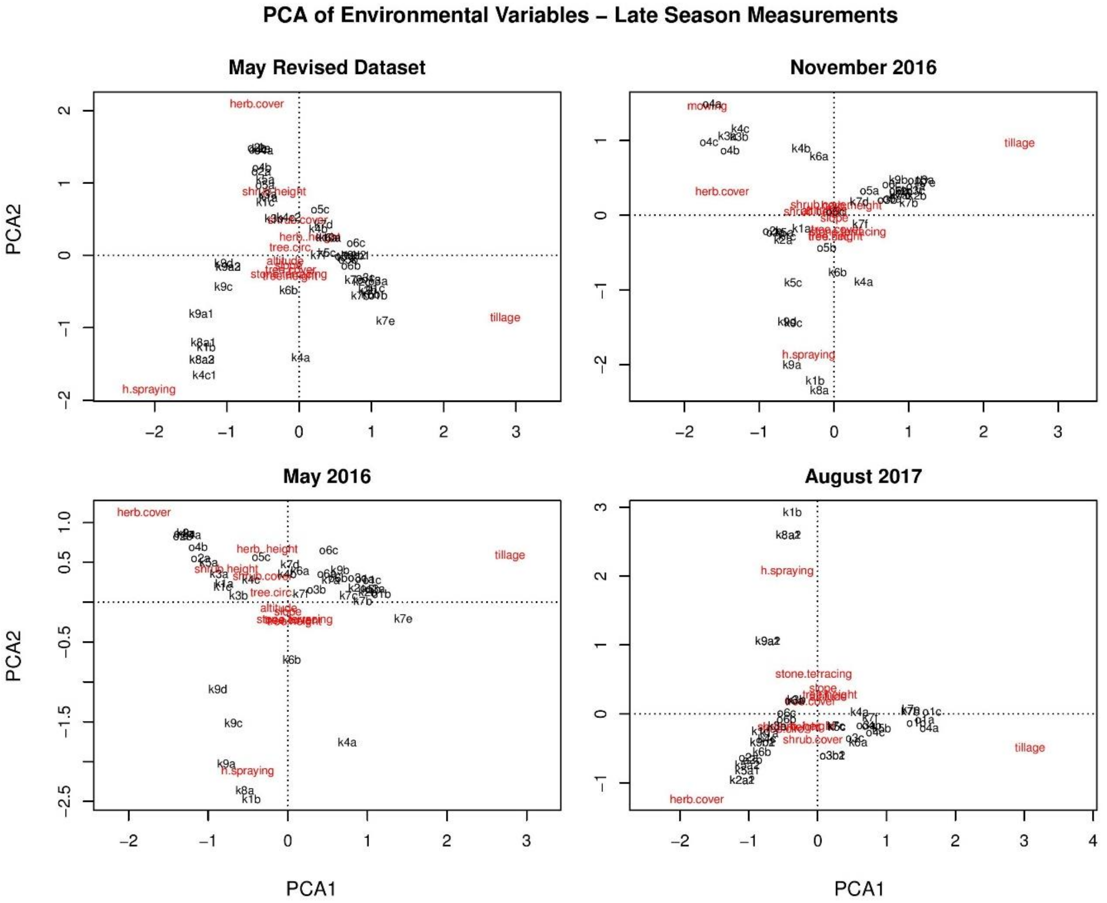


Figure 3. 15. PCA’s of environmental variables in late season

Another interesting finding from the PCA analyses was detection of habitat variables resulting from management practices with the strongest gradients in the datasets. This was not unexpected, as many of the habitat variables were limited in their variability during site selection procedures; and it also indicates that we can concentrate on the impact of management directly, rather than habitat variables, which was the main aim of our perspective.

3.7.2. Spider Communities – Environment Interactions

Direct gradient analyses were performed by using two types of multivariate methods, RDA and CCA modelling, with no significant difference between each other. Therefore, here we have reported the results only for the RDA models. These models plot sampling units in a multidimensional space (two dimensions were used here) in a similar fashion to ordination, but this time there is more weight to axis directions for interpretation of the results, rather than only the distance between points of sampling units.

RDA models were summarized in tri-plots, which indicate species (red dots) and environmental parameters (blue arrows) as well, besides the sampling units (green labels). Models were studied by using all of the environmental variables available, but we used only the significant variables for plotting, to be able get highly readable graphs. Distribution of species and sampling units in the two dimensional ordination plane were used to assess artifacts by the statistical methods (like the arc-effect), which were always resulted in cloud shaped distributions indicating that the evaluations were safe.

All RDA models were significant in the 0.05 significance, except one, which was significant in the 0.1 interval. None of the models explained more than 50% of the constrained variation, indicating that spider communities studied were effected by other variables as well, other than that we have measured. This is acceptable since our approach in this study was observational and descriptive, rather than experimental. More than 40% of the variation was explained by the first two PCA axes, therefore we have concentrated on these axes for interpretations of our data.

Table 3. 5. Summary of RDA statistics with given spider community and environmental datasets

Spider Community	Environment Dataset	Constrained Variation Prop.	Prop. Explained by First Two Axes****	Model Significance. (# perm: 999)
Tree Canopy	April 2017	0.38	0.50	0.07 ·
Herb. 2016	April 2016	0.32	0.44	0.01 **
Herb. 2017	April 2017	0.35	0.43	0.001 ***
Herb. 2017	November 2016	0.39	0.41	0.001 ***
Soil - Synthesis	Soil – Synthesis	0.32	0.44	0.001 ***
Soil 2017	August 2017	0.39	0.44	0.001 ***
Soil 2016	May 2016	0.37	0.46	0.002 **
Soil 2015.2	May 2015	0.31	0.44	0.037 *
Soil 2015.1	May 2015	0.33	0.42	0.001 ***

**** proportion of explained variation by the first two axes of ordination was given relative to total constrained variation.

Herbaceous vegetation covered surface area (herb.cover) was the most important predictor of the spider communities, as it was the only significant parameter in all of the models, except for the tree canopy spider communities. Soil surface spider communities unexceptionally effected by the herb. cover, but as well as from tillage and herbicide use too. Other significant parameters were tree cover (a proxy for canopy closeness), herbaceous vegetation height, slope and altitude.

It was not particularly surprising to find out tilled and herbicide sprayed surface area as significant predictors for soil surface spider communities, since these variables were natural negative correlates of the herb. cover in the dataset.

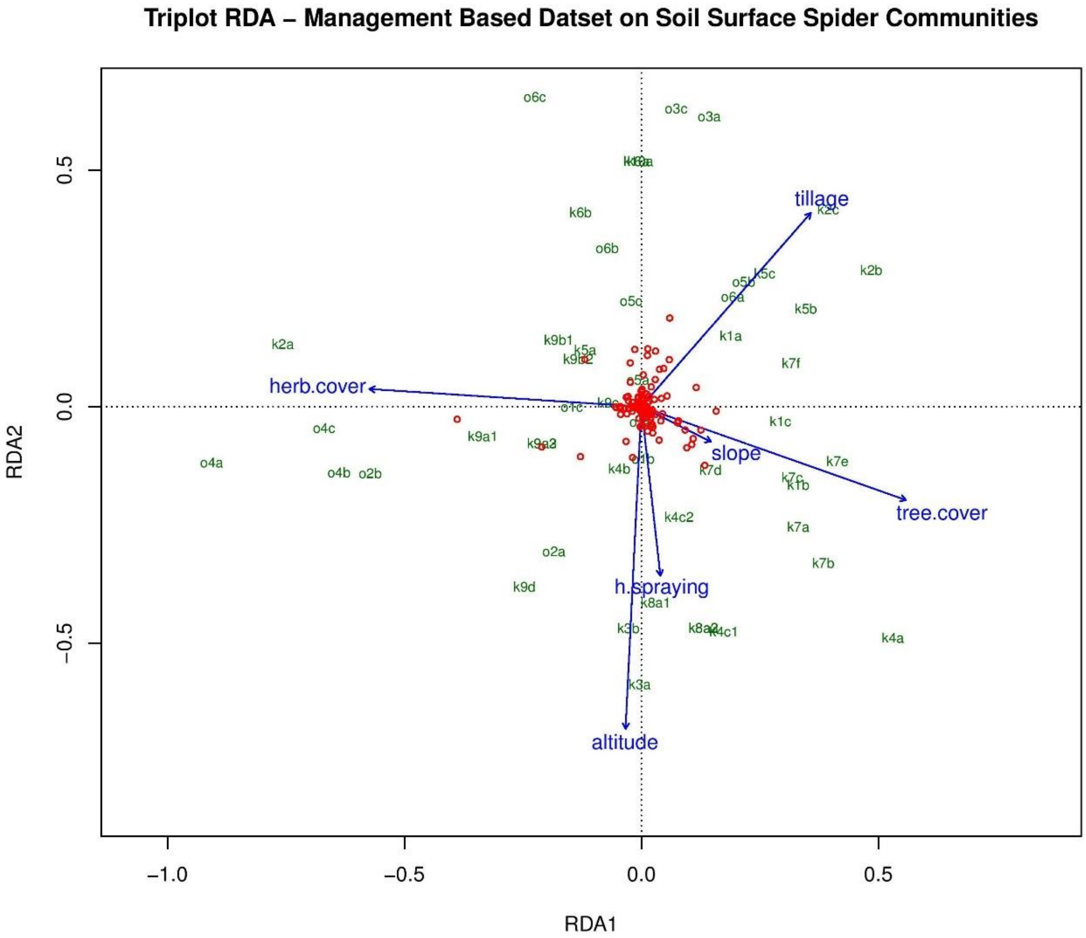


Figure 3. 16. Triplot RDA model for the management dataset, soil surface spider communities, also referred as synthesis

Furthermore, it has been already reported in the literature that tillage and herbicide use applications have significant impacts on terrestrial arthropod communities directly or indirectly (Thorbek & Bilde ,2004; Sunderland & Samu, 2000; Birkhofer et al., 2008b). What was new and interesting in our results was the finding that tillage and herbicide spraying had different impacts on the spiders, resulting in different spider communities on the soil surface.

Spider communities from other microhabitats were effected from different types of predictor variables. Shrub covered surface area was apparent in both of the microhabitats as a significant parameter. Herb. cover, tillage, tree cover and altitude were still significant predictors for herbaceous vegetation spider communities, whereas they were not significant for tree canopy. Tree canopy was the most distinct microhabitat in terms of significant predictor variables. Traits of olive trees, tree circumference and height appeared as important predictors on the spider communities, as well as shrub cover. Herb. cover was not a significant predictor for tree canopy spider communities, but included in the graph for comparison purposes.

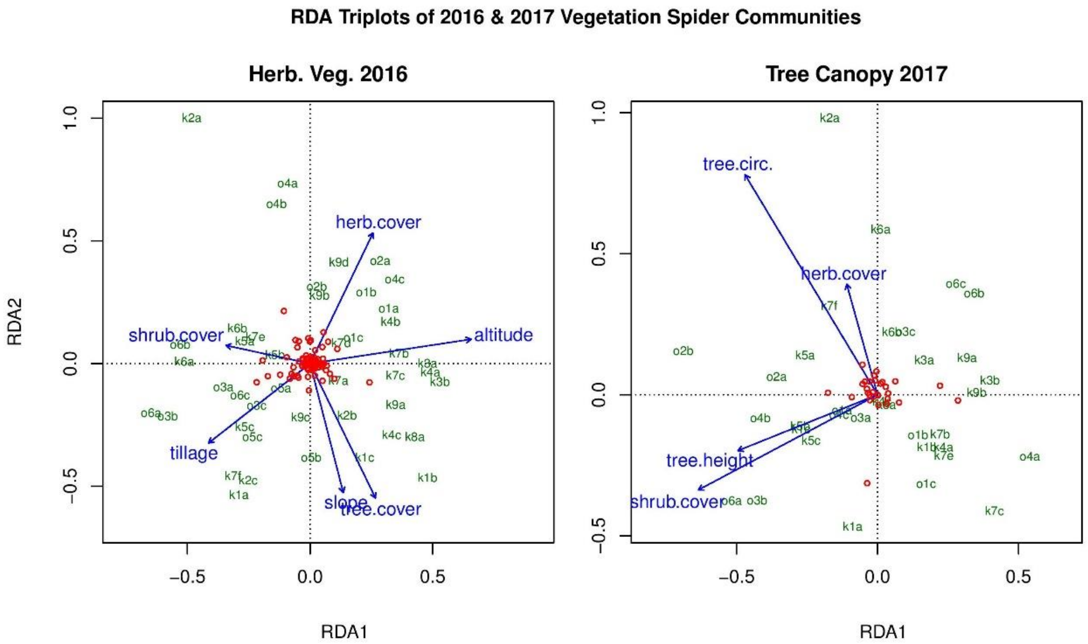


Figure 3. 17. Triplot RDA models for the herbaceous vegetation and tree canopy spider communities

Slope and altitude were appeared as significant predictors in most of the RDA models performed. These were the most difficult habitat parameters to justify by their impact on spider communities, since their variability and range were quite low or narrow between sampling units to cause distinct ecological conditions. Their role as predictors might have been caused from artifacts however, since some of the other variables were correlating with altitude and slope, as a result of anthropogenic activities. Some examples could be given as follows: a) herbicide use was more concentrated on sloping land due to difficulties associated to tillage at high slopes; b) olive trees were younger and densely planted at higher altitudes, whereas older with thicker trunks and low density at lower altitudes; c) some of the high slope or high altitude olive groves had lower management intensities, resulting in high shrub cover. Examples can be multiplied. Since it was difficult to control for these side effects, we fear that appearance of slope and altitude as important predictors was due to confounding effects from other variables

CHAPTER 4

CONCLUSION

With this study, we have performed one of the most effort intensive and detailed inventory studies on spiders ever made and discovered one of the richest spider communities that have ever been reported in the literature.

Knowledge and a better understanding on spider fauna of the study area had many important aspects, due to many reasons. Therefore, this knowledge can be used to answer a plenty of questions… Studied ecosystem is an agroecosystem and spiders are natural predators of insect pests and regarded to possess a huge potential for biological control. Study area has been regarded as a Pleistocene glacial refuge, which imply presence of many endemic species, which has never been studied in any arthropod group in detail before. Aegean Sea is one of the richest places in the world in terms of islands, but it has only recently become a focus of biogeographical research in terms of terrestrial arthropods.

We can count a plenty of more reasons and topics to emphasize the importance of knowledge on fauna and such work, which can be used to answer many more questions. But here in this work, we have limited our focus on two main questions: how spider communities vary naturally and what is the impact of anthropogenic activities on spider communities. We asked this question in the Mediterranean, and in an agroecosystem which has been under management for a long time, but still possessing natural value by harboring wildlife and ecosystem services, yet fragile and threatened in many aspects.

Olive has been one of the oldest known cultivated perennial cultivars at the Mediterranean coasts of Anatolia, but olive groves spread and reached their current distribution range during the last hundred years, as a result of governmental and agricultural policies of the Republic of Turkey. The range expansion took place by replacement of the native biotopes, such as maquis and pine forests. Even though olive is a natural component of the Mediterranean native vegetation, we have shown that the spider communities differ from the natural biotopes. By considering spiders as an indicator taxonomic group, it is possible to imply this pattern for the entire terrestrial arthropod taxa or biodiversity as a whole. This result must be an important consideration for biodiversity conservation at the Anatolian coast of the

Mediterranean. Maquis harbor evidently a unique type of biodiversity and must be protected by law, which is not currently protected outside of the protected areas.

We have shown how diverse spiders can be in an agricultural landscape, as well as how agricultural management can impact the spider communities directionally, by changing community composition and acting as an artificial force important on species sorting in the agri-environment. Management, as understood as main management types – organic or conventional – did not differ from each other, or at least we have failed to show the differences, as a result of lack intensive use of pesticides in the study area. However alternative weed control practices, tillage, herbicide use and mowing or no-management were found to have be strong impacts on the spider communities, acting as a disturbance.

Difference of olive grove spider communities from those of natural biotopes, or change in spider communities in olive groves by management intensification might be indicating increasing dominance of agrobiont species in the more intensive alternative. Agrobiont nature of spiders must be further examined by considering distributional patterns observed in the spiders. Chorotype concept can be an alternative way to study agrobioncy, but modern types of evaluations are also available, such as distribution modelling. Identifying agrobiont species from native or threatened ones will provide a new perspective on conservation decision giving, as well as it can be used for selecting and propagating biological control agents in the agroecosystems.

Spiders have been regarded with an obvious indicator role as previously mentioned, which has not yet been fully recognized. Use of spiders as indicators is still a developing field, and we need to find cheap alternative ways to use them for this purpose. This may include identifying certain taxonomic or functional groups in the spiders, which can be used as proxies for definite purposes. Population level evaluations are also considered necessary but missing alternatives. Comparative work must be performed together with other organism groups to evaluate indicator values or find out interactions which could be used as proxies for indicatorship.

All the results produced and discussed in this study were based on taxonomic diversity. Even though it has been the most fundamental approach in diversity studies, accepting all species as equal could hinder some of the most important answers, which could be provided by weighting species according to their roles. This problem can be resolved by means of functional ecology, which was a missing part of this work, but also one presumed to be addressed in the future efforts by the thesis author.

Finally, main approaches in this work have been designed to be observational and descriptive, which were resulted in significant findings, but confounded with many uncontrolled variables acting as sources of errors or reducing the amount of explained variation. Only way to get thorough these problem is controlled testing with experimentation. It is crucial to design and perform experiments testing and repeating the results reported here, before accepting them as fundamental rules

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APPENDICES

A. List of Species and Morphotypes Identified in This Study

Taxon or Morpho-Type Name	Chorotype
Agelenidae	
Agelena orientalis C. L. Koch, 1837	CEM
Maimuna vestita (C. L. Koch, 1841)	EME
Tegenaria dalmatica Kulczyński, 1906	MED
Tegenaria parietina (Fourcroy, 1785)	CEM
Tegenaria sp.1	Na
Amaurobiidae	
Amaurobius erberi (Keyserling, 1863)	EUR
Anyphaenidae	
Anyphaena sabina L. Koch, 1866	TEM
Araneidae	
Agalenatea redii (Scopoli, 1763)	ASE
Araniella cucurbitina (Clerck, 1757)	ASE
Araneus diadematus Clerck, 1757	SCO
Araniella inconspicua (Simon, 1874)	ASE
Araniella opisthographa (Kulczyński, 1905)	ASE
Argiope lobata (Pallas, 1772)	SCO
Cyclosa sierrae Simon, 1870	TUE
Gibbaranea bituberculata (Walckenaer, 1802)	ASE
Glyptogona sextuberculata (Keyserling, 1863)	EME
Hypsosinga albovittata (Westring, 1851)	ASE
Mangora acalypha (Walckenaer, 1802)	ASE
Neoscona adianta (Walckenaer, 1802)	ASE
Neoscona subfusca (C. L. Koch, 1837)	CEM
Zilla diodia (Walckenaer, 1802)	EME
Clubiobidae	
Clubiona vegeta Simon, 1918	CAM
Ctenizidae	
Cyrtocarenum cunicularium (Olivier, 1811)	PELO
Dictynidae	
Altella lucida (Simon, 1874)	TUE
Archaeodictyna consecuta (O. Pickard-Cambridge, 1872)	ASE
Dictyna kosiorowiczi Simon, 1873	WME
Dictynidae sp	Na
Lathys sp. n.	ANNW
Marilynia bicolor (Simon, 1870)	CEM
Nigma puella (Simon, 1870)	EME
Scotolathys simplex Simon, 1884	MED
Dysderidae	
Dasumia sp.	ANNW
Dysdera krisis Komnenov & Chatzaki, 2016	PELO
Dysdera lata Reuss, 1834	EUR

Appendice A. continues…

Dysdera longirostris Doblika, 1853	EEU
Dysdera romantica Deeleman-Reinhold, 1988	PELO
Dysdera sp. n. 1	ANNW
Dysdera sp. n. 2	ANNW
Harpactea alexandrae Lazarov, 2006	MACE
Harpactea camenarum Brignoli, 1977	PELO
Harpactea ice Komnenov & Chatzaki, 2016	PELO
Harpactea sp.1	Na
Eresidae	
Eresus walckenaeri Brullé, 1832	EME
Eutichuridae	
Cheiracanthium mildei L. Koch, 1864	TEM
Cheiracanthium pennyi O. Pickard-Cambridge, 1873	ASE
Filistatidae	
Filistata insidiatrix (Forsskål, 1775)	MED
Pritha nana (Simon, 1868)	MED
Gnaphosidae	
Anagraphis ochracea (L. Koch, 1867)	MACE
Anagraphis pallens Simon, 1893	SWA
Aphantaulax cincta (L. Koch, 1866)	TUM
Berinda ensigera (O. Pickard-Cambridge, 1874)	PELO
Callilepis cretica (Roewer, 1928)	PELO
Civizelotes caucasius (L. Koch, 1866)	ASE
Cryptodrassus creticus Chatzaki, 2002	PELO
Cryptodrassus hungaricus (Balogh, 1935)	SEU
Drassyllus crimeaensis Kovblyuk, 2003	EME
Drassodes cupreus (Blackwall, 1834)	EUR
Drassyllus jubatopalpis Levy, 1998	EME
Drassodes lapidosus (Walckenaer, 1802)	ASE
Drassodes lutescens (C. L. Koch, 1839)	CAM
Drassyllus praeficus (L. Koch, 1866)	ASE
Drassyllus pusillus (C. L. Koch, 1833)	ASE
Drassodes serratichelis (Roewer, 1928)	EME
Drassyllus villicus (Thorell, 1875)	EUR
Drassodes chybyndensis Esyunin & Tuneva, 2002	TUR
Gnaphosa lucifuga (Walckenaer, 1802)	ASE
Haplodrassus dalmatensis (L. Koch, 1866)	EUR
Haplodrassus signifer (C. L. Koch, 1839)	SCO
Lasophorus zografae Chatzaki, 2018	PELO
Leptodrassus albidus Simon, 1914	MED
Leptopilos levantinus Levy, 2009	EME
Micaria albovittata (Lucas, 1846)	ASE
Nomisia exornata (C. L. Koch, 1839)	TUE
Nomisia ripariensis (O. Pickard-Cambridge, 1872)	EME

Appendice A. continues…

Phaeocedus hebraeus Levy, 1999	ANAT
Poecilochroa furcata Simon, 1914	MED
Scotophaeus blackwalli (Thorell, 1871)	EUR
Scotophaeus scutulatus (L. Koch, 1866)	CEM
Setaphis parvula (Lucas, 1846)	MED
Talanites strandi Spassky, 1940	TEM
Trachyzelotes barbatus (L. Koch, 1866)	CEM
Trachyzelotes fuscipes (L. Koch, 1866)	CAM
Trachyzelotes lyonneti (Audouin, 1826)	CEM
Trachyzelotes malkini Platnick & Murphy, 1984	CAE
Trachyzelotes pedestris (C. L. Koch, 1837)	EME
Turkozelotes microb Kovblyuk & Seyyar, 2009	PELO
Zelotes babunaensis (Drensky, 1929)	PELO
Zelotes balcanicus Deltshev, 2006	EME
Zelotes cingarus (O. Pickard-Cambridge, 1874)	CAM
Zelotes harmeron Levy, 2009	EME
Zelotes prishutovae Ponomarev & Tsvetkov, 2006	EME
Zelotes cf. segrex	ANAW
Hahniidae	
Hahnia nava (Blackwall, 1841)	ASE
Linyphiidae	
Acartauchenius scurrilis (O. Pickard-Cambridge, 1873)	ASE
Agyneta pseudorurestris Wunderlich, 1980	MED
Alioranus pastoralis (O. Pickard-Cambridge, 1872)	EME
Araeoncus humilis (Blackwall, 1841)	ASE
Araeoncus rhodes Tanasevitch, 2011	PELO
Bolyphantes lamellaris Tanasevitch, 1990	EME
Canariphantes nanus (Kulczyński, 1898)	EME
Canariphantes zonatus (Simon, 1884)	MED
Centromerus albidus Simon, 1929	EUR
Ceratinella brevis (Wider, 1834)	ASE
Diplocephalus graecus (O. Pickard-Cambridge, 1873)	MED
Erigone dentipalpis (Wider, 1834)	ASE
Erigonoplus globipes (L. Koch, 1872)	EME
Erigonoplus spinifemuralis Dimitrov, 2003	EME
Frontinellina frutetorum (C. L. Koch, 1834)	CEM
Maculoncus parvipalpus Wunderlich, 1995	EME
Mecopisthes nasutus Wunderlich, 1995	PELO
Megalepthyphantes turkeyensis Tanasevitch, Kunt & Seyyar, 2005	ANAW
Neriene furtiva (O. Pickard-Cambridge, 1871)	EME
Oedothorax apicatus (Blackwall, 1850)	ASE
Palliduphantes byzantinus (Fage, 1931)	EME
Pelecopsis laptevi Tanasevitch & Fet, 1986	CAE
Prinerigone vagans (Audouin, 1826)	CEM

Appendice A. continues…

Tenuiphantes tenuis (Blackwall, 1852)	ASE
Theonina cornix (Simon, 1881)	EME
Trichoncoides piscator (Simon, 1884)	CEM
Walckenaeria alticeps (Denis, 1952)	ASE
Erigonolophus sp1	Na
Linyphiidae sp. 1	Na
Linyphiidae sp. 2	Na
Linyphiidae sp. 3	Na
Linyphiidae sp. 4	Na
Liocranidae	
Agraecina lineata (Simon, 1878)	MED
Agroeca parva Bosmans, 2011	EME
Mesiotelus scopensis Drensky, 1935	EME
Mesiotelus tenuissimus (L. Koch, 1866)	CEM
Liocranidae sp.1	Na
Liocranidae sp.2	Na
Liocranidae sp.3	Na
Lycosidae	
Alopecosa albofasciata (Brullé, 1832)	CEM
Arctosa tbilisiensis Mcheidze, 1946	TUE
Aulonia kratochvili Dunin, Buchar & Absolon, 1986	SWA
Hogna graeca (Roewer, 1951)	EME
Hogna radiata (Latreille, 1817)	CAE
Lycosa praegrandis C. L. Koch, 1836	CAE
Pardosa atomaria (C. L. Koch, 1847)	EME
Pardosa proxima (C. L. Koch, 1847)	CEM
Pardosa roscai (Roewer, 1951)	EME
Trabea paradoxa Simon, 1876	SEU
Mimetidae	
Ero aphana (Walckenaer, 1802)	CAE
Mimetus laevigatus (Keyserling, 1863)	CAM
Miturgidae	
Zora prespaensis Drensky, 1929	MACE
Nemesiidae	
Brachythele varrialei (Dalmas, 1920)	ANAW
Oecobiidae	
Oecobius maculatus Simon, 1870	MED
Oecobius rhodiensis Kritscher, 1966	EME
Oonopidae	
Oonopinus ionicus Brignoli, 1979	PELO
Orchestina sp.1	
Oxyopidae	
Oxyopes globifer Simon, 1876	CAM

Appendice A. continues…

Oxyopes heterophthalmus (Latreille, 1804)	CEM
Oxyopes lineatus Latreille, 1806	CEM
Oxyopes mediterraneus Levy, 1999	MED
Oxyopes nigripalpis Kulczyński, 1891	MED
Palpimanidae	
Palpimanus uncatus Kulczyński, 1909	EME
Philodromidae	
Philodromus cespitum (Walckenaer, 1802)	SCO
Philodromus krausi Muster & Thaler, 2004	PELO
Philodromus lunatus Muster & Thaler, 2004	EME
Philodromus rufus Walckenaer, 1826	SCO
Pulchellodromus medius (O. Pickard-Cambridge, 1872)	EME
Pulchellodromus pulchellus (Lucas, 1846)	MED
Rhysodromus hierosolymitanus (Levy, 1977)	CAM
Thanatus atratus Simon, 1875	ASE
Thanatus imbecillus L. Koch, 1878	TEM
Thanatus vulgaris Simon, 1870	SCO
Tibellus macellus Simon, 1875	CAE
Pholcidae	
Holocnemus pluchei (Scopoli, 1763)	EME
Pholcidae sp.1	ANNW
Spermophora senoculata (Dugès, 1836)	SCO
Phrurolithidae	
Phrurolithus sp. n.	ANNW
Pisauridae	
Pisaura orientalis Kulczyński, 1913	EME
Salticidae	
Chalcoscirtus infimus (Simon, 1868)	TUE
Cyrba algerina (Lucas, 1846)	CAM
Euophrys gambosa (Simon, 1868)	MED
Euophrys rufibarbis (Simon, 1868)	CAM
Euophrys sulphurea (L. Koch, 1867)	MED
Evarcha jucunda (Lucas, 1846)	MED
Habrocestum papilionaceum (L. Koch, 1867)	PELO
Heliophanus equester L. Koch, 1867	EME
Heliophanus kochii Simon, 1868	TEM
Heliophanus melinus L. Koch, 1867	SEU
Heliophanus tribulosus Simon, 1868	TUE
Icius hamatus (C. L. Koch, 1846)	MED
Leptorchestes berolinensis (C. L. Koch, 1846)	TUE

Appendice A. continues…

Macaroeris nidicolens (Walckenaer, 1802)	TEM
Menemerus semilimbatus (Hahn, 1829)	CEM
Menemerus taeniatus (L. Koch, 1867)	CEM
Neon convolutus Denis, 1937	CEM
Pellenes allegrii Caporiacco, 1935	TUR
Pellenes diagonalis (Simon, 1868)	EME
Pellenes flavipalpis (Lucas, 1853)	PELO
Philaeus chrysops (Poda, 1761)	CEM
Phlegra fasciata (Hahn, 1826)	ASE
Phlegra lineata (C. L. Koch, 1846)	MED
Pseudeuophrys obsoleta (Simon, 1868)	CAE
Pseudicius picaceus (Simon, 1868)	MED
Saitis tauricus Kulczyński, 1905	EME
Salticus noordami Metzner, 1999	EME
Salticus propinquus Lucas, 1846	MED
Synageles dalmaticus (Keyserling, 1863)	MED
Talavera aequipes (O. Pickard-Cambridge, 1871)	ASE
Thyene imperialis (Rossi, 1846)	CAM
Scytodidae	
Scytodes thoracica (Latreille, 1802)	CEM
Sicariidae	
Loxosceles rufescens (Dufour, 1820)	MED
Sparassidae	
Eusparassus walckenaeri (Audouin, 1826)	EME
Micrommata ligurina (C. L. Koch, 1845)	CAM
Theridiidae	
Anatolidion gentile (Simon, 1881)	MED
Crustulina scabripes Simon, 1881	MED
Dipoena galilaea Levy & Amitai, 1981	EME
Dipoena melanogaster (C. L. Koch, 1837)	EME
Enoplognatha afrodite Hippa & Oksala, 1983	SEU
Enoplognatha gemina Bosmans & Van Keer, 1999	MED
Enoplognatha latimana Hippa & Oksala, 1982	SCO
Enoplognatha macrochelis Levy & Amitai, 1981	EME
Enoplognatha mariae Bosmans & Van Keer, 1999	EME
Enoplognatha thoracica (Hahn, 1833)	SCO
Episinus truncatus Latreille, 1809	EUR
Euryopis episinoides (Walckenaer, 1847)	CAM
Euryopis sexalbomaculata (Lucas, 1846)	EME
Heterotheridion nigrovariegatum (Simon, 1873)	CAE
Kochiura aulica (C. L. Koch, 1838)	EME

Appendice A. continues…

Lasaeola convexa (Blackwall, 1870)	MED
Lasaeola minutissima Wunderlich, 2011	MED
Neottiura herbigrada (Simon, 1873)	CAM
Neottiura uncinata (Lucas, 1846)	MED
Parasteatoda tepidariorum (C. L. Koch, 1841)	SCO
Phylloneta impressa (L. Koch, 1881)	SCO
Platnickina nigropunctata (Lucas, 1846)	MED
Simitidion agaricographum (Levy & Amitai, 1982)	EME
Simitidion simile (C. L. Koch, 1836)	SCO
Steatoda albomaculata (De Geer, 1778)	SCO
Steatoda paykulliana (Walckenaer, 1806)	CEM
Theridion adrianopoli Drensky, 1915	EME
Theridion cinereum Thorell, 1875	CAE
Theridion cf. cyprusense	PELO
Theridion melanurum Hahn, 1831	EME
Theridion mystaceum L. Koch, 1870	ASE
Theridiidae sp.1	Na
Theridiidae sp.2	Na
Theridiidae sp.3	Na
Theridiidae sp.4	Na
Thomisidae	
Heriaeus simoni Kulczyński, 1903	EME
Monaeses israeliensis Levy, 1973	TUR
Ozyptila confluens (C. L. Koch, 1845)	MED
Ozyptila sp.1	ANAW
Ozyptila sp.2	Na
Ozyptila sp.3	Na
Pistius truncatus (Pallas, 1772)	ASE
Runcinia grammica (C. L. Koch, 1837)	CEM
Synema globosum (Fabricius, 1775)	CEM
Synema plorator (O. Pickard-Cambridge, 1872)	CEM
Thomisus onustus Walckenaer, 1805	ASE
Tmarus piochardi (Simon, 1866)	MED
Xysticus acerbus Thorell, 1872	ASE
Xysticus caperatus Simon, 1875	MED
Xysticus cribratus Simon, 1885	CEM
Xysticus kochi Thorell, 1872	ASE
Xysticus laetus Thorell, 1875	TUE
Xysticus tenebrosus Šilhavý, 1944	EME
Xysticus thessalicus Simon, 1916	EME
Xysticus thessalicoides Wunderlich, 1995	PELO

Appendice A. continues…

Xysticus tristrami (O. Pickard-Cambridge, 1872)	SWA
Xysticus sp.1	Na
Titanoecidae	
Nurscia albomaculata (Lucas, 1846)	CEM
Nurscia albosignata Simon, 1874	EEU
Uloboridae	
Uloborus walckenaerius Latreille, 1806	CAE
Zodariidae	
Palaestina expolita O. Pickard-Cambridge, 1872	EME
Zodarion beroni Komnenov & Chatzaki, 2016	PELO
Zodarion bigaense Bosmans, Özkütük, Varli & Kunt, 2014	ANNW
Zodarion morosoides Bosmans, 2009	PELO
Zodarion thoni Nosek, 1905	EME
Zoropsidae	
Zoropsis lutea (Thorell, 1875)	EME
Unidentified sp.1	Na
Unidentified sp.2	Na
Unidentified sp.3	Na

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WORK EXPERIENCE

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FOREIGN LANGUAGES

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PUBLICATIONS

In International Peer Reviewed Journals

1) Kunt, K. B., Özkütük, R. S., **Elverici, M.** 2011 A survey of East Mediterranean *Dasumia* Thorell, 1875 (Araneae, Dysderidae) with description of new species. Zookeys, 137: 89–101.
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4) Kunt, K. B., Kaya, R. S., Özkütük, R. S., Danışman, T., Yağmur, E. A., **Elverici, M.** 2012 Additional notes on the spider fauna of Turkey (Araneae). Turkish Journal Of Zoology, 36: 637-651.
5) Ribera, C., **Elverici, M.**, Kunt, K. B., Özkütük, R. S. 2014 *Typhlonesticus gocmeni* sp n., a new cave-dwelling blind spider species from the Aegean region of Turkey (Araneae, Nesticidae). Zookeys, 419: 87–102.
6) Kunt, K. B., Özkütük, R. S., **Elverici, M.** 2013 *Harpactea ballarini* sp nov., a new dysderid (Araneae, Dysderidae) spider from Turkey. Turkısh Journal Of Zoology, 37: 238-241.
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9) Özkütük, R.S., Marusik, Y.M., **Elverici, M.**, Kunt, K. B. 2016 A new species of *Lathys* from Turkey (Araneae, Dictynidae). ZooKeys, 1(632), 35-45.
10) Özkütük, R.S., Kunt, K.B., Marusik, Y.M., **Elverici, M.** 2017 Taxonomic notes on two sibling species of Metellina from Asia (Araneae, Tetragnathidae). ZooKeys, 2(695), 75-88.
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12) Özkütük, R. S., **Elverici, M.,** Kunt, K.B., Kılıç, G. 2017 First faunistic record of *Tegenaria vankeerorum* Bolzern, Burckhardt Hänggi, 2013 (Araneae: Agelenidae) from Turkey with description of unknown female. Biological Diversity and Conservation, 3(10).

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1) Kunt, K.B., Yağmur, E.A., **Elverici, M.** 2008 The cave dwelling arthropods of Dim cave (Turkey: Antalya: Alanya). Munis Entomology & Zoology 3 (2): 682-690.
2) Özkütük, R.S., Kunt, K.B., **Elverici, M.** 2012 *Dictyna uncinata* Thorell 1856, a new record for spider fauna of Turkey (Araneae, Dictynidae). Biological Diversity and Conservation, 5 (3): 24-27.
3) Özkütük, R.S., Kunt, K.B., **Elverici, M.** 2013 First record of *Centromerus albidus* Simon, 1929 from Turkey (Araneae: Linyphiidae). Munis Entomology & Zoology, 8 (1): 502-504.
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6) Özkütük, R.S., Kunt, K.B., **Elverici, M.** 2013 Türkiye Faunası İçin Yeni Bir Örümcek Türü; *Agroeca cuprea* Menge, 1873 (Araneae:Liocranidae). Biyoloji Bilimleri Araştırma Dergisi, 6 (1): 29-32.
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2) **Elverici, M.**, Kunt, K.B. 2011. *Cyrtocarenum cunicularium* (Araneae, Mygalomorphae, Ctenizidae) Türünün Mevsimsel Aktivitelerinin Belirlenmesi ve Yaygın Yer Örümcekleri ile Olan İlişkisinin Ekolojik Niş Hipotezi Kapsamında Değerlendirilmesi. 10. Ulusal Ekoloji ve Çevre Kongresi, Çanakkale, 4-7 Ekim 2011, poster presentation.
3) Özkütük, R.S., Kunt, K.B., **Elverici, M.** 2011. Türkiye'den yeni bir *Amaurobius* (Araneae, Amaurobiidae) türü. 10. Ulusal Ekoloji ve Çevre Kongresi, Çanakkale, 4-7 Ekim 2011, poster presentation.
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2) **Elverici, M.**, Bilgin, C., Kence, A. 2014 Compositional Changes in Ground Spider Fauna over the Seasons: An Additional Case from the Mediterranean. 1. International Ecology and Evolutionary Biology Symposium, İstanbul, 12-13 July 2014, oral presentation.
3) **Elverici, M.**, Bilgin, C., Kence, A. 2014 Compositional Change in a Mediterranean Ground Spider Community Over the Seasons. 28. Avrupa Araknoloji Kongresi, Torino, İtalya, 24-29 August 2014, oral presentation.
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5) **Elverici, M.**, Karaçetin, E., Bilgin, C. 2017 Comparison of Spider Communities from Olive Groves and Native Mediterranean Biotopes, New Data from the Aegean Coast. XIII. Environment and Ecology Congress with International Participation, Edirne, oral presentation.
6) Karaçetin, E., Balkız, Ö., Bilgin, C., **Elverici, M.**, Güçlü, H.,,Karabacak, P., Turak, A., Demirbaş-Çağlayan, S., Vural, M., Bekyürek, Y. 2017 Organik vs Conventional? How do butterfly communities respond to farming type and agricultural practices in olive groves in Ayvacık, Çanakkale, Turkey?. 4. International Ecology and Evolutionary Biology Symposium, oral presentation.
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9) **Elverici, M.**, Karaçetin, E., Bilgin, C. 2018 Exploring the Little Known Spider (Arachnida: Araneae) Community of the Olive Grove Ecosystem from Southern Foothills of Kaz Mountains, Çanakkale & Balıkesir, Turkey. International Ecology Symposium, Kastamonu, oral presentation.
10) **Elverici, M.**, Karaçetin, E., Bilgin, C., Balkız, Ö., Demirbaş-Çağlayan, S., Vural, M. 2018 Drivers of Spider (Arachnida:Araneae) Biodiversity in an ‘Olive Grove Ecosystem’ from North-Western Anatolia (Çanakkale and Balıkesir, Turkey). 4. International Ecology and Evolutionary Biology Symposium, oral presentation.

HOBBIES

Hiking, Swimming, Snorkelling, Climbing, Herpetology, Speleology, Aquarium Keeping